1.0 **ABSTRACT**

The aim of this project was to identify a suitable solution to the problem of landfill leachate at the North Katoomba landfill site. The landfill is situated within the environmentally sensitive area of the Blue Mountains, near the township of Katoomba, approximately 120 km west of Sydney. Options were affected by a range of constraints including economics, location and the intrusion of ground water into the landfill.

Blue Mountains City Council requested a method be devised to treat large volumes of landfill leachate, with the initial goal being to contain and treat the leachate on site and the eventual target to discharge into the nearest receiving waters. A constructed wetland option was devised and researched and Council gave the project approval. Research involved identifying the major pollutants contained within the leachate, developing a concept design and estimating the likely removal efficiencies expected from a constructed wetland.

Investigations identified the primary pollution parameters as microbial and nutrients. Metals were found to be low in concentration in the leachate although the wetland has the capacity to deal with these pollutants should they become part of the pollution plume.

A bench scale constructed wetland system was developed to give an indication of the removal efficiencies. Results were a 66% reduction in suspended solids, a 95% reduction in TKN, a 55% reduction in TP and a >99% reduction in faecal coliforms. The results suggest that a constructed wetland system would be appropriate for treating landfill leachate at the North Katoomba site. It is recommended that a constructed wetland be established in the field to determine the long term treatment prospects and the potential management problems in a practical application.
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## 1.0 ABSTRACT

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Jennifer E. Scott
University Western Sydney - Hawkesbury
1st November, 1994
"One of the penalties of an ecological education is that one lives alone in a world of words. Much of the damage inflicted on the land is quite invisible to laymen. An ecologist must either harden his shell and make believe that the consequences of science are none of his business, or he must be a doctor who sees the marks of death in a community that believes itself well and does not want to be told otherwise”

- Aldo Leopold, ecologist

from 'Wisdom of the Elders',

Peter Knudtson and David Suzuki
2.0 INTRODUCTION

2.1 Background and Significance of Research Project

The North Katoomba landfill facility receives general municipal wastes and serves a significant percentage of the residents of the Upper Blue Mountains region. The history of the site is sketchy and based mainly on anecdotal evidence of long term Council employees. This evidence suggests that the landfill has received waste material from such diverse sources as the local hospital, animal carcasses, and sewage sludge from floor scrapings at the local sewage treatment plant.

The site has been used as a landfill for in excess of 20 years, other past activities on the site have been identified only as the local dog pound. Currently the site is under the ownership and control of the local authority, the Blue Mountains City Council.

Historically, the site has been unsecured and monitoring of the type of wastes disposed by residents and commercial operations only intermittent. As a result, it is open to conjecture as to the nature of the waste components deposited at the site. Current practices now ensure the site is monitored during opening hours.

In 1986, plans were put forward to expand the useable landfill area to ensure its maximum potential. The plans detailed 9 future stages of work. Stages 1-6 are to occur over the following 20 years and the three final stages implemented when the need for further cells develops.
The progress of the construction was proposed to occur in the following sequence by the Department of Engineering and Parks, Blue Mountains City Council, in 1985.

**Stage 1:** (1986-1987)
- construction of a temporary leachate pond
- construction of leachate drains
- relocation of truck wash bay and water tanks
- seal off, fill and vegetate old fill areas

**Stage 2:** (1988)
- excavation of new fill areas
- plant trees and batters around old fill areas
- create tree disposal area
- maintenance of existing truck wash bay facility
- leachate drains and pond
- construction of an open leachate drain

**Stage 3:** (1989)
- construction of larger leachate dam
- stage 2 fill area sealed, topsoiled and revegetated
- leachate diverted into new dam
- stages 3 and 4 fill areas concurrently being excavated

**Stage 4:** (1990-1994)
- excavation of stage 4 for cover material
- construct leachate drain around stage 4
- seal off, topsoil and revegetate stage 3
- maintain leachate dam and pond
Stage 5: (1994-1999)
- seal off, topsoil and revegetate stage 4
- maintain leachate dam and drains
- stage 5 excavation for cover material

- maintenance of leachate dams and drains
- extend open leachate drain
- stage 4 filling
- alternative arrangements to be made for truck wash down facility
- relocation of tree disposal area
- excavation of stage 4 for cover material

Stage 7: (2003-2006)
- stage 7 filling
- stage 8 excavation for cover material
- move wash bay and tank to new location
- seal off, topsoil and revegetate stage 6
- maintain leachate dam and drains

Stage 8:
- excavate ahead for cover material
- construct new leachate and stormwater drains
- seal off, topsoil and revegetate stage 7
- construct new leachate sediment pond

Stage 9:
- possible end use, netball courts, rugby field, cricket pitch
- seal off final areas, topsoil and revegetate
- construct amenities block
- maintain leachate dam and drains

From the above plan it is clear that the local authority expects the landfill to continue producing leachate for many years to come. However, they have not identified in the plan an end point for the leachate other than collection in the dam.
Management practices at the site have developed rapidly since the commencement of this research project. A strategy was needed to solve the problem of disposing of large volumes of leachate forming from the infiltration of ground water into the waste material. The leachate migrates to the surface of the landfill and flows into a collection pond. In times of heavy rainfall the pond would easily be breached and flow into nearby Yosemite Creek.

A solution to the problem was needed and action precipitated when Council was prosecuted in the Land and Environment court by the Environmental Protection Authority for contaminating the creek in December, 1990. The site is licensed by the EPA and a condition of the licence is zero discharge.

Ground water infiltration of landfill sites is not an uncommon problem (Kean and Jern, 1985; Sanford et al., 1990) and several engineering strategies exist to prevent the problem occurring in future landfill sites. Many old landfills were constructed with little or no thought given to the environmental consequences and are today churning out highly toxic pollutants despite the fact they may have been closed for many years (Donnelly and Scarpino, 1984; Moshiri and Miller, 1991). This research project aimed at identifying and assessing the feasibility of a low cost, low maintenance method of dealing with landfill leachate that could have widespread application throughout Australia.

2.2 Historical Leachate Management Practices

In an effort to protect the surface waters surrounding landfill sites there is a need to collect and dispose of leachate. Surface et al. (1991) cite a traditional management practice as transportation off site and depositing at the local sewage treatment plant or expensive on site disposal methods using high level technology that often requires a large amount of capital to set-up and skilled specialist to maintain.
Many landfills are situated in isolated areas away from the population and present distinct management difficulties. Power can frequently only be applied to a system if a portable generator is maintained on site. Being isolated means the equipment is susceptible to theft and vandalism as security is inadequate.

Sanford et al. (1990) claim that leachate is the major reason for forced enclosure of landfills by government authorities. In the USA, Sanford et al. (1990) found that the most common leachate management practice is removal to a sewage treatment plant which is expensive, hazardous and disturbs the normal biological processes at the plant.

The preferred method at the North Katoomba landfill site in the past has been to collect the leachate and spray it back over the capped area of the landfill site to suppress dust and provide moisture for vegetation for use minimising surface erosion. Associated with this method are several potential problems.

i) apart from the loss to evapotranspiration the volume on site is ever increasing

ii) saturation of the landfill will eventually occur

iii) the potential exists for increasing the concentration of the pollutants each time the water passes through the waste

iv) the potential for ground and surface water contamination still exists

The immediate task at the North Katoomba landfill site is volume reduction and secondly a decrease in the bacterial contamination level of the leachate. The possibility of isolation of ground water from the refuse was not considered an option by the local authority.
2.3 Location, Description and Current Management Practices of the Study Area

Katoomba is located approximately 120 km west of the city of Sydney in an area of New South Wales known as the Blue Mountains. The township has a population of around 8,000 people although the Council services an area of 1405 square kilometres with a population of 64,000. This region is characterised by steeply sloping valleys and highly erodable Narrabeen sandstone consisting of sandy, yellow, leached gradational soils with ironstone gravel and grey brown and yellow brown uniform sands. (Native Plants of the Sydney District, Kangaroo Press, 1989, Atlas of Aust. Soils, Soil Conservations Service NSW 11390/SHEET 1).

The elevation of the site ranges from 940 m to 920 m above sea level. Rainfall occurs mainly during the warmer months of the year that is from November to April. Temperatures range from 14 °C to 35 °C in summer and 2 °C to 17 °C in winter (City of Blue Mountains publication no. ISSN 0725-2307).

(See figures 2.3A, 2.3B, 2.3C, 2.3D and 2.3E )

The landfill site covers an area of 7.2 hectares and is bounded by an access road to the south, National Park to the east and Yosemite Creek to the north and west. The creek has been impacted by siltation and severe weed infestation along its banks. Further downstream the creek flows through a public recreation area, over Minnie Ha Ha Falls, into the Blue Mountains National Park, finally flowing through the Grose Gorge to the Hawkesbury River. Adjacent to the site is bushland, with the nearest human activity being some 150 meters upslope.
WATER SUPPLY CATCHMENTS
SCHEDULE 1 AREAS

Showing AREAS PROHIBITED AND RESTRICTED TO THE PUBLIC

Boundary of Special Areas proclaimed under Sec. 21 Water Board Act 1987

Entry to the areas coloured pink is entirely prohibited except as indicated below.

Entry for bushwalking purposes is permitted within that part of Warragamba Catchment area more than 3 km from stored water, subject to strict observance of the Board's Regulations which relate to prevention of pollution of the catchment area.

A written permit need not be obtained.

Entry for bushwalking within 3 km of stored water is permitted only at the following locations:
- Siloan Pass
- Bonang - Goombung - Bellbird Pass

Vehicular access within 3 km is only permitted to McIvor's Lookout and Burswood Lookout.

For those parts of the area within the Blue Mountains and Kanangra-Boyd National Parks, the regulations of the NSW National Parks and Wildlife Act must also be observed.

Members of the public are requested to co-operate with the Rangers in the protection of the water supply.

For all catchment areas, the restrictions do not apply to persons lawfully on Private Property or Public Roads (incl Fire Roads).

National Parks shown above.

Shoalhaven Scheme shown below.

Compliance only is permitted at specified points for fishing, boating, non motorized rowing and other passive recreation purposes.

To assist in maintaining purity of the water supply, all members of the public are expected to co-operate fully in matters of bushfire prevention, sanitation and preservation of flora and fauna.

To protect wildlife, foxes must not be carried.

Fig 2,3A: Catchment Map, Blue Mountains City Council, (1990)
Fig. 2.3C Local Map, (indicating landfill site), Blue Mountains City Council (1990).
Fig. 2.3D Temperature and Rainfall Graph for Council Region, Blue Mountains City Council (1990).

**MAXIMUM TEMPERATURES 1991**

- **HIGHEST**
- **LOWEST**
- **AVERAGE**

**RAINFALL**

- **Millimetres**

**YEARLY RAINFALL**

- **Millimetres**

11
## STATISTICAL INFORMATION
### BLUE MOUNTAINS CITY AREA

**AREA - 1405 sq km**

<table>
<thead>
<tr>
<th>TOWN</th>
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<td>454</td>
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<td>BELL</td>
<td>1085m</td>
<td>29</td>
<td>32</td>
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Management of the site was formalised in 1986 by Blue Mountains City Council to reduce the pollution potential of the landfill. These regulations were instituted to achieve this aim.

Site Requirements, Blue Mountains City Council, 1986

- cover all wastes within 12 hours of dumping and all burning prohibited
- waste only to be deposited on the working face
- litter fences to be erected and regular collection of wind blown material
- daily cover to a minimum of 150 mm thickness, final cover 600 mm after compaction
- surfaces to be graded with a minimum slope of 1% to prevent ponding on surface
- maximum height of waste filling lifts to 2 m
- working face to have a minimum slope of 1 in 3 and face to be worked progressively along in 2 m lifts
- for maximum site useability compaction must be done by all available methods initial compaction should have a minimum bearing capacity of 60 kPa
- the length of the working face should be kept to a minimum
- traffic must be segregated, that is trucks from cars roads must be kept in good condition to prevent ponding and dust suppression methods are to be applied
- each completed area to be sealed with an impervious layer to prevent ingress of surface water
- revegetation to be carried out as soon as an area is complete
- recover soil/ash/humus from tree disposal for topsoil
- any material that could be used for topsoil should be stock piled on site
- leachate and sediment dams must be kept operational
- construct temporary drains to divert leachate into permanent drains
- monitor leachate quality at 3 monthly intervals
These management strategies appear to have had a limited effect on the volume leachate being produced from within the landfill. Hydrology advice to Council is that the formation of leachate is due to infiltration of ground water and little or none of the management strategy has been directed at preventing the infiltration of ground water. It would therefore be reasonable to predict that the ground water contamination of the site and subsequent production of leachate will continue unabated.

2.4 Formation and Characteristics of Leachate

Applications for the treatment of landfill leachate are limited by several factors including cost, land availability, and maintenance capability (Scott, 1990). Before any treatment system can be applied a knowledge of the characteristics of the waste water is essential, not only to assist in design considerations but also to measure performance.

Influences on the characteristics of leachate in a landfill site are numerous due to the highly varied and complex nature of the materials contributing to the leachate (Mullins and Sommers, 1983). The dilution factor that occurs through the variation in the volume of ground or surface water contacting the waste is an additional problem when trying to predict either concentrations or flow rates (Baumann, 1981).

Soil type and the amount of organic matter present within the landfill, the temperature both within the refuse and externally are further examples of influences upon the character of the leachate produced (Mullins and Sommers, 1983). Management strategies as described in section 2.3, above, are aimed at minimising the production of leachate.

When the leachate does enter the environment its potential impact is governed by many variables such as biological assimilation potential, bio or chemical
oxidation, adsorption and ion exchange mechanisms (Lakshman, 1981). The dry weight of refuse that is leachable was determined by Lakshman (1981), to be between 3 and 5% of the total mass. Characteristically this relatively small portion of the waste material contains high concentrations of organic and inorganic substances, dissolved chemicals and a range of microbial populations. Leachate production is influenced by the nature and amount of the parent material. Landfills which are not saturated by ground water may still produce considerable volumes of leachate. Table 2.4 details the composition of municipal refuse (similar to the North Katoomba landfill) on a percent wet basis. Organics account for more than 70% of the total mass deposited in landfills.

Table 2.4 Waste Composition - municipal refuse composition based on percentage wet weight basis. (Adapted from Emcon and Associates, 1980)

<table>
<thead>
<tr>
<th>Site no. =&gt;</th>
<th>7</th>
<th>16</th>
<th>17</th>
<th>39</th>
<th>40</th>
<th>43</th>
<th>47</th>
<th>47</th>
<th>47</th>
<th>47</th>
<th>48</th>
<th>49</th>
<th>49</th>
<th>54</th>
<th>mean</th>
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<td>cell no =&gt;</td>
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<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>refuse↓</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Food</td>
<td>10.1</td>
<td>12</td>
<td>15.3</td>
<td>14.5</td>
<td>16</td>
<td>22.5</td>
<td>12</td>
<td>48</td>
<td></td>
<td>10.7</td>
<td>12</td>
<td>.5</td>
<td>15.7</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Garden</td>
<td>5.2</td>
<td>9</td>
<td>13.8</td>
<td>12.5</td>
<td>9</td>
<td>-</td>
<td>12</td>
<td>9.6</td>
<td>70</td>
<td>69</td>
<td>10.4</td>
<td>36</td>
<td>64</td>
<td>16</td>
<td>25.9</td>
<td></td>
</tr>
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<td>42.5</td>
<td>48</td>
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<td>40.6</td>
<td>30</td>
<td>30</td>
<td>40</td>
<td>43.9</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Plastic/</td>
<td>2</td>
<td>3</td>
<td>1.8</td>
<td>4</td>
<td>2</td>
<td>2.5</td>
<td>2.4</td>
<td>-</td>
<td>2.2</td>
<td>.2</td>
<td>4.6</td>
<td>-</td>
<td>1.3</td>
<td>2.6</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rubber</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Textiles</td>
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<td>1.6</td>
<td>2</td>
<td>1</td>
<td>4</td>
<td>.6</td>
<td>-</td>
<td>1</td>
<td>1.5</td>
<td>1.7</td>
<td>1</td>
<td>1</td>
<td>1.5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wood</td>
<td>1.4</td>
<td>2</td>
<td>1.2</td>
<td>2.5</td>
<td>2</td>
<td>3</td>
<td>2.5</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1</td>
<td>2</td>
<td>1.9</td>
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<td>6.7</td>
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<td>8</td>
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<td>14</td>
<td>8.7</td>
<td>10</td>
<td>9</td>
<td>6</td>
<td>-</td>
<td>9.8</td>
<td>8.7</td>
<td></td>
</tr>
<tr>
<td>Glass/</td>
<td>5.1</td>
<td>7</td>
<td>10.1</td>
<td>10</td>
<td>6</td>
<td>6</td>
<td>6</td>
<td>6.1</td>
<td>11</td>
<td>11</td>
<td>10.9</td>
<td>4</td>
<td>-</td>
<td>10</td>
<td>8.6</td>
<td></td>
</tr>
<tr>
<td>ceramic</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ash/dirt/</td>
<td>6.6</td>
<td>7</td>
<td>7.2</td>
<td>-</td>
<td>8</td>
<td>7.5</td>
<td>11</td>
<td>8.5</td>
<td>-</td>
<td>-</td>
<td>2.8</td>
<td>-</td>
<td>1</td>
<td>1.7</td>
<td>6.1</td>
<td></td>
</tr>
<tr>
<td>rock</td>
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<td></td>
<td></td>
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</tr>
<tr>
<td>Fines</td>
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<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>3</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>8.3</td>
<td>7</td>
<td>-</td>
<td>-</td>
<td>6.1</td>
</tr>
<tr>
<td>Miscellaneous</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>.5</td>
<td>-</td>
<td>7.1</td>
<td>7.4</td>
<td>-</td>
<td>2</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>4</td>
<td></td>
</tr>
</tbody>
</table>

a. The mean composition for all sites and cells
Leachate may vary considerably but they tend to have some consistent characteristics such as they are usually anoxic, have high BOD values, and have high concentrations of organic carbon, nitrogen, chloride, iron, manganese and phenols (Sanford et al., 1991).

A more detailed description of the chemical, physical and biological characteristics of leachate can be found in the Section 3, the literature review.

2.5 Application and performance of potential constructed wetlands

The most common application of constructed wetlands has been in the treatment of municipal waste waters. Several hundred of these units exist in Europe alone (Brix, 1992). The majority are designed for sub surface horizontal flow and have returned good results for suspended solids and BOD reduction but relatively poor for nutrients.

However the performance of constructed wetlands in regard to nutrients appears to be highly variable (dependant upon loading and effluent type). For example Jukwarkar et al. (1992) found their constructed wetland treating municipal wastewater reduced nitrogen from 30.8 mg l$^{-1}$ to 9.5 mg l$^{-1}$ and phosphorus from 14.9 mg l$^{-1}$ to 9.6 mg l$^{-1}$.

Brix (1992) reports reductions of 40% for nitrogen and 30% for phosphorus in municipal wastewater, Tanner (1992) reports reductions of 40-90% for total nitrogen and 30-80% for total phosphorus in dairy waste and Perfler and Haberl (1992) found long term removal rates in constructed wetlands to be in the vicinity of 45% for total nitrogen and 54% for total phosphorus.
Commonly, enteric pathogenic populations in waste water are substantially and consistently reduced by constructed wetland systems to approximately 90-99% of the original value. Jukwarkar et al. (1992) report reductions of 93-98% in the European systems. Brix (1992) states an average reduction of 99.9% in vertical flow systems. Venus and Oldcorn (1992) reporting on a constructed wetland in New Zealand found a faecal coliform reduction of 99.35%. Roser et al. (1987), in a treatment system for sewage, found a reduction in the order of 99% consistently in a gravel based constructed wetland.

With regard to the performances of wetlands where the application is more specific to this project, several investigations have been conducted into the suitability of using constructed wetlands to treat landfill leachate. New landfill sites are now subject to far more stringent regulations and conditions than in years past both in Australia and overseas.

To reduce the pollution potential of these areas, measures such as lining the site with an impermeable membrane and employing treatment systems such as constructed wetlands to treat leachate, are being implemented. Moshiri and Miller (1991) designed a landfill site utilising recycling, volume reduction, and wetland leachate treatment in Florida, USA.

Robinson et al. (1991) review the implementation of an aerobic treatment pond combined with a constructed wetland to treat high strength landfill leachate in the UK. Another pilot scheme to test the success of treating landfill leachate with constructed wetlands is being conducted in Slovenia where many small landfill sites are causing serious environmental problems both to surface and ground waters (Urbanc-Bercic, 1992).
Perhaps amongst the most advanced research in the use of constructed wetlands for leachate treatment has been by Surface et al. (1990) and Sanford et al. (1990) where experimental rock-reed filters have been used to treat leachate from a municipal solid waste facility. Table 2.5A is a summation of the results to date of the experiments performed in this specific field.

Data is expressed in percentage reduction of pollutants. It is interesting to note that it was difficult to locate exact figures for influent and effluent and that most results for the treatment of landfill leachate were expressed as percentages. The type of effluent treated by the systems described in Table 2.5A is leachate produced from municipal refuse.

