WETLAND WASTEWATER TREATMENT SYSTEMS:

A NEW ZEALAND BASED REVIEW

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- by -

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ABSTRACT

Natural and constructed wetlands have proven capable of providing a high standard of renovation to wastewater and their use as engineered treatment systems is rapidly developing in New Zealand and overseas. This thesis begins by examining the nature of the wetland environment and outlines the renovating processes it contains. The different types of wetland treatment systems are reviewed. Discussion continues into design and management principles which include physical and process design and hydrological and biotic considerations. Finally a review is made of the application of wetlands to the treatment of domestic sewage, non-point pollution, and industrial wastewaters. The conclusion is reached that wetland treatment systems offers an innovative and appropriate solution to a wide range of New Zealand's waste treatment problems and are likely to become an increasingly common treatment option in this country.

PREFACE AND ACKNOWLEDGEMENTS

This thesis concludes approximately eighteen months of reviewing literature, finding and interpreting data and interviewing prominent workers in this field. I sincerely thank all those people who sent me their reports and papers, and those who gave me their time and shared their knowledge and experiences.

My thanks to Dr Rao Bhamidimarri, my supervisor, for guiding me through my masterate studies, and to both the Department of Biotechnology at Massey University and the Waste Tecynology Group at MAF Invermay for support and assistance. Also special thanks to all those who helped type and prepare this work.

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INTRODUCTION

Conventional waste treatment technology has developed in the metropolises of Europe and the United States. Primarily designed to reduce organic pollution within the confines of limited space, they are intensive processes necessitating skilled design and careful operation. They have been generalised as concrete and steel structures (USEPA, 1988) with a high degree of mechanical engineering and significant civil engineering requirements (Smith, 1989). Because these processes are well developed and extensively proven in the field, there has been a tendency for engineers to apply such systems universally to all wastewater problems. Today, however, there is a growing awareness amongst engineers that these conventional technologies do not always offer the best solution. There is an increasing recognition of the need, and indeed the benefits of applying innovative and alternative waste treatment technologies. One alternative, growing in popularity, is the use of wetland treatment systems.

Wetlands have been receiving wastewater discharges for up to one hundred years (Venus, 1987). However it is only recently that their potential as efficient and cost effective treatment systems has been realised (Smith, 1989). In 1953, Kathe Seidel first presented research into the use of aquatic plants for pollution control in a report published by the Max Planck Institute (Seidel, 1976). Since this time the research and development of wetlands as waste treatment systems has rapidly increased. Even so, wetlands are still acknowledged to be a developing technology, with many of the key processes and design considerations poorly understood. Nevertheless, confidence in these systems is such that hundreds of them are now in operation around the world.

Wetlands have been proven capable of providing a high degree of secondary and tertiary treatment and appear to have a number of attributes which make them ideal for application to the New Zealand situation. Indeed the number of these systems that are already in operation in New Zealand is comparatively high by international standards. Although there is significant interest in this technology in New Zealand there has been very little information published that clearly outlines how to design these systems, and practically no reports have been forthcoming on just how successful the existing New Zealand systems are.

This thesis begins with a review of the wetland environment, and the chemical, physical and biological characteristics that make it unique. The second chapter examines how these characteristics are able to renovate the wastewater that is passed through the wetland. This is followed by an introduction and description of the various types of wetland treatment systems. Chapters four, five, six and seven then discuss the design principles and considerations required to successfully create and operate a constructed wetland treatment system. Case studies are reviewed in chapter eight, where the efficiencies that have been achieved and the performance levels of wetlands with various types of wastes are highlighted. Finally the use of wetland treatment systems is summarised and conclusions are drawn as to their application in New Zealand.

By investigating what has already been achieved in this country and reviewing the latest results and findings from international research and development, this thesis provides up-to-date design guidelines and examines the suitability of wetland treatment systems to the New Zealand situation. It is hoped that this independent university based assessment will aid in the ongoing development of wetland treatment systems in New Zealand.

1. THE PROFILE OF A WETLAND ENVIRONMENT

Wetlands are naturally occurring ecosystems. Their use as waste treatment systems represents a new approach in wastewater treatment technology. Most conventional systems involve developing a naturally occurring renovating process into a high rate, highly engineered waste treatment system. A constructed wetland, on the other hand, involves the engineering of an entire natural environment which has been found capable of providing excellent treatment for wastewaters.

Although a wetland is inexpensive to build and simple to operate, understanding the host of characteristics and mechanisms that result in its wastewater treatment is complex. To date most of the literature reports that wetlands have proven successful for waste renovation, but there is limited information as to why this is so. Consequently many engineers are designing and constructing wetland systems whilst not fully understanding the processes of these systems. An optimal design can only be achieved if the concept of a wetland is understood in its entirety, including knowledge of the components and characteristics that are considered integral to the successful treatment of wastewaters. This chapter discusses the general wetland environment, its importance, and the features that make wetlands unique. The objective of this being to provide a basic understanding of this renovating environment.

1.1 Defining a Wetland Environment

Wetlands are areas transitional between aquatic and terrestrial environments, the water level being at or near the soil surface (USFWS, 1979). Consequently wetland soils are subjected to either saturation or flooding long enough to assume hydric characteristics.

Hydric soils are types of soils that lack atmospheric gases for extended periods of time. However, the presence of these soils does not necessarily infer a wetland environment. The area must also be inhabited by water-loving or hydrophytic vegetation (NWF, 1987).

A commonly used scientific definition of a wetland is "land that has a predominance of hydric soils and that is inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances does support, a prevalence of hydrophytic vegetation typically adapted for life in saturated soil conditions" (The US Environmental Protection Agency and the US Army Corps of Engineers, 1977, in NWF, 1987, p4).

1.2 Types of Wetlands

The variation of environments that may be defined as wetlands is enormous. The USFWS (1979) classification system is the most commonly used and defines five major groups as Marine, Estuarine, Riverine, Lacustrine and Palustrine, as illustrated in figure one.

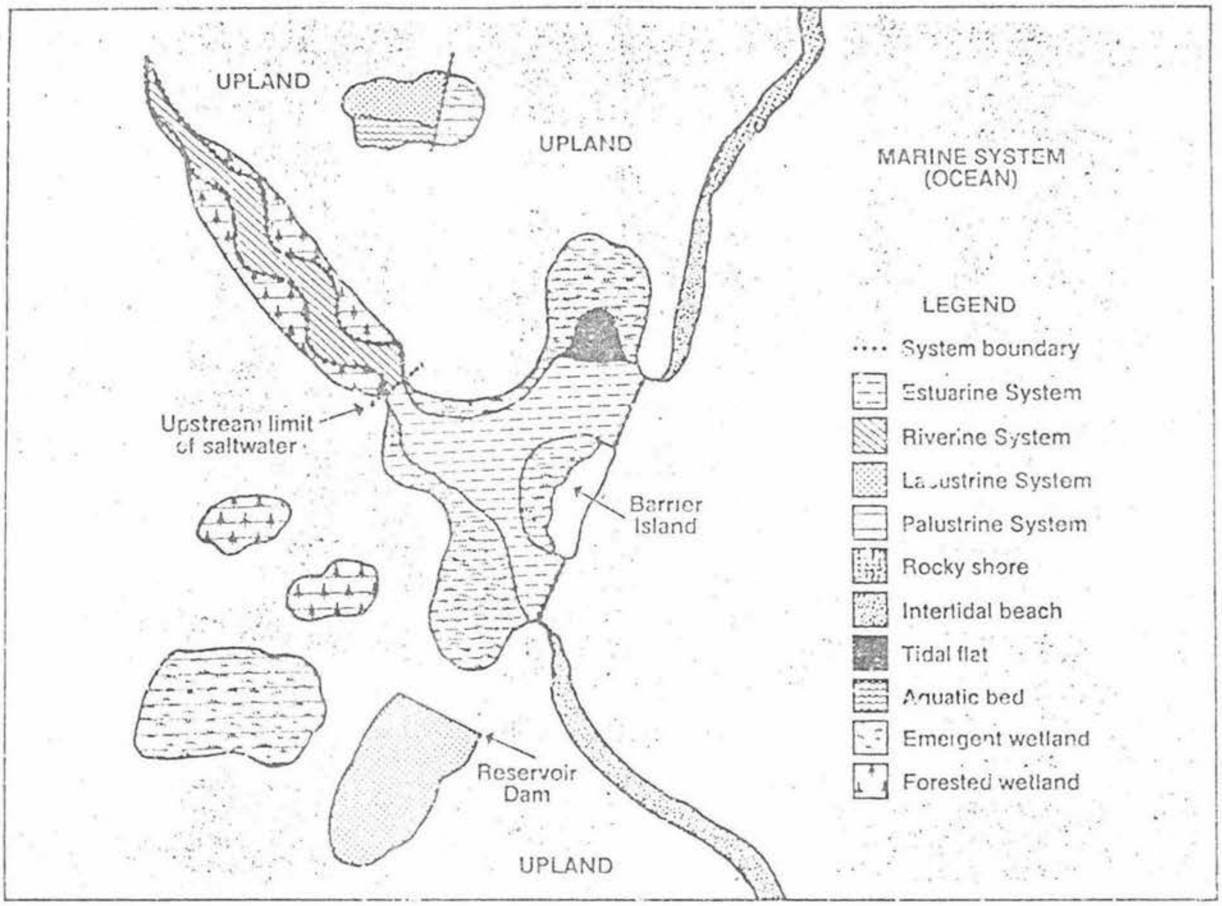

Figure 1. Major Wetland Systems

(Tiner, 1984, in NWF, 1987, p16)

Estaurine and Palustrine wetlands are the most abundant. According to NFW (1987), Estaurine areas encompass most coastal brackish wetlands, for example, tidal salt marshes, mangrove swamps, intertidal flats and coastal rivers. Palustrine systems account for the majority of freshwater wetlands such as marshes, bogs, wet meadows, swamps and shallow ponds.

Emergent wetlands are found in both Estuarine and Palustrine systems. When found in the latter they are more commonly known as marshlands. These emergent wetlands are typified by erect, rooted, herbaceous flowering plants. The construction or modification of marshlands in order to facilitate wastewater renovation is what is generally referred to as a wetland treatment system.

1.3 Importance of Wetlands

Wetlands are a vital element of the greater environment in which they occur. They are one of the most prolific parts of the food chain. With high nutrient levels (washed in from the surrounding land) and abundant water, they are highly productive. This has, however, made them very popular for conversion to farmland (Hammer and Bastian, 1989). This abundance of food production naturally supports an abundance of consumers. Wetlands teem with consuming animals from microbes to mammals, which feed on the vegetation and on the other consumers (Hammer & Bastian, 1989).

Wetlands are rich in biological diversity being an essential element of the lifecycles of countless animals and plants. However vast areas of this environment both here and overseas have disappeared due to drainage and land reclaimation (Johnson and Brook, 1989). Within the last one-hundred years, over 70% of New Zealand's natural wetlands have been drained or grossly modified (Tanner & Clayton, 1990). This destruction has had a devastating effect on many of the wetland dependent species. For example, in the United States an alarming 31% of all the threatened and endangered bird species' habitats are in some way dependant on wetlands (Williams & Dodd, 1979, in Feierabend, 1989). Clearly the wetland habitat is of great importance in the conservation of biological diversity.

The diverse vegetation and wildlife of wetlands makes them popular for a number of recreational pursuits. Johnson and Brooke (1989) refer to them as containing "much to explore, discover, interpret, and enjoy" (p.21), and indeed they are popular for photography, nature studies, and birdwatching. Additionally wetlands offer several sporting activities. Fishing for eels and whitebait, and hunting for waterfoul, in particular duck shooting, are popular pursuits in the wetlands of New Zealand. The importance of wetlands for recreation can perhaps be quantified by the fact that in the United States they form the basis of over $10 billion in annual expenditures on nature study, fishing, hunting and other outdoor recreational pursuits (NWF, 1987).

Wetlands also have a number of important hydrological attributes. Because of their large volume they basically act as an uncontrolled reservoir and, as such, modify floods travelling down river channels. The flood flow enters the system creating a rise in water level and an increase in outflow. Because of the storage capacity the outflow rate is less than the inflow which effectively reduces the downstream peak level. This is known as flood routing. The NWF (1987) reports that a large natural wetland upstream of Boston, Massachusetts is now preserved for this very purpose after its loss was calculated to be likely to result in a cost of $1.7 million in annual flood damage.

Finally, a number of wetlands provide essential recharges to ground water reservoirs. A large wetland outside the American city of Amherst, for example, recharges an aquifer at a rate of eight million gallons per day providing most of the water supply for the city (NFW, 1987).

1.4 Aquatic Environment

The aquatic environment of a wetland is very quiescent in nature. There are three main reasons for this. Firstly, as inflowing water enters the comparatively huge volume of a wetland its flow velocity decreases proportionally to the increase in cross-sectional area of flow. Secondly, the macrophytes slow the current velocity by their frictional drag (Gaudet, 1974). And finally, the thick plant coverage shelters the water column from wind mixing, and absorbs and dissipates any wave action that does form during flood flows (Gosselink and Turner, 1978).

Because of the quiescent environment and dense vegetation, solids in the runoff accumulate in the wetland system due to sedimentation and filtration. Thus wetlands are naturally rich in organic matter and, with its subsequent decomposition, are nutrient enriched.

The degree of inundation varies throughout the range of wetland environments. Swamps and marshes, for example, are practically continually flooded, whereas riparian wetlands on the edges of streams and rivers may have infrequent and/or short periods of saturation (Faulkner and Richardson, 1989).

The main oxygen pathway into a wetland is via atmospheric diffusion, but some oxygen also enters in the inflowing water, and via plant rhizomes. As the decomposition of the organic material in the aquatic environment exerts a biochemical oxygen demand, the dissolved oxygen concentration in the water is reduced (Sloey et al., 1978). Thus dissolved oxygen concentrations are generally rather low in wetlands. However, it has been determined that the oxygen content is directly related to flow velocities. In general, velocities > 10mm/s tend to retain a low level of oxygen saturation, while at velocities below 4mm/s the oxygen tends to be depleted (Sparling, 1966). Low oxygen concentrations lead to the aquatic environment becoming partially anaerobic. The resulting production of methane gas is actually a distinctive characteristic of marshlands as it creates an illuminesence known in folklore as will-o'-the-wisp.

1.5 Wetland Soils

Wetland substrata range in particle size from clays, silts and sands to gravels and boulders (Haslam et al., 1982). The organic content also varies considerably, with some peatlands, for example, being practically totally organic. The pH of the soil in wetlands generally tends to the acidic, reading as low as 3.2 for a highly organic bog to a reasonably neutral 6.4 for typical Typha and Scirpus marshes (Richarson et al., 1978).

The flooded state of wetland soils creates an anoxic environment which results in a larger variety of chemical and biological characteristics than would be found in acidised soils (Patrick and Mikkelsen, 1971, in Gosselink and Turner, 1978). Although the general soil environment is anaerobic and reducing, there are a number of thin, oxidised, aerobic layers such as (a) over the solid surface, (b) in zones of high porosity (Aomine 1962, in Chamie and Richardson, 1978), and (c) around the oxygen leaking roots. In fact the roots are often stained brown by Fe^{3+} hydroxy-oxides which have been formed by oxidation of Fe^{3+} by the leaking oxygen (Etherington, 1983).

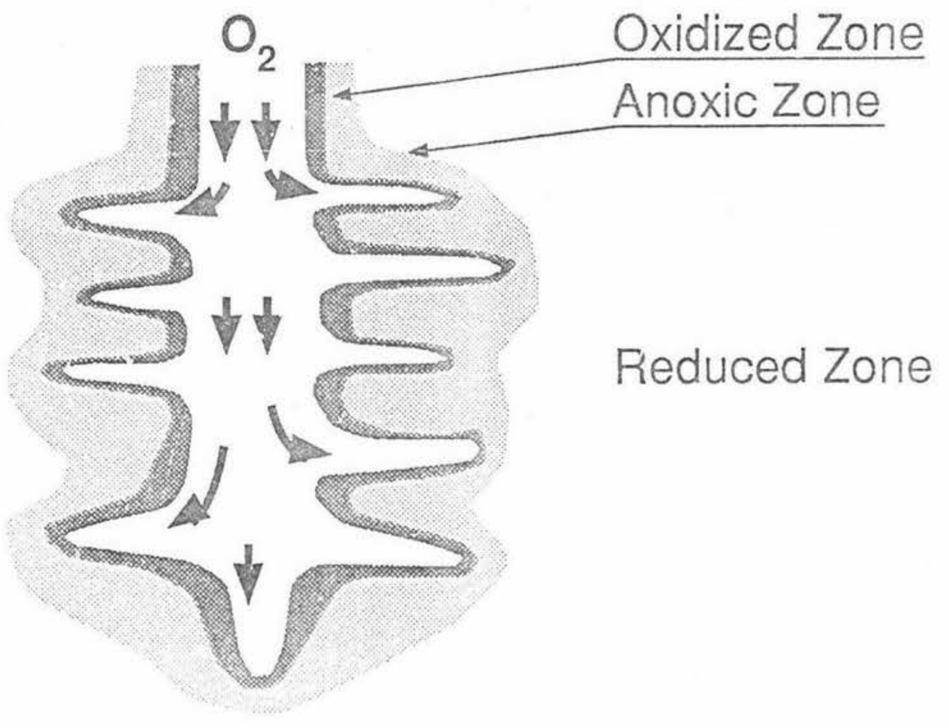

Figure 2. Illustration of Redox-Conditions Around Roots of Wetland Plants.

(Brix, 1987, p110).

1.6 Wetland Vegetation

Over five thousand species of plants inhabit wetlands (NWF, 1987). Wetland species have evolved from terrestrial plants that modified their nature in order to colonise the wetland niche. Except for woody species, the emergent macrophytes are the least modified, with floating and then submerged plants, respectively, having higher numbers of modifications and fewer of the structural and functional characteristics of the ancestral terrestrial plants (Guntenspergen et al., 1989). Emergent, floating and submerged plants have all been tested and/or used for wastewater treatment (discussed further in Chapter 3). It is, however, emergent plant species that have been used in what are commonly known as wetland treatment systems.

Because the flooded condition of wetland soils result in an anoxic environment wetland plants have developed alternative methods of respiring or obtaining oxygen (Gosselink and Turner, 1978). One of the characteristic features of aquatic plants is the formation of lacunae and/or aerenchyma tissue (Gore, 1983). Although lacunae provide support, their main functional significance is as a pathway for oxygen transportation throughout the plant, and in particular, the submerged or buried parts of the plant (William and Barber, 1961, in Brix 1988). Brix (1988) investigated this mechanism by extracting gas samples of the lacunal air in *Phragmites australis.* The results showed a steep oxygen gradient from top to bottom, and an opposite gradient for CO_2 and N_2 which is consistent with the concept of gas transport by passive diffusion. However, Brix (1988) found the effects of sunlight accelerated the process. Grosse (1988) documents a phenomenon termed thermoosmosis which Brix (1988) believes is responsible for this increase in ventilation. Grosse (1988) has discovered many wetland species use thermoosmosis which is basically driven by a temperature differential. As a consequence of this internal gas transport and the resulting high internal pressures (up to 1.1 KPa) (Stengel and Schultz-Hock, 1989), oxygen leaks from the roots into the surrounding substrate. This creates the oxidized and anoxic zones discussed and illustrated in the previous section.

Emergent wetland plants have extensive root systems. The roots and hollow rhizomes of *Phragmites australis*, for example, grow very densely and penetrate up to 1.5 metres into the substratum (see Figure 3). This in combination with the effects of the leaking oxygen creates a very large volume of active rhizosphere.

Figure 3.The Roots and Rhizones of *Phragmites australis*

(Brix, 1989, p102).

The stems and foliage of the emergents rise up from the bed, through the water column and into the atmosphere. As mentioned, wetland species are vigorous growers and some species can grow so dense that it actually blocks water flow through the wetland. However, growth rates and indeed species diversity varies depending on the site conditions and the climate.

In classifying wetland plants Haslam et al (1982) refers to five key factors that influence the success of a particular species:

(i) flow (water velocity);

(ii) type of water body (pond, stream, marsh and so on);

(iii) nutrient availability;

(iv) soil type;

(v) site (depth, light availability, wind exposure and so on).

Some species are particularly well adapted to certain environments and therefore tend to dominate. This is often the case with *Phragmites spp.* and *Typha spp.* in swamps and marshlands (Mitchell, 1974).

1.7 Wetland Chemistry

Apart from the biochemical reactions (discussed in section 1.8), adsorption and precipitation reactions are of the most interest, in relation to waste treatment.

1.7.1 Adsorption

Adsorption reactions in wetlands involve the attachment or binding of an absorbate compound in the aquatic solution onto an absorbent surface within the wetland. Adsorption rates are, however, strongly dependent on the pH and temperature. In general it is found that as temperature increases adsorption decreases. It is possible that with increased temperatures substances absorbed at lower temperatures may be desorbed back into the solution if they have not been immobilised or degraded. The optimum pH for adsorption of a given contaminant is when the pH is equal to the dissociation constant, the pKa. The optimum pH for many substances is around 7, and since the wetland is a biological system and, as such, has a natural buffering ability, adsorption rates can be high.