**Table 2.5 A % Reduction of Pollutants by Constructed Wetland Performance Treating Landfill Leachate Overseas**

<table>
<thead>
<tr>
<th>Reference</th>
<th>Country</th>
<th>SS</th>
<th>BOD</th>
<th>TN</th>
<th>TP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urbanc - Bercic (1992)</td>
<td>Slovenia</td>
<td>73%</td>
<td>32%</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
<td>Surface et al. (1990)</td>
<td>USA</td>
<td>70%</td>
<td>--</td>
<td>--</td>
<td>high</td>
</tr>
<tr>
<td>Sanford et al. (1990)</td>
<td>USA</td>
<td>75%</td>
<td>90%</td>
<td>99%</td>
<td>52-88%</td>
</tr>
</tbody>
</table>

Surface et al. (1990) noted that volume reduction of up to 40% in summer and 20% in winter occurred during the time the leachate was detained within the wetland.

**2.6 Legislative water quality standards**

Any treatment system for waste water will be required to meet a certain standard under law before it can be discharged into the environment. Although the regulations are currently under review by the EPA NSW, the Clean Waters Regulations (1972) remain in force as a guide to discharge requirements. Variations occur depending upon the nature of the water body receiving the
effluent. The classification of the aquatic ecosystem in NSW is a two tier system with waterbodies being classified as either Level 1 or Level 2.

Level 1 is the maximum level of protection afforded to an aquatic ecosystem and as such should not receive any waste discharges. The only ecosystems eligible to receive such classifications are a small percentage of waters in NSW that exist in undeveloped areas and are free from current human impact. Schedule 15 of the Clean Waters Regulations (1972) lists the relevant criteria for such ecosystems.

Level 2 is described as a practical level of protection that recognises that a certain level of impact is inherent in developed areas. These ecosystems "have been degraded and modified such that their natural state is of limited relevance" (Clean Waters Regulations, 1972). These impacts are generally considered irreversible and management objectives must be determined according to the ecosystem that is desired rather than restoration to a pristine, natural state.

The NSW EPA acknowledge that classification of waterways will often be very difficult due to the lack of data regarding the current condition of the water and extent of the bio diversity existing within that waterway.

Schedule 15 of the Clean Waters Regulations (1972) list water quality criteria for aquatic ecosystems classified as Level 1.

Table 2.6 A  NSW EPA Water Quality Criteria for Level 1 Waters

<table>
<thead>
<tr>
<th>Indicator</th>
<th>Marine</th>
<th>Freshwater</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dissolved Oxygen</td>
<td>6 mg l⁻¹</td>
<td>6 mg l⁻¹</td>
</tr>
<tr>
<td>pH</td>
<td>6.5 - 8.5</td>
<td>6.5 - 8.5</td>
</tr>
<tr>
<td>Suspended solids</td>
<td>5 mg l⁻¹</td>
<td>25 mg l⁻¹</td>
</tr>
<tr>
<td>Total nitrogen</td>
<td>0.002 mg l⁻¹</td>
<td>0.200 mg l⁻¹</td>
</tr>
<tr>
<td>Total phosphorus</td>
<td>0.20 mg l⁻¹</td>
<td>0.02 mg l⁻¹</td>
</tr>
</tbody>
</table>
Schedule 16 of the Clean Waters Regulations (1972) list water quality criteria for aquatic ecosystems classified as Level 2.

Table 2.6 B  NSW EPA Water Quality for Criteria for Level 2 Waters

<table>
<thead>
<tr>
<th>Indicator</th>
<th>Marine</th>
<th>Freshwater</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dissolved Oxygen</td>
<td>4 mg l⁻¹</td>
<td>4 mg l⁻¹</td>
</tr>
<tr>
<td>pH</td>
<td>6.0 - 9.0</td>
<td>6.5 - 8.5</td>
</tr>
<tr>
<td>Suspended Solids</td>
<td>20 mg l⁻¹</td>
<td>400 mg l⁻¹</td>
</tr>
<tr>
<td>Total Nitrogen</td>
<td>0.5 mg l⁻¹</td>
<td>0.05 mg l⁻¹</td>
</tr>
<tr>
<td>Total Phosphorus</td>
<td>0.05 mg l⁻¹</td>
<td>0.05 mg l⁻¹</td>
</tr>
</tbody>
</table>

USEPA standards are applied for other pollutants including metals, pesticides, organic chemicals and miscellaneous substances such as Cyanide and Ammonia-N. Schedule 17 and 18 list these substances and their acceptable level in both freshwater and marine ecosystems.

The leachate at the North Katoomba landfill will need to meet the discharge requirements as set down by the EPA of NSW. The waterbody that would receive the treated effluent is Yosemite Creek which flows around the base of the landfill site. This creek is not likely to be classified Level 1 as it flows through a residential area and light industrial complex before reaching the landfill site.

To accurately assess the requirement for discharge, the current water quality of Yosemite Creek should be determined, and if a pristine creek exists in the region then background levels will give an indication to what extent Yosemite Creek has been impacted. If water quality is monitored above and below the landfill site an indication will be given as to the extent the landfill contributes to pollution in the creek (Simmons, personal communication, 1991) (see 4.2.1).

The EPA NSW expects to release new guidelines for receiving waterbodies in September, 1994.
3.0 LITERATURE REVIEW

Leachate percolating from landfill sites may have a significant effect upon not only the surrounding environment but also many of the natural systems in areas remote to the landfill site (Kean and Jern, 1985). Modern society has developed a disposable mentality which has amplified through the progression of the 20th century. This disposable material discarded over many decades has been subject to traditional disposal methods, that is placement in the ground or incineration.

The generation of refuse varies from country to country, but in modern consumer oriented cities such as Singapore the generation of waste per capita in 1985 had reached 1.14 kg d\(^{-1}\) compared with 1974 when the figure ranged around 0.58 kg d\(^{-1}\). The doubling of waste generation in a 10 year period is expected to continue at least into the 21st century and beyond (Kean and Jern, 1985). Consumer demand, packaging material and methods are also a process of and subject to, economic influences. As third world nations move from being predominantly producers to consumers so the global waste generation grows.

As a result of landfills being the most economic method of waste disposal, landfills have appeared at hundreds of thousands of locations around the world. The leachate produced at these sites has polluted subterranean aquifers, freshwater and marine bodies (Scott, 1990).
3.1.1 Leachate Characterisation

As water percolates through the refuse within a landfill it comes into contact with many substances. During infiltration the influent acquires or deposits components depending upon the physio chemical and biological environment (Mullins and Sommers, 1983).

One of the major difficulties in dealing with landfill leachate is the variability of the components and their concentrations (Williams, 1982). Table 3.1A details typical variations found in leachate samples.

Table 3.1 A Range of Values Found in Municipal Landfill Leachate
(from Williams, 1982)

<table>
<thead>
<tr>
<th>Substance</th>
<th>Range</th>
<th>Substance</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>6.6 - 7.6</td>
<td>Calcium Ca</td>
<td>165 - 1,150 mg l⁻¹</td>
</tr>
<tr>
<td>Chemical Oxygen Demand COD</td>
<td>66 - 11,600 mg l⁻¹</td>
<td>Magnesium Mg</td>
<td>13.3 - 480 mg l⁻¹</td>
</tr>
<tr>
<td>Biological Oxygen Demand BOD</td>
<td>2 - 8,000 mg l⁻¹</td>
<td>Iron Fe</td>
<td>1 - 370 mg l⁻¹</td>
</tr>
<tr>
<td>Total Organic Carbon TOC</td>
<td>28 - 44.4 mg l⁻¹</td>
<td>Manganese Mn</td>
<td>0.2 - 26.5 mg l⁻¹</td>
</tr>
<tr>
<td>Ammonia nitrogen NH³</td>
<td>0.9 - 780 mg l⁻¹</td>
<td>Chlorine Cl</td>
<td>70 - 2,777 mg l⁻¹</td>
</tr>
<tr>
<td>Potassium K</td>
<td>20 - 600 mg l⁻¹</td>
<td>Sulfates SO₄</td>
<td>55 - 456 mg l⁻¹</td>
</tr>
</tbody>
</table>

3.1.2 Leachate Formation

The evolution of leachate is a complex interaction of biological, chemical and physical components. Figure 3.1.2A, adapted from Rees (1982), describes the evolution of leachate within a municipal landfill.
The volume of leachate and the subsequent concentration of the components is determined by several inter-relating factors.

- hydro geology of the site involving soil type, degree of compaction
- operational and management techniques such as daily capping and sorting of wastes
- topography of the site
- rainfall and or presence of ground water
- seasonal variations due to temperature and evaporation rates
- age of the landfill site and extent of decomposition of the waste
- the type and volume of the waste

Minimising the production of the leachate within a landfill is linked to the construction and operational methods used. In a survey conducted by Baumann (1975) of putrescible landfill sites in the Sydney region it was concluded that landfills constructed in flat areas were less hazardous than those constructed in valleys.
Flat sites produce the small quantities of leachate whereas sites such as brick pits or abandoned quarries yielded the greatest volume of pollutants. Baumann (1975) assessed the pollution potential for these sites by comparing the Biochemical Oxygen Demand for flat, valley and quarry locations. Table 3.1B gives the results.

Table 3.1B  Comparison of Topographical Difference and Leachate Biochemical Oxygen Demand.  (from Baumann, 1981)

<table>
<thead>
<tr>
<th>Location</th>
<th>BOD (av. mg l⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flat (av. BOD)</td>
<td>0.39</td>
</tr>
<tr>
<td>Valley (av. BOD)</td>
<td>5.6</td>
</tr>
<tr>
<td>Quarry (av. BOD)</td>
<td>36.4</td>
</tr>
</tbody>
</table>

3.1.3  The Effects of Leachate on Natural Systems

In response to the need for environmental protection strategies by identification of point source and non-point source pollution and preventing degradation of the natural biome, landfill sites and the leachate they produce are causing increasing concern (Surface et al., 1991; Robinson et al., 1991). Leachate presents a threat to a natural aquatic system in several ways.

1)  Biochemical Oxygen Demand  -  BOD is a pollutant measure which reflects the concentration of available biological oxidisable organic matter entering a waterway. The organic matter is degraded by micro-organisms which utilise the dissolved oxygen in the water depleting it until the water is no longer able to sustain aerobic organisms.

The biochemical oxygen demand is a measure of the amount of oxygen required to completely oxidise the organic substances in the water (Kimball, 1983). The higher the BOD value the less oxygen present for use by aquatic organisms. It is common for landfill leachate to give a BOD value of between 2-8,000 mg l⁻¹. A reduction in the BOD is essential if the leachate is to come into contact with a natural aquatic system (Williams, 1982).
2) **Nutrients** - The principle nutrients for plant growth are nitrogen and phosphorus, and as such may be referred to as growth limiting factors. These plant nutrients only become pollutants when they exceed background (natural) concentration levels, or if greater than would normally occur in a waterway (Simmons, personal communication, 1991). In elevated concentrations they enhance plant growth and consequently the plant biomass, usually kept in check by natural constraints, chokes the waterway. As a result of the natural growth and decay processes oxygen depletion occurs as the amount of decaying organic matter increases. Organisms responsible for the decay processes rapidly consume the available oxygen. Eventually eutrophication occurs and aerobic aquatic life ceases to exist (Kimball, 1983).

3) **The Nitrogen Cycle** - Most organisms require nitrogen for growth in one of its many forms, either as a gas or as a compound. Plants must 'fix' the nitrogen or incorporate it into a compound before the plant is able to use this element for the purposes of manufacturing protein. Nitrate ions in the form of $\text{NO}_3^-$ and ammonium ($\text{NH}_4^+$) are absorbed through the rhizomes of the plant (Kimball, 1983) and utilised for plant growth and repair.

As much as one third of all nitrogen fixation occurs as a result of artificial or man-made processes such as in the production of chemical fertilisers. Certain micro-organisms are capable of fixing atmospheric nitrogen (Kimball, 1983), for example Clostridium and Azotobacter. The first stable product for this process is ammonia which is quickly absorbed into the plant material and also into the microbial population for protein production.

As the organic matter decay process continues, ammonification occurs where nitrogen loaded molecules are broken down into ammonium to benefit the micro-organisms involved in the decay process. Most ammonium produced during the
decay process is converted into nitrates. This occurs through two distinct phases. Firstly bacteria such as Nitrosomonas oxidise \( \text{NH}_4^+ \) to \( \text{NO}_2^- \) that then oxidise to nitrates (\( \text{NO}_3^- \)) by a different bacterial group, Nitrobacter. Such bacteria are characterised as chemoautotrophs and it is through their action that nitrogen is capable of being absorbed into the roots of the plants.

1. During the denitrification process (where nitrogen is released into the atmosphere in its gaseous form), bacterial organisms that are naturally found in aquatic sediments may use the nitrates in the water as an oxygen source. In doing so they reduce the \( \text{NO}_3^- \) to its original state, a gas (\( \text{N}_2 \)) and this is released back into the atmosphere (Kimball, 1983).

Water pollution caused by nitrogen is a result of several processes

- hydrodynamics
- physio chemical processes
- biological transformations

(Chetboun and Bachmat, 1981)

These processes involve hydrodynamic mass transport (i.e. movement of the nutrient in the waterway), chemical and biological mass transport (where the nutrient is translocated by formation of compounds and incorporated into the biota) and physical mass transport at interface surfaces (i.e. absorbed onto particles or into sediments) (Chetboun and Bachmat, 1981).

The second growth limiting nutrient is phosphorus. In the phosphorus cycle this element is made available to living organisms for use in the formation of nucleic acids that store and translate the genetic code. Phosphorus plays an important role in the formation of ATP which forms the basis of high energy bonds and
creates the mechanisms for energy production and cellular respiration during the photosynthesis process in plants (Kimball, 1983).

Limiting the availability of one or the other of these nutrients may prevent blooms of aquatic weeds occurring. 'Algal blooms' may occur in nutrient rich waterways and are often the precursor to the destruction of aerobic aquatic life. The figures contained in Table 3.1A, 'Leachate Characteristics' note the levels of nutrients and pollutants that are present and should be reduced before the leachate contacts a natural water body to reduce impact (Williams, 1982; Kimball, 1983).

4) **Trace Metals** - Metals contamination of landfill leachate is generally fairly low in concentration from municipal landfill sites (Scott, 1990) with iron often being the metal in greatest concentration (Sanford *et al.*, 1990). However, concentrations in leachate can vary as reported by Williams (1982).

Concentrations may vary both over time and between locations on the same site. Loch *et al.* (1981) established a link between the mobility of metals in leachate and the presence of fatty acids. Consequently metals may either remain in landfill because of chemical reaction with other agents and precipitate out of the leachate or remain as ions which can undergo a complexation process to produce ligands that are soluble and are transported out in the leachate (Mullins and Sommers, 1983).
Once in the environment, metals such as Hg$^{2+}$ or Pb$^{2+}$ are absorbed into cells where they may act as 'irreversible inhibitors' (Bohinski, 1987). An irreversible inhibitor is a substance that bonds with the active site on an enzyme and prevents a reaction with the enzymes' natural substrate, inhibiting the production of the cells' normal metabolic process (Bohinski, 1987). The inhibition of this metabolic activity can lead to the rapid destruction of the organism, as the metabolic pathway may involve oxygenation of cells or perhaps the translocation of nutrients. The reduction of metals in leachate will be a function that any leachate treatment unit must be able to perform.

5) Pathogenic Micro-organisms - Pathogenic micro-organisms are often a component of landfill leachate (Scott, 1990) and treatment of leachate must have the capacity to reduce microbial numbers. It is of vital concern for public health that the microbial load be reduced prior to leachate migrating into the surrounding environment. Generally waters that are nutrient rich will contain elevated numbers of microbes in suspension (Tortora et al., 1986).

Faecal contamination of landfill leachate is not uncommon (Scott, 1990) and the landfill leachate that is the subject of this research recorded very high levels of faecal coliform contamination ($10^{5}$cfu/100 ml) similar to levels present in raw sewage. Faecal coliforms indicate the possible presence of pathogenic micro-organisms (Tortora et al., 1986), the greater the concentration the more likely the presence of pathogens. Diseases such as cholera and typhoid may be transmitted through contact with water contaminated by faecal matter. For drinking water the only acceptable safe level of contamination is zero faecal coliforms.

The survival time of bacteria and viruses in water varies according to the species, temperature, chemical environment and presence of autothonic
populations of micro-organisms (Mathess and Pedkeger, 1981). Important pathogenic bacteria capable of causing disease in humans via contact with contaminated water are *Salmonella* sp, *Shigella* sp, *Vibrio cholera*, *Yersinia entercolitica*, *Y.pseudo tuberculosis* and *Leptospira* sp to name but a few.

It is well established that bacterial and viral pathogens are responsible for serious human diseases and their presence gives rise to a significant threat to the health of the community. It is important that contamination by effluent containing these pathogens be prevented. Entry of landfill leachate containing high faecal coliform numbers into natural waterways may represent a serious community health problem.

Pathogen elimination or die off rate is dependent upon physical, chemical and biological parameters. Published values for reduction are highly variable, for example a 99% reduction of *Salmonella typhi* has been reported to occur between 2 and 107 days (Mathess and Pekdeger, 1981).

Physical, chemical and biological effects have been found to have a combined interaction with each other and are often complementary. Physical effects are those relating to temperature, biological effect is the presence or absence of autochthonous bacteria and bacteriophage. Chemical effects relate to the pH of the water and organic content. Bacteria and viruses survive best at a neutral pH, high organic compound content and low oxygen concentrations (Mathess and Pekdger, 1981).

Donnelly and Scarpino (1984) investigated landfill micro-organisms to determine the presence of human pathogens and their survival in leachate. It was found that faecal coliform numbers decreased rapidly over time and distance, but faecal streptococci decreased at a much slower rate. After one year, indicator
organisms were below the level of detectability, but some viable pathogens were still identifiable after that period of time. After two years, several pathogenic bacteria and fungi could still be found in the leachate samples.

The researchers concluded that even when only low faecal coliform numbers are present, it could not be assumed that bacteria were not remaining active within the landfill site. Moreover, it was confirmed that bacteria were capable of surviving in the long term within the landfill site. Although higher numbers were detected in active landfills, viable microbial populations were still present years after the closure of the landfill.

3.2 Wetland Treatment Systems

3.2.1 Potential for Waste Water Treatment by Natural Systems
Wetland treatment systems mimic the processes of decay and utilise the natural removal mechanisms found to naturally occur in wetlands (Robinson et al., 1991; Wood, 1990). The potential for waste water treatment involves utilising natural responses in a managed system. Such responses include gravity, sedimentation and actions of biological organisms (Reed et al., 1988). A feature of these systems is that no external energy source is required and so are economic, robust, reliable and easy to maintain.

Wetlands may be defined as ‘land in which the water table is at or above the ground surface long enough each year to maintain saturated soil conditions and growth of related vegetation’ (Reed et al., 1988). Constructed wetland treatment units are usually gravel and/or sand filled trenches planted with a suitable species of vegetation. Within the wetland, mechanisms exist that promote the removal of pollutants (Roser et al., 1987; Wood, 1990).
Aquatic plant systems may be categorised under three basic classifications:

1) **Emergent plants** - these plants are rooted in a substrate such as gravel and emerge from the substrate or grow on the surface. Emergent plants may emerge through water but they remain rooted in the substrate.

2) **Floating plants** - as the name indicates these plants float on the surface of the water. This basis for categorising aquatic macrophytes is determined their growth characteristics and location of their photosynthetic parts.

3) **Submerged plants** - survive and grow beneath the surface, this term is usually used to describe those plants suspended in the water column. These plants are not attached to a substrate and the roots float below the surface of the water as do the other plant parts.


Natural wetlands may contain all three types of macrophytes. In a natural system distribution of these plants depends upon factors such as the amount of light able to penetrate the water, salinity and the depth of the water body. Generally in constructed wetlands only one or two species is used.

Pollutant removal mechanisms are the features that promote much interest in constructed wetlands. These mechanisms may involve a combination of factors including physical, chemical and biological interactions and associations (Roser *et al.*, 1987). Constructed wetland designs generally involve dispersion of effluent through a series of treatment beds.
Treatment units are categorised by Bavor et al. (1987) in the following mani

a) Emergent macrophytes in lagoon systems
b) 'Root-zone' treatment systems
c) Floating plant systems
d) Emergent plants in gravel trenches.

The system proposed for this research project involves emergent macrophytes in gravel substrate. Aquatic, emergent macrophytes are capable of oxygenating the sediment surrounding them, producing an oxygenated zone in the usually anoxic substratum (Roser et al., 1987; Wood, 1990; Robinson et al., 1991). It is thought that this feature may enhance the microbial mechanisms of pollutant removal or immobilisation within the sediment. Aerobic and anaerobic zones are established within the substrata allowing a broad microbial population to establish, capable of breaking down organic matter (Davies, 1988; Robinson et al., 1991).

3.2.2 Organic Load and BOD
Organic matter in leachate occurs as a result of the decomposition of matter such as vegetation, food wastes, paper and the like deposited in the landfill. Particles are leached out as water percolates through the waste matter. These particles remain in suspension and progressively settle out of the effluent according to the particle size. The amount of material in suspension will contribute to the total levels of carbon and nitrogen in the water.

Soluble and insoluble organic carbon may be measured as the Biochemical Oxygen Demand for the wastewater (Bavor et al., 1988). Filtering out and biological oxidation of organic matter in the wetland will assist in the reduction of BOD levels.
Suspended solids are filtered out of the leachate through entrapment by the plant biomass and the porous substratum, removing suspended particles that may then be subject to degradation via the actions of the microbial population. Micro-organisms attach to plant roots or onto the surface of the substrate particles creating a large surface area and correspondingly potentially a large microbial population.