1.7.2 Precipitation

Contaminants in the ionic form react with compounds present in the wetland system to form an ionic precipitate. Phosphorous, for example, may be removed by chemical precipitation with various multivalent metal ions such as calcium (Ca^{2+}), aluminium (Al^{3+}), and iron (Fe^{3+}), as illustrated below (Metcalf & Eddy, 1985, p 262).

Calcium :

$$10Ca^{++} + 6PO_4^{3-} + 2OH^- \rightleftharpoons \underset{\text{Hydroxylapatite}}{Ca_{10}(PO_4)_6(OH)_2}$$

Aluminum :

$$Al^{3+} + H_nPO_4^{3-n} \rightleftharpoons AlPO_4 + nH^+$$

Iron :

$$Fe^{3+} + H_nPO_4^{3-n} \rightleftharpoons FePO_4 + nH^+$$

As precipitates are solids they are subsequently removed by sedimentation and filtration. Increases in temperature speed up these reaction rates. Thus the reduced adsorption capacity at higher temperatures is offset to some degree by the resulting increase in precipitation.

1.8 Wetland Microbiology

The organically rich wetland environment abounds with microbiology. Microbes play an essential role in completing the nutrient and energy flows of a wetland ecosystem. By consuming and decomposing the waste organic material microbes free nutrients and consume vegetable energy, forming the base of the animal food chain (Etherington, 1983).

Where oxygen is constantly present obligate aerobic bacteria such as *Pseudomonas spp.* will be found decomposing organic material. However, where these strict aerobes are eliminated due to anoxic conditions, their place is taken by facultative or obligate anaerobes (Etherington, 1983).

A requirement of normal bacterial respiration is the presence of an electron acceptor. This is provided by:

a) Using oxygen present in the water entering the wetland, oxygen diffusing in from the atmosphere, or oxygen leaking from plant roots. In these zones ammonia (NH_3) is also oxidised to form nitrite (NO_2) or nitrate (NO_3) by *Nitrosomonas* and *Nitrobacter* bacteria.

b) Utilising the combined oxygen of other compounds in the wetland as electron acceptors, resulting in their regeneration.

Bacteria such as *Achromobacter* and *Bacillus*, for example, are active denitrifiers using nitrate nitrogen as an oxygen source reducing it to nitrous oxide (NO) or molecular nitrogen (N_2), both of which can then escape from the wetland environment as gas. However, free oxygen strongly inhibits these bacteria, even in concentrations as low as 0.1 to 0.2% V/V the rate drops to 10% of that in a fully anaerobic environment (Etherington, 1983). In anaerobic environments many anaerobes also respire via a fermentative mechanism.

A large proportion of microbial respiration in dry soils is attributable to filamentus fungi. However, most fungi are obligate aerobes and thus there are very few in a wetland. Their effects are greatly outweighed by those of the bacteria (Brock, 1966).
Algaes are also found in wetlands. However, their populations are very limited due to the thick vegetative cover which limits the amount of available sunlight.

Many Protozoa, Rotifers, and Nematoda are also found in the water column, saturated soil and litter of the wetland. These small invertebrates consume and comminute other micro-organisms, vegetable detritus and the faeces produced by higher animals.

2. THE RENOVATING PROCESSES

The previous chapter has provided a basic insight into the wetland environment and the chemical, physical and biological characteristics and processes that make this environment unique. The aim of this chapter is to examine how particular aspects of this environment facilitate the renovation of wastewater.

2.1 The Renovating Medium

In order to appreciate the treatment processes in a wetland and their significance to wastewater renovation, the important aspects of the wetland environment that are encountered by the waste stream must be reviewed.

Wetland treatment systems generally have theoretical hydraulic resident times of 6 to 7 days. During this time the effluent enters the system and becomes a calm and slow moving flow. The dense vegetation and substrate within a wetland creates a 'packed media' treatment system. In a natural or free surface wetland the flow encounters an environment thickly packed with stems and foliage. A layer of litter lines the bottom of the wetland and there are also pockets of litter entangled in the vegetation above. In a subsurface wetland the packing is even more dense with the effluent flowing through a bed of rock and/or soil substrate consolidated with the roots and rhizomes of the wetland plants.

Media encountered by the waste stream is both chemically and biologically active. Many types of substrate and organic compounds within the system have the potential to react with, and retain elements of the waste flow via precipitation and adsorption reactions. In addition, the environment is teeming with biological activity, including aerobic and anaerobic bacterial slimes that colonise and cover the vegetative and physical media. These microbes consume pollutants which results in their stabilisation.

2.2 Organic Decomposition

As previously discussed, wetland microbes and animals feed on the organic debris naturally found in a wetland. The populations of these decomposers expand proportionally to consume the additional organic load of the wastewater.

The organic load of the wastewater consists of both solid and dissolved organic material and is often quantified as BOD (biochemical oxygen demand). BOD is a measure of the potential oxygen depletion via biochemical oxidation of organic matter by micro-organisms in the water column (Metcalf & Eddy, 1985).

The solid organic material is colonised by bacteria and entrapped in the wetland by the processes of sedimentation or filtration. Larger organic particles may be scavenged by wetland animals resulting in further stabilisation and comminution.

Dissolved organics are adsorbed and consumed by both aerobic and anaerobic species of bacteria. Wetlands are normally 'plug-flow' in nature and, as such, little contribution is made by single bacterial cells and bacterial flocs suspended in the water column. This is because without recycling, they are washed through the system before developing significant concentrations. Instead the bacteria grow as attached films on the physical and vegetative media in contact with the waste stream.

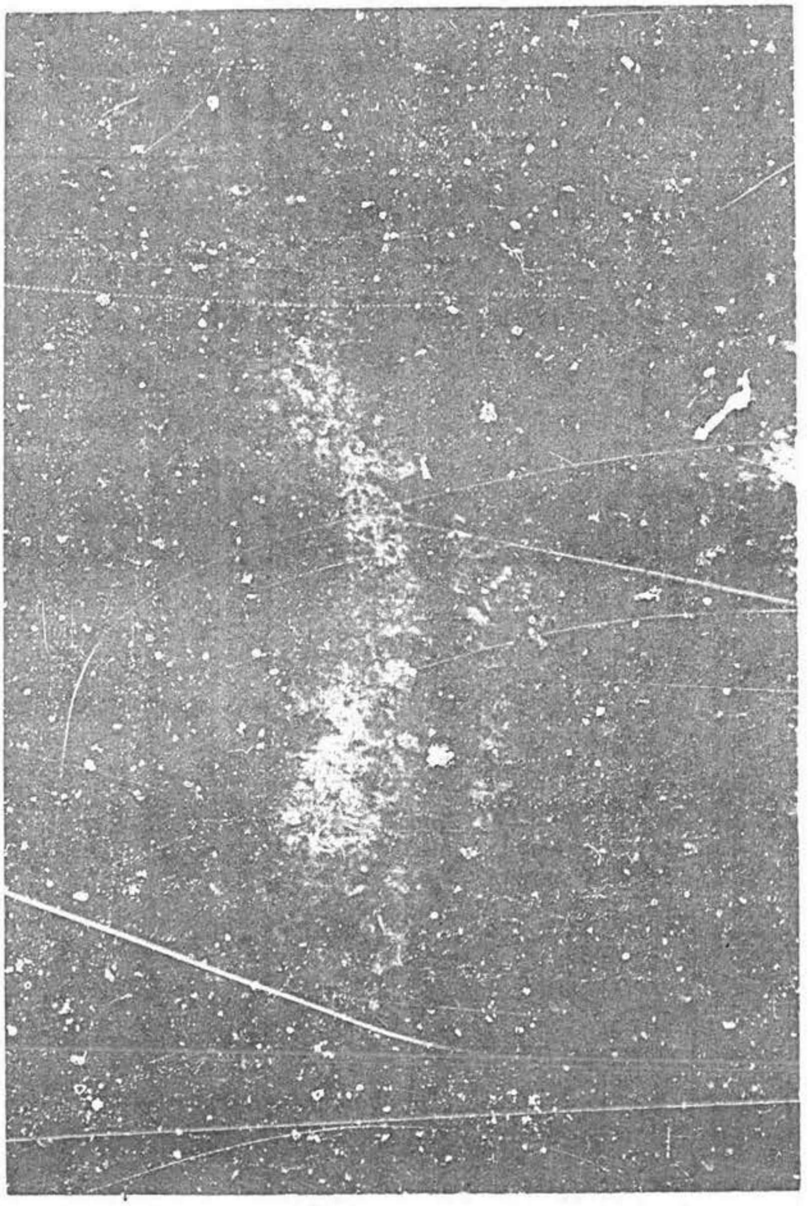

Figure 4. Microbial Colonisation of the Submerged Section of a *Typha* Shoot

Due to the flow being slow and quiescent, the system is packed with the bacterial covered media and there is good opportunity for contact and diffusion of the soluble organics into the bacterial film. In this bacterial film the organics are degraded into stable waste products which in turn diffuse out. A portion of the energy released goes towards new cell growth (approximately 48-52% for aerobic bacteria, but only 3-5% for anaerobic bacteria), the remaining energy being dissipated from the system as it is consumed for maintenance metabolism (Bhamidimarri, 1989).

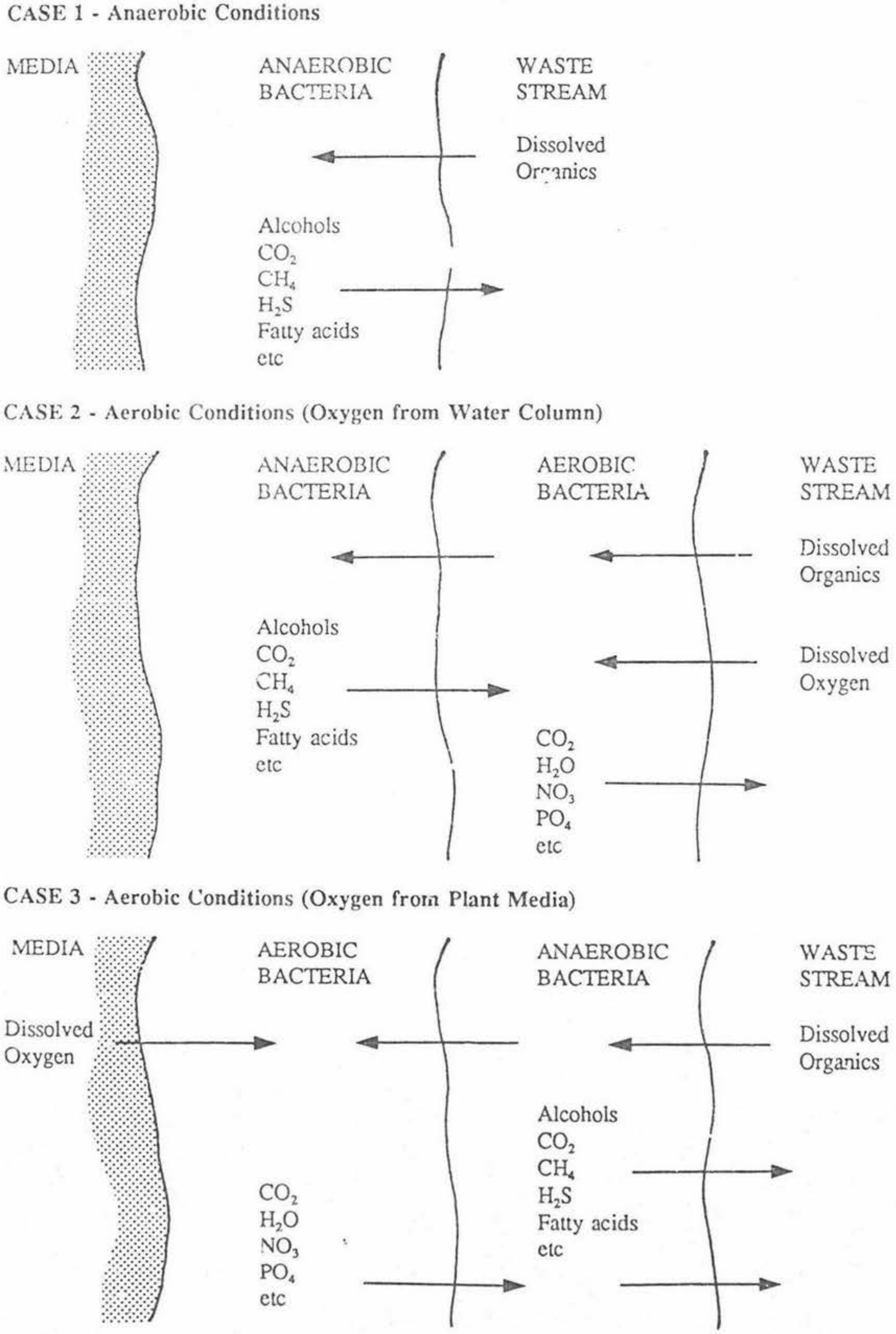

Figure 5. **The Three Variations of Organic Stabilisation by Bacterial Films in a Wetland**

As the bacterial films grow they become less stable until they eventually slough off the surface to which they were attached. They are then subject to the same process of degradation as organic solids.

2.3. Oxygenation

Dissolved oxygen is essential for aerobic organisms with habitats below the water level. Aerobic bacteria consume organics at about ten times the rate of the anaerobes and, because of this, are the key species in many of the conventional biological waste treatment processes. However, as aerobic bacteria have high cell mass yields the disposal of this excess biological cell mass can be a problem. Anaerobes produce about one tenth of the yield, but they are less effective at reducing BOD, produce some foul waste products and are inherently less stable because of complex metabolic reactions. In addition to organic decomposition aerobic bacteria also facilitate the nitrification of organic nitrogen. Thus oxygenation of the wetland water medium is an important concern in order to support aerobes that aid the waste treatment processes.

Dissolved oxygen enters a wetland via several pathways. These can be seen as follows:

(i) influent;

(ii) diffusion across air/water interface;

(iii) algae respiration;

(iv) loss of oxygen from root zone.

The effluent entering a wetland may contain a significant level of dissolved oxygen. This varies depending on the type of waste and the extent and method of pretreatment. This contribution is easily measured at the entrance to the wetland system.

Oxygen diffusion across time air/water interface appears the most significant oxygen pathway in surface flow wetlands. Because an oxygen concentration differential exists across the air/water interface, oxygen dissolves into the water column and diffuses down through the aquatic environment. At the same time, because the higher concentration of oxygen near the surface is more efficient at reducing the BOD, dissolved organics diffuse upwards. This same process will occur in sub-surface wetlands where a wet film from the water level is in contact with the atmosphere. However, as this pathway is more restricted, it is less efficient than a surface flow wetland.

Algae respiration is an important source of oxygen in pond systems. However, algae growth is limited in wetlands due to screening of the sunlight by the dense plant cover (USEPA, 1988), and filtration by the wetland media.

The loss of oxygen from the rhizomes of wetland plants, as discussed in the previous chapter, is a particularly unique and interesting method of oxygenation. This pathway was reputed to be the major oxygen source for subsurface wetlands (USEPA, 1988). This pathway has been thoroughly discussed in the literature as being one of the key elements of the wetland renovation process. However, Brix (1990) studied the oxygen balance about *Phagmites australis* in a three year old reed bed system and found that the "respiratory oxygen consumption of the roots and rhizomes almost perfectly balanced the oxygen influx through the culms", and concluded that the "macrophyte-induced rhizosphere oxygenation was therefore of no quantitative importance for aerobic BOD degradation and microbial nitrification" (p40). However these findings conflict with data published by Moorhead and Reddy (1988) who report oxygen transfer rates of up to 1.39 g 0_2 kg^{-1} root mass h^{-1} for cattails and up to 1.72 g 0_2 kg^{-i} root mass h^{-1} for pickerelweed. From their studies they also concluded that "as the root mass increased (with growth) the 0_2 transport rate decreased" (pp 142). This may therefore explain Brix's findings, and if correct indicates that oxygenation via the rhizomes is likely to become less significant in terms of wastewater renovation unless regular replantings are undertaken.

From consideration of these pathways a hypothesis can be reached that a wetland will have the highest dissolved oxygen (DO) levels near the water surface, around the entry of the influent, and surrounding oxygen leaking roots. The ratio of the aerobic zone over the anoxic zone will vary between different systems and depends on the BOD loading, and the method and rate of oxygenation. An effective design requires a good balance of the two. The aerobic zone will achieve high organic reduction and nitrify ammonia while oxidising (and thus avoiding the release of) unpleasant anaerobic waste products. The anaerobic zone will slowly degrade the litter and trapped organic waste, reducing build up of sediments and also contributing to BOD reduction. Furthermore, an anaerobic environment is necessary for successful denitrificatiion.

2.4 Filtration and Sedimentation of Suspended Solids

Filtration and sedimentation are the processes which enable wetlands to successfully remove fine particles of organic and inorganic material that are suspended in the waste stream. These are commonly termed suspended solids. Apart from contributing to the BOD and nutrient loads, suspended solids cause receiving waters to become cloudy and can smother the delicate environment found at the bottom of lakes, rivers and streams.

The dense media encountered by the suspended solids forces them to follow a "tortuous path" around the many obstacles (Venus, 1987, p22). The particles collide with the substrate, the vegetation, or with other particles, becoming entangled or losing momentum and settling out of the flow. Stowell et al (1981) believes they may also develop bacterial growths that assist in settling the very fine collodial particles. The effect the vegetation filtration has on the suspended solids removal has been documented by Gearheart et al. (1982 in Venus 1987), who discovered that suspended solid removal increased with plant growth. However, Venus (1987) notes that while wetlands are effective at trapping suspended solids, they also produce small quantities themselves. Plant detritus and other natural wetland wastes will emerge from the wetland system. It is important to distinguish between the "wetland derived" and "effluent derived" wastes particularly in terms of water right applications and in dealing with the general public (Venus, 1987, p22). When added to existing pond systems, wetlands have been found to be useful at removing BOD and suspended solids loads produced by the growth of algae. In Figure 6 the jar in the middle contains effluent from an oxidation pond prior to entering a wetland while the jar on the right contains water leaving the wetland.

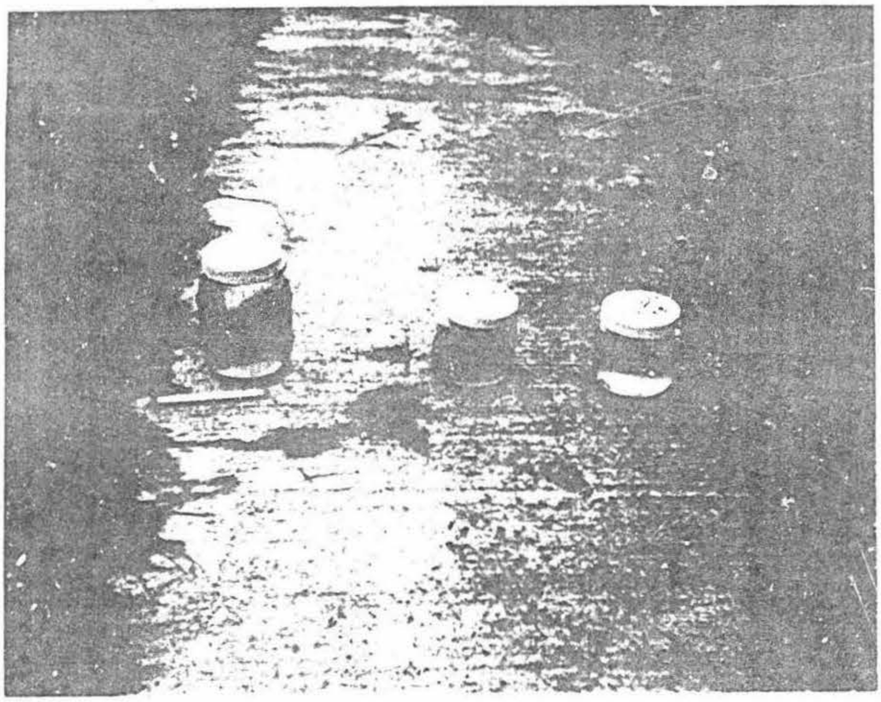

Figure 6. Algae Removal by a Wetland

2.5 Nitrogen Removal and Stabilisation

Nitrogen removal in a wetland is believed to be primarily due to the nitrification/ denitrification reactions (Stowell et al., 1981). In addition to nitrification/denitrification there are several secondary removal mechanisms as listed below:

(i) Uptake by the subsequent harvesting of plant biomass (discussed further in Chapter 6).

(ii) Volatilisation of ammonia. Ammonia exists in a liquid state in the wastewater and at normai operating temperatures is prone to volatilisation to gas, the gaseous state therefore escaping into the atmosphere. However, this process is only significant for effluents with a pH greater than 7 (Metcalf and Eddy, 1985).

(iii) Entrapment of solids. The physical entrapment of solid wastes by filtration and sedimentation remove nitrogen and other nutrients that are bound to or are actually part of the solid material. However, if these solids are biodegradable the removal is only temporary as the nutrients will eventually be released back into the flow.

As discussed in chapter one, the wetland environment contains nitrifying bacteria in the aerobic regions and dentrifying bacteria in the anaerobic regions which oxidise and reduce nitrogen leading to its eventual escape as gas. These reactions are detailed in Figure 7 which illustrates the nitrogen flow through a wetland.

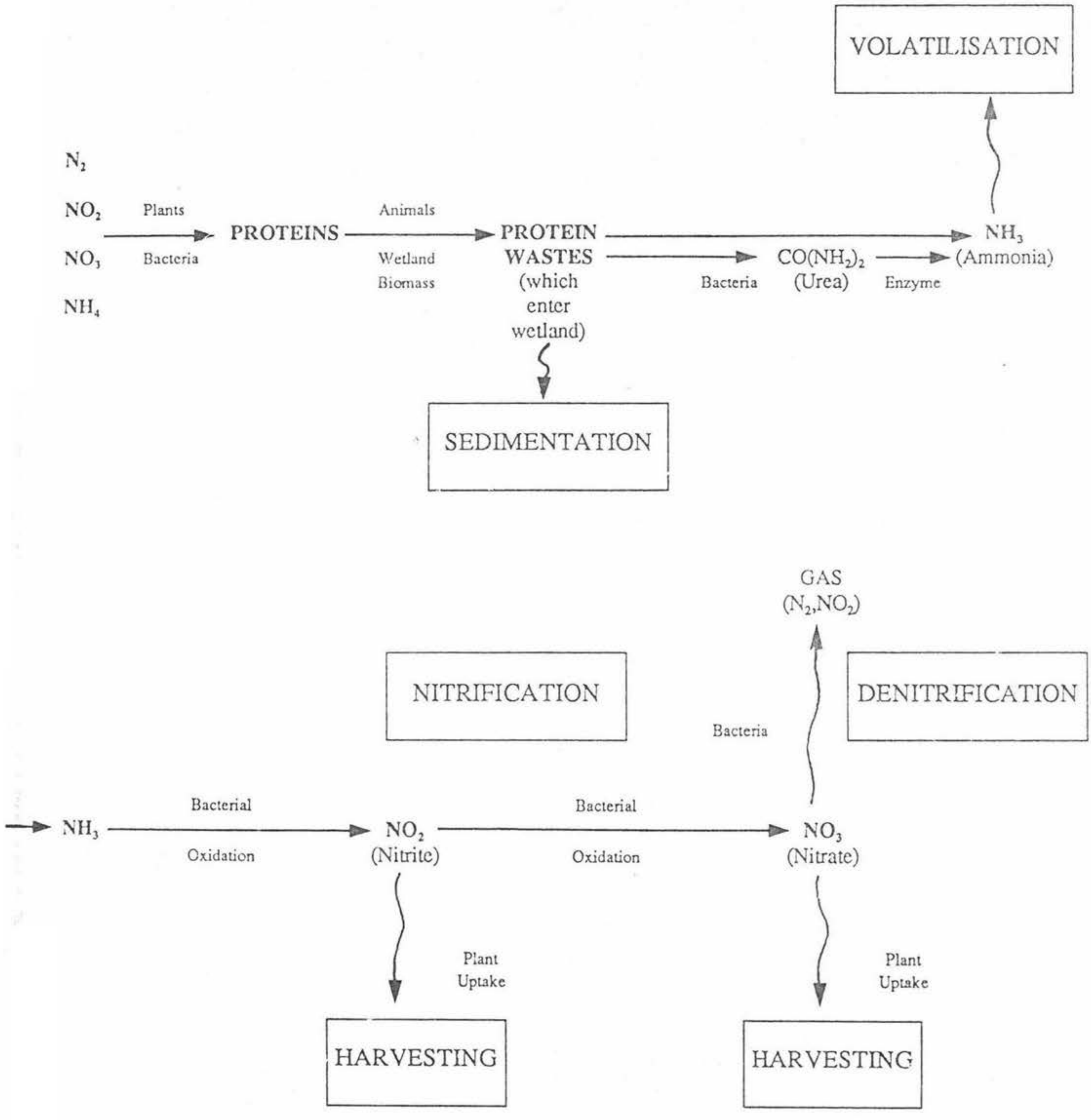

Figure 7. The Nitrogen Fow Through a Wetland Treatment System.

2.6 Phosphorus Removal

In a wetland phosphorous is removed from the waste stream by three methods:

(i) plant uptake and subsequent harvesting (discussed further in chapter 6);

(ii) entrapment of solids (as for nitrogen); and

(iii) adsorption and precipitation reactions (as discussed in chapter 1)

Of these three methods the last, adsorption and precipitation, is believed to be the most effective.

Unlike biomass harvesting or the denitrification of nitrogen, adsorption and precipitation reactions do not actually remove the phosphorous from the wetland. Rather they remove it from the waste flow and bind it in storage within the wetland system. This implies that the phosphorus removal capacity of these systems is finite. Yet, at a rapid infiltration scheme that has been disposing of raw sewage for 88 years, the phosphorous retention capacity of the soil is still capable of producing excellent results (Baillod et al., 1977). This implies that soils have a more than adequate retention potential for phosphorous removal.

Stowell et al., (1981) state that the phosphorus removal potential of wetland systems is "variable and transitory" (p923). Venus (1987) reports the removal of phosphorus in wetlands to be "intermittent" (p41). Indeed, phosphorous removal rates in twenty one wetland systems documented by Venus varied between 0% and 83%. Although, in land treatment systems the phosphorus removal by soils is excellent. Typical examples are 98% retention in a system spray irrigating secondary domestic effluent (Quin, 1984), and 89% at the aforementioned rapid infiltration scheme (Baillod et al., 1977).

As noted, the primary mechanisms of phosphorus removal appear to be adsorption and precipitation. Therefore, if a wetland is performing poorly several hypotheses can be reached:

(a) there is a lack of adsorption sites, and/or reactive ions for precipitation; and

(b) environmental conditions are unfavourable, causing slowed reaction rates or resorption back into the water

In order to increase the phosphorus removal potential of a wetland treatment system it would seem prudent to incorporate suitable soils into the system and encourage good contact between the soils and the flow.

2.7 Removal of Toxic Contaminants

Toxic contaminants are becoming increasingly common constituents of waste streams. Metals, trace organics, herbicides, and pesticides are all potentially lethal pollutants.

2.7.1 Metals

The conventional methods of removal used in industrial applications are expensive, with high energy requirements. In wetlands, however, the removal of heavy metals can be achieved by basically the same mechanisms as for phosphorous. With good design, wetlands are capable of producing outstanding results and have proven particularly successful in the treatment of acid mine drainage (Brodie et al, 1988). Heavy metals react with organic minerals forming complexes with phynols, hydroxl groups and so on. Because they are tightly bound there are no problems with subsequent desorption.

As for phosphorous, the contaminants are not easily degraded into stable products but are removed from the water and stored in concentration within the wetland treatment system. Ultimate removal of these materials from the wetland treatment system is possible by harvesting, dredging or by chemical resolubilisation. However, unless they are to be removed for actual treatment they may as well remain in the wetland (provided it is sealed) as they will be effectively constrained from contaminating the greater environment.

2.7.2 Trace Organics

A number of synthetic organic compounds resist removal by conventional waste treatment systems. Some of these compounds are toxic, and to the alarm of researchers, appear to be accumulating in the food chains as the majority are fat soluble.

Biological degradation of the easily degraded organic compounds is considered the foremost removal mechanism. However chemical, photochemical and physiochemical processes are also important (Gieger and Roberts, 1978). Adsorption of trace organics by organic matter and clay particles is believed to be the primary physicochemcal mechanism in wetland treatment systems. No data is available for the efficiency of wetlands, but it could be expected to be comparable to those obtained for the water hyacinth treatment system listed in Table 1.

Table 1.

Trace Organic Removal in Pilot Scale Water Hyacinth Basins

(Reed et al., 1987)

	Conventional, µg/L	
Parameter	Untreated Wastewater	Hyacinth Effluent
Benzene	2.0	Not Detected
Toulene	6.3	Not Detected
Ethylbenzene	3.3	Not Detected
Chlorobenzene	1.1	Not Detected
Chloroform	4.7	0.3
Chlorodibromomethan	5.7	Not Detected
1,1,1-Trichloroethane	4.4	Not Detected
Tetrachioroethylene	4.7	0.4
Phenol	6.2	1.2
Butylbenzyl phthalale	2.1	0.4
Diethyl phthalale	0.8	0.2
Isophorone	0.3	0.1
Naphthalane	0.7	0.1
1,4-Dichlorobenzene	1.1	Not Detected

2.8 Pathogen Treatment

Pathogen treatment in wetlands is particularly successful. Normally a double order of magnitude reduction in indicator organisms can be expected. The pathogens of concern in wetlands are parasites, bacteria and viruses. However, studies on the transmission of parasitic diseases to animals and humans on systems comparable to wetlands indicate that the potential for serious health problems does not seem likely (USEPA, 1988).

Pathogenic bacteria and viruses are removed by the same mechanisms found in oxidation ponds and similar systems. Physical effects of the environment are significant, these include temperature ranges that are unfavourable to cell production, and the lethal ultra-violet rays of sunlight (USEPA, 1988). However, much of the success of the pathogen treatment can be attributed to the development of highly complex microbial and unicelluler communities, elements of which have the capability to consume or out-compete the pathogens.

3. TYPES OF TREATMENT SYSTEMS

There are many variations of natural wetlands, as discussed in chapter one. Freshwater marshes and swamps inhabited by emergent macrophytes are invariably the ones that are used as, or constructed for wetland treatment systems. The use of other types of aquatic vegetation for wastewater renovation is discussed in the latter part of this chapter.

3.1 Natural Wetlands

Natural wetlands have in the past been used as wastewater disposal sites, and more recently, as managed treatment systems. A natural wetland offers a well developed ecosystem, rich in biological and chemical diversity, and has proven to be a very effective and inexpensive wastewater treatment system. Kloosterman and Griggs (1989) report that such systems have operated for several years in New Zealand. Parker (1989) indicates there are also other semi-wetland semi-soakage field systems with more than ten years of operation. However, contemporary researchers are now arguing against further development of these environments for waste treatment (Hammer and Bastian, 1989). As mentioned earlier, natural wetlands have many important social, environmental and economic values. To believe they are best used only as treatment systems is described by some as narrow sighted (NWF, 1987). Clearly due consideration must be given to the impact of the wastewater on the natural wetland environment.

When a natural wetland receives wastewater, the nature and make-up of that wetland will change dramatically from that of its undisturbed state (Wentz, 1989). Scultharpe (1967) discusses how recolonisation of a stream takes place after receiving a sewage discharge. Although some aquatic plants will thrive in the organically and nutrient enriched water, others will die off.

All wetlands have a finite capacity to absorb and renovate, thus the overloading and/or failure of a wetland is always a possibility. Total, or even temporary failure of a natural wetland is perceived as either a loss of or pollution of part of the natural environment. On the other hand constructed wetlands offer improved operational control and are seen more as waste treatment systems.

3.2 Constructed Wetlands

Constructed wetlands are designed and built with the sole purpose of creating an environment conducive to wastewater renovation. A constructed wetland is a "man-made complex of saturated substrates, emergent and submergent vegetation, animal life and water", the elements of which have been selected to optimise the treatment potential of the system (Hammer and Bastian, 1989, p12). Kloosterman and Griggs (1989) report that in New Zealand there are presently twenty constructed wetlands in operation, consisting of eleven surface flow, four subsurface flow, and five combinations of the two. There are also several more wetlands under construction. The majority of wetland treatment systems in use today both, in New Zealand and overseas, are constructed rather than natural. It seems likely that this trend will continue in the future.

3.2.1 Surface Flow Wetlands

Surface flow wetland systems are the most common type of wetland treatment and are the most similar in structure to a natural marsh.

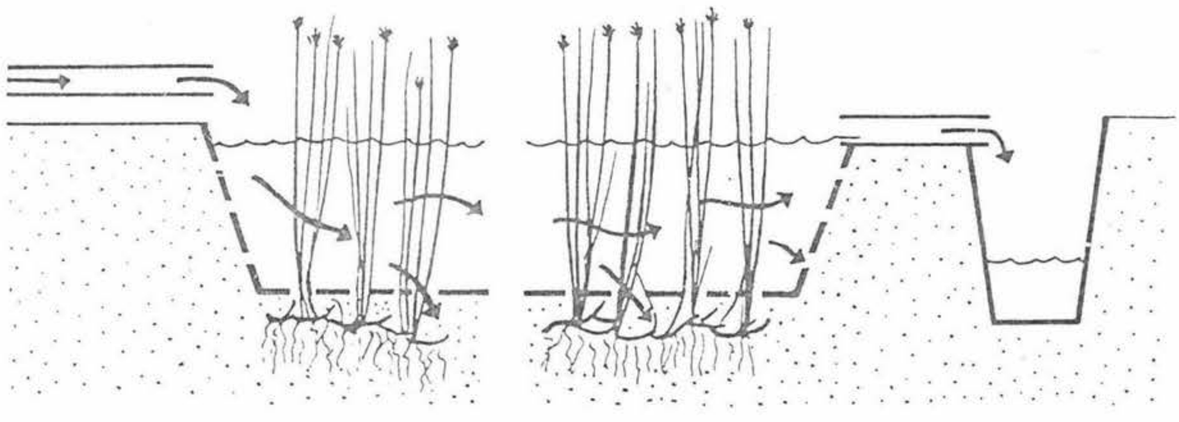

Figure 8. Surface Flow Wetland

(Brix and Schierup, 1989, p103).

Surface flow wetlands are basically shallow, plug-flow ponds packed with vegetative media covered in microbial slime. Like lagoon systems, the predominant oxygen pathway would appear to be by diffusion across the air/water interface.

3.2.2 Subsurface Flow Wetlands

The subsurface flow wetland is also in common use. In a subsurface flow wetland the wastewater is confined within the substrate. Substrate materials vary from clay, silt or sand through to gravel and crushed rock. Systems using soil are often referred to as the 'root-zone method'.

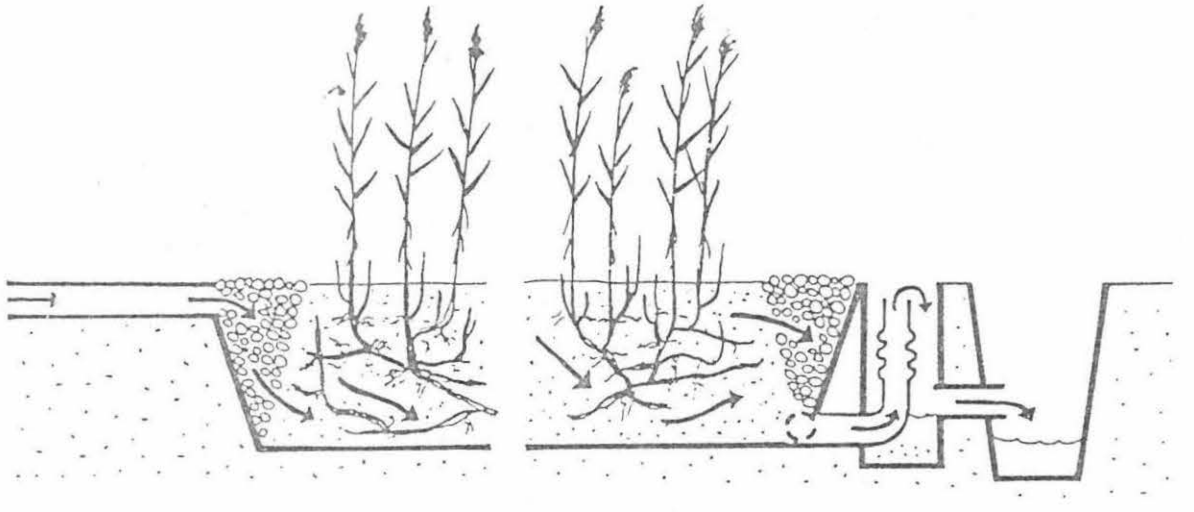

Figure 9. Subsurface Flow Wetland

(Brix and Schierup 1989, p103)

These systems have greater aesthetic appeal. With the effluent being beneath the surface all that is seen is a soil or rocky bed of vegetation. However, in order to achieve their design potential the effluent must not reach the surface and short circuit the treatment process by overflowing the system. This, unfortunately, is a fairly common phenomenon. Thus attention to bed hydraulics is a particularly important design consideration, especially when using less permeable soils.

3.2.3 Vertical Flow Wetlands

Vertical flow wetlands are also known as the Max Planck Institute Process. This type of wetland system is designed so that the wastewater percolates down through the root zone to a subsurface drain. This system would appear to have potential, however research literature is scarce, and there have been no reports of full scale systems.

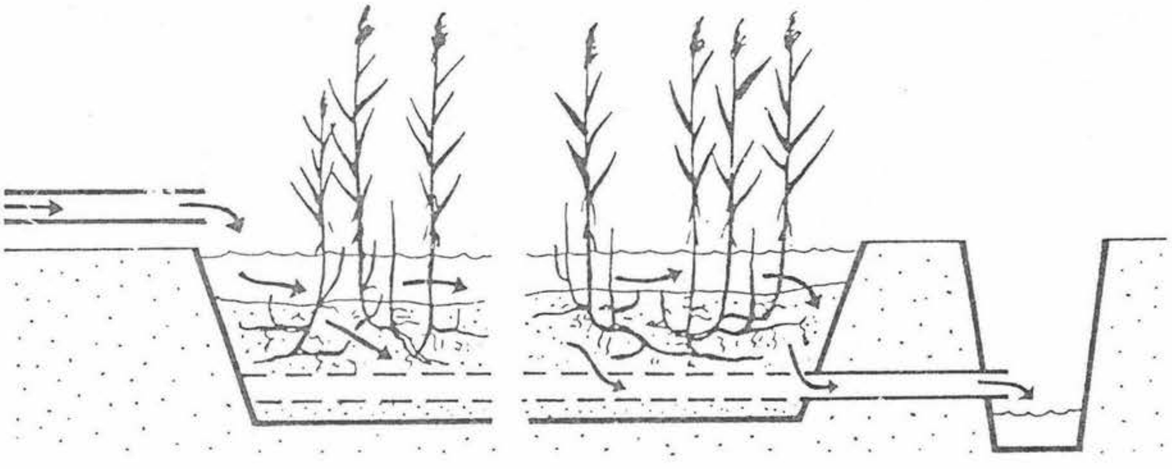

Figure 10. Vertical Flow Wetland

(Brix and Schierup 1989, p103)

A variation of this type of wetland involves the upward percolation of the effluent through the bed under pressure. This process is being developed in Australia by Dr Mitchell at CSIRO. The process is reported to be effective, but the appropriateness of using a pressure system is questionable.

3.2.4 Surface Flow Versus Subsurface Flow Wetlands

Selection of a surface flow or a subsurface flow systems requires consideration of a large number of factors, some of which are discussed below.

(i) **Land area.** A larger land area is required for a surface flow wetland in order to achieve the same treatment as a subsurface flow wetland.

(ii) **Construction costs.** Earthworks comprise the major cost in constructing a wetland treatment system. If the site is not level, subsurface systems will be less expensive by virtue of their smaller area. However, subsurface systems require more precise grading because of hydraulic considerations, and often a selected substrate has to be purchased and trucked in (Steiner and Freeman, 1989).

(iii) **Treatment ability.** Subsurface systems are more effective at removing fine suspended solids, phosphorus and metals because of better contact with the substrate and roots. These systems are also reported to be less susceptible to reduced treatment ability in cold climates, and to freezing over (Steiner and Freeman, 1989). However, because of their nature they are likely to become clogged by wastes with high solids contents. Additionally as surface systems have better oxygenation they should be capable of superior nitrification and microbial decomposition of BOD.

I believe the most effective overall treatment is achieved by a combination of both these systems as will be discussed in the next chapter.

3.3 Aquatic Plant Systems

The term aquatic plant systems encompasses all wastewater treatment systems that use aquatic vegetation (including wetland species) as an integral part of achieving wastewater renovation.

Aquatic plant systems originate from a) the concept of achieving resource recovery of nutrients and energy via plant uptake and harvesting, b) from the use of plants to upgrade stabilisation ponds, and c) simply as the progression from using natural aquatic plant environments as disposal sites to their use and construction as treatment systems.

Aquatic plants can be broadly categorised into emergent plants, floating plants, and submerged plants. Treatment systems using emergent macrophytes and other species planted in the substrate, rising through the water column, and emerging into the atmosphere are known as 'wetland treatment systems'. However floating and submerged plants have also proved to be capable of enhancing wastewater treatment.

This can cause some confusion as systems using other types of wetland species are commonly referred to as 'floating plant systems' or 'submerged plant systems' rather than as wetland systems (USEPA, 1988). 'Plant systems' have basically the same treatment processes and mechanisms as wetlands and therefore warrant discussion.

3.3.1 Floating Plant Systems

Floating plant systems are similar to surface flow wetlands, except that the plants float freely on the water surface as opposed to being planted in the substrate. The root systems of floating plants extend down into the water column.