In the oxygenated root area, micro-organisms are able to oxidise and breakdown organic matter (Roser et al., 1987; Wood, 1990). Living matter is generally comprised of approximately 18% carbon (Kimball, 1983). Carbon atoms have the potential to bond to each other and create organic material combined with other elements. Carbon is cyclic, meaning it is capable of being degraded and reformed (Kimball, 1983). In the natural world carbon, containing material is subject to the decay processes facilitated by decay micro-organisms such as bacteria and fungi.

In aquatic systems, removal of settleable solids will reduce the BOD. The rate of biodegradation is dependent upon the type of material and its bonds, temperature, oxygen concentration, nutrient availability, pH, substrate concentration and the presence of toxins (Wood, 1990).

Solids are removed through physical filtration and then degraded in the substrata by the micro flora attached to the sediments and roots of the macrophytes. Fisher (1985) states that removal of colloids and soluble BOD in a constructed wetland is primarily based upon the microbial population and their metabolic activity.

In aquatic plants soluble organic components can be absorbed into plant tissue, translocation within the plant is via biological pathways that lead to the
production of new plant tissue (Fisher, 1985). It is a combination of removal mechanisms that generates a significant reduction BOD and organic content. The task of the designer to allow for the creation and interaction of each of these mechanisms.

### 3.2.3 Nitrogen Removal Mechanisms

Nitrogen has the ability to alter character because of the variety of oxidation states it can adopt. These alterations in oxidation result in a complex chemistry that is a result of the utilisation of nitrogen by living organisms (Fisher, 1985). All living organisms require nitrogen, but nitrogen in its free form (N₂) is an inert substance requiring substantial amounts of energy to break its bonds and allow it to react with other elements.

Naturally occurring nitrification processes include -

1) **Lightening** - may combine nitrogen with O₂ molecules in the atmosphere, this forms nitrates which dissolve in rain and washes into the soil for plant utilisation

2) **Legumes** - are plants that are able to 'fix' nitrogen directly from the atmosphere through the actions of micro flora associated with the root systems of these plants. The ability to fix nitrogen extends to other prokaryotic bacteria such as Clostridium and Azotobacter that are found in the soil.

In modern society industrial processes are responsible for much of the nitrogen entering the soil, for example manufactured agricultural fertiliser compounds (Kimball, 1983). Nitrogen is an essential growth nutrient as it has an integral role in the synthesis of proteins. There are three principle mechanisms that enable aquatic plants to remove nitrogen from waste water.
a) **Hydrophyte absorption** - where aquatic plants absorb fixed forms of nitrogen into the biomass for protein synthesis

b) **Bacterial mechanisms** - these allow nitrification ($\text{N}_2$ from atmosphere to plant), denitrification (decay process from plant to atmosphere), and volatilisation of ammonia. (Ammonia is a term used to cover all nitrogen existing as free ammonia ($\text{NH}_3$) and the ammonium ion ($\text{NH}_4^+$) (Fisher, 1985).

Brix (1991) states that the primary source of organic nitrogen for aquatic macrophytes is in the form of ammonium ions facilitating rapid incorporation into the plant. Also ammonium is seen as the primary source of nitrogen for plants in water logged soils. Fisher (1985) notes the rate at which ammonium ions are created in the water surrounding the plant is dependant upon the rate at which the plant assimilates the nitrogen into the biomass. However, Wood (1990) claims that ammonia volatilisation is not likely to be a significant pathway for removal under normal circumstances as a neutral pH is usually maintained in a wetland. Actively growing plants producing biomass will remove nitrogen at the maximum rate.

Removal rate is affected by the availability of oxygen in the wetland as biological nitrification and denitrification are key processes for long term removal in a wetland (Wood, 1990). The stage of growth of the plant, senescence, harvesting, plant species and temperature will all affect removal rates (Wood, 1990; Fisher, 1985).

The presence of specific nitrogen transforming bacteria in the wetland will also influence removal performance. Bacterial action by micro-organisms such as *Nitrosomonas* sp oxidises ammonium into a nitrite.

$$2\text{NH}_4^+ + 3\text{O}_2 \rightarrow 2\text{NO}_2^- + 4\text{H}^+ + 2\text{H}_2\text{O} + \text{generation of new cells.}$$
A subsequent bacterial species, *Nitrobacter sp* may then oxidise the nitrite to form the compound nitrate.

\[ 2 \text{NO}_2^- + O_2 > 2 \text{NO}_3^- + \text{generation of new cells}. \]

The oxygen demand created by these reactions is equivalent to a carbonaceous oxygen demand. The O\(_2\) remaining after the nitrifying process will be consumed perhaps partially or wholly by organic BOD reactions (Fisher, 1985; Wood, 1990).

The final mechanism for removal of nitrogen is that of denitrification (Kimball, 1983). Heterotrophs living under anoxic conditions are capable of using the oxygen molecule in the nitrate as a terminal electron acceptor if the surrounding environment is deficient in O\(_2\). This denitrification process results in the formation of N\(_2\).

The following equation describes the process (Fisher, 1985).

\[ \text{NO}_3^- + \text{(organic carbon)} > \text{N}_2 + \text{CO}_2 + \text{H}_2\text{O} + \text{OH}^- + \text{new cells}. \]

Micro-organisms such as *Pseudomonas*, *Achromobacter*, *Bacillus* and *Micrococcus* are examples of denitrifying bacteria that require organic carbon to allow them to carry out their processes. It is worth noting that Fisher (1985) states the denitrifying process occurs much more slowly in acidic conditions than alkaline.

### 3.2.4 Phosphorus Removal Mechanism

The most readily available form of phosphorus for biological metabolism is the orthophosphate species. Some phosphorus will be organically bound in certain industrial waste waters but the most common form of removal has traditionally been through the addition of chemicals that form precipitates (Cooksey and Cheng, 1989). The rate of removal has been found to vary considerably
according to geographic location and the subsequent diurnal and seasonal changes within the wetland (Raper, 1988; Fisher, 1985).

Occasionally soils are dosed with phosphorus absorbing compounds such as iron, calcium or aluminium salts to enhance the removal capabilities of the substrate (Davies, 1988). Under natural conditions a wetland has a finite capacity for phosphorus utilisation and exports will occur when supply exceeds demand/deposition rate (Fisher, 1985).

Davies (1988) states that mechanisms for the uptake of phosphorus are less numerous than for nitrogen and that variability in uptake can be reduced by the choice of macrophyte species and operating techniques. The reduction in the quantity of phosphorus absorbed by plants as compared to nitrogen can be expressed as a ratio of P:N. Ratios of 1:5 - 1:10 have been determined in aquatic macrophyte species (Fisher, 1985). This may account for the relative amounts of nutrients absorbed by plants. It is suggested by Fisher (1985) that the uptake of phosphorus may be dependant upon the concentration of nitrogen in the waste water with the phosphorus, and that the nitrogen concentration may be a limiting factor.

Aquatic macrophytes assist in removing phosphorus from the sediment but once the sediment is saturated plant uptake may not be sufficient to release an adequate number of reactive sites.

Chemical precipitation is another phosphorus removal mechanism existing within natural systems (Cooksey and Cheng, 1989). In wetlands it has been determined that phosphorus may be removed from the influent through chemical precipitation. This may be attributed to the formation of iron phosphate resulting from the soluble phosphorus contacting iron oxides where iron is present within
the wetland. Hard water will precipitate phosphorus out of solution. Calcium carbonates react with phosphorus forming a precipitate.

Physical absorption into the sediment may also occur, however the phosphorus can also be quickly re-released from the sediment because of the relatively weak bonds (Fisher, 1985). Phosphorus uptake may be rapid initially but due to the finite absorptive capacity of the substrate the predominant mechanism will eventually be precipitation once the reactive sites in the substrate are exhausted (Wood, 1990).

The redox potential of the soil combined with the effects of the pH will determine the substrates ability to absorb phosphorus. Wood (1990) reports the mechanism identified in the absorption into the substrate involves the exchange of a phosphate ion with an OH⁻ of a metal hydroxide, MOH = Fe or Al and is considered to be usually irreversible at iso-pH, constant ionic strength, and in an environment where there are no other competing ions.

Depending upon the concentration of phosphorus in the waste water, the sediment may be dredged and replaced perhaps in two or three years or ten years. In a constructed wetland consisting of sequential units this may be done in phases so as not to interfere with or disrupt the system.

Micro-organisms living in the root zone will also utilise small amounts of phosphorus (Fisher, 1985). Phosphorus is essential in the synthesis of proteins and it was noted by Bavor et al. (1987) that high levels of phosphorus retention in a system may correlate to high rates of metabolic activity. Incorporation into the biomass is a significant removal mechanism and according to Bavor et al. (1987) the filtering, adsorption and sedimentation of particulate matter are other
significant removal mechanisms that may be associated with type and character of the substrate material used.

3.2.5 Pathogen Removal Mechanisms

In constructed wetlands faecal coliform reduction has been very successful (Roser et al., 1987; Bavor et al., 1987). A consistent removal rate of up to 99% has been achieved in trials where a six day retention period of sewage effluent in a constructed wetland was maintained. Faecal coliform die-off increases as the effluent moves through the treatment beds. This may due to several mechanisms combined with other factors that are not necessarily a direct role of the wetland e.g. temperature (Bavor et al., 1987).

Removal of pathogenic micro-organisms is essential due to the potential effects these organisms may have on the health of the community (Fisher, 1985). The length of time bacteria and viruses may survive in effluent is variable and subject many environmental conditions. In a U.S. E.P.A. study, Donnelly and Scarpino (1984) found bacteria and fungi to have survived for over two years within a landfill site.

Mechanisms involved for the removal of pathogens are a combination of natural die-off over time and distance, sedimentation, and action of the microbial population in the rhizosphere (Fisher, 1985) including the presence of bacteriophage and parasitic bacteria (Mathess and Pekdger, 1981).

Natural die-off occurs because the pathogens are being exposed to physical, chemical and biological changes that will cause their destruction. Bacteria are stressed and die when exposed to elements such as ultra-violet radiation, changes in temperature to non-optimal and predation by protozoa (Fisher,
These components form the natural mechanisms that combine together to produce efficient microbial reduction.

Fisher (1985) suggests other mechanisms may also assist in the reduction of microbial numbers for example evidence suggests that hydrophyte plants excrete an antibacterial agent from the roots. *Schoenoplectus, Juncus* and *Phragmites* have been suggested to exhibit this ability.

The nature of the bacteria or virus also determines its ability to survive. Some species survive changes in environmental conditions by producing endospores to encase and protect themselves while other species may develop another resistant character such as being able to survive in a range of pH conditions.

Substrate conditions will contribute to microbial removal. Adsorption onto the substrate surface plays a significant role in the removal of viruses. The efficiency of this mechanism is subject to the texture of the sediment, presence of organic substances, humic acids, cations, pH ionic strength and permeability (Gerba and Keswick, 1981).

Adsorption of viruses onto the surface of the substrate is governed by Van der Waals forces where an electrostatic double layer interaction occurs. The surface charge on the virus and the surface charge on the substrate particle and surrounding environmental factors will determine the net charge and the ability for the virus to be absorbed onto the particle (Gerba and Keswick, 1981). In general sandy, organic soils are poor absorbers as opposed to clay which has substantial absorptive capacity.

Filtration and interception are other mechanisms that assist in reducing microbial numbers. Micro-organisms may be entrapped by the pore cavity in the
substrate or again by electrostatic or Van der Waals forces. Van der Waals forces are comparatively weak and only effective over a short distance. In an aquifer, the solid particles are usually negatively charged, the same as bacteria and viruses. Negatively charged particles remain in suspension in sand filters, as the repulsive electrostatic forces are stronger than the bonding Van der Waals forces (Gerba and Keswick, 1981).

Dissolved cations in the effluent decrease the repulsive forces on sediment particles as the monovalent cations are bonded with the substrate solids and so diminish their charge deficiency allowing the electrostatic forces to bond. To artificially increase bacteria and virus adsorption the addition of a substance such as calcium will enhance this mechanism.

In slow sand filters, a biologically active layer may develop at the sand/water interface. Microbial reduction can occur here through the mechanisms of adsorption due to the small particle size of the substrate, an increased surface area is made available.

Microbial slimes and bacteriophage (a virus that attacks bacteria and destroys them) are also promoted at the interface although it requires a certain length of time to allow this biologically active layer to establish (Mathess and Pekdger, 1981). Bacterial removal may be through a combination of adsorption and to a lesser extent filtration, but virus removal occurs only through adsorption (Gerba and Keswick, 1981).
3.2.6 Trace Metal Removal

Heavy metals and other inorganic substances may leach out of the landfill in solution and exhibit a highly varied character and concentration. Many trace elements are essential to plant growth particularly in the translocation of nutrients, for example iron is a minor nutrient element. Plants may flourish where an excess of an element exists but fail to survive a deficiency (Brook, 1971).

Accumulation and adsorption of ions in plants enter through root hair cells. The differential permeability of the membrane will impose a significant degree of selectivity in which ions will be incorporated into the plant tissue. The particle size of the molecule involved is generally the determining factor. Plants have the ability to accumulate certain ions in their cells, which only occurs when plants undergo vigorous respiration as substantial amounts of energy are required for this function. A respiratory inhibitor such as Cyanide will limit O₂ supply and therefore prevent the uptake of metals ions (Brook, 1971).

Monovalent anions and cations are absorbed more easily and efficiently than divalent species. Antagonism in a plant cell structure may account for the character of the uptake of trace metals. A solution containing only one metal salt will plasmolyse a plant, however a solution containing two metals salts will not. Single metal salts adversely affect the plant protein of the cell membranes altering their permeability, but two salts maintain permeability at a balanced level (Brook, 1971).

Under certain conditions the following elements have been found to have a stimulating effect on plants - aluminium, cadmium, chromium, cobalt, codeine, lead, selenium, and silicon (Curtis and Clark, 1950). Plants absorb particular metal ions to enhance their metabolism, for example aluminium acts as a
catalyst. Metal uptake in plants is related to metal activity rather than the concentration of the metal in the waste water (Mullins and Sommers, 1983).

In a study of the chemical speciation of landfill leachate by Mullins and Sommers (1983), it was determined that if only a few metal ions were present in the waste water is was generally due to metal ions within the landfill reacting with organic compounds to complexes. Ions that were commonly found in the effluent were those that could exist in the free ion form such as K, Ca, Mg, and Na.

Mullins and Sommers (1983) state that the more soluble metals were found in the effluent whereas the less soluble (eg PbCO₃) remained within the landfill as a precipitate. Most heavy metals are only slightly soluble and so tend to remain within the waste material and do not figure significantly as a precipitate. The behaviour of metals can be predicted from their position on the periodic table.

- **Alkaline Metals** - occupy the first column on the periodic table. These metals have a tendency to remain in solution as ions. Sedimentation through chemical removal is not possible as they do not form precipitates, however biologically mediated removal may still occur.

- **Alkaline Earth Metals** - occupy the second column on the periodic table. These metals react with inorganic compounds such as CO₃, they do not form ligands but usually form phosphates and carbonates.

- **Transition Metals** - these include heavy metals and will form co-ordination compounds or complexes (ligand and metal). A complex is a co-ordination compound with a charge.
Mullins and Sommers (1983) found that over time the free ion form of metals in
the leachate increased as the chemical reactions in the landfill were gradually
exhausted. Heavy metals may be soluble when bonded to fatty acids which may
permit the metals to be transferred from the landfill in the leachate (Loch et al.,

Metals may become entrapped into the sediment of the treatment beds by
adsorption onto colloidal clay particles and organic matter. There is a low
reversibility for this reaction because of the preferential nature of the chemical
bond. Precipitation with phosphates, carbonates, and sulphates are a useful
mechanism to entrap heavy metals in a wetland treatment unit (Loch et al.,
1981). Plant uptake and microbial utilisation will be finite given the minor
concentrations plants and microbes require in these metals. If saturation does
occur the same technique that is recommended for phosphorus may be effective
ie remove the substance from the treatment beds and replant. Done in
succession and over a period of time it is an effective long term management
strategy.
3.3 Designing an Constructed Wetland

3.3.1 Overview

Bavor et al. (1987) cite the following considerations when designing an constructed wetland.

a) Construction requirement including the layout of the system, substratum to be incorporated, the loading/hydraulics, and bed liner selection

b) Plant species, their selection, criteria for growth and establishment period plus the relevant chemical, physical and biological features of the system.

c) Potential operational difficulties and management guidelines

d) Limitations to the design

When establishing the optimum design the project should be subject to continual review. The specific requirements of the system are determined by the effluent characteristics and concentration. Brix (1992) reviews several design options such as overland flow systems, sub-surface horizontal flow systems and vertical flow and combined systems. Hammer (1992) lists surface and sub-surface flow systems as the primary design options. Selection depends upon the effluent and the level of pollutant removal necessary.

Selection of macrophytes and identification of long term management strategies are needed before commencement (Fisher, 1985). Integral to the design is the standards to be meet on discharge. The economics of the project, climatic conditions and topography, hydrology and use of the surrounding area all play a determining role (Bavor et al., 1987). Pilot scale studies are useful and demonstrate potential efficiency (Fisher, 1985).
3.3.2 Construction Requirements

An overall concept is formulated depicting the general form and location of the unit. Bavor et al. (1988) state that it is important to be aware that many of the design features are evolutionary and flexible and should be able to be easily adaptable over time. Critical information in planning the wetland are the expected loading on the system and optimising the potential for good hydraulic conductivity (Mackney, 1990).

The size of the system is governed by the area available and the anticipated volume to be treated. Bavor et al. (1988) determined that for 10 mg l$^{-1}$ (BOD) loss from sewage effluent a loading of 1 Ml ha$^{-1}$ d$^{-1}$ was necessary to give consistent results. For adequate nitrogen and coliform reduction a loading in the vicinity of 0.4 Ml h$^{-1}$ d$^{-1}$ is optimal.

Maintenance of the hydraulic gradient is vital to the performance of the system (Hammer, 1992). The flow can not short circuit parts of the treatment system by developing preferential flow paths. These problems can be overcome by manipulation of the length to width ratio, modification of the design, use of baffles and ensuring linear flow is compatible to the substrate permeability coefficient (Bavor et al., 1988). If management strategies such as harvesting of macrophytes is considered necessary the design must allow for access to these areas with sealed pathways to prevent further particulate matter from entering the system (Bavor et al., 1988).

Solids loading and hydraulic conductivity have a direct relationship (Hammer, 1992; Bavor et al., 1988). Hammer (1992) considers that sub surface flow systems should receive only tertiary treated effluent for this reason and if suspended solids are likely to be significant then a surface flow system should be used.
Solids accumulating at the inlet or the water/gravel interface reduce the hydraulic conductivity and may lead to short circuiting and reduced retention period (Brix, 1992). Sufficient gradient must exist to prevent hydraulic head loss that may reduce the amount of effluent in contact with rhizosphere and so impair macrophyte growth in sub surface flow systems. Placing different grades (diameter size of particles) of substrate at the inlet, centre and outlet zones of the beds can maintain conductivity and help prevent clogging (Bavor et al., 1988).

A substratum with large macropores (void volume between the solid particles) will increase the permeability of the system. Gravel, as opposed to soil or sand, with a 5-10 mm diameter has been reported to give a flow rate of 0.3 ms⁻¹ after 2-3 years of operation with a gradient of < 5% (Bavor et al., 1988).

The adsorptive capacity of gravel is less than that of soil (Brix, 1992), however the addition of Al or Fe may enhance the uptake of phosphorus (Bavor et al., 1988; Brix, 1992). Hammer (1992) comments the primary role for the substrate is to provide physical support for the plants, create a surface area for complexing ions, anions and some compounds and for the attachment of microbial populations.

Protection of the surrounding environment from contamination of the effluent requires the treatment beds be lined with an impermeable membrane barrier. Liners may be made from clay material or a synthetic substance such as PVC or rubber which has been UV stabilised.

Designs generally source the energy to move the effluent through system by utilising the influences of gravity. In systems with multiple units interconnecting pipes or channels move effluent from one unit to the next. Overflow weirs, riser
pipes and sumps can regulate the flow and distribution of water into and between units.

3.3.3 Macrophyte Selection

It is a systematic process to determine which genus of macrophyte is best suited to the system. Fisher (1985) lists several criteria plants must meet to qualify.

a) Growth Rate - fast growth means optimum capacity will be reached more quickly, planting should be planned for spring for maximum growth in the warmer months.

b) Pollution Absorptive Capacity - some species have proved more effective in pollutant reduction particularly for nutrients. Trials where a comparison is drawn between species have demonstrated that one may return consistently better results. For example Bavor et al. (1987) found Typha superior to Myriophyllum.

c) Bacterial Biomass Enhancement - mechanism for microbial action are integral to pollutant removal (Hammer, 1992; Fisher, 1985). Plant species must encourage bacterial growth. Plants with extensive rhizosphere, oxygenation capabilities and resistant to a wide range in effluent strength are best.

d) Endemic and non-noxious species - native species adapted to the natural climatic conditions of the region are unlikely to cause down stream disturbance, particularly if the species exists in the area naturally (Fisher, 1985).

e) Adaption to Variations - all plant species have tolerance limits to a range of chemical, physical and biological conditions. These include concentrations of oxygen, nutrients, organic matter, turbidity and inorganic salts (Fisher, 1985). Climate range is also an influential factor.
on species selection. A rapid growth phase and resilience to environmental conditions give plants the adaptability to survive and flourish. *Typha* has been found a suitable genera for waste water applications (Bavor *et al.*, 1987).

### 3.4 Potential Operational and Management Problems
The efficiency of constructed reed beds is dependent upon creating a healthy and mature crop of macrophytes with a deep root system, this may possibly require 3-5 years to establish (Brix, 1992). Bavor *et al.* (1988) comment that a shallow bed (0.2-0.3 m) will ensure root contact with the effluent.

Problems associated with bed design are oxygen diffusion, vertical mixing of effluent, use of gravels with a reasonable cation exchange potential, variations in volume of effluent and possible harvesting of macrophytes or removal and replacement of substrate.

Harvesting on an annual basis in Autumn has proved beneficial, although Hammer (1992) comments that plants take only very small quantities of pollutants, for example <5% of nutrients. Operating expense involved in harvesting the plants often outweighed the benefit and resulted in only slight increases in pollutant removal.

Weed control may be necessary from time to time depending upon the weed infestation in the surrounding district. Application of a herbicide or physical removal of weeds twice per year has been found adequate. Herbicides such as 'Roundup' (Monsanto Inc,) when applied to the weeds directly will not damage the surrounding macrophytes (provided there is no direct contact).
Options that may exist for the harvested plant material would be dependent upon plant tissue analysis. Options such as landfill, silage, burning, composting or reincorporating into the system through breakdown on the surface of the treatment beds (Bavor et al., 1988) are all possible.

Bavor et al. (1988) cite maintenance of optimal hydraulics as the primary management aim, as it is vital to the performance of the unit. To achieve a satisfactory hydraulic regime the following criteria may be applied.

- reduce the hydraulic gradient
- solids loading must be regulated
- 'pseudo-plug flow' maintained
- continue to maximise void volume

Management strategies such as maintaining even inlet dispersion, prevention of solids accumulation, effluent contact with rhizosphere and monitoring macrophytes for evidence of stress will assist in preventing these potential problem areas from developing (Bavor et al., 1988; Brix, 1992).

Flow checks over short and long distances and periodic monitoring of flow rates enhance operations. Tracer dyes may be used for these tests. Shorter retention periods may indicate zones within the system are being bypassed.

The working life of constructed wetlands will vary and an approximate figure has yet to be determined (Bavor et al., 1988). Some European systems have been in use for many decades treating municipal waste water (Brix, 1992). If in the construction phase of the project sediment from disturbed bare earth surfaces is a difficulty, the initial stage of design can incorporate sumps or drainage reticulation systems (Bavor et al., 1988).
3.5 Performance of Existing Constructed Wetland Units

Constructed wetlands have predominantly been used for treating municipal waste water and urban stormwater pollution (Bavor et al., 1988; Fisher, 1985). Other more varied applications for point source and non-point source pollution are being examined. Landfill leachate has been subject to considerable attention in many countries particularly where ground water contamination presents a health threat (Robinson et al., 1991; Sanford et al., 1990; Moshiri and Miller, 1991).

Research by Moshiri and Miller (1991) lists the objectives of the development landfill regulations. Firstly to construct and operate lined landfills with a leachate collection system that will prevent ground and surface water contamination. Secondly, to reduce volumes of waste being transported to landfill, thirdly to increase significantly the suitable life of landfills through volume reduction and engineering design and fourthly to keep the cost of solid waste disposal as low as possible.

Constructed wetlands have been trialed for their suitability to treat landfill leachate in several countries including the USA and UK (Surface et al., 1991; Robinson et al., 1991; Sanford et al., 1990).

- **United Kingdom** - Robinson et al. (1991) report the progress of a 5 year study into the treatment of landfill leachate to surface discharge water standards. The constructed wetland was the second treatment phase, the first being an aerobic lagoon plant. The objectives of this research program were to design and construct a reed bed within a secure site and to further investigate the effluent from the aerobic lagoon, to seal the base of beds with an appropriate liner, prepare substrate and plant macrophytes, and to design, construct and...
install a holding tank for the leachate, put in a pump for priming the tank from the main treatment plant, install an irrigation system within the beds and sample points for flow measurement and control, rainfall monitoring and final effluent disposal (initially to sewer).

**Compton Bassett Landfill** - a leachate treatment plant design consisted of a 1300 m³ aeration lagoon which removed solids and degraded organics through microbial action. The plant typically treats 30 to 50 m³ of leachate per day. In the aeration unit reduction of BOD, COD and ammonia was excellent. The reed bed could not accommodate the entire leachate volume and was control dosed with the effluent. It was calculated that the bed could be dosed with around 6.5 m³ per day. The substrate used was shingle with a 42% porosity, giving a hydraulic capacity of 34 m³ and 80 m³ volume occupied by the substrate. The macrophyte selected was *Phragmites australis*. Whole plants were planted the day after extraction by hand from a sandy lagoon. A total of 452 plants were put in and within 5 weeks (in summer) active growth was occurring in over 80% of the plants. At the time of publication the system was ready to receive leachate, however the authors anticipate that as the high ammonia levels decrease as the landfill ages, eventually the aerobic lagoon will not be necessary.

The USA has been investigating the use of constructed wetlands for landfill leachate treatment for several years. Surface *et al.* (1991) and Sanford *et al.* (1990) discuss the research in Tompkins County in New York State.

**United States** - objectives were to examine the efficiency of landfill leachate treatment as a function of substrate material, plant growth and
seasonal changes, examine the effects of leachate on the hydraulic characteristics of the bed and examine the chemical, biological and physical processes by which nutrients, metals and organic compounds are removed from leachate flowing through a porous media.

Each bed measured 3 m wide, 30 m long, and 0.6 m in depth, with a slope of 5% or less. Beds had different grades of gravel within the substrate and two of the four beds were planted with *Phragmites australis* and the other two left unplanted.

Leachate application began in July, 1990 at a rate of 2,000 litres per day per plot with a 15 day retention period. Removals for BOD and $\text{NH}_4^+$ ranged from 45-65% and 55-70%, respectively. Sand/gravel substrate demonstrated the best removal capability and planted to unplanted bed comparison showed little difference.

Removal of Fe and Mn was around 70-90% and results showed the uptake of metals and P was not seasonally affected. Metals were noted to accumulate in or on the root surface but were not translocated to other parts of the plant tissue.

Plant biomass was found to positively affect evapotranspiration rates, with the coarse gravel beds exhibiting the highest rate continually. Bed conductivity was related to the particle and pore size.

The physical/chemical process dominated the removal of P and metals from the leachate. The presence of macrophytes may increase volume reduction but did not modify hydraulic conductivity or increase root zone aeration.

In an earlier study in New York, USA Trautmann *et al.* (1989) studied the use of constructed wetlands to treat landfill leachate as a cost effective alternative to road transport to a sewage treatment plant. The waste
disposal site covered 3.6 hectares and had been in operation for 28 years, accepting only domestic waste. Leachate is collected in 5 ponds, at this point the effluent exceeded discharge limits for ammonia, iron, manganese, nitrates and benzene.

The treatment unit consisted four beds, 3.7 m x 30.5 m with a 60 ml geomembrane liner on a 61 cm clay base. The first two beds were designed as a pre-treatment phase followed by an overland flow to a coarse gravel bed planted with cattails and the final bed contained no vegetation and was used as a control. The pre-treatment beds were designed to precipitate the iron and manganese out of the solution to prevent clogging the wetland units and in addition to volatilise ammonia to nitrates and to volatilise benzene.

Laboratory experiments were conducted to examine and quantify the ability of the wetland system to oxidise the root zone, measure iron oxide formation and relate to $O_2$ transport in the roots and establish if various rhizomes enhance hydraulic activity. Results are yet to be published but Trautmann et al., (1989) commented that this research was a result of legal and economic pressure.

**Sweden** - although this study is not looking at landfill leachate disposal the aim of the research by Lowengren et al. (1989) was to discharge wastes into a natural system without causing a high increase in algal growth and other aquatic organisms. In this wetland system pre-sedimented waste water was applied by surface irrigation to reed sweet grass growing in loamy sand and harvested for nutrient recovery. Reductions in BOD to tertiary treatment levels were achieved using this system with a similar result being experienced for phosphorus. Nitrogen
was removed more efficiently by this treatment application than by conventional waste water treatment methods.

Investment and operational/maintenance costs for wetland filters were considered to be substantially lower than existing technology. Potential revenues from the utilisation of the biomass created and energy savings by integrating the system into existing farm infrastructures and requirements also exists.

3.6 Landfill Site Design and Operation

Ideally minimising the volume of leachate produced from the landfill site is the optimum management strategy. Loch et al. (1981) consider it unlikely that the production of leachate can be prevented altogether, but comments that by careful management of the waste material in the landfill the toxicity of the leachate may be reduced. Loch et al. (1981) and Mullins and Sommers (1983) discuss the chemical reactions that trap toxins in the site or conversely dissolve them into solution for migration. Precipitation and absorption of metals occurs in presence of fatty acids, if landfill cells contain wastes of known components the leachate resulting from controlled interactions should have a predictable toxicity.

Several guidelines have been created to prevent landfill leachate generation including impermeable barriers, cell discretion and security. Control of surface flow from storm water through diversion channels is a significant leachate reduction factor. Sealing refuse with a water proof cap of clay on a daily basis is essential to prevent rainwater from percolating through the refuse. The capped area can quickly be revegetated to prevent wind and water erosivity.
Recirculating leachate back through the waste to increase the metabolic processes of the microflora within the landfill is a current management method. While this method may reduce the concentration of the leachate over time by creating an anaerobic filter, the many variations in conditions within the landfill makes it difficult to predict or establish values for leachate with this method (Palmer, 1982).

A second leachate management strategy involves the spraying of leachate over soil supporting vegetation adjacent to the site. Volume reduction generally occurs through transpiration and evaporation. If a sufficiently large area of land is available, spraying may eliminate the waste volume completely (Palmer, 1982). If this method is to be used the environmental impact must be carefully investigated.

Other disposal options such as transport to a sewage treatment plant, incineration or high tech filtration are expensive both in capital and maintenance costs. Further research in to the mechanism of constructed wetland technology is essential if these system are to achieve optimum, long term service. Current research indicates performance is variable often influenced by geographic location, climate and the types of waste involved (Davies, 1988).

Bavor et al., (1988) and Davies (1988) comment that these systems have proved beneficial in the reduction of nutrient and microbial pollutants. Landfill leachate application is a new area that is giving promising results (Surface et al., 1990; Sanford et al., 1991; Robinson et al., 1991). With the ever increasing number of instances of pollution of aquifers from landfill leachate (Loch et al., 1981; Wegman et al., 1981) leachate can no longer be considered a problem of little consequence.
Constructed wetland treatment units should be further investigated for potential when applied to this problem both as a preventative measure and to ameliorate existing pollution plumes from current and closed landfill sites.

Table 4.0 in 'materials and methods' summarises the method used in this project to analyse and design the constructed wetland.
4.0 MATERIALS AND METHODS

In determining the appropriate materials and methods for the project an holistic approach incorporating a 'cradle to the grave' strategy was formulated. This strategy evolved as a flow chart detailing the perceived progression of one phase to the next to achieve the goal of a functioning constructed wetland.

Table 4.0

CONSTRUCTED WETLAND RESEARCH DESIGN METHODOLOGY

<table>
<thead>
<tr>
<th>Establish performance standards and discharge requirements</th>
</tr>
</thead>
<tbody>
<tr>
<td>Commence monitoring of leachate</td>
</tr>
<tr>
<td>Determine pre-treatment requirements</td>
</tr>
<tr>
<td>Assess physical constraints of the site</td>
</tr>
<tr>
<td>Select suitable macrophyte species and substrate material</td>
</tr>
<tr>
<td>Determine preliminary design and performance estimations</td>
</tr>
<tr>
<td>Commence pilot scale study</td>
</tr>
<tr>
<td>Modify design according to pilot study data</td>
</tr>
<tr>
<td>Build closed* loop constructed wetland</td>
</tr>
<tr>
<td>Establish monitoring regime to determine system efficiency</td>
</tr>
</tbody>
</table>

Allow system to open* - dependant upon performance and surrounding ecology

* open and closed refers to the effluent from at the end point of the wetland either being re-circulated into the leachate pond (closed) or allowed to drain into the surrounding environment (open).

Preliminary testing indicated that a primary pollutant was the bacterial component of the leachate and the bacterial concentration varied seasonally. The sampling conducted during this period indicated that the future sampling regime for this project should be designed to include the influence of seasonal temperature variations on coliform numbers. Preliminary sampling also suggested that nitrogenous compounds were a significant pollutant and their reduction should be a principle target of the constructed wetland.
4.1 Discharge requirements

Currently the EPA of NSW has placed a zero discharge condition on the license held by the local authority for the site. However if any treatment process of the leachate proves consistent and adequate then this condition may be reviewed.

To evaluate appropriate discharge guidelines required for landfill leachate two sources were consulted. Firstly the Environmental Protection Authority of NSW published an Environmental Protection Policy (State Pollution Control Commission, SPCC) 1989) to govern discharges to natural wetlands. This paper discusses the protection of wetlands designated to receive a variety of effluent.

As a possible scenario for the final discharge of the treated leachate at the North Katoomba site it could be dispersed to an adjacent hanging swamp. This may constitute an ideal receiving body for further polishing of the effluent provided it maintained an adequate treatment standard. The EPA policy (SPCC, 1989) disallows discharges into natural wetlands to "contain high loads of nutrients or suspended material" and recognises that certain wetlands may benefit from the reception of treated effluent. A natural wetland is described in this document as

"An area where the water table is at or above the land surface for long enough each year to support the growth of aquatic vegetation, much of which is emergent". (SPCC, 1989)

Although a hanging swamp is the result of a perched water table and does not necessarily contain aquatic vegetation the plants growing in the area are usually those associated with saturated soils.

Before effluent can be discharged into a natural wetland the EPA requires sufficient evidence to demonstrate that the extent of the probable ecological
changes will be such that the existing wetland characteristics and values are not
degraded. In addition to this study the formulation of a management/monitoring
plan for the wetland is essential. Several other government instrumentalities
have policies regarding wetlands including the Water Resources Commission,
NSW, and the Water Board, Sydney. These documents should also be reviewed
to ensure compliance.

The second source consulted as to discharge requirements was the authority
responsible for the landfill site, that is Blue Mountains City Council. The landfill
site is currently subject to licensing restrictions which includes zero emissions to
the surrounding environment. However, once the constructed wetland has
developed sufficiently to exhibit a consistent effluent standard and a regular
monitoring program commenced the local authority may wish to make
application to have this restriction varied.

Schierup et al. (1990) note the Danish official regulations for discharge from
constructed wetlands are BOD 20 mg l\(^{-1}\) and Suspended Solids 20 mg l\(^{-1}\).
Although this regulation pertains to sewage effluent it supplies a parameter for
waste water discharge into the environment.

4.2 Data collection and sampling methods
Preliminary sampling results combined with the historical site sampling by the
Council again suggested a significant pollutant was the bacterial component
(Scott, 1990; Blue Mountains City Council, 1990). Sampling regimes were
conducted on a periodic basis rather than the storm event scenario. Automatic
sampling devices were not considered appropriate for the site due to the
vandalism experienced in the area and the history of destruction to equipment
left unattended at the landfill.
Data gathered by Blue Mountains City Council prior to the commencement of this research indicated that analysis should include nutrients. Sanford et al. (1990) consider metals, nutrients, BOD, and COD the appropriate indicators for landfill leachate evaluation, however Surface et al. (1991) include iron and manganese due to their high concentration in their project leachate from a New York landfill currently being researched.

The preliminary data from the Blue Mountains site also suggested a significant dilution of the leachate was occurring, due possibly to the infiltration of ground water, the surface area of the leachate collection pond, and surface storm flow. This observation cannot be confirmed by records as staff at the site have not recorded or maintained any data in regard to when the collection pond was pumped out or the volume removed. Research is currently underway to determine the source of the ground water and the extent of the infiltration into the landfill.

Sampling to determine the nature and concentration of pollutants was performed twice per month over a period of one year. During periods of heavy rain or substantial storm events in the region leachate would flow directly from the working face of the refuse.

Sites for collection of leachate flowing from the refuse changed according to accessibility and the flow regime both of which altered frequently as the management of site progressed. For several months of the sampling period only the leachate collection pond provided sufficient volumes for analysis. The collection pond remained consistently around maximum capacity and frequently required pumping out to evaporation ponds to prevent breaching.
Table 4.2 Sample analysis methods and source for the biological, physical and chemical parameters for landfill leachate.

Table 4.2 A Analytical Methods

<table>
<thead>
<tr>
<th>Substance</th>
<th>Method</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Faecal coliforms</td>
<td>Membrane filtration</td>
<td>*Standard methods 1992</td>
</tr>
<tr>
<td>Total phosphorus</td>
<td>Digestion/Spectrophotometer</td>
<td>*Standard methods 1992</td>
</tr>
<tr>
<td>Total Kjedahl Nitrogen</td>
<td>Digestion/Spectrophotometer</td>
<td>*Standard methods 1992</td>
</tr>
<tr>
<td>Biochemical Oxygen Demand</td>
<td>Winkler method</td>
<td>*Standard methods 1992</td>
</tr>
<tr>
<td>Metals-Pb,Cu and Zn</td>
<td>Stripping voltammetry</td>
<td>PDV 2000 Chemtronics</td>
</tr>
<tr>
<td>Suspended solids</td>
<td>Filtration and dry mass assay</td>
<td>*Standard methods 1992</td>
</tr>
<tr>
<td>pH</td>
<td>portable pH meter</td>
<td>*River study manual</td>
</tr>
<tr>
<td>Conductivity</td>
<td>portable conductivity meter</td>
<td>*River study manual</td>
</tr>
<tr>
<td>Temperature</td>
<td>portable temperature meter</td>
<td>*River study manual</td>
</tr>
</tbody>
</table>

* Full citation is given in references

4.2.1 Background Pollutants

Background pollutant levels were measured to assess the extent to which a natural stream flowing at the base of the landfill had been impacted. A stream in the vicinity was tested for conductivity levels at a point prior to flowing into any area where it would be subject to stormwater or other pollutant infiltration to provide a background pollutant reference. As the region is predominantly sandstone, soils would yield few dissolved ions (Simmons, personal communication, 1992).

Any significant ion concentration should occur from pollutants rather than the surrounding natural environment. These figures provided a comparison for samples taken from upstream and downstream of the landfill and indicated whether the leachate was affecting the water quality of surrounding surface water bodies.
4.2.2 Soils

Characterisation of the soils present at the site of the future constructed wetland was integral to the inclusion of certain design parameters such as the need for an impervious liner.

Soils were analysed for particle density and bulk density. Sampling sites were selected immediately beneath the treatment beds, 10 m down slope and finally immediately below the existing leachate pond which generally retained the effluent satisfactorily. The information from this analysis would provide information on the capability of the soil to absorb treated effluent should that become a long term management option. Brouwer and Bugeja (1985) recommend investigation into the absorptive capacity of soils to predict infiltration rates and the potential for pollution plumes to migrate.

4.3 Determining the optimum design:

Constructed wetlands are designed to be conducive to the biochemical mechanisms that exist in the natural biome. Translocation of macronutrients, precipitation of metals, oxygenation of the waste water, and an enhanced microbial population assist in the pollutant removal potential of the wetland. Volume reduction is another significant role wetlands are capable of performing, all of which is achieved by recreating and enhancing natural physical and biochemical mechanisms (Hammer, 1992; Schierup et al., 1990; Wood, 1990).

Landfill leachate is a heterogenous combination of organic, inorganic, and sometimes toxic substances (Sanford et al., 1991). Composition and concentration vary according to a range of conditions and are difficult to predict with any accuracy. As the North Katoomba landfill accepts only municipal waste the range in concentration may be reduced as compared to leachate produced in a hazardous waste site. Characteristically leachate exhibits a high BOD
value, is usually anoxic, and high in nitrogen, iron, manganese, chloride, and phenols (Kean and Jern, 1985; Sanford et al., 1991).

4.3.1 Factors Influencing the Design

Baumann (1981) suggests the following factors affect the composition of leachate, design and management of the site, hydrogeology of the region, topography, climate, seasonal variations in temperature and rainfall, age of the landfill and depth and composition of the waste.

The treatment system for the leachate must accommodate these variations in composition in addition to significant changes in flow rates (Sanford et al., 1991). Constructed wetlands may incorporate various design modes to accommodate the particular features found in the waste water and often consist of a series of treatment beds or trenches.

The configuration of the system will be determined by pre treatment options and the relative performance of the different macrophyte systems available (Fisher, 1985). Bavor et al. (1988) cite the major considerations in determining the optimum design are solids loadings and hydraulic conductivity. Other factors considered by Bavor et al. (1988) are the area of land available with appropriate topography, land cost, strength of the effluent, evapotranspiration and rainfall of the region, degree of treatment required before discharge, construction, operational and management costs, local technical expertise, budget constraints, substratum availability, local soil permeability, and finally variations in the expected demand on the facility.

Topography may not be a critical constraint provided the hydraulic gradient can be maintained at a level to optimise plant growth (Roser et al., 1987). Configurations of treatment beds can be created to suit the topography or other
site constraints. Bavor et al. (1988) regard configurations such as serpentes, terraces, or the use of baffles as useful in overcoming site difficulties.