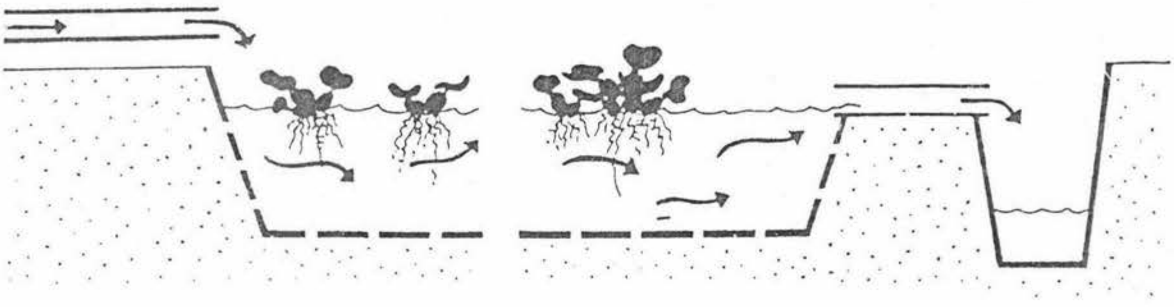

Figure 11. Floating Plant System

(Brix & Schierup, 1989, p103)

These systems are becoming increasingly popular overseas. Although no such systems have been built in New Zealand, Dr Alan Graham of the Waste Technology Group centred at MAF Invermay in Dunedin is currently undertaking a three year research study into their use in this country. The study initially aims at assessing their feasibility. Trials will be undertaken that will yield the necessary data required to develop process designs for the New Zealand situation. It is expected that a full scale trial will be operational in one to two years.

Water hyacinth and duckweed species have proven to be the most popular plants for use in these systems, although the use of water lettuce, salvinia, and water cress has also been investigated. As opposed to wetland species many of the floating plants have high nutritional value and achieve significant nutrient uptake, which makes harvesting of the biomass a worthwhile practice in terms of both enhancing treatment and achieving resource recovery (Graham, 1990).

Dr Graham believes the New Zealand climate is even more conducive to the growth of some floating plant species than in areas overseas where they have already proven successful, and thus believes that these systems are likely to develop further in New Zealand.

3.3.2 Submerged Plant Systems

As the name suggests, these plants are rooted into the soil with the vegetation floating up into the water column. At present these systems are still very much in the experimental stage (Brix & Schierup, 1989).

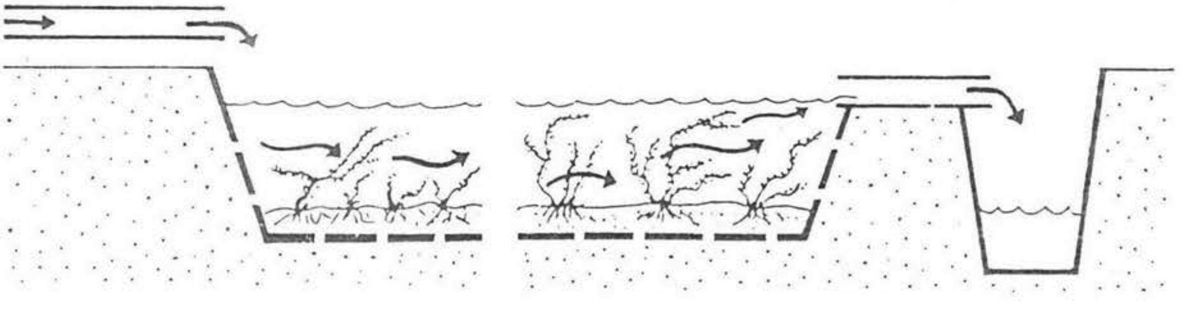

Figure 12. Submerged Plant System

(Brix and Schierup, 1989, p103)

These plants only grow in oxygenated water which limits their use to polishing systems. Their photosynthesis increases the dissolved oxygen and depletes the dissolved carbon dioxide which results in an increased pH. The higher pH creates optimal conditions for volatilisation of ammonia and chemical precipitation of phosphorus (Brix and Schierup, 1989). However, problems arise due to algae growth which 'shade out' the submerged plants (Gaudet, 1974), the pronounced diurnal effect which in turn may create anaerobic conditions which damage the plants (USEPA, 1988).

4. PHYSICAL AND PROCESS DESIGN

4.1 Site Evaluation

The degree of engineering investigation is constrained by the size of the project. A practical programme of site evaluation should include the following steps:

1. Clearly determine the design objectives and the performance efficiencies required to deal with the waste problem.
2. Collect sufficient data, such as waste strengths, flow rates, seasonal temperatures and so on, in order to develop a preliminary design so that the approximate area requirement and type of system can be assessed.
3. Interpret aerial photographs and conduct a basic field survey to identify potential sites and familiarise oneself with aspects of the local environment such as geography, vegetation, and land uses.
4. Investigate potential sites to quantify soil types, relative elevation in comparison to source, extent of earthworks required, ownership and cost of land.

The size of the project will influence the extent of the site evaluation and the degree of engineering that is affordable. Furthermore, project size influences factors such as the site position because the costs of conveying the wastewater to the wetland or building the access road, for example, become increasingly more significant as the size of the project decreases.

An additional constraint is that as a wetland is a waste treatment system, a buffer zone from local households or other land users is necessary. In New Zealand, Stevenson (1976), in the Interim Guide for Land Application of Treated Sewage Effluent, recommends a buffer clearance of no less than 300 metres from the boundary of the nearest subdivision and no less than 150 metres from the closest isolated dwelling.
Although intended as an abstract guideline for spray irrigation systems, this would also seem to be applicable for wetland treatment systems.

4.2 Design Equations and Evaluation

A number of empirical models have been presented to model the BOD removal in a wetland. In general, they originate from the sound assumptions that being a predominantly biological process BOD removal will be first order, and temperature dependent:-

$$\frac{Ce}{Co} = \exp(-K_T t)$$

where

$\frac{Ce}{Co}$ = fraction of soluble BOD removed

K_T = Temperature dependent rate constant

t = reaction time (theoretical retention time)

This equation is then developed for use by using results from trials which have recorded the external parameters of influent concentrations, effluent concentrations, temperature, hydraulic loading, and system dimensions (the black box approach). Equations produced to date have tended to either be based on a number of similar trials at a particular site, or have been based on limited results from a collection of wetlands which are quite different in configuration, design and management. These types of equations take little or no account of the actual processes and mechanisms that contribute to the wastewater treatment, they are useful for providing a guide, but should not be used as a basis of design. There is a danger that engineers will place too much confidence in them for general application.

Design of wetland treatment systems is still a developing technology and instead of using a design manual or a textbook equation the designer needs to make a general evaluation of the treatment system. As indicated at the beginning of this thesis, many wetlands have been constructed by engineers who recognised that the wetland as an entity achieved waste treatment but who took little or no account of the actual treatment mechanisms.

The most appropriate design approach at this stage of wetland development is to size the wetland in terms of flow, depth, retention time and hydraulic loading (discussed further in the following sections). This keeps the calculations at a basic level, but requires the designer to study what has been achieved in practice, and to assess the effects the key parameters have on the overall treatment. The emphasis should then be on developing and refining the design by examining the individual treatment mechanisms and designing the systems' configuration to cater for these processes thereby optimising the treatment potential.

As noted in the introduction, the use of wetlands as waste treatment systems represents a move away from highly engineered processes. For this reason I believe that the design emphasis should remain on evaluation rather than empirical modelling, and that future research and development should reflect this emphasis.

Much of the literature published to date has concentrated on "black box" trials of wetlands and has demonstrated that wetlands have great potential for waste treatment. Ideally, future research should aim at examining and quantifying the actual mechanisms of treatment that occur within a wetland.

4.3 Design Parameters

The size of the wetland is determined by selection of several key parameters including flow, retention time, depth, and the hydraulic loading rate. Selecting appropriate values for these parameters requires the designer to research, inspect and evaluate existing wetland systems which have similar waste inputs and treatment objectives to the system under consideration.

4.3.1 Flow rate

The design flow rate is the flow rate at which the effluent is retained within the wetland for the minimum acceptable time to achieve the required treatment (design retention time). Determination of this parameter requires consideration of the wastestreams' hydrograph. Designing for the peak flow is not necessarily justified. When a design flow is selected and the systems' dimensions calculated, the design flow should be rechecked by calculating the resultant retention times for the average and low flow periods. Consideration must also be given to the environmental effects on the water balance and the buffering capacity of the wetlands volume (both are discussed in chapter five).

4.3.2 Retention Time

Stephenson et al. (1980 in USEPA, 1988) report a retention time of six to seven days to be optimal for treatment of primary and secondary wastewater. The USEPA (1988) believes that a shorter period does not allow sufficient time for pollutant degradation, while a longer time can lead to stagnant, anaerobic conditions. Watson et al. (1989) recommend that retention times should be, at least, greater than five days. Values of eight to ten days are also common in the literature.

Under design will lead to poor treatment, while over design rapidly increases construction costs, especially for subsurface flow wetlands where the bed material has to be brought into the site. The retention time used for design should fit into the range of values given above, although it is likely to be finally determined when it is balanced against the other key parameters when sizing the wetland system.

4.3.3 Water Depth

In surface flow wetlands the selection of the water depth influences the oxygenation of the water and plant growth. Reported depths range from 0.1 m (Watson and Hobson, 1989), to a recommended maximum of 0.6 m to ensure adequate oxygenation of the water column (USEPA, 1988). Fisher (1990) believes 0.45 m is typical for municipal systems. However, these shallow depths result in a large land area requirement and small hydraulic loading rates. Depending on treatment objectives and site constraints, depths of up to and exceeding 0.8 m can and have been used. These greater depths are more common in subsurface wetlands where loading rates are, on the whole, greater than surface flow wetlands. However, in a subsurface wetland, the depth should not exceed the reach of the wetland plants root zone as the presence of the root network has been shown to significantly improve the efficiency of the treatment system (refer to chapter six). For design purposes the design depth of a subsurface wetland is the depth of the cross-sectional area of flow leaving the system.

Little comment has been made in the literature about the fate of the sludge (both inorganic and biological) that must accumulate within the packed bed. In the "trickling filter" treatment system the media is large enough to allow the sludge to wash through. However, in a submerged wetland system the media is typically smaller, and the majority of these systems flow horizontally rather than vertically which encourages the settling and entrapment of the sludge within the system. Therefore, it would appear prudent to apply a safety factor in terms of an increased water depth, and perhaps use strips of chunky rock running across the width of the beds at regular intervals. They would be significantly more porous than the surrounding media thereby providing space for sludge accumulation and degradation. They would also have the benefit of arresting any short circuiting within the cell by acting as an exit and re-entry system.

In most wetland treatment systems the water depth is held constant by virtue of a static exit level. It is, however, advisable to design the system to allow for fluctuations of this level. Obviously the system should be able to be fully drained. In addition the water depth should be readily adjustable to promote plant growth and so on. "Water level control is needed for control of weeds and may be useful to encourage rhizome penetration" (Hobson, 1988, p4).

Provision of adequate freeboard (allowance for extra water height) is recommended. This will allow for fluctuations and management of the water level. For example, at the system at Rawine in Northland, extra freeboard allowed the operators to raise the water level after prolific growth and dieback of the wetland vegetation clogged the system (Parker, 1989).

4.3.4 Loading Rate

Watson et al. (1989) have prepared an excellent review of a large number of wetland systems, listing their configurations, hydraulic loading rate, waste strength and resultant treatment performance as well as a specific review of loading rates. These are provided in the Appendix. Some examples of loading rates used in New Zealand are given in Chapter 8. As an initial guideline, Watson et al. (1989) suggest the following ranges of hydraulic loading.

Table 2. Guideline Range of Hydraulic Loading Rates for Domestic Sewage
(adapted from Watson et al., 1989, p238)

	Surface Flow m/d	Subsurface Flow m/d
Primary	N/A	0.023-0.063
Secondary	0.012-0.029	0.047-0.187
Tertiary	0.019-0.094	0.047-0.187

The obvious method of selecting a design loading rate is to evaluate wetland systems operating under similar conditions. However, there are several important considerations to take into account:

1. Characteristics of wastewaters. If the characteristics of the wastewaters are not the same, consideration should be given to the effects of the differences in concentration, biodegradability and bioavailability.
2. Complexity of system. A number of modifications can be made to the design to increase treatment effectiveness (see sections 4.5 to 4.8).

A further level of design sophistication can be achieved by selecting an effluent loading rate based on the most critical loading rate of all the important waste constituents. This involves determining the maximum organic loading, maximum nitrogen loading and so on. The design is then determined by the waste constituent which has the lowest loading rate and thus needs the largest surface area to distribute the loading. This practice is sometimes used in the design of land treatment systems. However, at this stage there is insufficient data available on the maximum loading rates acceptable for wetland treatment systems for this to be readily used.

4.4 Sizing

The first step in the physical design of the wetland system is to calculate the area required to withhold the flow for the required time. Once the initial parameters for the retention time and depth have been selected, the area required can be simply calculated as the flow multiplied by the retention time over the depth.

However, this assumes the wetland is like an open pond. Allowance must be made for the volume occupied by the vegetation and the substrate contained within the flowpath (this is particularly significant for subsurface wetlands). The porosity of the wetland is defined as the volume of voids over the total volume. Watson and Hobson (1989) report typical porosities for surface flow wetlands as being: *Typha* 0.90, *Scirpus* 0.86, *Pharagmites* 0.98 and *Juncus* 0.95. Although these values will obviously vary depending on the growth rate and biotic management at the site. In subsurface systems the porosity of the selected substrate is easily measured. Approximate values include 0.20 for a sand/gravel mix, 0.25 for gravel, 0.35 for sand, and 0.45 for clay (Linsley and Franzini, 1984). Extra allowances must also be made for the growth of the root biomass, however, this varies for different species and for the different mediums they are grown in. Volume loss due to sediment build up should also be considered (as discussed in Section 4.3.3).

In sum, based on the initial choices of retention and depth, and allowing for the porosity, the land area required for the design flow can be calculated as follows:

$$\textbf{Surface Area } (\mathbf{m^2}) = \frac{\textbf{Flow } (\mathbf{m^3\ d^{-1}}) \textbf{ x Retention Time (d)}}{\textbf{Depth (m) x Porosity}}$$

Alternatively the surface area required may also be calculated by use of the hydraulic loading rate as shown below:

$$\text{Surface Area } (m^2) = \frac{\text{Flow } (m^3\ d^{-1})}{\text{Hydraulic Loading Rate } (m\ d^{-1})}$$

These two calculations involve different parameters and the required surface areas will differ. Some, or all of the key parameters will need to be altered and the calculations reiterated until a balance is achieved. For example, depth may be increased in order to maintain the retention time while decreasing the area, thereby resulting in a higher loading rate. In order to finalise the design the designer must assess the effect such alterations have on the treatment objectives while also allowing for the physical constraints of land area and cost.

4.5 Shape

The shape of the wetland should encourage plug-flow in an effort to avoid stagnation and short-circuiting of effluent. Steiner and Freeman (1989) recommend a length to width (L/W) ratio of at least 10:1 for a surface flow wetland. However, if the influent is sufficiently strong, it is important for it to be distributed over an adequate width so that localised organic overloading does not results (this is discussed further in section 7.2.3.). Because of the high L/W ratios surface flow wetlands are often built in a "serpentine" shape so as to fit into a compact area.

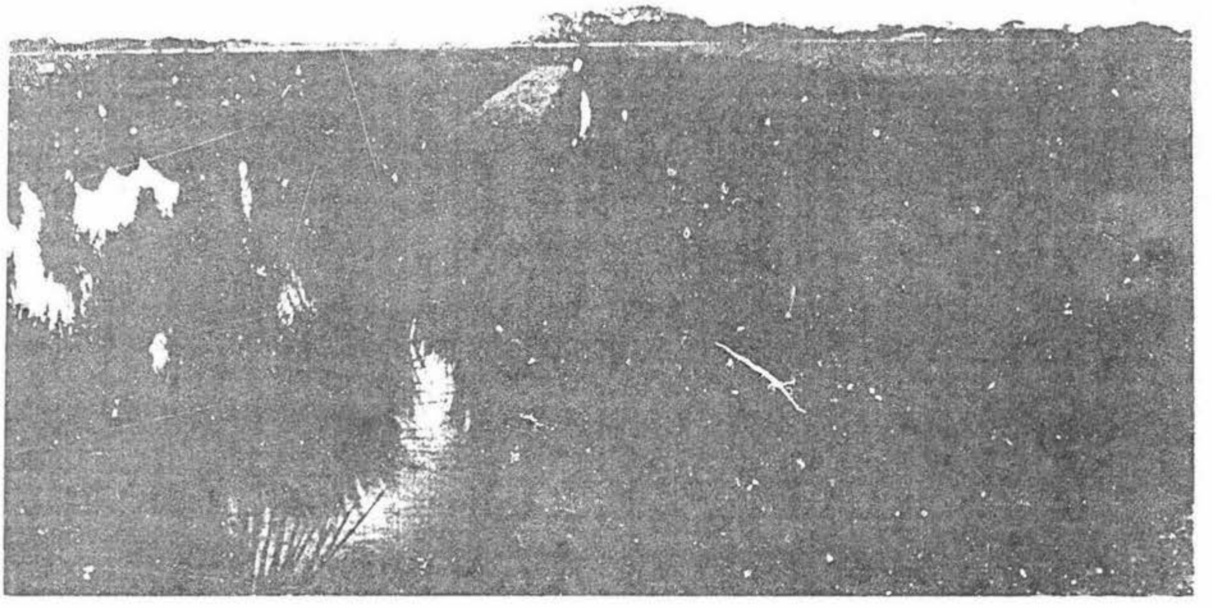

Figure 13. Serpentine Surface Flow Wetland

In subsurface wetlands, length is often restricted by hydraulic considerations (discussed in the next chapter). Additionally increasing the bed width distributes the solids over a wider area reducing the probability of clogging. L/W ratios may be as low as 1:1, although beds with course materials such as gravels can be greater, up to 3:1 (Watson et al. 1989).

Sloping the bottom of the wetland bed has been commonly practiced in order to aid the hydraulic gradient. However it has no effect on the hydraulic gradient and simply serves to maintain the design depth through subsurface beds, and allows the drainage of the system.

4.6 Bed Configurations

There are several ways of arranging a constructed wetland treatment system. Firstly, it is advisable to split a wetland up into a number of parallel cells. This increases operational control and enables close down for maintenance.

Since the potential of different renovating mechanisms varies in different types of systems, series combinations can prove effective. A surface flow wetland should be used first because:

a) it appears to have more reliable sources of oxygenation and is therefore more efficient for aerobic BOD removal and amonia oxidation; and

b) it is significantly more porous and thus less susceptable to clogging by the entrapment of solids.

The surface flow wetland should be followed by a gravel bed. Although the front of the bed will be partially aerobic as the remaining dissolved oxygen (DO) is utilised, the rest of the bed will be an anoxic/anaerobic environment except for the thin films of aerobic zones around the roots (Fisher, 1989). This bed will serve to:

a) provide secondary BOD removal by both aerobes and anaerobes;

b) further oxidise ammonia;

c) dentrify the nitrates; and

d) filter and absorb the finer organic solids, including entrapment and degradation of the biological cell mass from the aerobic process.

As a final polishing system the wastewater should then enter a root zone bed, containing a soil such as a silty loam with some clay content. This bed will:

a) complete dentrification;

b) precipitate, adsorb and retain phosphorous, trace metals, refractory organics and so on; and

c) provide microstraining of any remaining suspended solids.

As most of the solid material will have been removed by this stage, the root zone bed will not be subjected to clogging and the adsorption sites will not be saturated by organics.

Kloosterman (1989) has noted that for farily raw organic wastes, such as those from septic tank overflow, the use of rock/gravel subsurface systems is less likely to cause problems like odours, mosquitoes, flies and so on, that may occur in a surface flow wetland. However, careful consideration needs to be given to clogging and organic overloading which may result in fully anaerobic conditions.

4.7 Piping Configurations

A number of design improvements are possible by the use of piping systems, as listed below.

(i) **Recycle.** Recycling the treated effluent back to the inlet creates a greater flow through the system which dilutes strong wastes. A larger wetland is created with a resultant lower hydraulic loading which effectively increases the treatment. Furthermore, recycling can be used to balance the hydrological budget if excessive water is being lost from the system.

(ii) **Carbon Supply for Dentrification.** As discussed in chapter two, denitrifying bacteria requires a carbon supply to perform effectively. The easiest way of supplying this is by bypassing a portion of the influent around the carbon stabilising surface flow wetland and into the anaerobic rock/gravel bed. Because the effluent will be falling at grade, this operation would not even require pumping.

(iii) **By-pass.** It should be possible to by-pass the flow around any cell. The by-pass for root zone beds and gravel/rock beds will need to be sized to accomodate the predicted flood flow, as overflow will damage the bed surfaces (Hobson, 1988).