4.3.2 Wetland Types

Fisher (1985) cites three types of wetlands used for waste water treatment, they are natural wetlands, constructed wetlands and aquatic macrophyte lagoons. Brix (1991) nominates four distinct types of macrophyte systems:

a) free floating - such as water hyacinths
b) emergent macrophytes
c) Submergent water plants
d) multi-stage systems that consist of a combination of the above types.

Roser et al. (1987) separates the systems according to the location of the photosynthetic parts of the plant and the result is very similar to the categories established by Brix (1991). Often natural systems may contain all four types but a constructed wetland may be only restricted to one. A commonly used type utilises sub surface flow through the plant rhizosphere. These constructed wetlands may be designed as a single unit, series of units, along contours, stepped or flat (Wood, 1990).

A subsurface flow system incorporating emergent macrophytes was selected as appropriate for the North Katoomba site. Several reasons governed this choice, such as the evidence available detailing the ability of these systems to treat landfill leachate (Surface et al., 1990; Sanford et al., 1991), prevention of pooling surface water to minimise breeding of disease vectors, and constraints of the site.
4.3.3 Site Constraints

The North Katoomba landfill is situated in a steeply sloping valley where land availability for a constructed wetland was restricted to approximately 2,400 m². Rainfall in the region is approximately 1500 mm per annum and generally falls in the warmer months of the year. The site has been used as a municipal landfill for the past 15 years and is anticipated to reach capacity in another 15-20 years depending upon increases in local population and the success of such projects as recycling schemes.

The design of the wetland needed to accommodate the restrictions of land availability, cost, operational difficulties such as vandalism of equipment left on site, shortage of skilled maintenance staff, no power source, highly erodible soils, a shallow water table, highly variable waste water volumes, zero site discharge licence, and future site development.

4.3.4 Waste water Influence

Factors to be determined before the configuration was established were the proposed gravel/void volume, area, depth and volume required for an adequate retention period, rainfall average, effluent volume estimation and estimations of suspended solids, TKN, Total Phosphorous, and the bacterial load.

4.3.5 Pre treatment

Fisher (1985) and Wood (1990) discuss the necessity of pre treatment and cite the settling of solids as an essential pre treatment consideration. At the North Katoomba landfill site the leachate is produced principally through ground water contact with the waste material. The leachate is collected and channelled into a pond which traditionally relied on evaporation or pump out to road tanker as the management strategy.
Soils at the site are light and friable and easily eroded which has required soil stabilisation by use of organic cover and seeding a cold resistant grass species. The suspended matter in the leachate and the material collected in the pond from surface flow is readily settled with a fairly low colloidal particle load. It is anticipated that the collection pond will provide adequate pre treatment for the leachate.

4.3.6 Hydraulic Conductivity

Maintenance of the hydraulic conductivity is a major concern and blockages at the inlet are a common cause for hydraulic failure (Schierup et al., 1990). Surface et al. (1991) link conductivity to particle size of the substrate. Waste water with a substantial suspended loading will cause blockages at the inlet through the entrapment of suspended matter leading to head loss of the effluent.

In addition to the settling aspect the pipe taking the effluent from the collection pond into the wetland will float on the surface of the pond ensuring that only the most clarified water enters the wetland. The first half meter of each treatment bed will contain a substrate of a particle diameter in excess of 50 mm which will conduct the waste water into the smaller diameter gravel substrate without headloss, channelling, or build up of suspended matter. Wood (1990) recommends 60-100 mm stones at locations where conductivity could be a problem.

To reduce the possibility of short circuiting and facilitate an even distribution of waste water across the width of the treatment beds the effluent is channelled into a concrete or plastic gully box drain at the leading edge. The drain has slits cut in the base of varying widths, that is narrow in the centre and wider at the outer edges. This design assists to promote the even distribution of water across the entire leading edge of the bed. Roser et al. (1987) indicate that
manipulation of the gravel porosity as suggested will assist in the vertical flow of the effluent through the substrate and promote contact of effluent with the rhizosphere.

**Fig. 4.3.6. Gully Dispersal Design**

top view - gully with slots cut to allow dispersal along to the edges. In very low flows, the gully may be raised in the centre to disperse effluent to the edges more effectively.

Hydraulic head loss can be minimised by compartmentalising the system as opposed to use of barriers in continuous single unit systems. To achieve a constant hydraulic gradient a method of grading the substrate in the treatment beds to ensure accuracy and uniformity is necessary. Again compartmentalising will afford this objective and enhance the precision potential during construction.
4.3.7 Aeration

Sub surface flow systems benefit from an aeration step to improve levels of dissolved oxygen transfer within the substrate (Hammer, 1992). To aerate the effluent, the gravel beds will be positioned 20-50 cm below the box drain. The movement of water through air enhances the potential for maximum aeration of the waste water and promotes the oxidation processes at the aerobic/anaerobic interface within the substrate.

During the early phases of preliminary data gathering when the leachate flow rate was determined the rainfall was unseasonably high. However as heavy rain events are likely to reoccur sometime in the future the capacity and retention figures were based upon the flow rates of 2000-8000 litres of waste water per day. These figures do not represent the amount of leachate produced within the landfill but rather a mixture of leachate, ground water and surface runoff channelled into the collection pond.

It may be advantageous to periodically allow the wetland to drop in volume to promote super aeration of the substrate as void spheres empty of water and fill with air. To permit the draining of the treatment beds a length of agricultural pipe will be positioned at the base on the end of each bed. A suction device can then be used to drain the water from the treatment beds when necessary.

Bavor et al. (1988) consider a plug flow regime will enhance the oxidation processes within the treatment beds. The plug flow can be achieved by having a series of smaller treatment units in consecutive phases, this will permit one bed to drain into the next. The flow into each bed can be manipulated to vary retention periods or volume and promote plug flow. Compartmentalisation can prevent flooding and desiccation of the rhizosphere.
Chudoba et al. (1991) state that it is a well known fact that chemical reactions of the first and higher orders are more effective in a plug flow regime, whereas completely mixed biological systems have been known to be less effective than compartmentalised bioreactors. Completely mixed systems are a disadvantage as they may lead to promotion and growth of filamentous microorganisms that contribute to biomass bulking.

4.3.8 Configuration Options

Three configurations were devised for assessment of their potential, they were Rectangular, Diamond and Circular (see Appendix 1). The configurations were considered for suitability with respect to hydraulic gradient, plug flow, area required, and financial and operational constraints. The following issues contributed in determining the optimum design.

a) **Length/Width Ratio**: Bavor et al. (1988) indicate that a length width ratio of 10:1 will allow for reduced cross sectional loading of solids. Multiple units rather than long, narrow beds provide promotion of plug flow.

b) **Retention Period**: to ensure a minimum retention period of five days a gradient less than 5%, that is almost flat is needed. The capacity of the void volume should be sufficient to retain the waste water generated over a 5 day period of maximum flow (8,000 l d⁻¹) which should permit adequate pollutant removal to occur.

4.3.9 Liner and Substrate Requirements

Ground water pollution is considered a problem by staff at the North Katoomba landfill and lining the constructed wetland is not only recommended because of this but also with regard to the highly porous nature of the soil. Either a clay liner
or an artificial polymer compound creating an impervious barrier between the leachate and surrounding soil is proposed as part of the design.

Sanford *et al.* (1990) found gravel superior in removal of iron, manganese, BOD, and nitrogen than sand or sandy loam. Wood (1990) comments that the choice of substrate affects microbial attachment, plant growth, physical and chemical functions, hydraulic conductivity, opportunity for organisms to contact the waste water, and oxygen availability.

4.4 **Selection and propagation of aquatic macrophyte species**

Selection of suitable macrophytes is based on several criteria. Cullen and Lambert (1990) state there are several rooted emergent macrophyte species native to Australia that have proven useful in the treatment of waste water and water quality improvement. Selected plants should ideally be native to downstream regions to prevent introduced species creating an ecological impact. By deleting exotic species the number to select from is considerably reduced.

Another selection criteria is the ability of the plant to survive a variety of waste water concentrations and variations in flow rates. Nutrient removal capacity is another significant factor as are growth rates. Fisher (1985) found growth rates may vary in some species according to the season some more so than others. These seasonal growth patterns are attributed to temperature and light intensity variations. Macrophytes must demonstrate an ability to provide support to the bacterial population in the substrate by generating an extensive root system. Some macrophytes are salt tolerant and others cold tolerant, the choice of plant will also be determined by the physiochemical conditions that prevail (Fisher, 1985).

Cullen and Lambert (1990) detail characteristics of certain macrophyte species
found in natural wetlands to provide further information for use in selecting appropriate macrophyte species.

**Schoenoplectus** (*S.fluitans, S.productus and S.validus*)

**Great Bulrush or River Clubrush** -

- emergent inflorescences
- favours fast flowing clear streams
- native submerged
- best in depth <1 m
- perennial
- grows to 3m with 1mm diameter stems.
- native
- growth period October to April
- best in shallow water, 50-100 cm
- margins swamps, will not extend inland
- invasive, seeds germinate easily in water
- spread by birds
- slow to colonise, forms dense stands
- rhizomes less active than Typha
- faster growth in silt than clay, poor growth in low nutrient and sandy soil
- senece in winter
- useful in erosion control and as a wildlife habitat

**Typha** - (*T.domingenis, narrow leaf and T.orintalis, broad leaf*)

**Cumbungi or Cattail**

- rhizomatous
- to 4 m in height
- invasive - seed dispersal by wind
- seed bulk may cause allergies and aesthetically undesirable
propagates from seed

grows in depth up to 1.5 m

forms extensive underground rhizosphere network producing shoots

slow growth in winter but shoots year round

growth can be controlled by burning every 3-4 years

prefers slow or stationary flow

senesce in winter

useful as a pollutant trap in natural waterways especially for contaminants such as oil

good for erosion prevention and in providing shelter for birds.

*Phragmites* - *(Phragmites australis)*

Common Reed or Cane Grass

- extensively used in treating waste water in Europe and U.S.A.
- rhizomes resist penetrating anaerobic zone and generate formation of a mosaic of aerobic and anaerobic zones
- robust
- emergent perennial
- grows to 3 m
- exhibits senescence but slow growth in winter
- grows up to 2 m in depth
- salt and cold tolerant
- native
- grows in areas where frequent inundation occurs.
**Parrots Feather** - (*Myriophyllum aquaticum*)

- Introduced species
- Submerged stems
- Propagation via fragmentation
- Slow moving or static water up to 2 m in depth
- Best growth in waters with high nitrogen content
- Grows in either mud or gravel

*Reed et al.* (1988) formulated a table for selecting macrophytes

#### Table 4.4A  Emergent Aquatic Plants for Waste water Treatment
(from Reed et al., 1988)

<table>
<thead>
<tr>
<th>Common name Scientific name</th>
<th>Distribution</th>
<th>Temperature °C Desirable/ Optimum *</th>
<th>Maximum salinity tolerance ppt ‡</th>
<th>Optimum pH</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cattail Typha spp</td>
<td>worldwide</td>
<td>10-30/12-24</td>
<td>30</td>
<td>4-10</td>
</tr>
<tr>
<td>Common reed Phragmites communis</td>
<td>worldwide</td>
<td>12-33/10-30</td>
<td>45</td>
<td>2-8</td>
</tr>
<tr>
<td>Rush Juncus spp</td>
<td>worldwide</td>
<td>16-26/----</td>
<td>20</td>
<td>5-7.5</td>
</tr>
<tr>
<td>Bulrush Scirpus spp.</td>
<td>worldwide</td>
<td>16-27/----</td>
<td>20</td>
<td>4-9</td>
</tr>
<tr>
<td>Sedge Carex spp.</td>
<td>worldwide</td>
<td>14-32/----</td>
<td>----</td>
<td>5-7.5</td>
</tr>
</tbody>
</table>

‡ppt = parts per thousand  
* Temperature range for seed germination, roots and rhizomes can survive in frozen soil

From this table temperature, salinity and optimum pH range give additional information to assess the species suitable to a specific instance. Development of plant diversity in constructed wetland is an accepted and desirable practice, (Fisher, 1985; Reed et al., 1990).
Bavor et al. (1989) trialed 3 emergent macrophytes for sewage treatment in constructed wetlands. The species T.orientalis, S.validus, and M. aquaticum were trialed for their pollutant removal capacity with M. aquaticum proving the least effective. Schoenoplectus returned satisfactory treatment values but Typha was the most successful macrophyte.

Smith et al. (1982) found Typha to be the most effective in the treatment of secondary sewage effluent. This species was very productive and removed the most nutrients per unit area. Smith et al. (1982) quote studies from the USA, Canada, New Zealand and Australia that concluded Typha removed nutrients more successfully than any other plants trialed.

Roser et al. (1987) noted that particular species of macrophytes such as Phragmites spp and Schoenoplectus spp may enhance bacterial die-off in waste water through the secretion of antibacterial agents in their root systems.

Bavor et al. (1989) and Roser et al. (1987) discuss seasonality as a factor in constructed wetlands with macrophytes, however Tsiprijan and Kravets (1988), found that by using a combination of plant species successful treatment can be a year round phenomena.

Combinations of Typha and Phragmites complement the actions of one another as does Typha and Myriophyllum. Reed et al. (1988) found that many European reed bed treatment systems utilise Typha and Phragmites with Typha generally being placed at the inlet zone due to its ability to clump together rapidly and withstand strong effluent concentrations and enhance hydraulic conductivity.

The species selected for the North Katoomba wetland were Typha cumbungi and Phragmites australis as each should be able to remain viable given the

75
physiochemical conditions of the leachate and the environment. The characteristics of the two species should also maximise the pollutant removal capacity of the wetland.

Finally in regard to plant selection, the macrophytes decided upon should not be a species declared noxious to the local area. Several species of aquatic plants are declared noxious in the Blue Mountains region. These plants are *Salvinia molesta*, Water Hyacinth (*Eichhornia crassipes*), Alligator weed (*Alternanthera philoxeroides*), Water lettuce (*Pistia stratiotes*), Largarosiphon (*Largarosiphon major*) (Blue Mountains City Council-Weed control, 1992).

In summary the optimum design and most appropriate macrophyte species was determined by site constraints, by the characteristics of the surrounding environment and the pollutants found in the leachate. The various configurations reviewed can be found in Appendix 1.

4.5 Field Techniques and Data Analysis

Sampling periods were disturbed by several influences, some were predictable, such as site maintenance activities however the single most disruptive influence to the project was an extended delay in the building of the constructed wetland by Council. This delay resulted in the post commissioning sampling period to be significantly reduced in an attempt to remain within a reasonable time frame for the project.

4.5.1 Sample Collection

Samples were taken of both dilute and undiluted leachate. The undiluted leachate was collected from small rivulets flowing directly from the working face of the landfill. Point sources of ground water infiltration were difficult to determine due to the constantly changing form of the site. The leachate was
generally flowing directly from the refuse after periods of heavy rain and collection was performed by channelling the flow to create a reasonable volume which could then be collected in sterile containers. The containers were placed in an insulated box for transport to the lab.

The diluted samples were retrieved directly from the leachate pond which was a mixture of storm flow and leachate. Transport of the samples was the same as for undiluted samples.

4.6 Pilot Experiments
To assess the potential of the design of the constructed wetland a pilot experiment was conducted. The aim of the experiment was to simulate conditions the leachate would be subjected to when introduced to the gravel/macrophyte beds. The results obtained from this experiment would provide an indication for the capacity of gravel/macrophyte beds to treat landfill leachate and give an indication as to where possible problems may arise with the design.

Three polystyrene boxes measuring 53 cm x 34 cm x 15 cm were lined with a polymer waterproof membrane and then filled with washed river gravel, 0.5 - 0.8 cm in diameter.

In September, 1992 these gravel filled boxes were inundated with tap water, 2 were planted with macrophytes that is one with Typha and the other with Phragmites. The third box was left with gravel only.

The boxes were maintained in an unheated glass house for six months to allow the plants to establish an extensive rhizome system and an active microbial
population. The plants were propagated from cuttings and twelve of each species were planted in their respective containers.

During the six months establishment period the plants experienced vigorous growth during the warmer weather (summer season) and began to senescence in late February, early March (autumn). In March the boxes were put into a step configuration and slightly sloped to allow the water to drain under the influence of gravity from the top box to the bottom box. This configuration simulated the design for the field application and also prevented the water from pooling on the surface of the gravel and ensured contact of the effluent with the rhizosphere.

After draining from the third box the water was channelled back to the container holding the original volume. This enabled easy and accurate estimates of loss through evapotranspiration and closed the system so samples could be removed when required. A peristaltic pump was used to take the water from the container back up to the first box.

To simulate the required retention period of 5-15 days within the gravel beds the flow rate was adjusted to introduce eight litres per day to the system. This gave a theoretical retention period of five days.

**4.6 Pilot Experiment**

**Calculations**

- **Volume of box** - 15 cm x 34 cm x 53 cm = 27,030 cm³
  
  \[
  27,030 \div 1,000,000 = 0.02703 \text{ m}^3 \\
  0.02703 \times 1,000 = 27.03 \text{ l box}^{1}
  \]

- **Void Volume** = 50% of total volume

- **Available volume for water** = 13.5 l

- **Total available volume for 3 boxes** = 13.5 x 3 = 40.5 l

- **Retention Period** = 40.5 l (one days' volume in total system)
for 5 days retention time - $40.5 \div 5 = 8.0 \text{ l d}^{-1}$
for 7 days retention time - $40.5 \div 7 = 5.7 \text{ l d}^{-1}$
for 10 days retention time - $40.5 \div 10 = 4.0 \text{ l d}^{-1}$
for 14 days retention time - $40.5 \div 14 = 2.8 \text{ l d}^{-1}$

Flow Rate - the peristaltic pump delivers the volume at a set rate. That is:
10 ml per 2 min and 20 s
5 ml per 1 min and 10 s
4.8 ml per 1 min and 0 s
the flow rate per 24 h = $4.8 \times 60 \times 24 = 6,912$ mls
Litres per 24 h = $6,912 \div 1,000 = 6.9$ l

the retention times calculated 6.9 l d$^{-1}$ will give a retention time of 5→7 which is within the range appropriate for the experiment.

e Analysis- to evaluate the design of the constructed wetland, samples I be analysed for the primary pollutants existing in the leachate from the Katoomba landfill. Sample analysis included microbial content, (faecal ms), total Kjedahl nitrogen and total phosphorus. Temperature, humidity -I will continue to be monitored throughout the experiment.
5.0 RESULTS AND DISCUSSION

5.1 Results of Leachate Sampling

The leachate was sampled in two forms. The samples consisted of undiluted leachate and samples taken from the leachate pond that were diluted with ground water and a small amount of storm flow. The concentrated leachate was only collected for short periods of time because as the site developed the surface flow of leachate was inaccessible, i.e. sub-surface flow.

Table 5.1 A represents sampling results for the dilute leachate samples. As the volume entering the constructed wetlands will be sourced from the leachate pond, so these figures represent more accurately the quality of the waste water entering the system.

<table>
<thead>
<tr>
<th>Date</th>
<th>Weather</th>
<th>Air temp °C</th>
<th>H2O temp °C</th>
<th>pH</th>
<th>Cond us/cm</th>
<th>TKN mg/l</th>
<th>TP mg/l</th>
<th>SS mg/l</th>
<th>F.C. cfu 100mls⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>13.6.91</td>
<td>fine</td>
<td>10.5</td>
<td>5.4</td>
<td>7.6</td>
<td>1.04</td>
<td>2.3</td>
<td>0.2</td>
<td>5</td>
<td>0.4</td>
</tr>
<tr>
<td>3.7.91</td>
<td>o'cast</td>
<td>7.8</td>
<td>7.8</td>
<td>1.6</td>
<td>1.30</td>
<td>55</td>
<td>2.2</td>
<td>200</td>
<td>2.7x10⁴</td>
</tr>
<tr>
<td>16.7.91</td>
<td>fine</td>
<td>10</td>
<td>8.5</td>
<td>7.6</td>
<td>1.04</td>
<td>74.8</td>
<td>1.1</td>
<td>160</td>
<td>900</td>
</tr>
<tr>
<td>31.7.91</td>
<td>fine</td>
<td>10</td>
<td>7.8</td>
<td>8.1</td>
<td>1.30</td>
<td>54.8</td>
<td>1.1</td>
<td>250</td>
<td>10</td>
</tr>
<tr>
<td>12.9.91</td>
<td>fine</td>
<td>10</td>
<td>7.0</td>
<td>7.6</td>
<td>1.31</td>
<td>55</td>
<td>2.2</td>
<td>200</td>
<td>2.7x10⁴</td>
</tr>
<tr>
<td>25.9.91</td>
<td>fine</td>
<td>10</td>
<td>7.8</td>
<td>1.6</td>
<td>1.31</td>
<td>55</td>
<td>2.2</td>
<td>200</td>
<td>2.7x10⁴</td>
</tr>
<tr>
<td>24.10.91</td>
<td>o'cast</td>
<td>11.5</td>
<td>16</td>
<td>7.8</td>
<td>1.73</td>
<td>45</td>
<td>2.2</td>
<td>150</td>
<td>2.7x10⁴</td>
</tr>
<tr>
<td>5.11.91</td>
<td>fine</td>
<td>23</td>
<td>19</td>
<td>8</td>
<td>1.60</td>
<td>30</td>
<td>4.3</td>
<td>30</td>
<td>5.7x10²</td>
</tr>
<tr>
<td>24.3.92</td>
<td>fine</td>
<td>20</td>
<td>25</td>
<td>8</td>
<td>1.61</td>
<td>30</td>
<td>4.3</td>
<td>30</td>
<td>5.7x10²</td>
</tr>
<tr>
<td>9.4.92</td>
<td>fine</td>
<td>14</td>
<td>17</td>
<td>7.7</td>
<td>1.68</td>
<td>30</td>
<td>4.3</td>
<td>30</td>
<td>5.7x10²</td>
</tr>
<tr>
<td>10.6.92</td>
<td>o'cast</td>
<td>7</td>
<td>6</td>
<td>8</td>
<td>1.82</td>
<td>30</td>
<td>4.3</td>
<td>30</td>
<td>5.7x10²</td>
</tr>
<tr>
<td>25.6.92</td>
<td>fine</td>
<td>9</td>
<td>9</td>
<td>8</td>
<td>1.40</td>
<td>30</td>
<td>4.3</td>
<td>30</td>
<td>5.7x10²</td>
</tr>
</tbody>
</table>

Baumann (1981) and Mullins and Sommers (1983) found numerous variations occurring with time in landfills. Therefore, to characterise the leachate it was essential to sample over an extended period of time. By doing so the influence of seasonal changes and the multiple variations peculiar to production of leachate and the breakdown of the refuse could be better analysed.

In terms of pathogen removal the results of this study are paralleled by other researchers such as Mathess and Pekedger (1981) and Donnelly and Scarpino
(1984) where it was suggested that the numbers of pathogenic organisms may vary over time according to the chemical and physical environment. Temperature also appears to play a significant role in the survival of pathogens. This relationship was further investigated in the bench scale pilot experiment. Suggested interactions and relationships abound in the chemical, physical and biological constituents (Mullins and Sommers, 1983) and the amount of moisture in contact with the refuse clearly affects the effluent quality.

5.1.1 Statistical Analysis

Graphs 5.1.1 investigate the correlations between faecal coliforms and the parameters of pH, conductivity and temperature. Regression statistics demonstrate a relationship between faecal coliforms and temperature with \( r = 0.84 \) (see appendix 5). This is a strongly positive relationship. Predicting when maximum concentrations of coliforms will enter the wetland will be important in monitoring the efficiency of the treatment system.

No significant relationship appears to exist between pH and coliforms, \( r = 0.51 \). The range of pH values generally remained neutral, but a peak value of 9 and minimum 7.4 indicates the effluent is quite alkaline. The reason may be associated with naturally occurring inorganic ions contained in the ground water infiltrating the site or may be associated with the leachate. Further testing of the leachate for calcium carbonate found a concentration of around 500 mg l\(^{-1}\) in both the leachate pond and leachate flowing directly from the landfill.

These values may reflect the leaching of remnant bags of lime put in the leachate pond in 1990 in an effort to raise the pH and destroy the bacteria. Conductivity does not appear to influence the concentration of faecal coliforms in the leachate, \( r = 0.46 \).
The volume of moisture coming into contact with the refuse in the North Katoomba landfill via infiltration of ground water is unknown. Although the leachate collection pond is pumped out regularly, no records of the volume removed have been maintained. Given the highly permeable nature of the soils on site and lack of an impermeable membrane liner in the existing leachate pond it is likely that infiltration of contaminated ground water to the surrounding environment is occurring.

5.1.2 Physical Testing

Physical testing included air and leachate temperature, turbidity and suspended solids.

a) Temperature - results of the temperature monitoring in the leachate pond revealed a variation of 10 °C over the sampling period. At an elevation of 1017 m above sea level this landfill site experiences minimum extremes of cold that may be below 5 °C at night. The summer daytime average is around 25 °C. Coolest months of the year are June, July and August and the warmest December, January and February.

Temperature can affect the function of an constructed wetland as many mechanisms within the wetland are subject to variations in rate according to temperature (Bavor et al., 1989). However Cooper and Findlater (1990) and Barnes et al. (1983) suggest temperature may only have a limited effect on conditions existing in the wetland bacteria (and by inference other parameters) due to the insulation by the leaf litter and plant cover accumulation. Microbial activity occurring within the rhizosphere may also assist in maintaining temperature by producing heat. Although temperature may influence the removal efficiency within the wetland (Bavor et al., 1989) other mechanisms that
are not temperature dependent such as sedimentation means removal capacity may be reduced but perhaps still remain sufficient.

b) Rainfall - annual rainfall in the region ranges from a minimum of 620 mm/annum to a maximum of 1850 mm/annum. An average value would be in the vicinity of 1400-1600 mm/annum.

The North Katoomba landfill site has an area of 7.2 hectares and the majority contains an extensive system of surface drains to minimise infiltration of storm flow onto the site. The capped area has the soil surface stabilised to prevent erosion and the vegetation cover is maintained by the practice of spraying the leachate back over the capped and vegetated landfill. Principle points for the storm water permeating and adding to the volume to be disposed of through the wetland are the surface area of the leachate pond, the working face of the landfill and the surface area of the treatment beds. This area will increase over time as the landfill progresses.

Anecdotal evidence from site staff and personal observation suggest a highly variable volume of ground water entering the landfill places an unknown quantity of water to be disposed of into the system and this factor warrants further study. Although the wetland is designed to treat 2000 l d\(^{-1}\) to 8000 l d\(^{-1}\) which should be sufficient to maintain the levels in a 1500 m\(^3\) pond (estimated volume) a second backup retention pond may be required to manage the volume in periods of prolonged heavy storm events as can occur from time to time in this mountain region particularly summer when the majority of rainfall occurs.

Site staff suggest rapid increases in volume during storm events are common but the pond volume did not necessarily correspond with wet periods and levels had been known to increase in dry weather.
When comparing the results for the concentration of constituents for the leachate flowing directly from the working face and the volume in the leachate pond a significant dilution factor is evident. Table 5.1 B gives a comparison between the leachate characteristics flowing directly from the working face compared with values for the volume contained in the leachate pond taken during the period 1990/91.

### TABLE 5.1 B
Comparison of Dilute and Undilute Leachate from the North Katoomba Landfill Site

<table>
<thead>
<tr>
<th>Temp air/H₂O</th>
<th>F. coli (cfu 100mls⁻¹)</th>
<th>SS (mg/l⁻¹)</th>
<th>Turbidity (ftu)</th>
<th>Conductivity (ms cm⁻¹)</th>
<th>NH₄⁺ (mg/l⁻¹)</th>
<th>P (mg/l⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Undilute</td>
<td>10.2 / 7.1</td>
<td>5.5x10²</td>
<td>200</td>
<td>&gt;200</td>
<td>1190</td>
<td>8.8</td>
</tr>
<tr>
<td>Dilute</td>
<td>10.2 / 8.1</td>
<td>4.3x10⁴</td>
<td>103</td>
<td>39</td>
<td>1610</td>
<td>43</td>
</tr>
</tbody>
</table>

Comparison dilute and undilute leachate from North Katoomba site reveals that faecal coliforms were generally elevated at the source of the leachate and figures were consistent, that is when concentrations at the source decreased the concentration of coliforms in the leachate pond were comparably decreased.

The source of the undiluted leachate samples ceased being accessible when the working face progressed and migration into the leachate pond became sub-surface. During the initial phase of this project before the working face was altered, the volume of leachate was directed and channelled into the collection pond at a single point. It was possible on several occasions to quantify the flow coming from the working face. This figure varied from 2,800-8,000 l d⁻¹. Figures established are estimates for maximum and minimum flow rates to be treated by the constructed wetlands.

Clearly the flow is highly variable and could be greater or less than the figures indicate. To overcome this difficult to estimate factor, in the design of a
treatment system the inflow to the first treatment bed will be controlled by pumping the leachate up to a header tank or pond immediately upslope of the treatment bed. From this tank the flow rate required could be controlled and should excess leachate accumulate in the pond a second pond could be used to hold the excess until it can also be pumped to the header tank. Volume reduction is an important goal for the constructed wetland. Surface et al. (1990) estimated that an evapotranspiration loss of between 20% in winter and 40% in summer could be expected for landfill leachate in an constructed wetland.

c) Suspended Solids

Reed and Brown (1992) expressed concern about potential clogging in constructed wetlands. Viswanatha and Boetcher (1990) suggest suspended matter can come from a range of sources including soluble iron forming complexes with organic matter in waste water, resulting in clogging of the wetland. Bavor et al. (1987) give a value of 15 - 70 kg h\(^{-1}\) d\(^{-1}\) as a limit for suspended solids loadings into an constructed wetland.

Leachate samples collected from the North Katoomba landfill site averaged 103 mg l\(^{-1}\) d\(^{-1}\) with a range of 20-160 mg l\(^{-1}\) d\(^{-1}\). This gives a suspended solids loading of 0.2 kg per 400 m\(^{2}\) d\(^{-1}\) or 5 kg h\(^{-1}\) d\(^{-1}\) during minimum flows of 2,000 l d\(^{-1}\) and during maximum flow a value of 20.5 kg h\(^{-1}\) d\(^{-1}\). However the inflow volume will be controlled at a prescribed rate and provided the volume per day is below 7,000 l d\(^{-1}\) the suspended solids load should be remain within recommended guidelines.

The main concern with suspended solids is clogging of the substrate at the inflow to the wetland, (Bavor et al., 1987), subsequent short circuiting and an inefficient flow regime. Considerable settling of solids currently occurs in the leachate pond and the influent volume will be drawn from the first few
centimetres of surface water ensuring only the smallest possible fraction both in terms of load and size of suspended matter enters the wetland.

Design parameters which have been included to prevent clogging include dispersal of effluent along the full width of each treatment bed (10 m), the use of crushed rock rather than river gravel in the first and last meter of each bed giving a larger void volume to facilitate the movement of the effluent through to the gravel. Crushed rock has a diameter exceeding 40 mm whereas the river gravel forms a smaller, denser matrix. Management strategies for the constructed wetland will need to include removal of the filtered and settled solids material deposited in the first bed. Head loss, short circuiting and changes in hydraulic gradient can occur because of accumulation of settled solids in the treatment beds.

Bavor et al. (1989) report a reduction of 98% in suspended solids in sewage effluent treated with a constructed wetland and found that problems associated with the suspended solids loading were overcome by increasing the diameter of the gravel at the inflow point from 5-10 mm to 30-40 mm. Roser et al. (1987) found that where influent suspended solids loads regularly exceed 100 mg l\(^{-1}\) the load was rapidly reduced by filtering effects of solid matrix (river gravel) combined with sedimentation in the void pore volume. Final effluent suspended solid discharges were reported to be in the range of 6-18 mg l\(^{-1}\).

Given the design considerations built into the proposed North Katoomba wetland and that the site management have spent considerable time and expense in stabilising site surface soils, the suspended solids loading should be a manageable aspect of the constructed wetland.

In section 2.6 the Clean Waters legislation requirements for discharge of waste water gives a maximum load for suspended solids of 400 mg l\(^{-1}\) into freshwater
systems. Surface et al. (1990) and Sanford et al., (1990) state that in landfill leachate experiments a reduction of 70-75 % occurred through the constructed wetland. If this system emulates those results, a reduction of approximately 73-78 mg l$^{-1}$ will occur giving a suspended load of around 30 mg l$^{-1}$ in the polished effluent. These figures will be further verified with the results from the pilot experiment.

The figure of 30 mg l$^{-1}$ is well below the limit set for freshwater discharge into Level 2 waterways, however the lower the load the better for the receiving waters.

d) Soils

The soils at the North Katoomba landfill site are typical of the Narrabeen sandstone group and characteristic to the region. Although the design for the constructed wetland recommends the use of an impermeable membrane liner in each of the treatment beds the soil characteristics of the site are nevertheless important to the future potential of the project.

After the performance values for treatment have been stabilised and are consistent it may be acceptable to discharge the treated effluent off site. The porosity and permeability of the soils will indicate the potential for water to move off site allowing absorption into the environment surrounding the landfill. Soils characterised by large continuous pore spaces allow for more rapid dispersal than compact soils with a smaller void volume for example when sand is compared with clay (Miller and Donahue, 1990). Clay will act as an efficient impermeable barrier to water migration because the small pore spaces are quickly filled and easily blocked and so prevent movement of the water.
Miller and Donahue (1990) categorised pores according to the average diameter of the soil particles.

\[
\begin{align*}
< 0.5 \text{ mm} & \quad = \text{very fine} \\
0.5-2 \text{ mm} & \quad = \text{fine} \\
2-5 \text{ mm} & \quad = \text{medium} \\
> 5 \text{ mm} & \quad = \text{coarse}
\end{align*}
\]

Water can move by gravitational force from pores larger than 30-60 \text{ um} with a particle size below 30 \text{ um} soils quickly become water logged. Miller and Donahue (1990) calculated the percentage of a given volume of soil occupied by pore space as $E_p$ minus the percentage of solid space.

\[
E_p = \% \text{ pore space} = 100\% - \% \text{ solid space}
\]
\[
E_p = 100\% - \text{bulk density divided by particle density } \times (100)
\]
\[
E_p = 100\% - \text{Pb divided by Pp} (100)
\]

Particle and bulk density calculations are determined using the method recommended by Miller and Donahue (1990).

i) \text{PP:} particle density is described by Miller and Donahue (1990) as the density of the solid particles of soil excluding any void space and existing moisture content. A standard value which can be used for the average particle density dominant for all soils and minerals is $2650 \text{ kg m}^{-3}$or $2.65 \text{ g cm}^{-3}$.

ii) \text{PB:} bulk density reflects the density of a given volume of soil in its natural state, that is it includes water, organic material, the void volume and particle density. For the purposes of the soil evaluation three sites were selected in and around the area of the future wetlands.

\begin{itemize}
  \item \textbf{Site 1} - immediately below the area of the future treatment beds
  \item \textbf{Site 2} - 10 m down slope of the treatment beds
  \item \textbf{Site 3} - below the existing leachate ponds
\end{itemize}
Results from Site 1 will assist in determining the need for an impervious liner in the treatment beds, Site 2 was selected to examine the potential for waste water to migrate through the soil. Site 3 will indicate whether or not the leachate could currently be migrating from the unlined existing leachate collection pond.

To calculate the bulk density the soil sample was oven dried at 100°C for at least one hour. The following calculations were used to determine the bulk of the sample.

\[
\text{Volume} = \frac{TT \times \text{(diameter)}^2 \times h}{2} \\
\text{cylinder height (h)} = .17.5 \text{ cm} \\
\text{internal cylinder diameter} = .2.3 \text{ cm} \\
\text{volume} = 3.14 \times (2.3)^2 \times 17.5 = 3.14 \times 1.32 \times 17.5 = 72.53 \text{ cm}^3
\]

Once the volume of the cylinder is determined the bulk density of each sample can be determined.

**Sample 1**

- Bulk density = soil mass
- soil volume
- oven dried mass = 101.8 g
- volume = 72.53 cm³
- Bulk density = \((101.3) \times (1 \text{ kg}) \times (1,000,000 \text{ cm}^3) / (72.53) \times (1000 \text{ g}) (1 \text{ m}^3)\) = 1403 kg m⁻³ or 1.4 g cm⁻³

**Sample 2**

- Bulk density = soil mass
- soil volume
- oven dried mass = 66.6 g
- bulk density = \((66.6) \times (1) \times (1,000,000 \text{ cm}^3) / (72.53) \times (1000 \text{ g}) (1 \text{ m}^3)\) = 918.2 kg m⁻³ or 0.9 g cm⁻³
Sample 3  

Bulk Density = soil mass  
soil volume  
oven dried mass = 64.7 g  
bulk density = \((64.7 \times (1 \times (1000,000 \text{ cm}^3))\)  
\(= 892 \text{ kg m}^{-3}\) or \(0.89 \text{ g cm}^{-3}\)

Having established a particle density standard value and a bulk density figure for each sample the percentage of void volume or pore space can be calculated.

Sample 1  

Bulk Density (Pb) = 1.4 g cm\(^{-3}\)  
Particle Density (Pp) = 2.65 g cm\(^{-3}\)  
% Pore Space (Ep) = 100% - Pb x (100)  
\[
\begin{align*}
\text{Pp} & = 2.65 \\
\text{(Ep)} & = 100 - 1.4 \times (100) \\
& = 74.6% \\
\end{align*}
\]

\(\text{Ep} = 47.1\%\) of total volume occupied by pore space.

Sample 2  

% Pore Space (Ep) = 100 - 0.9 x (100)  
\[
\begin{align*}
\text{2.65} & \quad (\quad) \\
\text{(Ep)} & = 100 - 33.96 \\
& = 66.03\%\text{ of total occupied by pore space.}
\end{align*}
\]

Sample 3  

% Pore Space (Ep) = 100 - 0.89 x (100)  
\[
\begin{align*}
\text{2.65} & \quad (\quad) \\
\text{(Ep)} & = 100 - 33.5 \\
& = 66.4\%\text{ of total occupied by pore space.}
\end{align*}
\]

The soils tested at each site varied only in one aspect. Site 1 (the site of the future wetland treatment beds) contained a higher organic content and exhibited a smaller pore volume. With a pore volume range from 47-66% the particle size would be coarse to medium and form large continuous pore spaces (Miller and Donahue, 1990). This would mean relatively free water movement through the soil would occur. Soils with this porosity are characterised by high erodability and are loosely packed.
An impermeable membrane liner has been incorporated into each of the treatment beds and also recommended for the leachate ponds both current and future.

If disposal of the polished leachate is eventually considered an acceptable strategy absorption into and migration through the soil should be effective. Surface pooling of the treated leachate is unlikely unless flow rates increase (caused by, for example an increase in ground water infiltration) or the soil becomes saturated via some extreme weather event. The area receiving the polished effluent should be protected from inundation by storm water with perimeter upslope diversion channels and barriers.

Brouwer and Bugeja (1983) calculate the capability of land to absorb effluent. The absorptive capacity is influenced by the volume of water being discharged (including rainfall), the area dedicated as an absorption field and the infiltration rate. Both infiltration and evapotranspiration rate must be determined for both summer and winter. Before a decision could be made regarding discharge acceptability further study of the surrounding environment must be undertaken.

Potential migration of leachate via absorption from the leachate pond should be of considerable concern to site management. Phosphorus has poor attenuation in sandy soils but owing to the high levels of aeration in lightly textured soils mobility of nitrates is good. However viruses and bacteria suffer little restraint in sandy soils (Brouwer and Bugeja, 1983). Lining the leachate collection ponds would therefore appear essential given the characteristics of the site and the nature of the effluent. To reduce the ground water infiltration problems the only alternative is to isolate the waste from the aquifer.
5.1.3 Chemical Analysis Results

a) Nitrogen

Analysis was performed to determine the levels of organic nitrogen using the Kjeldahl method as per Standard Methods for the Examination of Water and Wastewater (APHA, 1992).

TKN values in the leachate ranged from 30 mg l\(^{-1}\) to 78 mg l\(^{-1}\) in the pond and only one single reliable value was obtained from the undiluted leachate and that was 44 mg l\(^{-1}\). This figure is not likely to be representative given the size of the range in the leachate pond figures. From the milligram per litre value calculations for the load entering the full scale system can be estimated in terms of kilograms per hectare per day.

Nitrogen loading has been calculated using an average value of 54 mg l d\(^{-1}\)

Total area of constructed wetland -

\[10 \times 4 \times 5 + 10 \times 20 = 400 \text{ m}^2\]

Load entering wetland on daily basis

a) Wet Flow - 54 x 8000 = 0.432 kg 400 m\(^2\)d\(^{-1}\) or 0.172 kg h\(^{-1}\) d\(^{-1}\)

b) Dry Flow - 54 x 2000 = 0.108 kg 400 m\(^2\)d\(^{-1}\) or 0.043 kg h\(^{-1}\) d\(^{-1}\)

Sources of the organic matter in the leachate results from many possibilities such as detritus vegetative matter (a tree lopping disposal area forms part of the site) and many degradable organic compounds (Robinson et al., 1991; Sanford et al., 1990). The chemistry of nitrogen is complex owing to its various oxidation states. Fisher (1985) comments that removal of nitrogen from effluent is multifaceted owing to the variety of chemical states it can assume.

Incorporated into the design of the constructed wetland are several stages that encourage nitrification and denitrification of the effluent to promote the maximum
removal potential. By encouraging both aerobic and anaerobic zones within matrix the nitrogen will be removed by several mechanisms including plant and cell uptake and as a gas into the atmosphere. 

Between each of the 5 treatment beds the effluent will fall through the air approximately 15 cm into the matrix of the next bed. This design measure will encourage maximal aeration of the effluent. NOx values for the leachate directly flowing from the working face of the landfill ranged around 78 mg l\(^{-1}\) whereas the leachate in the pond was closer to 4 mg l\(^{-1}\).

b) Phosphorus

Phosphorus is frequently targeted in the field of freshwater ecology because it is a primary growth limiting nutrient (Richardson, 1990; Verhoeven and Van der Toorn, 1990). Cullen and Lambert (1990) identified suspended solids, phosphorus and microbial pollutants as the primary degraders of natural waterways.

The leachate produced at the North Katoomba site was examined for total phosphorus levels as per Standard Methods for the Examination of Water and Wastewater (APHA, 1992). The mean value for the total phosphorus in the pond leachate was 1.8 mg l\(^{-1}\) with a range from 1.0 mg l\(^{-1}\) to 3.7 mg l\(^{-1}\). Bavor et al., (1989) cited that influent concentrations of 0.33 mg l\(^{-1}\) to 1.2 mg l\(^{-1}\) experience a reduction of 67% in concentration after being polished through a constructed wetland.