(iv) **Step Feed.** Wetlands are basically plug-flow reactors and, as such, provide the bulk of the treatment at the front end, with the percentage removal falling rapidly off towards the rear. By diverting portions of the inflow further down the bed, the effectiveness of the cell is increased. As for 'carbon supply', the step feed pipes should be able to simply run at grade without requiring pumps.

4.3 Inlet Outlet Structures

The construction of good inlet outlet structures will encourage an even plug-flow thereby utilising the whole bed. Frequently a single inlet and outlet pipe is used which is likely to utilise only 70% of the bed at most. Ideally the effluent should be evenly distributed into the system and evenly collected at either end. This may be achieved by using a piped manifold, or a packed trench as illustrated in Figure 14.

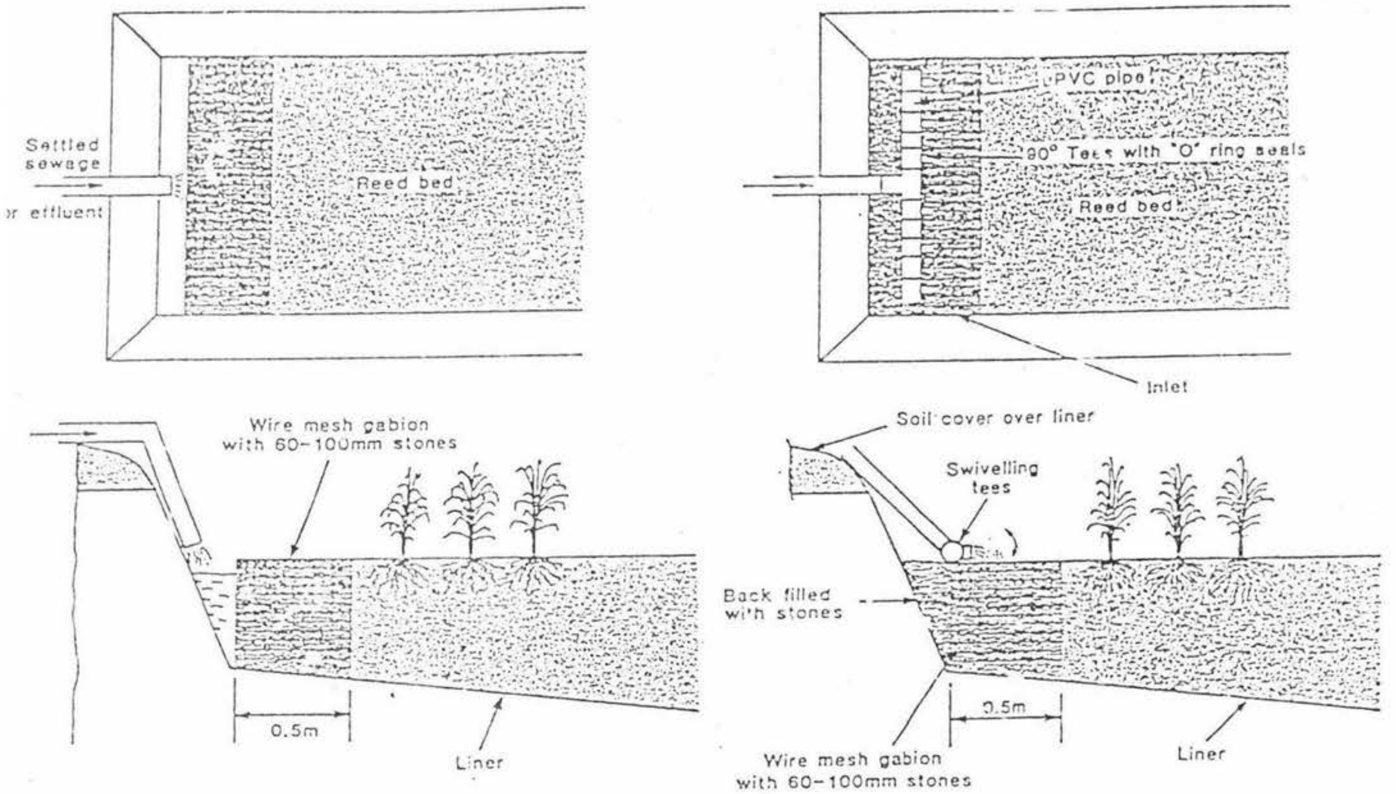

Figure 14.Inlet and Outlet Designs

(Copper & Hobson, 1987)

4.9 Selection of Substrate

As good plant growth is very important for effective treatment, it is essential to have a substrate that will support and nourish the wetland plants. Allen et al. (1989) recommend loamy soils as they are " soft and friable", promoting good root and rhizome penetration. Dense or hard clays and gravels, on the other hand, restrict penetration and establishment.

Often the design specifications of a wetland calls for the incorporation of a layer of upland or wetland topsoil. However, Allen et al. (1989) state there is little evidence to support the benefits of this practice and question its worth. Although peaty organic soils are commonly found in natural wetlands, they have been found to lack nutrients and provide weak support (Allen et al., 1989).

5. HYDRAULIC AND HYDROLOGICAL DESIGN

5.1 Wetland Bed Hydraulics

5.1.1 Surface Flow Wetlands

Reports in the literature often recommend the use of Mannings Equation (steady state, open channel flow) for modelling the hydraulics of flow in surface wetlands. In reality, the head loss across a properly managed system as predicted by this equation is negligible. A significant hydraulic gradient is only likely to occur in these systems if the wetland becomes clogged. This possibility should be allowed for by provision of adequate freeboard (as discussed in section 4.3.3).

5.1.2 Subsurface Flow Wetlands

A number of operational problems have been encountered with subsurface wetlands. These can be attributed to a lack of consideration of the flow hydraulics in the bed. The fundamental problem, reported at a number of wetlands overseas, is overflow across the surface at the bed. This causes short circuiting of the treatment process and a subsequent failure to meet the treatment objectives. This problem arises when the bed length is too long for the relative hydraulic conductivity of the substrate, causing a head loss greater than the actual wetland depth. The wastewater consequently backs up and overflows the bed (illustrated in figure 15).

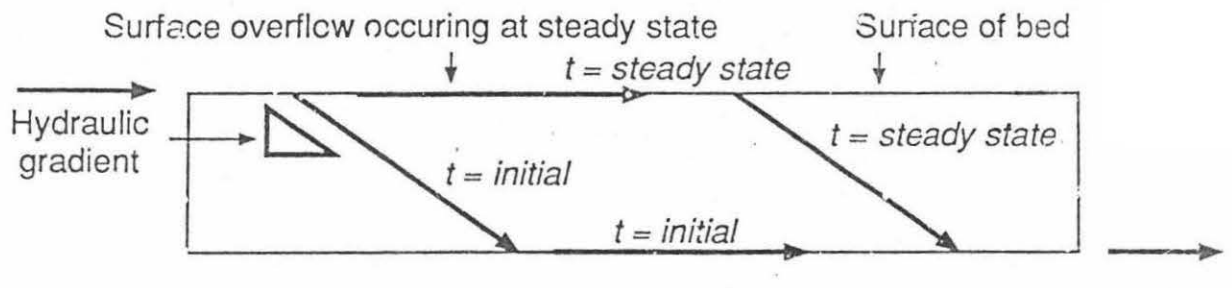

Figure 15. Schematic Representation of a Subsurface Wetland Bed with Overflow Conditions

Another problem can arise in beds that have a suitable balance between length and hydraulic conductivity, but are overall quite long. The resultant head loss towards the end of the bed puts the area of flow out of reach of wetland plants root zone. This impedes their growth and contribution to the treatment, and additionally, impedes passive diffusion of oxygen from the surface. A head loss of 300 mm or less should be acceptable. The obvious way of avoiding this problem is to slope the surface of the bed to a similar grade as the hydraulient gradient (but still allowing for an adequate margin of coverage). However, Watson and Hobson (1989) advise that a sloping surface should not be used, because if overflow does develop, it may not penetrate back into the bed. Even so, beds with lengths long enough for head loss to be a problem are generally constructed with coarse gravel and therefore surface overflow should easily re-enter even a sloping bed.

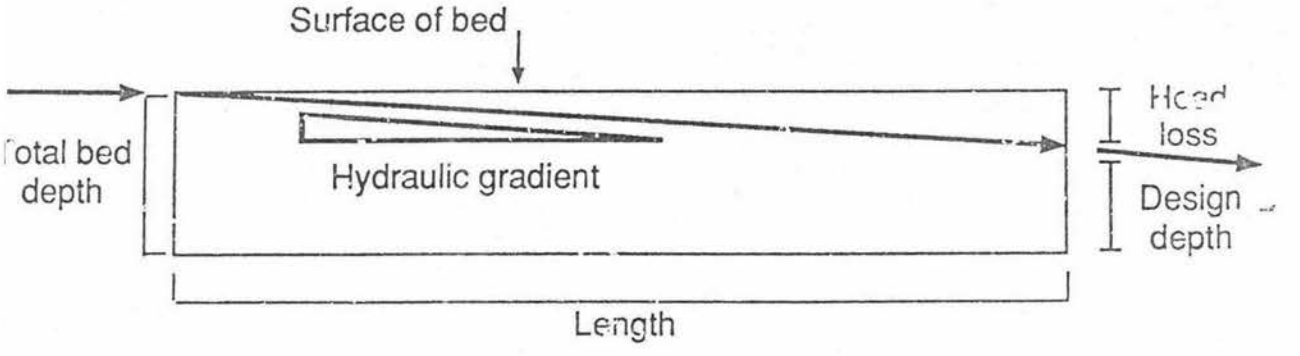

Figure 16. Schematic Representation of the Hydraulic Profile in a Sub-Surface Wetland

Kadlec (1989) recommends the use of Ergun's equation for modelling the hydraulic profile of a sub-surface bed. This equation adds a term of Darcy's law for turbulent flow. However, for turbulent conditions to arise in a wetland with a substrate 30 mm in diameter, the flow would have to move through a length of 100 metres in only 3.3 days (Kadlec, 1989). It would appear that in the great majority of wetlands Darcy's law is more appropriate. Darcy's law models the discharge velocity of water through saturated soils (Das, 1985):

$$v = ks$$

where

v = discharge velocity (ms^{-1})

k = hydraulic conductivity (ms^{-1})

and

s = hydraulic gradient

Now if the following expressions for flow are substituted into Darcy's law, a suitable expression defining the hydraulic gradient can be found.

$$Q = v\,A$$

and

$$Q = \frac{V}{t}$$

where

Q = flow rate ($m^3 d^{-1}$)

A = cross-sectioned area of flow (m^2)

V = volume of effluent in the wetland (m^3)

t = retention time (d)

then

$$\frac{Q}{A} = ks$$

and

$$\frac{V}{tA} = ks$$

since

$$V = A \times l$$

where

l = length

then

$$\frac{l}{t} = ks$$

or

$$s = \frac{l}{tk}$$

This final expression enables the calculation of the hydraulic gradient that will result for the selected values of bed length and retention time when the hydraulic conductivity has been determined.

It is therefore possible to select an initial length, calculate the resultant head loss and reiterate the calculations until a length is selected that allows the flow to remain submerged, and has a head loss that is not too great to restrict plant growth and oxygenation.

5.2 Wetland Hydrology

The unique hydrological characteristic of a natural wetland is its narrow range of water level fluctuations. These depths and durations of inundation create the conditions in which wetland vegetation and animals dominate. However, like any biological system, a wetland is in a delicate balance. Even minor changes in the water management regime can be enough to cause drastic changes in the biological communities of a wetland (Wentz, 1987).

The formation of a wetland hydrological system is dependent on regional climatic, and local site characteristics such as microtopography, soil, and vegetation (Duever, 1988). For example, differences in plant communities cause marked variations in the evapotranspiration flow rates between various types of wetlands. Duever (1988), in modelling wetland hydrology, concluded that the dominant processes included atmospheric circulation, precipitation, evapotranspiration and surface and groundwater flows as illustrated in Figure 17.

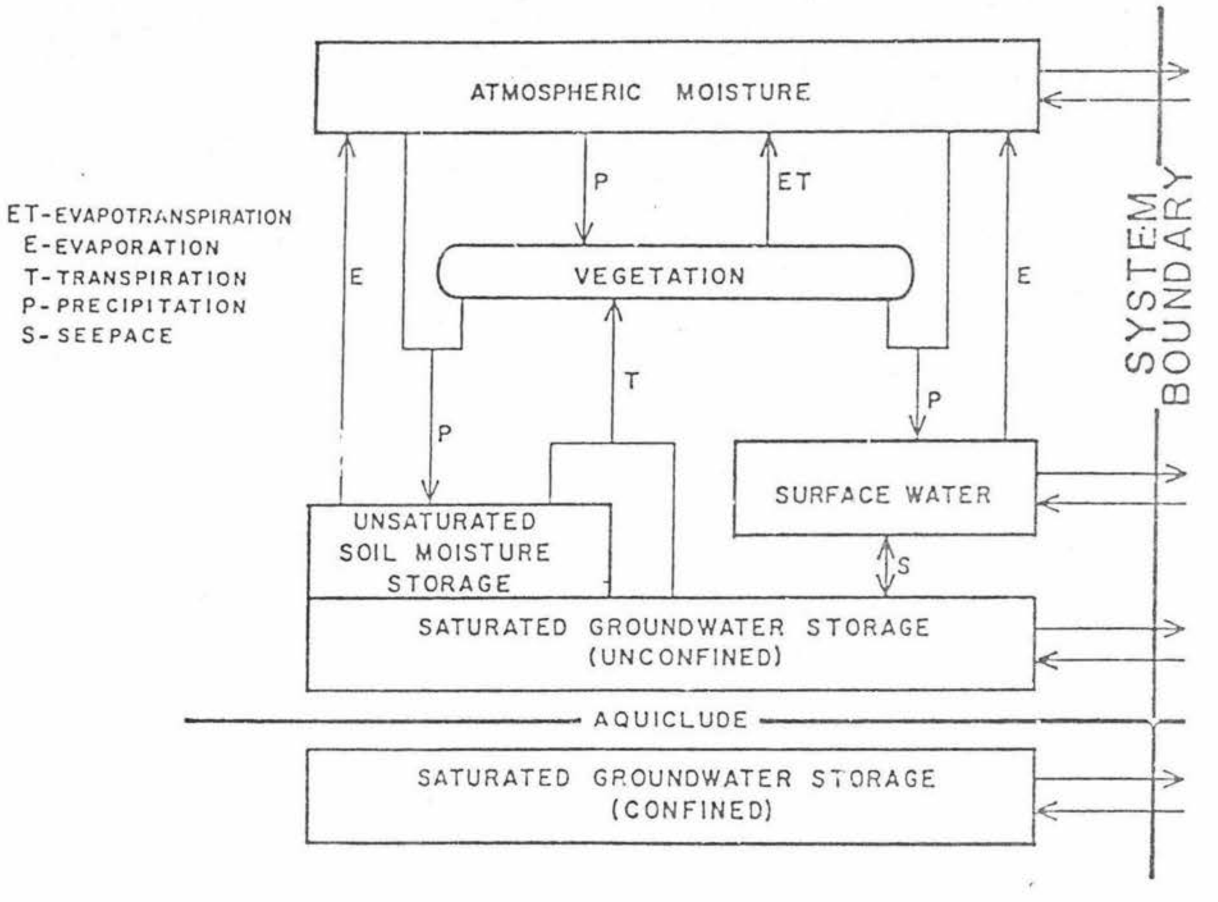

Figure 17. A Conceptual Model of Wetland Hydrology

(Duever, 1988, p11)

The hydrology of wetlands and shallow water bodies is depicted by the diagram in figure 17, the variations between different types of environments being of degree rather than of kind.

5.3 Hydrological Balance

The overall performance of a wetland treatment system is, in part, dependent on the system's hydrology. The removal efficiency of organics, nutrients, pathogens and so on, can be dramatically affected by changes in detention time or volume. Precipitation, surface/groundwater flow, evapotranspiration, or variation of the hydraulic loading rate or the water depth, alter the detention time and either concentrate or dilute the wastewater (Zirschky, 1986, in USEPA, 1988). Therefore, in order to design and operate a wetland treatment system effectively, the conceptual hydrological model in figure 17 needs to be developed further to include estimates of the flows across the system boundary for the site in question. For an artificial wetland this may be simplified into the following equation which models the rate of water volume change.

$$dV/dt = I - E + P - ET + G - S$$

where I = influent flow rate

E = effluent flow rate

P = precipitation

ET = evapotranspiration

G = groundwater inflow rate

S = seepage

V = volume

t = time

If the water table is low then it is likely that G can be neglected. If the system is sited in an impermeable soil, or if an impermeable subsurface barrier is incorporated into the design, then G and S can be neglected. The surface area over which the flow rate of precipitation is calculated must include any surrounding land that drains into the wetland. The choice of the time period over which to average the data requires consideration of the region's climate, the available data and so on. However, bi-monthly periods should account for seasonal fluctuations.

The values of P, ET and S can be determined by field tests and statistical manipulation of historical climatic data. These tests and analyses are common hydrological engineering practices. Investigation of these parameters will reveal any potential problems that may require the design to be altered. For example, at a wetland system in Northland, a daily ET loss of 10 mm per day coupled with times of low inflow and periods of low rainfall resulted in the system having a negative dV/dt value. A freshwater bore had to be provided to prevent the marsh drying up and dying during these dry periods (Parker, 1989).

5.4 Sealing the Wetland

In New Zealand none of the wetland treatment systems have been sealed to prevent seepage. However the design manual published by the USEPA (1988) favours sealing. There are two reasons for sealing a wetland, but for what is a basically low technology system this added expense requires justification by the designer. A wetland may be sealed to prevent contamination of the groundwater, and prevent water loss, each of which will be discussed in turn.

Prevent contamination of the groundwater

(i) If the wetland is infiltrating effluent into an aquifer or groundwater reservoir the resulting contamination of that water resource may be deemed to be unacceptable.

However, in New Zealand, after discussion with local water quality officers, I have found this to be practically a non-issue. It would appear that an investigation would only be instigated if a problem such as pathogenic contamination of a bore used for drinking water arose after the commissioning of the system (Gilliland, 1989). In the United States, groundwater is more commonly used as a source of drinking water and consequently protection of this resource is strongly advocated.

(ii) If the wetland is receiving significant levels of toxic pollutants (such as heavy metals in acid mine drainage) it is desirable to have a sealed system which will effectively isolate these pollutants from the greater environment.

Prevent water loss

(i) If the soil is porous the wetland system will take on the characteristics of an infiltration system. Significant seepage may cause problems in trying to balance the hydrological budget resulting in lower treatment efficiencies, as discussed earlier.

(ii) Alternatively in arid areas, renovated wastewater is regarded as a resource that should be preserved to boost low receiving water levels.

There are two methods of sealing a wetland that are economically practical:

(i) by using an impervious artificial membrane such as butynol; and

(ii) by importing and incorporating an impervious clay.

The most popular option being the use of a artificial membrane. Although the use of a local clay is the more appropriate alternative for developing countries.

5.6 Effluent Routing

Gearheart and Finney (1982) noted, following studies of a free surface wetland, that the system had to ability to dampen spikes in effluent characteristics from an oxidation pond producing a more stable and consistent effluent.

The large volumes of wetlands (and other large storage systems) would give them the ability to rout (as in flood routing by reservoirs) peaks in effluent discharges. Because the sizing of most treatment plants is based on the peak organic loading, wetlands can be designed proportionally smaller than systems without significant storage, and therefore the wetlands design flow may be less than the peak flow (as discussed in 4.3.1).

6. BIOTIC CONSIDERATIONS

It has been clearly demonstrated that the presence of the vegetation contributes significantly to the treatment potential of the wetland system. For example, Gersberg et al. (1987), in experiments on pathogen removal, found a significant difference between a vegetated and an unvegetated bed in a subsurface wetland system. Results from the same system at Santee, California (tabulated below) demonstrate the dramatic increases in removal of various parameters of pollution due to the presence of vegetation.

Table 3. **Comparison of Treatment between Vegetated and Unvegetated Wetland Beds.**

(Adapted from: Watson et al., 1989, pp322-326)

POLLUTANT	INFLUENT CONCENTRATION	EFFLUENT CONCENTRATION Vegetated Bed	EFFLUENT CONCENTRATION Unvegetated Bed
BOD	118.3 mg/l	5.3 mg/l	36.4 mg/l
SS	58.1 mg/l	3.7 mg/l	5.6 mg/l
Ammonia-N	24.7 mg/l	1.5 mg/l	22.1 mg/l
Coliforms	67 500 000 no./100 ml	577 000 no./100 ml	2 890 000 no./100 ml

In comparison to conventional waste treatment processes such as activated sludge, wetland treatment systems require a lower level of technology. One of the main technical challenges in wetland design is the selection and successful establishment of suitable plant species.

6.1 Plant Selection

In selecting plant species one should not limit the design to a monoculture (single species). The reasoning being that if a disease or pest attacks a particular plant species the result will be total die-off and a subsequent lowering of treatment efficiency. A multiple species system is obviously less susceptible to such events. The use of multiple species also has the benefit of creating a more diverse ecosystem which serves to broaden and enhance the treatment capabilities of the wetland. Furthermore, different plant species have differing tolerance levels to fluctuations in water level, pollutant concentrations, and changes in environmental conditions, thereby ensuring a more resilient and stable treatment system.

Even so, developing a multiple species wetland can, in practice, be difficult. Hammer and Baston (1989) report that three plants (cattail, bullrush, and giant reed) frequently used in wetland treatment systems, actually tend to create and/or maintain single-species stands by inhibiting or monopolising the other plant species.

There are two primary objectives in the selection of plant species for a wetland. Firstly, the plants must be capable of flourishing in the particular wetland environment being designed, and secondly, the characteristics of the plant selected should optimise the systems' treatment potential. Some of the plant characteristics that have been identified as being important are as follows:

(i) Rate of growth; preferably an active coloniser with a penetrating rhizome system.

(ii) Tolerance of temperature, pH, oxygen, water depth and concentrations of wastewater constituents,

(iii) Nutrient uptake, biomass production and usefulness of product,

(iv) Natural life span, degree of seasonal dieback and detritus production,

(v) Surface area for support of microbial populations.

6.2 Wetland Plant Species

There are at least five thousand plant species adapted to living in a wetland environment. As different species have different characteristics that influence wastewater treatment, as indicated, it is clear that performance of a wetland treatment system is influenced by the plant species which inhabit it. Some of the species commonly found in wetland treatment systems are discussed below.

6.2.1 *Scirpus* and similar species

Scirpus spp., or bullrush as it is more commonly known, grows naturally in a diverse range of inland and coastal waters. The USEPA (1988) reports they have a desirable temperature range of 16 to 27°C. They are found growing within a pH range of 4 to 9, and water depth of 5cm to 3m. Kloosterman (1987) considers *Scirpus* spp. to have "the greatest treatment potential in the New Zealand situation" (p299).

In New Zealand, Tanner (1990) believes *Bolboschoenus fluriatilis* and *Schoenoplectus validus* have greatest potential in this group of plants. *Bolboschoenus medianus* also has potential, as has *Bolboschoenus caldwellii* particularly in the South Island.

6.2.2 *Typha* spp.

Commonly know as cattails, or in New Zealand by the Maori name raupo. *Typha* is found almost throughout New Zealand growing abundantly in the fertile waters of shallow lakes, swamps and ponds (Johnson & Brooke 1989). It is a hardy plant with an optimal temperature range of 10 to 30°C, and an effective pH range of 4 to 10 (USEPA, 1988).

It is capable of growing from a planted rhizome to a dense stand within three months, and in fact grows so well in wetlands it tends to dominate at water depths of over 150 mm (USEPA, 1988). However, it is prone to significant dieback over winter. As mentioned earlier, this has been a particular problem at the Rawine wetland in Northland. Kloosterman (1988) also reports that *Typha latifolia* has died due to high ammonia levels and that *Typha* in general has "a relatively shallow root system and is susceptible to flattening by wind" (p299).

6.2.3 *Juncus* spp.

Commonly referred to as rushes, these plants have a desirable temperature range of 16 to 26°, and an effective pH range of 5-75 (USEPA, 1988). Sixteen native and thirty-one naturalised species of *Juncus* are found in New Zealand (Johnson and Brooke, 1989), although not all will survive in a wetland environment. Tanner (1990) recommends *Juncus pallidus* and *Juncus effusus* as the best choices for wetland treatment systems

6.2.4 *Glyceria* spp.

These are creeping aquatic grasses. *Glyceria maxima* is most commonly used in wetland systems and is found scattered about the length of New Zealand. *Glyceria* forms a thick mat and can have a tendency to clog wetlands. Although it is a vigorous grower, it is reported to be a seasonal plant (Smith, 1985).

6.2.5 *Phragmites* spp.

Known by the common name of reeds these species are extensively used in wetland treatment systems in Europe and North America. In New Zealand it is classified as a noxious weed and is therefore not available for use in wetland treatment systems (Kloosterman and Griggs, 1989).

6.2.6 *Baumea* spp.

Baumea spp. have been described as "erect sedges of swamps and swampy lake edges mostly with rush-like culms and leaves" (Johnson and Brooke, 1989, pp 120). Clunie (in Smith, 1985) suggests the use of both *Baumea rubiginosa* and *Baumea articulata*.

Tanner (1990) notes that although *Baumea* spp. do not thrive in high nutrient levels and have a moderate to low growth rate, they can be advantageous in that only a few of their shoots carry seed heads. As it is the shoots with seed heads that die back the most severely, *Baumea* spp. will continue to support healthy shoots during this period.

6.2.7 Other species

The species already discussed are the most popular selections for wetland treatment. These are but a few of the thousands of wetlands species found in the natural aquatic environment. Trialing of new species is always worthwhile. Inspection of the flora in natural wetlands in the locality of the proposed treatment site is a recommended practice especially when establishing the first wetland treatment system in a new area.

6.3 Establishment of Wetland Plants

Water level is the most critical factor affecting plant survival in the first year of growth (Allen et al., 1989). Therefore, careful management of the wetland's water level is vital. At a subsurface wetland system in Auckland lack of supervision of the water level following planting resulted in die off of the plants. This required substantial replanting, effectively setting the system back a whole year (Kloosterman, 1989).

Before planting is undertaken Parker (1989) recommends that the site be saturated several times with the wastewater in an effort to improve the nutrient quality of the substrate. During planting it is advisable for the substrate to remain wet but not to be flooded. Once the planting programme is completed, the plants must be kept saturated and allowed to grow sufficiently to "generate a stem with leaves that provide above the initial flooding height" and as growth proceeds the water level can be increased proportionally (Allen et al., 1989, p413).

In order to successfully establish the vegetation, planting must be undertaken at the correct time of year. Species such as bullrush (*Scirpus*), rush (*Juncus*) dock (*Rumex*), and arrowhead (*Sagittaria*) have been successfully planted in autumn. Species such as raupo (*Typha* spp.), sledges (*Carex* spp.) and manna grass (*Glyceria* spp.) have proven more successful when planted in spring (Allen et al., 1989). In general planting in the early spring appears to be the most effective. However, Venus (1988) suggests that year around planting is possible as long as the plants and, in particular, the rhizomes are constantly kept moist.

Suitable plant species can be obtained from specialist nurseries. Alternatively they can be collected from existing wetlands in the surrounding environs. In New Zealand, the general practice has been to use the nursery raised plants. However, nursery plants may be more difficult to establish successfully as they are "genetically and physiologically adapted to their particular growing site" (Allen et al., 1989, p410). Large latitudinal distances betweenthe nursery and wetland site will mean that plants must adjust to climatic changes in addition to the stress of being uprooted, packaged and transported.

Despite being well adapted to local environmental conditions, wild plants are logistically difficult to gather. For example, a large local supply must be assured, permission from the land owner and/or local authorities must be sought, and plants must be removed carefully from their location to ensure no permanent damage to the natural wetland environment is made. Care must also be taken to avoid the inadvertent collection of undesirable weed species. Such difficulties are likely to result in an increased cost per plant (Allen et al., 1989), compared to buying nursery plants.

Wetlands may be planted with either whole plants or simply with just the rhizomes and tubers of the plant. Allen et al. (1989) report that tubers having 20 - 25 cm of stem tend to be the most successful as the plants can obtain sufficient oxygen through the stem when the wetland is flooded. When handling planting stock it is essential that the plants and rhizomes remain moist as they are very sensitive to dessication (Venus, 1988). Once uprooted, the transportation and subsequent replanting of the plant stock must be undertaken as quickly as possible. Delays of as little as one day have been known to result in substantial or total plant mortality (Allen et al., 1989).

Although it is unrealistic to expect a 100% survival rate, Parker (1989) believes that with careful supervision a 90% survival rate is achievable. From initial planting an adequate stand of vegetation will take from six months to one year to develop (Smith, 1985). As the vegetation makes a significant contribution to the wastewater treatment, there will be a delay before full treatment efficiency is attained.

In New Zealand wetlands, a planting density of around four plants per square metre is normally used for all plant species (Venus, 1989). Allen et al. (1989) recommend that plants be allowed to become well established before the introduction of wastewater so as to allow the plants to overcome the planting stress. However, Parker (1989) has successfully introduced domestic sewage into a newly established wetland system thereby avoiding the expense of providing a temporary fresh water supply. Wetland environments are by nature rich in nutrients and organic material and unless the wastewater is particularly strong or a water supply was readily available, the use of fresh water appears unnecessary. The effect of nutrient rich effluent on a wetland is profound. The rate of growth increases markedly, with the size and strength of the plants being considerably higher than in natural environments. The effect of stagnation on plant growth is clearly illustrated in Figure 18 where the inlet was positioned about fifteen metres from the beginning of the wetland channel. This caused plant die off at the front of the system compared with vigorous growth in the rest of the system.

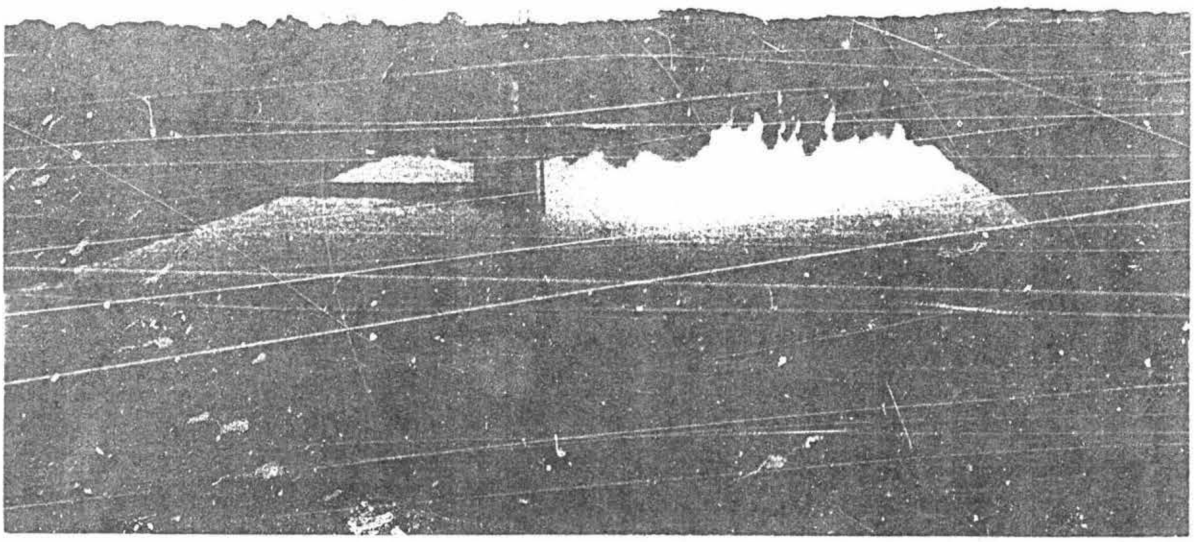

Figure 18. **Effect of Stagnation on Plant Growth**

6.4 Harvesting

Regular harvesting of plant biomass from wetland systems in order to achieve nutrient removal is not considered practical. A single late season harvest at Listowal removed 200 gm of plant material (dry weight) per cubic metre, however, this contained only 8% and 10% of the annual phosphorous and nitrogen loadings respectively (Miller et al., 1985 in USEPA, 1988).

It is, however, advisable to harvest the plants annually prior to their seasonal dieback, although the degree of dieback does vary between species and climates and, Tanner (1990) reports, the extent of dieback is reduced as nutrient enrichment of the water is increased. This regular harvesting will maintain the bed porosity and reduce the waste load of the dead plant material. As discussed in chapter two, oxygen transfer via the root zone decreases as the plants grow older. If the designer considers this process important in achieving the required treatment (for achieving nitrification in a gravel bed system used for polishing, for example), it is suggested that replanting of around a third of the wetland plants should be undertaken each year.

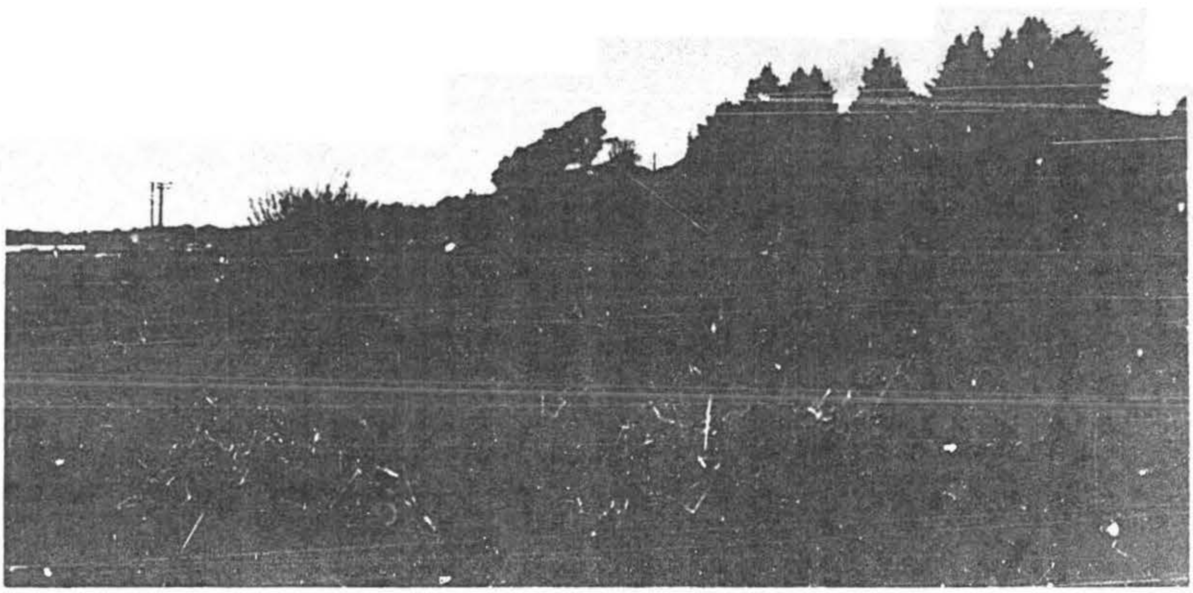

Figure 19. View of Dieback of *Typha* at Whangarei Trial Wetland

6.5 Design

As mentioned at the beginning of this chapter the selection and successful establishment of the wetland plants is one of the main challenges in the design of a wetland treatment system. There have been several instances of wetlands failing due to a lack of expertise in this area. It is therefore advisable that engineers involved in the design of wetlands should consult closely with aquatic botanists with knowledge in wetland species.

7. MANAGEMENT AND GENERAL CONSIDERATIONS

7.1 Public Perceptions and Project Planning

As discussed in chapter one, wetlands are places of significant biological importance and can be quite picturesque. However, people who have had little involvement with these environments often consider wetlands to be wastelands which are responsible for harbouring disease and producing mosquitoes. Generally public reaction to the concept of wetland treatment tends to border on surprise that a 'marshland' could improve water quality, and concern that depositing sewage into a wetland is in reality polluting part of the environment and is a potential health risk.

It must be recognised that waste disposal is a highly emotive issue, and wetlands, unlike enclosed conventional treatment systems, are usually clearly visible to the public. It is therefore essential to the overall success of a wetland project that issues of public perception are "addressed in the early planning stages" (Smardon, 1989, p289).

The key to gaining public support is through education, communication, and public participation. In the Levin Borough Council's sewage spray disposal project public support and approval was gained by:

(i) displays in the windows of the council building including photos of the site and a map and model of the proposed treatment system;

(ii) education via the local media of a) the need for the project (highlighting the pollution problems of Lake Horowhenua from existing waste disposal), b) the concept of the treatment system, and c) the benefits of the treatment system as against remaining with the status quo or alternative systems such as sea outfalls.

This public education programme, coupled with effective communication by the engineers, helped to overcome local citizens' initial qualms over health risks, unpleasant odours and so on, with the local community soon becoming firm supporters of the project. Although this represented a new concept in sewage disposal in New Zealand, the public appreciated that they were participating in the creation of an environmentally and socially sensitive waste treatment system (Shilton, 1987). Clearly similar methods should be used in the promotion of the development of a wetland system in a community.

After initial misconceptions are dispelled and the community becomes aware of the potential advantages of wetland systems, they usually receive strong public approval. The Whangarei wetlands are a prime example of this. The Whangarei City Council investigated the use of wetlands for secondary treatment (see Chapter 8), an option which soon gained widespread support from the community. Although the wetlands were not developed to the extent originally intended, a smaller 'polishing' wetland was constructed as the final stage of the treatment system apparently in recognition of the publics' desire for an alternative 'natural' type of waste treatment.

The design of this wetland places much emphasis on aesthetics and on the development of the 'natural' treatment concept. The land surrounding the wetland has been extensively planted, islands were built to provide refuges for birdlife, and walking tracks and lookouts have been incorporated throughout the area to promote and encourage public inspection of the system and its recreational use. The Whangarei scheme highlights the increasing involvement of the general public in waste treatment planning, and indeed, the growing awareness of planners and engineers of the importance of social and environmental issues.

In New Zealand, respect of the Maori community's spiritual values is an important aspect of the planning procedure. Venus (1988) reports that the general Maori concept of effluent disposal is that it be "disposed to ground where it is purified before entering any water from which food may be taken" (p37). Wetland treatment, according to Venus, closely mimics traditional waste disposal methods. However, he advises that it is important to stress that the wastes leaving the wetland are predominantly derived from the wetland (that is, vegetable detritus and so on) as opposed to effluent derived.

Venus (1987) asserts that the purification of wastes by wetland systems is generally supported by the Maori communities in which they are situated. However, the OPCE (Office of the Parliamentary Commissioner for the Environment) (1989) states that different tangata whenua groups have varying opinions on the cultural acceptability of wetland treatment. The Ngati Pihio in Rotorua, for example, do not find wetlands to be ulturally acceptable. Again, the process of discussion, public participation and education of the use of wetlands for waste treatment may overcome initial resistance to such systems and help to generate increased support of the local Maoris. Certainly wetlands are a more acceptable option of waste treatment than many other alternative methods, in particular direct discharges into surface waters, in particular.

In summary it can be seen that public opinions are very important in the development of innovative waste treatment systems. The concept of wetlands treatment has proved to be a particularly acceptable option, once some negative preconceptions have been overcome. Being a new technology the key to the successful implementation of a wetland project requires the enlightenment of the public via education, communication, and participation from the earliest stages of planning.

7.2 Aesthetics

7.2.1 Plants

Wetlands are frequently situated in full public view. Because of this it can be a worthwhile exercise to soften and beautify their appearance by undertaking a planting programme, as at the Whangarei wetland. The Whangarei City Council planted shrubs and trees extensively around the wetland, even installing an irrigation system to ensure their growth. Such planting acts to enhance the natural environment concept of the wetland system. They also function as wind breaks which can reduce wind mixing and serve as a buffer against any odours that may arise. Flowering plants, agapanthas and canna lillies for example, grow prolifically on the perimeter of wetlands. These not only augment the attractiveness of the wetland but also produce dense barriers of vegetation.

7.2.2 Odours

In a wetland odours are caused by mercaptons and skatoles (organic compounds which contain sulphur) and by hydrogen sulphide (H^2S) (Tchobanoglous, 1987). After the depletion of oxygen and nitrate, sulphate (SO_4^{2-}) will serve as an electron acceptor and is reduced to hydrogen sulphide. Tchobanoglous (1987) believes the key to avoiding odour problems rests with the avoidance of organic overloading.

7.2.3 Organic Overloading

Areas of a wetland will invariably become anaerobic. This is actually quite important in order to facilitate denitrification. If strong organic wastes are not contained in these areas no difficulties should be encountered as the anerobic zone will be overlaid by an aerobic zone near the surface which will oxidise the offensive byproducts of anerobic decomposition. However, anaerobic conditions resulting from organic overloading of strong wastes is likely to cause several problems (odours and mosquitoes, for example). The development of organic overloading occurs most frequently near the influent end of the wetland. For example, a plug flow wetland with a L/W ratio of 10:1, receiving an organic loading of 1g/m^{2}day has an actual loading of 10g/m^2.day over the first tenth of the system (Tchobanoglous, 1987).

To avoid this problem of overloading it is possible to set a limit for localised organic loading, and design the system to avoid this. Watson et al. (1989) suggest that the organic loading should not exceed a maximum value of 11g BOD/m^2.day. Thus the total surface loading should not exceed this value when concentrated over a length of one or two metres. A number of possibilities exist to avoid exceeding this localised loading limit. These can be seen as follows:

(i) use the limit to select the width of the wetland, however, this may result in a smaller than acceptable L/W ratio if the waste is strong.

(ii) reduce the organic loading before it enters the wetland by adding or increasing the pretreatment.

(iii) use step loading or dilution via recycling (as discussed in chapter 4).

7.3 Pest Control

The building of a wetland creates an organic and nutrient rich aquatic environment. It is therefore important to ensure that a pest problem does not occur and further, that pests do not interfere with the operation of the wetland itself.

7.3.1 Algae

Algae blooms will develop in these nutrient enriched waters when exposed directly to sunlight. Algae fixes carbon from the atmosphere which serves to increase the BOD loadings. However, this increase is partly offset by their production of oxygen. As algae grows freely suspended in the water column and as wetlands are plug-flow in design, this algae growth flows out of the system contributing to the effluent suspended solids. The diurnal respiration of algae may also create temporary anaerobic conditions during night time.