Sanford et al. (1990) stated that in their landfill leachate monitoring, phosphorus concentrations ranged from 0.01 mg l\(^{-1}\) to 2.7 mg l\(^{-1}\). Surface et al. (1990) found a mean value for phosphorus in landfill leachate to be from 0.77 mg l\(^{-1}\) to 0.28 mg l\(^{-1}\) over three separate sampling periods. Hence a mean value of 1.8 mg l\(^{-1}\) is
slightly higher than for other landfills. If a 67% reduction occurs through the wetland the resulting effluent should contain a phosphorus concentration in the vicinity of 0.6 mg l⁻¹. Legislative criteria limit the concentration of total phosphorus in effluent to be discharged into Level 2 waterways to 0.05 mg l⁻¹. The pilot experiment has been designed to determine the potential performance levels of the wetland. Future monitoring of the leachate will confirm the concentrations of total phosphorus and the range that could occur over an extended period.

### 5.1.4 1993/1994 Leachate Sampling Results

<table>
<thead>
<tr>
<th>Date</th>
<th>Conductivity ms cm⁻¹</th>
<th>pH</th>
<th>Nitrates mg l⁻¹</th>
<th>Ammonia mg l⁻¹</th>
<th>Reactive P mg l⁻¹</th>
<th>Zinc mg l⁻¹</th>
<th>Copper mg l⁻¹</th>
<th>Cadmium mg l⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>June 1993</td>
<td>1,610</td>
<td>7.9</td>
<td>1.1</td>
<td>43</td>
<td>0.07</td>
<td>&lt; 0.05</td>
<td>&lt; 0.02</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>July 1993</td>
<td>1,020</td>
<td>7.7</td>
<td>1.1</td>
<td>18</td>
<td>&lt; 0.2</td>
<td>0.06</td>
<td>&lt; 0.02</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>December 1993</td>
<td>970</td>
<td>7.5</td>
<td>&lt; 0.5</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

Table 5.1C documents further leachate sampling from the leachate pond during 1993. Phosphorus and metals have remained low as indicated in previous sampling periods. However ammonia is elevated indicating anaerobic conditions exist in the leachate pond. This result and the presence of a strong odour from the pond suggest an elevated oxygen demand that may be a result of a high organic content. Ammonia levels are similar to those found in raw sewage.

Results confirm that oxidation of the leachate is required to permit maximum removal efficiency of nitrogen. The July, '93 sample included tests for fluoride which were negative. This result suggests that mains water is not infiltrating the landfill site. Metals analysis during this period included Selenium, Arsenic, Mercury, Lead, Chromium, and Manganese all of which remained below levels of detectability (< 0.005 mg l⁻¹).
In 1994 several leachate outbreaks occurred in the batters around the disused portion of the landfill site (see appendix 4 and 5). Conductivity was very high at the lowest point of the batter (2,200 mS cm⁻¹) compared with upslope, however metals remained low. The lower outbreak also revealed elevated ammonia levels, the outbreak higher up in the batter was <0.5 mg l⁻¹ whereas the lower outbreak was 75 mg l⁻¹ and 50 mg l⁻¹ in the two separate samplings. This may be due to the proximity of the old tree dump where waste vegetation has been buried giving rise to a concentrated source of organic material.

5.1.5 Yosemite Creek Sampling

Appendix 4 show the location of Yosemite Creek which bounds the lower portion of the North Katoomba landfill site. Sampling of this creek was performed to determine if contamination from the landfill site was occurring. Sampling sites were in stream above and below the landfill site.

Results revealed elevations in nitrates (0.5:2.1 mg l⁻¹), phosphorus(0.04:0.06 mg l⁻¹), conductivity (84:120 ms cm⁻¹) and iron (0.89:7.7 mg l⁻¹). There was no change in the levels of ammonia. It is difficult to isolate the landfill as the only cause of the elevations because stormwater and surface flow from the slope on the opposite side of the creek may be contributing.

5.2 Results of Pilot Experiment

5.2.1 Pilot Experiment Design

The design of the pilot system mirrored a bench scale system the field design. Three polystyrene foam boxes lined with a heavy duty plastic liners were filled with washed river gravel.

Six months prior to the commencement of the pilot experiment one box was planted with *Typha orientalis* (12 plants) and a second box planted with the
same number of *Phragmites australis*. Both species had been cultivated from rhizome cuttings. The third and final box was filled with tap water and kept moist for the six months during which time the planted boxes were developing an extensive rhizosphere.

In the box filled with gravel moisture, warmth and light created an active aerobic and anaerobic zone. The interface would serve to help act in the removal of pollutants by promoting an active microbial population in the matrix.

During the six months the planted boxes flourished in the glass house conditions. The period of the pilot experiments was in both senescence and regrowth. The pilot experiment proved a useful guide to the potential efficiency of the field system. Although conducted under glass house conditions the variation in temperature was more extreme than the external environment. This was as a result of the glasshouse not being heated. During the day the glasshouse trapped heated air bringing the internal temperature to 20-30% above external readings. However the opposite effect occurred where heat was transferred through the glass to the cooler outside night temperature resulting in internal temperatures being proportionally reduced.

This greater variation in temperature was a benefit to the study as it permitted performance to be measured in extreme temperature conditions. Extreme weather conditions are not infrequent occurrences in the Blue Mountains particularly extreme lower temperatures during winter. (Refer appendix 2)

The design and concept of the system proved effective allowing for minor problems in the early stages such as leakages and equipment calibration variations. These variations were due to sunlight casting directly on the thermohydrograph measuring heat and humidity. Sunlight also caused the
blockage of tubes delivering leachate to the system through the build up of algae. After the tubes, thermohydrograph and leachate storage containers were covered to exclude the light the algae ceased to be a problem. In the full scale system light will be totally excluded from the pipes feeding the treatment beds and blockages should not occur.

With the full scale system as in the pilot experiment gravity is the energy source utilised to move the leachate through the treatment beds. To move the leachate up to the first treatment bed and possibly back to the leachate collection pond an external energy source such as a petrol pump is required. In the pilot experiment this function was performed by a peristaltic pump.
Fig. 5.2.1 A Pilot scheme design to examine potential of constructed wetlands to treat landfill leachate
5.2.2 Results of Pilot Investigations

The results of the pilot investigations are produced from two separate trials. In each trial the leachate was monitored for changes in physical, chemical and biological characteristics including evapotranspiration and mass balance of nutrients.

a) Microbial

The results of the pilot investigations suggest the constructed wetland will have the capability to reduce target pollutants. The faecal coliform population was significantly reduced in the first 5 days after the point of introduction into the initial treatment box.

Numbers of faecal coliforms were reduced from 4800 cfu 100 mls\(^{-1}\) to zero in the first 5 days. Monitoring continued up to 14 days and no regrowth was evident. In a second run of leachate the faecal coliform numbers were reduced from 80 c.f.u 100 mls\(^{-1}\) to zero in the time taken to circulate once through the system (i.e. 5 days). The leachate was monitored for a further 3 weeks to determine if regrowth was occurring but the figure for the faecal coliforms remained at zero, i.e. no colonies emerged from the aliquots filtered.

An experiment was conducted during the first phase of the project in 1991 to establish that the leachate from the North Katoomba landfill site could sustain faecal coliform growth. Leachate was sterilised then seeded with \textit{E.coli} and maintained at 44.5 °C. Results clearly demonstrated that the leachate provided a suitable medium for \textit{E.coli} survival and growth.

b) Nitrogen

Nitrogen reduction, measured as TKN-N, in the first experiment had an influent concentration of 29.24 mg l\(^{-1}\). After 5 days the concentration had been reduced to 2.86 mg l\(^{-1}\) and at 14 days to 1.47 mg l\(^{-1}\). These figures represent a reduction of 95% over a 14 day retention period. In the second trial through the system the influent concentration was 41 mg l\(^{-1}\). After 5 days concentration was reduced to
TKN. Later trials which included the measurement of oxidised nitrogen forms (NO₃) did not indicate a build up of NO₃. It is hypothesised that the TKN loss reflects total N loss, however it is recognised and emphasised that a complete nitrogen balance would need to be carried out to confirm nitrogen removal from the leachate.

c) Phosphorus

Phosphorus, measured as total phosphorus, was more difficult to reduce satisfactorily within the bench scale wetland. From the first pilot experiment, the influent concentration was a low recording of 0.13 mg l⁻¹. After 5 days the concentration had not changed and after 14 days a figure of 0.14 mg l⁻¹ revealed a slight rise in the total phosphorus in the treated leachate (see 5.2.3).

In the second trial the influent concentration was 0.135 mg l⁻¹ and after 5 days the concentration had been reduced to 0.030 mg l⁻¹. After three weeks the final concentration was 0.060 mg l⁻¹ giving a total reduction over the trial period of 55%.

d) Suspended Solids

The initial concentration of suspended solids was 30 mg l⁻¹ which is comparatively low as against the figures recorded for the leachate earlier in the study. However this is not unexpected as the site works undertaken by management have secured much of the loose surface soils surrounding the leachate collection pond. After five days the suspended solids concentration decreased to 10 mg l⁻¹ and the same figure was recorded after 14 days.

In the second trial the influent measured 110 mg l⁻¹ suspended solids. After seventeen days in the system this figure had been reduced to 64 mg l⁻¹. A reduction of approximately 60% could reasonably be expected from these gravel/macrophyte beds.
In the field system the remaining colloidal matter should be extracted by the sixth bed consisting of a shallow sand and grass matrix. Being 20 m in length this bed should be capable of trapping and sedimenting any remaining fine suspended matter.

**e) Evapotranspiration Rate**

Several trials were conducted to estimate the evapotranspiration rate using tap water and leachate. In the first trial leakages occurred, however the last 3 weeks of the trial all the leakages were contained and the system remained secure. Humidity varied according to temperature with the highest readings recorded during the hours 0800-1600 hours and lowest humidity from 2000-0400 hours approximately. Typical humidity measurements were in the vicinity of 70% during the day and down to 30% in the night. There should exist a relationship between the evapotranspiration rate and the temperature and humidity (see Graphs 5.2D and 5.2E)
Graphs 5.2D Evapotranspiration v. time and 5.2E Humidity v. Temperature for total system (planted and unplanted boxes) in pilot trials

**Evapotranspiration Rate v Time**

**Relative Humidity**

**Median Temperature**
Of prime interest was the evapotranspiration rate exhibited by the pilot scheme as volume reduction is an important function of the constructed wetland. Evaporation rates were recorded in the first trial over 16 days and a second trial over 22 days.

Results of the first trial where leakages occurred returned an evapotranspiration rate of 3.2 l d⁻¹ or 23% of the total volume lost per day. However leakages did give an unrealistic figure and this rate should be discounted.

In the second trial no leakages occurred and a far more realistic figure of 1.32 l d⁻¹ or 10% of the total volume lost per day was recorded. The mean maximum temperature for the period was 30.5 °C and minimum was 4.45 °C. Humidity returned mean values of maximum 97.3% and minimum 33.8%. Several nights during the trial period were heavy fogs with 100% humidity recorded. A third trial will determine whether or not 10% can be regarded as minimum figure for a winter period and better figures are recorded as temperature rises and humidity falls in the spring.

Given 10% reduction in total volume on a daily basis it is possible to relate the evapotranspiration loss value to the flow rates predicted for the North Katoomba landfill site. A loss of 200 to 800 l d⁻¹ can be predicted.
Table 5.2 A  Summary of Pilot Experiment Results

<table>
<thead>
<tr>
<th>Experiment</th>
<th>Pollutant</th>
<th>Influent</th>
<th>Effluent</th>
<th>% reduction</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>F. coliforms</td>
<td>4.8x10⁴ cfu 100 mls⁻¹</td>
<td>0 cfu 100 mls⁻¹</td>
<td>&gt;99%</td>
</tr>
<tr>
<td>2</td>
<td>F. coliforms</td>
<td>80 cfu 100 mls⁻¹</td>
<td>0 cfu 100 mls⁻¹</td>
<td>&gt;99%</td>
</tr>
<tr>
<td>1</td>
<td>Suspended solids</td>
<td>30 mg l⁻¹</td>
<td>10 mg l⁻¹</td>
<td>66.6%</td>
</tr>
<tr>
<td>2</td>
<td>Suspended solids</td>
<td>110 mg l⁻¹</td>
<td>64 mg l⁻¹</td>
<td>58.1%</td>
</tr>
<tr>
<td>1</td>
<td>TKN</td>
<td>29.2 mg l⁻¹</td>
<td>1.4 mg l⁻¹</td>
<td>95%</td>
</tr>
<tr>
<td>2</td>
<td>TKN</td>
<td>41.1 mg l⁻¹</td>
<td>1.7 mg l⁻¹</td>
<td>95%</td>
</tr>
<tr>
<td>1</td>
<td>TP</td>
<td>0.13 mg l⁻¹</td>
<td>0.14 mg l⁻¹</td>
<td>(+) 12%</td>
</tr>
<tr>
<td>2</td>
<td>TP</td>
<td>0.14 mg l⁻¹</td>
<td>0.06 mg l⁻¹</td>
<td>55.6%</td>
</tr>
</tbody>
</table>

5.2.3 Nutrient Budget

A mass balance calculation established the nutrient budget within the pilot system. A mass balance was determined by establishing the total amount of nitrogen and phosphorus entering system in terms of concentration multiplied by volume and subtracting from that figure the total amount of nutrients remaining in the leachate at the end of the trial. The amount remaining at the end of the trial was calculated by multiplying the final concentration with the volume remaining after evapotranspiration losses.

First pilot trial - TKN

Initial volume 45 l
Influent concentration 29.84 mg l⁻¹
Initial load 1.342 kg
Final volume 26 l
Final concentration 1.47 mg l⁻¹
Remaining load 0.03 kg

The amount of total nitrogen removed over the 14 days trial was 1.303 kg.
Total phosphorus
Influent concentration 0.1267 mg l\(^{-1}\)
Initial load 5.7 mg
Final concentration 0.1426 mg l\(^{-1}\)
Remaining load 3.7 mg

The amount of total phosphorus removed during the 14 day trial was 2 mg.

Second pilot trial - Total nitrogen
Initial volume 45 l
Influent concentration 41 mg l\(^{-1}\)
Initial load 1.845 kg
Final volume 26 l
Final concentration 1.2 mg l\(^{-1}\)
Remaining load 0.0312 kg

The amount of total nitrogen removed during the 21 days was 1.813 kg.

Total Phosphorus
Influent concentration 0.14 mg l\(^{-1}\)
Initial load 6.075 mg
Final concentration 0.060 mg l\(^{-1}\)
Remaining load 1.56 mg

The amount of total phosphorus removed over the 21 days was 4.5 mg.

These results demonstrate an overall reduction in the nutrient budget during both trial periods. In terms of percentage reduction the average removal efficiency for TKN in the pilot investigations was 97.5%. However, for phosphorus the results were less successful removing only 54.5% of the available nutrient.
5.2.4 Plant Nutrient Uptake

After the two pilot trials were completed the system was dismantled and the biomass removed from the substrate. The biomass was divided into shoot material and root material (above and below the surface of the gravel). The plant material was washed and allowed to dry, and then oven dried at 69 °C for 24 hrs and weighed. Both *Typha* and *Phragmites* were prepared separately. In the box planted with *Typha*, there were 30 plants in the box at the time of harvesting. In the box planted with *Phragmites*, 48 plants were harvested being the total number of plants contained in the box.

Table 5.2B Comparison of Plant Biomass used in Pilot Investigations by Species

<table>
<thead>
<tr>
<th>Species</th>
<th>Total Biomass Shoots</th>
<th>Total Biomass Roots</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Typha</em> spp</td>
<td>111.8 g</td>
<td>666.0 g</td>
</tr>
<tr>
<td><em>Phragmites</em> spp</td>
<td>383.2 g</td>
<td>750.0 g</td>
</tr>
</tbody>
</table>

The total biomass of *Typha* was 777.8 g and the total biomass of *Phragmites* was 1136.2 g giving a combined biomass of 1914 g. Nutrient analysis of the plant material was performed to determine the percentage of nutrients present in the plant tissue. These results were compared to studies by Breen (1992) to estimate the amount of nitrogen and phosphorus absorbed from the leachate by the plants.

Table 5.2C Comparison of % nutrients in plant biomass from plants in unpolluted water with those growing in landfill leachate

<table>
<thead>
<tr>
<th>Species</th>
<th>Location in Plant</th>
<th>Tap water %N</th>
<th>Tap water %P</th>
<th>Leachate %N</th>
<th>Leachate %P</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Typha</em></td>
<td>shoots</td>
<td>1.22</td>
<td>0.21</td>
<td>1.23</td>
<td>0.11</td>
</tr>
<tr>
<td><em>Typha</em></td>
<td>roots</td>
<td>0.65</td>
<td>0.18</td>
<td>1.4</td>
<td>0.31</td>
</tr>
<tr>
<td><em>Typha</em></td>
<td>rhizome</td>
<td>0.60</td>
<td>0.26</td>
<td>0.77</td>
<td>0.38</td>
</tr>
<tr>
<td><em>P. australis</em></td>
<td>shoots</td>
<td>2.49</td>
<td>0.29</td>
<td>1.03</td>
<td>0.13</td>
</tr>
<tr>
<td><em>P. australis</em></td>
<td>roots</td>
<td>1.28</td>
<td>0.18</td>
<td>1.03</td>
<td>0.11</td>
</tr>
<tr>
<td><em>P. australis</em></td>
<td>rhizome</td>
<td>1.14</td>
<td>0.20</td>
<td>N/A</td>
<td>N/A</td>
</tr>
</tbody>
</table>
Comparisons of plant parts demonstrate that shoots contain the highest concentrations of both nitrogen and phosphorus in the unpolluted water. Roots and rhizome material are similar in concentrations to each other.

In the *Typha* tissue, significantly more nitrogen was found in the root material than the shoots. This may have been due to the fact that the plant biomass experiment was not undertaken until well after the system had been permitted to dehydrate. Consequently with shoot mass dying off, translocation of the nutrients may have ceased before the roots had finished absorbing the nutrients remaining in the substrate material.

*Phragmites* shoot and root concentrations in the leachate are similar, suggesting that the shoot material may have withstood desiccation better than the *Typha*. It is evident from the percentage of nutrients in the plant tissue from the pilot experiment that the macrophytes may have had insufficient nutrients available during the start up time of the pilot experiment. This may explain the high level of removal efficiencies exhibited by the pilot experiment.

5.3 Influence of Sampling and Pilot Experiment on Optimum Design

The results of the leachate sampling and the pilot experiment have indicated the demands likely to be placed on the field system and the potential for this system to reduce the pollutant load. Areas of critical concern where design criteria dictate the potential performance of the system have been examined in light of experimental and sampling results.

5.3.1 Aeration

Sampling results confirmed the local authority's original analysis that faecal coliforms were a primary target pollutant. For effective reduction of coliforms to occur it is essential the aerobic\anaerobic interface within the gravel substrate
be promoted. At the North Katoomba landfill site a strong odour from decaying organic matter was always present around the leachate pond reflecting a high oxygen demand in the pond leachate.

Aeration of the leachate is proposed to enhance the removal of ammonia and organic compounds (Robinson et al., 1991; Brix, 1992). Microbially mediated oxidation also occurs within the substrate and oxidation of the leachate is integral to metals removal. Aerobic decomposition combined with anaerobic digestion of organic compounds will occur at the interface if sufficient oxygen exists within the substrate (Sanford et al., 1990; Brix, 1992).

Aeration of the effluent may be promoted by several design factors including incorporating an aeration facilitation stage between each of the 5 gravel treatment beds, the choice of macrophyte species and the subsequent rhizosphere created (Rodgers and Dunn, 1992; Robinson et al., 1991), hydraulic considerations such as a plug flow in the system to promote the aeration of the pore space in the substrate (Sanford et al., 1990) and the creation of macropores in the root-zone which will be a result of the species of macrophyte chosen (Brix, 1992; Roser et al., 1987).

Both Typha and Phragmites have been noted to contribute oxygen to the rhizosphere (Rodgers and Dunn, 1992; Robinson et al., 1991). Brix (1992) noted that macrophyte roots are hollow and contain air filled channels which help to oxygenate the rhizosphere. Although the roots consume a large proportion of the oxygen, sufficient oxygen leaks out of the roots to meet aerobic plant demand processes and help provide an oxic zone around the root surfaces.

Hammer (1992) notes that constructed wetlands utilising sub surface flow systems are unable to maintain adequate dissolved oxygen levels for ammonia
removal. The aeration stage between each treatment bed will permit the leachate to flow into a collecting channel and fall through the air (about 15 cm) to the next bed immediately below. This will facilitate the contact of the anoxic leachate with air and ensure the aerobic qualities of the treatment system are maximised.

Deeper down in the matrix, below the aerobic/anaerobic interface there exists an anoxic zone. In the anoxic region important processes of denitrification and fermentation of sediment organic solids occurs (Bavor et al., 1993).

5.3.2 Hydraulic Conductivity

Brix (1992) stated soil is a poor medium in constructed wetland beds as it restricts hydraulic conductivity. To enhance hydraulics the size of the media in the first half metre of each bed should be increased to >40 mm and the remainder of the substrate in each bed to be 5-10 mm river gravel. These design considerations will maximise the expansion rhizosphere and enhance aeration. Preferential flow paths where the effluent circumvents major areas of the treatment beds can also be avoided by using the larger diameter substrate material (Sanford et al., 1991).