Anabaena and *Closterium* species were found in the Coromandel subsurface wetland, but only in insignificant quantities (Hale and Scarlet, 1989b). For surface flow wetlands the USEPA (1988) believes that algae growth will not be a problem as long as the water is shielded from sunlight by surrounding vegetation. However to achieve this the wetland vegetation must be dense.

7.3.2 Floating Plants

Floating aquatic plants such as *Lemma* spp. (refer to photo below) can smother the water surface of the wetland restricting oxygen diffusion across the air/water interface, and thereby reducing the aerobic degradatin of the pollutants. As for algae control, floating plant growth should be restricted by shading if a dense cover of wetland emergent plants is established over the surface of the wetland.

7.3.3 Sewage Fungus

Sphaerotilus natans or sewage fungus has appeared in the Coromandel wetland when the BOD increased (Hale and Scarlett, 1989b). This filimentus bacteria has the potential to clog packed beds. However this problem was easily remedied by decreasing the BOD load by pre-aeration (Venus, 1987).

7.3.4 Insects

The organically rich quiescent water of a wetland is an ideal habitat and breeding ground for many insects, in particular, mosquitoes. Biological controls such as predation by birds and fish are normally adequate to keep the populations in check. A number of reports document the introduction and success of *Gambusia* spp. (mosquito fish) for this purpose (Venus, 1987).

Severe mosquito problems have been known to occur when a wetland has become organically overloaded and anaerobic conditions have developed (Stowell et al., 1984). Obviously the primary concern should be the prevention of these conditions (as discussed in section 7.2.3). Although chemical controls are not recommended for regular use an occasional application of an insecticide or a light oil which suffocates the larvae, will bring excessive insect populations back down to manageable levels.

7.3.5 Wildlife

In general, it is worthwhile to encourage the habitation of wildlife in and around a wetland in order to further the concept of a 'natural' treatment system. However, problems have occurred with rabbits, hares, and waterfowl attacking and eating young plants. Pukekos are by far the worst culprits and have been reported to cause significant problems in the establishment of new wetlands in New Zealand (Armstrong, 1989) Control is possible by shooting, poisoning or trapping.

7.4 Monitoring

Monitoring assesses the actual success and the impact of a wetland system. Basic monitoring of treatment efficiency is often required as a condition of being granted a water right. In addition, as wetlands are still in a developmental stage much valuable information about design concepts and their effectiveness can be gleaned from the monitoring of existing systems.

The scope of monitoring will depend on the proposed objectives of the wetland system. For example, constructing a wetland for a combination of water renovation and wildlife enhancement requires a monitoring strategy that will survey the success of both these objectives (Hicks and Stobler, 1989). Each objective requires a monitoring plan, for example, do the island refuges of the Whangarei wetland actually increase bird populations and diversity? Regular counts of bird number and types and occasional inspection of the islands and so on, will conclude whether the effort and cost of providing these islands enabled the objective to be achieved and in addition, may provide valuable insight into the future modification of the design.

The most common monitoring that is undertaken is of the treatment effectiveness, both because this is usually the key objective and, as mentioned, is normally a legal requirement. Typically the parameters of interest include BOD, suspended solids, and coliform bacteria, all of which are relatively simple to measure. However, where the emphasis of a wetland design is on nutrient removal or treatment of more toxic contamination the range of parameters would need to be expanded (Hicks & Stober, 1989). In monitoring the effluent from a wetland it is important to attempt to quantify the residue waste from the influent as opposed to wastes originating from within the wetland itself. For example, waterfowl can automatically increase coliform counts (Parker, 1989) and, as Venus (1988) suggests, a significant portion of the effluent BOD and suspended solids are wetland derived. These distinctions are of great importance in terms of public attitudes and water right allowances.

8. REVIEW OF WETLAND TREATMENT APPLICATIONS

Wetlands have a host of treatment mechanisms and are therefore successful at removing and/or treating a wide range of contaminants. Because of this treatment versatility wetlands can be used to treat a broad range of wastewaters, and can achieve a number of different treatment objectives.

8.1 Domestic Sewage

The major use of wetland treatment systems in New Zealand is for the treatment of septic tank effluent from small communities (usually preceded by pond treatment), and for the upgrading the effluent from existing treatment ponds.

8.1.1 Rawene, Hokianga County - Surface Flow Wetland

This wetland has now been in operation for seven years and was one of the first such systems constructed in New Zealand. The scheme receives septic tank effluent from the township of Rawene (current population 400). The effluent receives initial treatment by a series of two facultative ponds, the first having an area of 4150m^2, the second 2200m^2. At the design flow of 125m^3/day (design population 500) these ponds provide a sixty day retention period (Jacobson, 1989b). The wetland consists of a serpentine style marsh planted with raupo, with an area of 4430m^2 and a design depth of 0.3m. The design retention time in the wetland is fourteen days and thus the hydraulic loading rate is about 3cm/day. In 1983 this project cost only \$67,000 to complete (Jacobson, 1989b).

The designers (Fraser Thomas Partners) expected that other marsh species would "eventually root and prosper" (Jacobson, 1989b, p24). However, as discussed in chapter six, *Typha* spp. such as raupo will often dominate in marshlands. This was the case at Rawene, with subsequent plantings of other plant species proving unsuccessful. Although the raupo has grown vigorously and become very dense in areas, it unfortunately undergoes extensive dieback over winter. This has caused seasonal variations in treatment potential and has severely clogged the system with vegetative detritus, reducing the water depth and thereby shortening the retention time. Fortunately the system was built with adequate freeboard making it relatively simple to raise the water level and thus restore the design retention time.

The wetland system was primarily built to reduce the faecal bacterial pollution of the Hokianga Harbour (CILM, 1987). In order to supervise the scheme's water rights the Northland Regional Council monitored the system, the results of which are shown in Table 4.

Generally the marsh has proven capable of very good bacterial reductions, normally in the order of 90%. Parker (1988, in Jacobson, 1989c) attributes the drop to 66.5% removal to the reduction in retention time after the excessive litter build up. Once the water depth was returned to 0.3m the efficiency was restored to the 90% range. Although only limited data was collected on the nutrient removal, nitrogen removal was generally around 80 - 90%. The poor results recorded in 1989 may be attributed to the release of nutrients from the decomposition of the excessive detritus accumulation. However, on the whole the phosphorus levels were actually significantly increased as the wastewater passed through the wetland.

Table 4 **Monitoring data from the Rawene Wetland**
(Tabulated from Jacobson, 1989c)

		FAECAL COLIFORMS (per 100ml)	AMMONIA (mg/l)	NITRATE (mg/l)	REACTIVE PHOSPHORUS (mg/l)
6/8/85	**In**				
	Out	<20			
12/9/85	**In**	900	72	0.619	0.057
	Out	100(89%)	0.013(99%)	0.113(82%)	0.003(94%)
5/12/85	**In**	1850			
	Out	120(94%)			
13/3/86	**In**	200	0.130	0.171	0.420
	Out	60(70%)	0.009(93%)	0.040(77%)	0.477(-12%)
22/4/86	**In**	4100	0.016	0.094	0.656
	Out	20(99.5%)	0.001(99%)	0.015(84%)	1.356(-52%)
24/6/86	**In**	3700	0.035	0.583	0.061
	Out	240(93.5%)	0.016(54%)	0.010(98%)	2.150(-97%)
2/7/86	**In**	2600			
	Out	730(83.5%)			
25/8/87	**In**	17000			
	Out	500(97%)			
13/10/87	**In**	42000			
	Out	2900(93%)			
17/3/88	**In**	44000			
	Out	14700(66.5%)			
23/1/89	**In**	31300			
	Out	8000(74%)			
15/3/89	**In**	4300	3.420	0.010	3.545
	Out	3600(92%)	104.65(-97%)	0.140(-93%)	52.65(-93%)

Overall, Parker (1989) and Jacobson (1989a) are pleased with the performance of the wetland, and its success has led to several more wetlands being constructed in the county. Local Maori leaders have been especially interested in the scheme, and feel it provides a "satisfactory solution to the problem of the polluted harbour and contaminated seafood" (CILM, 1987, p24). It is clear that the Rawene wetland has had some faults in terms of plant selection and management, but these problems have led to design improvements in subsequent systems.

8.1.2 Coromandel Township - Subsurface Flow Wetland

Like the Rawene system, the Coromandel wetland was constructed to treat the septic tank effluent of a small township. In both systems poor soil absorption of the septic tank effluent was resulting in an effluent runoff into local streams and harbours resulting in unacceptable bacterial counts. In Coromandel this was of particular concern to the local aquaculture industry (Scarlet, 1989). The Coromandel system consists of an aerated facultative pond of $4100m^2$ (designed to reduce the BOD by 74% to 78mg/l), followed by two subsurface gravel wetlands ($6720m^2$), with final treatment being provided by ultra-violet sterilisation. The system has been designed for a peak flow of $680m^3$/day (design population of 4050) (Hale & Scarlet, 1989a) which equates to a hydraulic loading of 10cm/day. This scheme has been operational for two years and is performing extremely well.

The only problem encountered with the system to date has been the formation of sewage fungus in the bed when the aerator was turned off. The problem soon cleared when the aerator was reactivated (Venus, 1989).

Table 5. Monitoring Data from the Coromandel Wetland
(Tabulated from Hale and Scarlet, 1989b)

		10/12/86	11/01/89	23/02/89	24/02/89	27/02/89
Faecal	In	40,000	1.3×10^5	8×10^5	2×10^5	5×10^4
Coliforms	Out	<2,000	2.3×10^4	3×10^3	1.3×10^4	3×10^3
(per 100ml)	Change	95%	82%	99.6%	93.5%	94%
BOD	In	78	78	78	78	78
(mg/l)	Out	6.0	8.3	7.5	9.6	8.9
	Change	92%	89%	90%	88%	89%
Suspended	In		36.3			
Solids	Out	3.0	6.9	15.0	9.0	12.0
(mg/l)	Change		81%			

8.1.3 Matakana, Rodney County - Combined Subsurface/Surface Flow Wetland

This wetland receives raw septic tank effluent from a current population of 150 plus a small primary school, but is designed to accommodate a population of 300. The system consists of two parallel subsurface gravel wetlands (total of 400m^2) followed by a surface flow wetland split into four cells (total of 450m^2) (Armstrong, 1989b). With a design flow of around 50m^3/day the hydraulic loading rates of the wetlands are 12.5cm/day for the subsurface primary treatment and 11cm/day for the surface flow secondary treatment.

The system was planted in August 1988 and is subject to a comprehensive monitoring programme. To date nine samples have been taken over a period of around ten months and the results indicate outstanding treatment performances.

Table 6. Summary of Monitoring Data from Matakana Wetlands

(Adapted from Armstrong, 1989b)

	Range of Influent Concentration	Average Reduction
BOD	41 to 397 (mg/l)	96%
Faecal Coliforms	1.1×10^5 to 55×10^5 (per 100 ml)	99.89%
Ammonia	17 to 200 (mg/l)	85%
Nitrate	0.003 to 0.167 (mg/l)	-75%
Total Phosphorus	2.6 to 35.9 (mg/l)	90%

Faecal coliform removal is in the range of three orders of magnitude, BOD is in the 95% range, and the often difficult phosphorus contamination is being consistently reduced by around 90%. Nitrogen removal is also very good with influent ammonia being reduced around 85% from the range of 17 to 200mg/l. However, the nitrate levels are markedly increased through the system. As quantities of ammonia are being nitrified to nitrate there appears to be insufficient subsequent denitrification to result in a net nitrate loss. I believe this could be remedied by building a final surface flow system as these systems are generally anaerobic and therefore conducive to denitrification.

8.2 Non Point Pollution

Over the last few decades there has been significant progress in reducing pollution from municipalities and industry by the installation and upgrading of wastewater treatment prior to the discharge point. As these waste loads are reduced the contribution of non point pollution becomes increasingly more significant. Non point pollution involves the discharge or runoff of pollutants over a large area as opposed to the point discharges from reticulated sewerage systems.

In New Zealand there appears to be two main sources of non point pollution: septic tank effluent, and agricultural runoff and discharges. Small towns or households which rely on septic tanks can cause pollution if the raw effluent rapidly seeps through the soil (as in Rotorua), or runs off the land (as in Coromandel) into surface water. Agricultural pollution is a particular concern around intensively farmed areas such as the Waikato. Animal wastes and fertiliser applications are washed off the land and there are numerous small point discharges from the effluent ponds of milking sheds, piggeries and so on.

8.2.1 On Site Treatment of Domestic and Agricultural Wastewaters

As discussed earlier, wetlands have been successfully used for the treatment of septic tank effluent in small communities subsequent to the installation of sewerage reticulation. As wetlands are simple to construct and operate they have also proven successful as an extension to 'on site' treatment. Small wetlands are effective at up-grading the effluent from household septic tanks and farm effluent ponds. In this way the pollution is reduced before leaving its source.

Kloosterman and Griggs (1989) report five examples of New Zealand wetlands receiving domestic effluent flows of less than $30m^3$/day.

Table 7. Examples of On-Site Treatment in New Zealand

(Adapted from Kloosterman & Griggs, 1989, p21)

	Flow	Waste Type	Wetland Type	Hydraulic Loading
Hunua Youth Camp Franklin Country	$15m^3$/day	septic tank	subsurface	10cm/day
Mototapu Outdoor Ed. Centre (Auck)	$26m^3$/day	septic tank/ sand filter	natural	?
Tahuna Pa Papakainga Trust - Franklin County	$26m^3$/day	septic tank	subsurface	21.6cm/day
Kohukohu Hokianga County	$30m^3$/day	oxidation pond	surface	2.5cm/day
Te Kao School Mangonui County	$7.5cm^3$/day	aeration & settlement	surface	0.4cm/day

Discharges from farm oxidation ponds in New Zealand contain high levels of ammonia and are on average four to six times stronger than domestic sewage discharges (Tanner and Clayton, 1990). Upgrading the standard two stage pond system commonly used throughout the country by the addition of a wetland treatment system has the potential to significantly improve the water quality of water bodies polluted by farm discharges.

Chris Tanner and John Clayton of the MAF Ruakura Agricultural Centre in Hamilton are currently undertaking extensive research into the application of wetlands to agricultural wastes. Their work pre-dominantly involves the use of sub-surface flow wetlands and involves trials on piggery and dairy shed wastes, as well as investigations into the potential of different plant species and different design rates/configurations.

In the United States, Hammer et al. (1989b) also report on constructed wetlands being used in trials on the treatment of piggery effluent. The waste receives primary treatment by two lagoons (estimated to remove 60% of BOD) prior to entering the wetland system at a loading rate of $150m^2$/kgBOD/day. The wetland is comprised of three cells with an area of around 0.6ha. Hammer et al. (1989a) recommend loading rates of $130m^2/kgBOD_5$/day for dairy cattle wastes.

8.2.2 Renovation of Polluted Runoff and Receiving Waters

The most difficult form of non point pollution to treat is the contaminated runoff from farmland and urban areas. Wentz(1987) suggests that wetlands have great potential to deal with this problem by "strategic placement of numerous small wetlands" in areas where they will intercept and renovate contaminated runoff (p21).

Alternatively wetlands can be situated to intercept and renovate all or part of a contaminated stream, river or even lake. Such a programme has been undertaken in Illinois. A 182ha wetland has been created on the flood plain adjacent to a 5km stretch of river. The river, contaminated by up stream non point sources, will firstly flow through a quarry lake (sediment trap) before a portion is redirected through the wetlands (Smardon, 1989).

8.3 Industrial and Toxic Wastewaters

8.3.1 Pulp and Paper Mill Effluent

In 1934, Allender reported that his studies found that the use of wetland species "substantially improved water quality" of pulp and paper mill effluent (p305). More recently Thut (1989) conducted an investigation using eight trial marshes over a three year period to polish pretreated pulp effluent. Unfortunately the study failed to ratify earlier trials which suggested that wetlands had a capacity to remove the problem of the water discolouration. It did, however, show the wetland to be capable of providing polishing to what was already a high quality effluent, as detailed below.

Table 8. Polishing Efficiencies of Pulp Mill Effluent

(Adapted from Thut, 1989, pp241-242)

	Initial	Control	Cordgrass	Cattail	Reed
BOD	10-15mg/l	45%	44%	49%	36%
SS	5-8mg/l	46%	42%	58%	47%
Ammonia	1.5-3mg/l	24%	75%	79%	79%
Organic N	2.5-3.5mg/l	38%	36%	33%	37%
Phosphorus	0.5-0.8mg/l	8%	18%	41%	26%

It is interesting to note that the presence of the plants only affected ammonia and phosphorus removal. Based on a fifteen hour retention time Thut (1989) concluded that twenty to forty hectares would be suitable for the polishing of effluent from a typical

American pulp mill. This very short retention time may explain why better results were not obtained for the other waste parameters.

8.3.2 Acid Mine Drainage

The water drainage from coal mining operations is acidic, discoloured, and has high concentrations of iron, manganese, aluminium, calcium, and suspended solids (USEPA, 1971 in Brodie et al., 1987). The traditional method of treating this waste is by chemical dosing and use of sedimentation basins. In 1985, however, the Tennessee Valley Authority (TVA) opened the first of a series of wetlands designed to treat this wastewater. The 950 Mine wetland consists of a series of three surface flow wetlands receiving an average flow of 125m^3/day. The wetlands are 0.3 to 0.5m deep, have an area of around 0.43ha (corresponding to an average hydraulic loading rate of 2.9cm/day), and are planted predominantly with cattails (Brodie et al., 1989).

Table 9. Treatment Efficiencies of Acid Mine Drainage

(Tabulated from Brodie et al., 1989)

	Before	**After**
Acidity (pH)	6.1	6.9
Iron	14.4mg/l	0.8mg/l
Manganese	4.8mg/l	1.1mg/l
Suspended Solids	24mg/l	7mg/l
Downstream macro-invertabrate species	2	19

Although a number of studies have documented the effectiveness of wetlands in removing acidity, manganese, sulphates and other contaminants, there have also been several reports of trial wetlands failing to achieve their desired results, thus some caution is required in design (Brodie et al., 1986). The TVA predominantly used surface flow wetlands. Although these systems have lower capacity for surface adsorption than subsurface wetlands they do have two distinct advantages for treatment of this type of wastewater:

(i) they have better oxygenation and are generally more aerobic giving them greater capacity to oxidise dissolved iron and manganese resulting in its subsequent precipitation, and

(ii) they are markedly less susceptible to clogging which is particularly important because of the large quantities of solid precipitates produced in these systems.

8.3.3 Landfill Leachate

Trautmann et al.(1989) are investigating the use of wetlands to treat leachate from a municipal solid waste landfill. The system consists of two surface flow wetlands followed by two subsurface flow gravel wetlands. Trautmann et al. (1989) believe that the initial surface flow wetlands will remove the majority of the dissolved iron and manganese via oxidation and precipitation, as well as oxidising organics and nitrogen, and volatilising ammonia and benzene. The following two beds are intended to denitrify nitrates, further decompose organics, and absorb and precipitate phosphorus and metal ions.

This sort of application is perhaps starting to exceed the bounds of what can be reasonably expected from a wetland treatment system. Although wetlands have produced some excellent results for lower strength wastes, designers must be very careful not to over-estimate the mechanisms of treatment and should remember that it is essentially a living eco-system.

SUMMARY

The use of wetland treatment systems represents an innovative approach to wastewater treatment. Treatment by wetlands utilises a whole ecosystem, the diverse chemical, physical and biological characteristics of which have been found capable of very effective wastewater renovation. Although the mechanisms involved in wetland treatment are numerous and complex, wetland systems are simple to build and operate in comparison to the conventional 'concrete and steel' wastewater technologies.

Wetland treatment is a versatile technology which has been successfully applied to wastewater flows ranging from single houesholds up to cities and large industry. However, the potential application of these systems will always be limited by their relatively large land area requirement. Wetland treatment will not be a viable option where the terrain is hilly or uneven to the extent that the cost of the earthworks required to produce a level bed become prohibitive, and where high land costs or unavailability of sufficient land (such as near large cities) make the option unfeasable.

Constructed wetlands mimic natural marshes and need be no more complex to provide effective renovation, however certain bed and flow arrangements can be used to optimise the potential of a system. They can generally be built using local materials and labour, and require little or no machinery. As a consequence they are inexpensive to construct and operate and have a nil or minimal energy requirement. Additionally once they are established and operational they generally require very little supervision and technical input. Wetlands are obviously ideally suited for providing wastewater

treatment to the numerous small populations, farms and industries scattered throughout New Zealand.

The majority of the wetlands built in New Zealand to date have had the primary objective of reducing pathogen levels and/or the organic loading of the wastewater. However wetlands are also very effective at reducing the levels of secondary pollutants. Indeed, a number of wetlands have been constructed as extensions to existing treatment plants. Whereas nutrient stripping and toxic pollutant removal is complicated and expensive by conventional methods, wetlands can simply and economically provide effective treatment in situations where the pollution would otherwise have to go unchecked. Wetlands also provide a feasible method of combating the increasingly significant problem of non point pollution.

Although wetlands have proven capable of providing outstanding treatment to a wide range of effluents it is important not to overestimate the ability of these systems. With particular regard to organic loading, wetlands have only limited oxygenation and are simply unviable for high strength wastes without prior treatment. A severely overloaded system has the potential to create a number of very undesirable conditions and at this stage of wetlands development, could bring the system into general disrepute.

Wetlands offer an environmentally and socially sensitive approach to wastewater treatment. Not only does their construction provide an important habitat for a number of rare and endangered bird and wildlife species, but wetlands also offer a viable method of dealing with the problem of pollution by secondary contaminants such as nitrogen, phosphorus and metals, that are often neglected in conventional treatment systems. Additionally wetlands appeal to the greater public desire for a 'natural' but effective approach to waste treatment, in particular they receive the general support of the Maori community as a culturally acceptable waste treatment alternative.

A number of New Zealnd engineers and water quality officials have recognised the great potential that wetlands have in this country and have boldly taken the initiative to develop these systems in New Zealand. There are currently more than twenty wetlands in operation in New Zealand with several more under construction, a significant number by international standards. As could be expected, some operating problems have been encountered but these have been remedied with modifications to the design. Overall the New Zealand wetlands have achieved their design objectives, with many of them producing unprecedented results.

It is my belief that as research and development of these systems progress, awareness of the benefits, and confidence in these systems will grow and wetland treatment systems will become a popular and increasingly common method of treating New Zealand's wastewaters.

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APPENDIX

1. Performance and Hydraulic Loading Rates of Constructed Wetland Systems for Small Municipal Systems in North America and Europe.

2. Hydraulic Loading Rates for Constructed Wetlands in North America.

3. TVA Acid Drainage Wetlands Treatment Summary/Performance of Constructed Wetlands for Acid Mine Drainage.

Source: Watson et al (1989)

Performance and Hydraulic Loading Rates of Constructed Wetland Systems for Small Municipal Systems in North America and Europe

North American Systems	Ref.	Period of Record	System Type	Hydraulic Loading Rate (cm/day)	Biochemical Oxygen Demand Influent Conc. (mg/L)	Biochemical Oxygen Demand Effluent Conc. (mg/L)	Biochemical Oxygen Demand Removal Efficacy (%)	Suspended Solids Influent Conc. (mg/L)	Suspended Solids Effluent Conc. (mg/L)	Suspended Solids Removal Efficacy (%)
Listowel, Ontario	15	9/80–8/84	Surface							
			System 3	1.4[illegible]	19.6	7.6	61	22.8	9.2	60
			System 4	1.40	56.3	9.6	83	111.1	8	93
Arcata, California	16	9/80–8/82	Surface							
			Cells 1–4	22.3	26	12.8	51	37	5.4	85
			Cells 5–8	11.2	26	13.3	49	37	5.7	85
			Cells 9–12	5.61	26	10.7	59	37	6.1	84
Brookhaven NL, New York	17	8/75–8/76	Surface Marsh/Pond	3.35	170	19	89	353	43	88
Santee, California	18, 19	8/83–12/84[d]	Subsurface Gravel							
			Bulrush	4.68	118.3	5.3	96	58.1	3.7	94
			Reed	4.68	118.3	22.3	81	58.1	7.9	86
			Cattail	4.68	118.3	30.4	74	58.1	5.5	91
			Control	4.68	118.3	36.4	69	58.1	5.6	90
Village of Neshaminy Falls, Pennsylvania	20	9/79–7/82	Subsurface Sand Marsh/Pond/Meadow[e]	1.26	187	8	96	193	12	94
Iselin, Pennsylvania	21	3/83–9/85	Subsurface Sand/Gravel							
			Marsh/Pond/Meadow	1.47	140	7.4	95	380	19	95
			Marsh	5.28	140	17	88	380	53	86
			Meadow	10.57	20	7.4	64	61	19	69
Benton, Kentucky	22	3/88–11/88	Surface/Subsurface							
			Surface Cattail	4.15	23	10	57	60	15	75
			Surface Woolgrass	4.27	23	11	52	60	20	67
			Subsurface Bulrush	7.97	23	8	65	60	7	88

North American Systems	Ref.	Period of Record	System Type	Hydraulic Loading Rate (cm/day)	Ammonia Nitrogen Influent Conc. (mg/L)	Ammonia Nitrogen Effluent Conc. (mg/L)	Ammonia Nitrogen Removal Efficacy (%)	Total Nitrogen Influent Conc. (mg/L)	Total Nitrogen Effluent Conc. (mg/L)	Total Nitrogen Removal Efficacy (%)
Listowel, Ontario	15	9/80–8/84	Surface							
			System 3	1.40	7.2	3.8	47	12.3	6.3	49
			System 4	1.40	8.6	6.1	29	19.1	8.9	53
Arcata, California	16	9/80–8/82	Surface							
			Cells 1–4	22.3	12.8	9.8	23			
			Cells 5–8	11.2	12.8	11.6	9			
			Cells 9–12	5.61	12.8	9.6	25			
Brookhaven NL, New York	17	8/75–8/76	Surface							
			Marsh/Pond	3.35	8.4	3.5	58	19.7	6.8	65
Santee, California	18, 19	8/83–12/84[d]	Subsurface Gravel							
			Bulrush	4.68	24.7	1.5	94			
			Reed	4.68	24.7	5.4	78			
			Cattail	4.68	24.7	17.7	28			
			Control	4.68	24.7	22.1	11			
Village of Neshaminy Falls, Pennsylvania	20	9/79–7/82	Subsurface Sand							
			Marsh/Pond/Meadow[e]	1.26	26.1	6.4	75			
Iselin, Pennsylvania	21	3/83–9/85	Subsurface Sand/Gravel							
			Marsh/Pond/Meadow	1.47	30	3.3	83			
			Marsh	5.28	30	13	56			
			Meadow	10.57	5.2	3.3	36			
Benton, Kentucky	22	3/88–11/88	Surface/Subsurface							
			Surface Cattail	4.15	6.2	7.6	–23	14.8	10.9	26
			Surface Woolgrass	4.27	6.2	5.8	6	14.8	9.3	37
			Subsurface Bulrush	7.97	6.2	11.2	–81	14.8	13.0	12

North American Systems	Ref.	Period of Record	System Type	Hydraulic Loading Rate (cm/day)	Phosphorus[a] Influent Conc. (mg/L)	Phosphorus[a] Effluent Conc. (mg/L)	Phosphorus[a] Removal Efficacy (%)	Coliforms[b] Influent Conc. (No./100 mL)	Coliforms[b] Effluent Conc. (No./100 mL)	Coliforms[b] Removal Efficacy (%)
Listowel, Ontario	15	9/80–8/84	Surface							
			System 3	1.40	1	0.5	50	1,736	53	97
			System 4	1.40	3.2[c]	0.6	81	222,990	121	100
Arcata, California	16	9/80–8/82	Surface							
			Cells 1–4	22.3				3,183	389	88
			Cells 5–8	11.2				3,183	584	82
			Cells 9–12	5.61				3,183	367	88
Brookhaven NL, New York	17	8/75–8/76	Surface Marsh/Pond	3.35	7.2	2.1	71	1,560	50	97
Santee, California	18 and 19	8/83–12/84[d]	Subsurface Gravel							
			Bulrush	4.68				67,500,000	577,000	99
			Reed	4.68						
			Cattail	4.68						
			Control	4.68				67,500,000	289,000	96
Village of Neshaminy Falls, Pennsylvania	20	9/79–7/82	Subsurface Sand Marsh/Pond/Meadow[e]	1.26				1,290,600	5,600	100
Iselin, Pennsylvania	21	3/83–9/85	Subsurface Sand/Gravel							
			Marsh/Pond/Meadow	1.47	13	2.6	80	1,400,000	150	100
			Marsh	5.28	13	4.2	69	1,400,000	3,700	100
			Meadow	10.57	3.4	2.6	23	2,100	150	93
Benton, Kentucky	22	3/88–11/88	Surface/Subsurface							
			Surface Cattail	4.15	6.0	5.3	12	3,940	515	87
			Surface Woolgrass	4.27	6.0	4.9	18	3,940	94	98
			Subsurface Bulrush	7.97	6.0	5.1	15	3,940	157	96

European Systems	Ref.	Period of Record	System Type	Hydraulic Loading Rate (cm/day)	Biochemical Oxygen Demand Influent Conc. (mg/L)	Biochemical Oxygen Demand Effluent Conc. (mg/L)	Biochemical Oxygen Demand Removal Efficacy (%)	Suspended Solids Influent Conc. (mg/L)	Suspended Solids Effluent Conc. (mg/L)	Suspended Solids Removal Efficacy (%)
Gravesend, England	23	4/86–1/88	Subsurface Gravel							
			Bed 1	8.16	237	84	65	131	64	51
			Bed 2	8.16	237	90	62	131	58	56
			Bed 3	8.16	237	63	73	131	38	71
Marnhull, England	23	5/87–1/88	Subsurface Soil							
			Bed 1	4.46	87	13	85	74	23	69
			Bed 2	6.90	87	17	80	74	20	73
Holtby, England	23	7/86–1/88	Subsurface Soil	4.90	223	49	79	163	24	85
Castleroe, England	23	4/87–1/88	Subsurface Gravel							
			Cell 1	4.34	157	53	66	73	30	59
			Cell 2	4.34	157	70	55	73	29	60
			Subsurface Soil							
			Cell 1	4.34	157	37	76	73	33	55
			Cell 2	4.34	157	60	62	73	37	49
Middleton, England	23	6/87–1/88	Subsurface Sand/Gravel	8.89	11.0	3.0	73	30	8	73
Bluther Burn, England	23	5/87–3/88	Subsurface							
			Fine Fly Ash	10.76	207	49	76	167	26	84
			Coarse Fly Ash	6.24	207	43	79	167	17	90
			Unclassified Gravel	9.93	207	30	86	167	26	84
			Gravel	10.09	207	51	75	167	17	90
Little Stretton, England	23	7/87–12/87	Subsurface Gravel	26.0	140.9	32.9	77	127	20	84
Ringsted, Denmark	24	9/84–10/84	Subsurface							
			Gravel	5.70	189	11	94	243	6	98
			Clay	1.71	189	15	92	243	23	91

European Systems	Ref.	Period of Record	System Type	Hydraulic Loading Rate (cm/day)	Ammonia Nitrogen Influent Conc. (mg/L)	Ammonia Nitrogen Effluent Conc. (mg/L)	Ammonia Nitrogen Removal Efficacy (%)	Total Nitrogen Influent Conc. (mg/L)	Total Nitrogen Effluent Conc. (mg/L)	Total Nitrogen Removal Efficacy (%)
Gravesend, England	23	4/86–1/88	Subsurface Gravel							
			Bed 1	8.16	43	37	14			
			Bed 2	8.16	43	36	16			
			Bed 3	8.16	43	36	16			
Marnhull, England	23	5/87–1/88	Subsurface Soil							
			Bed 1	4.46	28.9	26.4	9			
			Bed 2	6.90	28.9	28.2	2			
Holtby, England	23	7/86–1/88	Subsurface Soil	4.90	34.9	32.9	6			
Castleroe, England	23	4/87–1/88	Subsurface Gravel							
			Cell 1	4.34	19.1	14.5	24			
			Cell 2	4.34	19.1	17.3	9			
			Subsurface Soil							
			Cell 1	4.34	19.1	13.9	27			
			Cell 2	4.34	19.1	18.2	5			
Middleton, England	23	6/87–1/88	Subsurface Sand/Gravel	8.89	2.8	1.5	46			
Bluther Burn, England	23	5/87–3/88	Subsurface							
			Fine Fly Ash	10.76	27	24	11			
			Coarse Fly Ash	6.24	27	20	26			
			Unclassified Gravel	9.93	27	20	26			
			Gravel	10.09	27	22	19			
Little Stretton, England	23	7/87–12/87	Subsurface Gravel	26.0	9.8	10.2	–4			
Ringsted, Denmark	24	9/84–10/84	Subsurface							
			Gravel	5.70	34	15	56	48	29.6	38
			Clay	1.71	34	15.3	55	48	18.3	61

European Systems	Ref.	Period of Record	System Type	Hydraulic Loading Rate (cm/day)	Phosphorus[a]			Coliforms[b]		
					Influent Conc. (mg/L)	Effluent Conc. (mg/L)	Removal Efficacy (%)	Influent Conc. (No./100 mL)	Effluent Conc. (No./100 mL)	Removal Efficacy (%)
Gravesend, England	23	4/86–1/88	Subsurface Gravel							
			Bed 1	8.16	12.6	7.2	42			
			Bed 2	8.16	12.6	7.5	40			
			Bed 3	8.16	12.6	5.0	60			
Marnhull, England	23	5/87–1/88	Subsurface Soil							
			Bed 1	4.46						
			Bed 2	6.90						
Holtby, England	23	7/86–1/88	Subsurface Soil	4.90	7.79	6.82	12			
Castleroe, England	23	4/87–1/88	Subsurface Gravel							
			Cell 1	4.34	5.0	4.0	20			
			Cell 2	4.34	5.0	4.0	4			
			Subsurface Soil							
			Cell 1	4.34	5.0	2.5	50			
			Cell 2	4.34	5.0	5.6	–12			
Middleton, England	23	6/87–1/88	Subsurface Sand/Gravel	8.89	10.8	7.6	30			
Bluther Burn, England	23	5/87–3/88	Subsurface							
			Fine Fly Ash	10.76	10.49	3.64	65			
			Coarse Fly Ash	6.24	10.49	1.71	84			
			Unclassified Gravel	9.93	10.49	0.93	91			
			Gravel	10.09	10.49	3.14	70			
Little Stretton, England	23	7/87–12/87	Subsurface Gravel	26.0						
Ringsted, Denmark	24	9/84–10/84	Subsurface							
			Gravel	5.70	15	9.6	36			
			Clay	1.71	15	6.0	60			

[a]All data are for total phosphorus except the English systems reported orthophosphate (as P).
[b]All data are for fecal coliforms except the Santee system reported total coliforms.
[c]Alum treatment provided prior to wetlands.
[d]Period of record for coliform data was January through December 1985.
[e]Influent data are for raw sewage. The sewage receives primary treatment (aeration cell) prior to the marsh. Effluent data are for the final effluent after chlorination except for the fecal coliform data, which are for the meadow effluent.

Hydraulic Loading Rates for Constructed Wetlands in North America

Project	Ref.	Configuration	Type	Number of Cells	Flow (1000 m³/day) Design	Flow (1000 m³/day) Actual	Hydraulic Loading Rate (cm/day) Design	Hydraulic Loading Rate (cm/day) Actual
Anne Arundel County, MD	60	series		3				
Freshwater wetland			surface		4.27		15.1	
Peat wetland			percolation		3.46		10.2	
Offshore wetland			surface		1.89		1.09	
Arcata, CA (Pilot)	61	parallel/series	surface	12	1.07	0.119	24.0	2.66
Benton, KY	62	parallel		3	4.16	2.32	9.48	5.30
Cell 1			surface	1	1.04	0.604	7.11	4.15
Cell 2			surface	1	1.04	0.620	7.11	4.26
Cell 3			subsurface[a]	1	2.08	1.16	12.2	7.97
Brookhaven, NY	17	marsh/pond		2				
Marsh			surface			0.0379		2.26
Pond			surface			0.0757		4.53
Cannon Beach, OR	63	single	surface	1	3.44		5.71	
Cobalt, Ontario, Canada	27	serpentine	surface	1		0.0186–0.093		2.0–10.0
Collins, MS	64	serpentine	surface		1.56		3.84	
East Lansing, MI	26	marsh/ponds	surface			2.84		1.80
Emmitsburg, MD	61	single	subsurface	1		0.110		16.4
Foothills Pointe, TN	22	single	subsurface	1	0.0676		4.68	
Fort Deposit, AL	65	parallel	surface	2			1.51	
Gustine, CA	61	parallel	surface	24	3.79		3.80	
Hardin, KY	62	parallel	subsurface	2	0.379		5.92	
Harriman, NY	22	marsh/pond/meadow	surface	2		0.114		
Houghton Lake, MI	66	single	surface	1		10.0		0.143
Incline Village, NV	67	series/serpentine	surface	4	6.24		1.27	
Iron Bridge, FL	65	parallel/series	surface	2			1.56	0.615
Iselin, PA	21	marsh/pond/meadow		5	0.0454	0.0257	1.83	1.47[b]
Marsh		parallel	subsurface[a]	2	0.0454	0.0257	4.68	5.28[b]
Meadow		parallel	subsurface[a]	2	0.0454	0.0257	9.37	10.7[b]

Project	Ref.	Configuration	Type	Number of Cells	Flow (1000 m³/day) Design	Flow (1000 m³/day) Actual	Hydraulic Loading Rate (cm/day) Design	Hydraulic Loading Rate (cm/day) Actual
Lake Buena Vista, FL	65		surface			14.0		3.99
Lakeland, FL	65	series	surface	7	53.0	29.4	0.936	0.520
Listowel, Canada	15	parallel/series	surface	5	4.54			
System 1		marsh/pond/meadow	surface	7				2.01
System 2		single	surface	1		0.040		1.89
System 3		series	surface	5		0.040		1.31
System 4		series	surface	5		0.030		1.31
System 5		single	surface	1		0.030		1.78
Mountain View Sanitary, CA	68	marsh/pond	surface	5	6.05	2.65	7.80	3.46
Neshaminy Falls, PA	20	marsh/pond/meadow			0.114	0.587	2.43	1.26
Marsh		parallel	subsurface[a]	4	0.114	0.587	4.63	2.39
Meadow		parallel	subsurface[a]	4	0.114	0.587	9.26	4.78
Orange County, FL	65		surface	3		75.7		1.53
Orlando, FL	69	series	surface	3	75.7	30.3	1.56	0.623
Paris Landing State Park, Paris, TN	22	series	subsurface	2	0.284		15.0	
Pembroke, KY	62	marsh/pond/meadow		6	0.341		2.25	
System A		series	surface	3	0.170		2.25	
System B		series	subsurface	3	0.170		2.25	
Phillips High School, Bear Creek, AL	22	single	subsurface	1	0.0757		3.74	
Santee, CA	18	parallel	subsurface	4	0.0122	0.0122	4.68	4.68
Silver Springs Shores, FL	65		surface			2.65		1.24
Vermontville, MI	26	parallel	surface	3		0.640	1.38	

[a]Plugging has resulted in substantial surface flow.
[b]Based on the use of only one of the two sets of marshes and meadows.

TVA Acid Drainage Wetlands Treatment Summary

		Influent Concentration			Effluent Concentration			Removal Efficacy (%)		Average Flow	Treatment Area m2/mg/min	
Wetlands System	Area m_2	pH	Fe (mg/L)	Mn (mg/L)	pH	Fe (mg/L)	Mn (mg/L)	Fe	Mn	L/min	Fe	Mn
King 006	9,300	6.0	170.0	4.9	2.9	17.0	3.8	96	22	1,374	0.05	1.4
WC 018	4,800	5.6	150.0	6.8	3.9	6.3	6.4	96	59	170	0.2	4.2
Imp 2	11,000	3.1	40.0	13.0	3.1	3.4	14.0	92	−8	646	0.4	1.3
Imp 4	2,000	4.9	135.0	24.0	5.0	2.3	3.5	90	85	42	0.4	2.0
Imp 3	1,200	7.1	28.7	8.8	6.8	0.7	1.3	98	85	83	0.5	1.6
RT—2	7,300	5.7	45.2	13.4	6.9	0.6	0.9	99	93	192	0.8	2.8
Imp 1	5,700	6.0	72.8	9.6	6.5	0.7	2.0	99	93	60	1.3	9.9
950 NE	2,500	6.0	11.0	9.0	7.2	0.5	0.4	96	96	75	3.0	3.7
950–1 & 2	3,400	5.7	12.0	8.0	6.5	1.1	1.6	91	80	83	3.4	5.1
Col 013	4,600	5.7	0.7	7.0	6.7	0.7	3.0	0	57	496	13.2	1.3

Performance of Constructed Wetland Systems for Acid Mine Drainage

	Mean	Range[a]
Size (m^2)	1,550	93–6,070
Number of basins (ponds)	3	1–7
Size of basin (m^2)	795	19–6,070
Flow rate, L/s	1.3	0.06–12.6
Water depth, m	0.3	0–2
Water chemistry ($n = 11$)		
Inflow:		
pH	4.9	3.1–6.3
Acidity, mg/L	170	ND–600
Fe, mg/L	33	0.4–220
Mn, mg/L	26	8.7–54
SO_4, mg/L	950	270–1,600
Outflow:		
pH	6.0	3.5–7.7
Acidity, mg/L	40	ND–140
Fe, mg/L	1.2	0.05–7.3
Mn, mg/L	15	0.3–52
SO_4, mg/L	740	160–1,500
Construction cost	$10,000	$1,500–$65,000

Note: ND = Below detection limits
[a]Twenty sites.