Hydraulic conductivity in the pilot system was maintained throughout the trial period with no surface pooling and the effluent maintaining contact with rhizosphere. Accumulation of algae in tubes feeding the leachate to the system and at the points where the leachate flowed from one treatment bed to the next was evident in the pilot system.

Algal growth may restrict the hydraulic conductivity so measures such as limiting exposure to light and easy maintenance of the dispersion channels for cleaning purposes will be needed.
Short circuiting via preferential flow paths should be prevented by the length to width ratio of 1:3 combined with the use of dispersion channels and the larger substrate material. As the configuration of the pilot system proved successful the original configuration of the field system will be implemented.

Solids accumulation at the inflow points will be managed by the use of larger crushed rock in the first half metre of each bed. Given the suspended solids loading the use of the >40 mm crushed rock will minimise effects of accumulation although a long term management strategy for removal of accumulated solids will need to be devised.

5.3.3 Phosphorus Removal
Total phosphorus levels in the leachate ranged around 0.12 mg l\(^{-1}\) to a maximum of 3.7 mg l\(^{-1}\) and orthophosphate from 0.01 mg l\(^{-1}\) to 0.07 mg l\(^{-1}\). As described in the results from the pilot system, the nutrient budget for phosphorus was favourable using river gravel as a substrate combined with macrophytes in a sub-surface flow system.

Bavor et al. (1987) found that phosphorus removal in constructed wetlands varied with loadings and occasional flushes may occur. Roser (1987) notes that principle removal mechanisms of phosphorus in constructed wetlands included chemical precipitation, adsorption into sediments, incorporation into the microbial population and uptake by macrophytes. Complete and permanent elimination from the system requires sediment removal and possibly plant harvesting at intervals, ranging to decades depending on loadings, to prevent the flushes described by Bavor et al. (1987).

The long term management strategy for the system will need to identify an appropriate method of permanent removal, options may include nutrient
recovery or utilisation. Nutrient recovery options may include plant harvesting and mulching for use on revegetated areas of the landfill or extraction of sediment which could be used as future cover on the landfill site in revegetation areas.

However other substances incorporated into the sediment and plant biomass would also need to be identified, for example trace metals, particularly if the material was to be transported off site for use elsewhere. Recovery for other uses is ideal but cost or lack of available technology may preclude this option at least in the short term.

To enhance phosphorus removal by a reactive substrate such as ironstone is an option. In the North Katoomba system, ironstone will be recommended for use as the larger crushed rock media in the first half metre of each bed. All beds will be planted with macrophytes to maximise the potential for plant uptake. Sanford et al. (1990) note nutrient removal from landfill leachate was more effective in constructed wetland treatment beds planted with macrophytes as opposed to gravel only.

5.3.4 Nitrogen Removal

TKN values ranged from 30-74 mg l⁻¹, nitrates tested by an independent laboratory (see appendix) remained around 1.1-1.2 mg l⁻¹. The pilot experiment returned reduction values of up to 90% over a 15 day retention period. NOx values were also determined for the second pilot experiment. Influent NOx value was 7.73 mg l⁻¹ and after a 21 day retention period, a concentration of 1.783 mg l⁻¹. These results give a 77.06% reduction in NOx over the trial period.

Bavor et al. (1987) attribute the limitation on nitrogen transformation to insufficient oxygen supply, Roser et al. (1987) state nitrogen removal occurs via
incorporation into the microbial population, plant uptake, biologically mediated nitrification and denitrification and conversion to ammonia. Surface et al. (1991) indicate temperature influences removal capability with NH$_4^+$ ions being released back into the effluent in periods of low temperatures.

Incorporation of measures to aerate the effluent will maximise the potential of the aerobic/anaerobic interface and subsequently the biologically mediated nitrification and denitrification processes. To minimise the influence of low temperatures, the long term management strategy will specify that plant biomass be accumulated on the surface of the gravel beds to act as an insulator between the air and the substrate.

5.3.5 Metals Removal

At the North Katoomba landfill site the leachate was tested for Cd, Zn and Pb and consistently exhibited very low levels. Independent analysis (BMCC) tested leachate for Cu, Zn, Cd, Cr, Pb, Hg, As and Se and all were below the limits of detectability in July 1990 and June 1993.

Metals analysis was excluded from the pilot experiment in view of the field sampling results, however should flushes of metals occur in the leachate, the constructed wetlands contain mechanisms capable of removal. Flushes may occur when metals trapped within the landfill are mobilised by complexation with fatty acids in solution, produced as organic matter breaks down within the landfill (Loch et al., 1981; Mullins and Sommers, 1983).

In a study on landfill leachate Sanford (1990) found zinc, iron, manganese and aluminium to be the most abundant metals to be found in leachate from a municipal waste site in solution. Removal efficiencies within constructed
wetlands is noted by Sanford et al. (1990) to be >70% for Zn, 9-67% for Al, 89-98% for Fe and Mn.

Removal of metal peaks in the leachate should be within the capacity of the constructed wetland. Metals are removed through adsorption onto colloidal surfaces in suspension and subsequently trapped within the wetland through sedimentation. Precipitation reactions with carbonates and sulfides can also trap metals in the sediment (Loch et al., 1981). At the North Katoomba site independent results reveal elevated levels for total hardness in the form of CaCO₃(530 mg l⁻¹) in the leachate retention pond.

Mullins and Sommers, (1983) noted inorganic complexes can be formed by metals combining with Ca and Mg within the landfill. It is possible given the level of CaCO₃ concentration in the leachate that complexes have been produced and this may explain the low levels of metals in solution.

Long term management strategy would require the removal of sediments periodically from the constructed wetland treatment beds and an acknowledgment of the potential for the translocation of metals into the plant biomass. Ultimate disposal of the sediment and biomass may vary from secure landfill to metals recovery if economically feasible. Sanford et al. (1990) in a study comparing treatment units that were planted with macrophytes or gravel only found a significant difference in the uptake of some metals but not others. For example iron was more readily removed in unplanted beds whereas zinc was removed more effectively in planted units.

5.3.6 Microbial reduction

Field sampling indicated a seasonality existed in microbial numbers with peaks in late summer. Faecal coliforms ranged from 10³ to 10⁶ cfu 100 ml⁻¹ in warmer weather down to 0 cfu 100 ml⁻¹ in winter. Initial concentrations in the pilot
experiments were 4800 cfu 100 ml\(^{-1}\) and 80 cfu 100 ml\(^{-1}\). In both trials numbers were reduced to 0 cfu 100 ml\(^{-1}\) within 5 days of start up.

Mathess and Pekdeger (1981) found survival time in ground water varies according to the type of virus or bacteria. Clearly within the North Katoomba landfill site the correct environmental conditions of temperature, moisture and substrates for nutrient utilisation must exist. Sampling by the local authority of ground and surface water entering the site gave no evidence of contamination (Skinn, personal communication, 1991).

Donnelly and Scarpino (1984) conducted trials to determine whether pathogens could survive within landfills and if they drain out with the leachate. They determined that pathogenic bacteria and viruses could survive environmental conditions in landfills with some bacteria being noted to survive over two years. However the majority of microbes exhibited rapid die-off when exposed to external conditions.

Baumann (1981) surveyed leachate being produced from twenty municipal landfill sites within the Sydney basin. At Menai garbage depot faecal coliform numbers ranged from 470-21,000 cfu 100 ml\(^{-1}\). Testing indicated rapid die-off on entering receiving surface waters, that is within two kilometres a 99.8% die-off occurred.

_E. coli_ is noted by Mathess and Pekdeger (1981) as having survived in the environment 8 to 23 days; Gerba and Keswick (1981) found _E. coli_ and faecal streptococci exhibited a 99.99% reduction within 20 days of exposure to the environment. The 99% reduction within the pilot scheme is reasonable taking into account natural die-off and the mechanisms which exist within the constructed wetlands that reduce microbial numbers in the leachate.
Removal mechanisms include the presence of bacteriophage and parasitic bacteria such as Bdellovibrio bacteriovorus and species of photolytic bacteria (Mathess and Pekdeger, 1981) filtration and adsorption (Gerba and Keswick, 1981). Variation in survival rates by pathogens of the same species can be attributed to factors such as the presence of organics, pH, encounters with microbial antagonists and desiccation (Gerba and Keswick, 1981). In addition the natural survival capabilities of the particular pathogen will dictate ability to endure ie encapsulated or not.

In constructed wetlands removal mechanisms consist of sedimentation, natural die-off and exposure to hostile chemical, biological and physical conditions, and predatory microbial groups (Hammer, 1992; Fisher, 1985). Certain macrophytes are also noted to excrete an antibacterial solution from their roots, for example Phragmites (Roser et al. 1987).

Hammer (1992) states the principle function of a constructed wetland is to create additional environments for microbial populations. Plant rhizosphere and sediments can contribute an enlarged surface area for the attachment of predatory microbial groups.

The original design of the wetland treatment unit has been proved effective by the pilot experiments in reducing the microbial population within leachate by promoting the optimal environment in the rhizosphere for removal mechanisms to operate.

5.3.7 Substrate and Macrophyte Selection
As referred to under hydraulics the choice of substrate is vital to the successful function of the constructed wetland. In the pilot experiment 5-10 mm river gravel was used in all 3 boxes. The field design calls for a sixth treatment bed
described as a shallow grass filter 10 x 20 m. The substrate selected for the sixth bed is a coarse sand, the purpose of which will be to act as a final filter and provide substantial surface area for microbial attachment.

This sixth bed will act as a final polish for the effluent and will assist in managing sustained flushes of pollutants. In the unlikely event of clogging or short-circuiting suddenly occurring the shallow grass filter would provide a back-up to the system to prevent untreated effluent leaving the system. If insufficient Phosphorus has occurred at this point an option is the addition of chemicals that will promote the precipitation of phosphorus e.g. Al, Fe or Ca salts (Brix, 1992).

Surface et al. (1990) state substrate choice has a distinct effect on growth form for reasons not fully explained. In the pilot experiment plant growth was compared to plants not subject to leachate. Neither the gravel substrate nor the leachate appeared to have any negative effect on plant growth. The macrophytes used in the pilot experiment were Phragmites australis and Typha orientalis. Both are emergent, have extensive rhizosphere and are unaffected by landfill leachate (Surface et al. 1992; Robinson et al., 1991; Sanford et al., 1990).

The characteristics exhibited by the macrophytes selected for the pilot experiment include enhancement of aerobic/anaerobic interface (Hammer, 1992; Robinson et al., 1991; Sanford et al., 1990), creation of an increased void volume and subsequent formation of macropores (Hammer, 1992), increased area for microbial attachment and provision of a large adsorptive capacity for translocation of nutrients and radial oxygen transfer into the rhizosphere (Robinson et al., 1991). With no significant management difficulties exhibited in the pilot system (other than leakages in the start up period of Pilot 1) the original
design criteria for substrate and macrophytes should be successful in the field system.

5.4 Evaluation of Original Design

After examination of all the design concepts the rectangular configuration was considered the most practical in terms of area required and hydrodynamics, (see fig. 5.4)

Fig. 5.4 Final Design Configuration (for statistical information see Appendix 6)

- crushed rock
- dispersal gully

Primer tank containing leachate from pond

slope < 5%

Phragmites

20m

Typha

4m

Phragmites

10m

Typha

coarse sand bed planted with rye or torpedo grass
The fundamental concepts of the original design, the primary objective of disinfection, and secondary objective of nutrient removal have remained unaltered in view of the results of the pilot investigations. The physical constraints of the site and economic constraints of the site managers reduced the number of available design options.

The system will consist of small discrete units in which the hydraulic regime will be more manageable than larger capacity treatment units. Although sub-surface flow systems have proved difficult in sustaining an acceptable hydraulic regime this system is sufficiently flexible to alter primary hydraulic indicators such as matrix characteristics quite easily.

Performance indications from the pilot system suggested small alterations in design could enhance the potential of the system. Areas such as increasing the possible aeration of the leachate, maximising the contact of the effluent with aerobic/anaerobic interface and enhancing the aerobic capabilities of the system were manipulated by changes in design.

Over the period the project has been conducted the site managers have periodically altered their view on the location of the constructed wetland. Due to pressure from local environmental groups and other management difficulties at the site, council staff have decided to locate the wetland at another landfill site within their boundaries.

The new site will require some investigation into the physical and other site constraints in addition to analysis of the leachate to be treated. Council records suggest little difference in the leachate characteristics and aspect, soils, climate and the economic constraints are likely to be very similar. The existing design
may require some alteration however they should be primarily based upon capacity when the volume of leachate produced from the new site is quantified.

Basic concepts from the original design can be common to most landfill sites. When the longterm results from this constructed wetland are available some variation in design may be necessary to make longterm management strategies effective and easy to administer.

5.5 Constructed Wetland Management Practices

Hydraulic Conductivity - Bed conductivities reflect particle and pore size (Surface et al., 1991). Monitoring of hydraulic performance to ensure pre-treatment mechanisms are effective and subsequent pollutant removal capability will be optimised is essential.

Promotion of a 'plug flow' regime will be part of the ongoing management program for the constructed wetland. Plug flow will be promoted by controlling the flow of leachate from the header tank through a feeder manifold draining the leachate into the wetland.

Flow control is also useful in enhancing rhizomal growth. By decreasing the water level in each treatment bed the rhizomes can be encouraged to grow deeper down into the matrix and so increase the effective surface available for microbial attachment and growth (Robinson et al., 1991).

Nutrient and Metals Removal - Bavor et al. (1987) note periodic flushes of phosphorus from the wetland may occur. Nutrient flux can be prevented by periodic removal of sediments trapped within the treatment units.
With the very low levels of phosphorus in the North Katoomba leachate fluxes through re-release into the effluent within the treatment units should not occur in the short term. However this aspect must be addressed in the longterm management strategy. Sampling of the sediment in each bed should be carried out at least every six months, unless pollutant removal efficiency is variable, indicating sampling needs to be performed on a more frequent basis.

**Biomass Removal** - Fluxes in nutrients and metals and generation of secondary SS and BOD pollution can be a result of the decay of accumulated biomass (Bavor *et al*., 1987) in addition to fluxes within the system.

Management will require periodic removal of accumulated biomass on the surface. However, to maintain the optimum temperature within each bed accumulated biomass should remain during the winter to act as an insulator from the external extremes of cold evident in this mountain region. Temperature has a direct influence on the performance capability within the wetland (Bavor *et al*., 1987; Surface *et al*., 1991; Hammer and Knight, 1992) particularly with regard to the removal of NH$_4^+$. 

Biomass and sediment removal are the only permanent removal processes available at this point. Biomass and sediment can be buried within the landfill site and if an impermeable membrane liner is used to limit the transgression of pollutants into the saturated landfill matrix then the potential threat to the surrounding environment will be eliminated.
5.6 Assessment of existing landfill management practices and the potential integration of constructed wetlands

5.6.1 Site Selection -
Traditionally selection of sites for landfills has been based primarily on economic rather than environmental considerations. Many landfills have been used to reclaim marginal land for the use of future development. Little thought appears to have been given, in many designs, to factors such as soil type, hydrogeology of the region, or suitable management strategies to prevent generation and migration of leachate.

Baumann (1981) states the relationship between site selection and leachate production is not clear because of other influential factors such as rainfall and site management. However, Baumann (1981) concludes that flat sites produced the smallest quantities of leachate and sites located in valleys generated a high risk for localised water pollution as the water courses receiving the leachate had little or no assimilative capacity.

5.6.2 Operational Strategies -
Leachate can continue to be generated long after the closure of the landfill. This may occur either through saturation of the waste material from an external source (Sanford et al., 1990), or from the volume of water contained in the waste at the time of disposal. Mulvey (1993) and Kean and Jern (1983) comment that most industrial and putrescible landfills develop a water table with the fill due to the moist condition of the material. Up to 60% of the total waste mass is attributed to water.
Production of leachate may be minimised through correct operational procedures such as daily capping, surface flow diversion, lining of cells with an impermeable material and revegetating capped areas as quickly as possible. Efficiency of existing leachate management controls at existing landfill sites, particularly older sites, leachate control appears a consistent management difficulty. Table 5.6.2 refers to landfills sites examined for this research and type of environment in which they were located. Sensitivities were subjectively assigned based on the level of protection designated to the region or waterway for reasons of water use, aesthetics, heritage protection or tourism value.

**Table 5.6.2 Location and Description of Landfill Sites Visited**

<table>
<thead>
<tr>
<th>LOCATION</th>
<th>RECEIVING WATERS</th>
<th>SENSITIVITY OF SURROUNDING BIOME</th>
</tr>
</thead>
<tbody>
<tr>
<td>Redhill (W. Australia)</td>
<td>freshwater creek</td>
<td>medium</td>
</tr>
<tr>
<td>Rottnest Island (W. Australia)</td>
<td>ocean (Indian)</td>
<td>medium</td>
</tr>
<tr>
<td>Hamilton Island (Queensland)</td>
<td>ocean (Barrier Reef)</td>
<td>high</td>
</tr>
<tr>
<td>Marulan (New South Wales)</td>
<td>freshwater creek</td>
<td>high</td>
</tr>
<tr>
<td>(Sydney water catchment area)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Blaxland (New South Wales)</td>
<td>freshwater creek</td>
<td>high</td>
</tr>
<tr>
<td>(Hawkesbury River)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Windsor (New South Wales)</td>
<td>freshwater creek</td>
<td>high</td>
</tr>
<tr>
<td>(Hawkesbury River)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Kellyville (New South Wales)</td>
<td>freshwater creek</td>
<td>high</td>
</tr>
<tr>
<td>(Hawkesbury River)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ku-ring-gai (New South Wales)</td>
<td>freshwater creek</td>
<td>high</td>
</tr>
<tr>
<td>(Hawkesbury River)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Some sites such as Redhill in Western Australia were recent developments and had incorporated technology such as a sub surface leachate collection system that drained into a collection pond. However, from the collection pond the effluent was pumped back over the site, removed or drained off site.

Redhill has a methane gas collection system incorporated into the disposal area. By spraying the leachate back over the buried refuse it maintains the saturation of the waste material and maximises the production of methane. This is an adequate disposal method, however problems arise when the volume of leachate produced exceeds the evaporation potential and biological requirements. Disposal of excess leachate remains a problem at the Redhill site as it does for many landfills around Australia.

At other locations inspected leachate accumulated at low points on the site where it was permitted to drain untreated (except in one instance a few straw bales) into the surrounding environment. The receiving waters ranged from freshwater creeks to the marine environment. Evidence of the impact was visually evident through changes in turbidity, attached algal growth, increased biomass and staining.

5.6.3 Potential of Constructed Wetlands -

At each of the landfill sites visited, there was sufficient area to incorporate a constructed wetland to treat the leachate. As the leachate is currently impacting the surrounding environment, any form of treatment would assist in reducing the pollution potential. Sites were generally in shallow valleys, although the Rottnest Island site was situated in a sand flat surrounded by surface water. The Rottnest site would require the use of macrophyte species that are salt tolerant as all
surface water is very saline. All other sites, with the exception of Hamilton Island, were situated well inland from the coast.

Operational management standards varied considerably, from an uncovered open dump to lined cells with daily capping and gas collection. Migration of surface water onto the site appeared to be a problem not well addressed. Robinson et al. (1991) detail the introduction of rock reed filters to treat leachate as a measure to meet obligations to protect groundwater.

Sanford et al. (1990) and Surface et al. (1991) record the use of constructed wetlands to treat landfill leachate as a response to stringent regulations by both state and federal governments to reduce the contamination of ground and surface water.

Moshiri and Miller (1991) found serious environmental problems were associated with landfill sites particularly surface and ground water pollution. Legislation enforcing landfill controls require all future landfills to be lined and leachate collected and treated.

Given the need for on site treatment and the apparent success of constructed wetlands in treating landfill leachate (Surface et al., 1991; Sanford et al., 1990; Robinson et al., 1991) and the physical characteristics of the sites visited, incorporation of this type of low level biotechnology appears appropriate.

5.7 Assessment of Future Research Potential

It is apparent from the research conducted for this project that landfill leachate is a well recognised problem that has very few adequate solutions. Given that the constraints to disposal appear a common factor all over the world, the preferred treatment option must address these constraints.
Constructed wetlands have demonstrated a potential for treating a wide range of pollution problems both point source and non-point source. This low cost eco-technology has both advantages and disadvantages. The advantages have already been documented in this paper, however the disadvantages are more often than not the determining factor to decision makers.

Little data exists on the long term efficiency or management difficulties associated with constructed wetlands treating landfill leachate. It will be some years before projected management strategies are proven effective.

Another concern has been the constant question of final disposal, that is what to do with the pollutants trapped within the wetland. The view is that while the pollutants remain on site they remain a problem. However if these aged landfills could be isolated from the surrounding environment they would not be a problem, therefore it is the migration of the pollutants off site in the leachate that must be controlled.

New landfills may still require leachate management strategies even if they are well designed and located in a suitable area. This will depend upon the nature of the waste being accepted in the landfill, if the percentage of water within the waste is around the possible figure of 60% and environmental conditions do not promote losses through evaporation then constructed wetlands should be incorporated into the design.

Capital costs of new landfills incorporating these pollution prevention strategies will be much greater than in the past. However the benefit is the cost of cleaning up the pollution in the surrounding environment which should be zero providing operational practices are also improved.
In older landfills where it is not an option to extract the refuse and line the site then put the refuse back, the strategy must be to prevent the pollutants from migrating off site by treating the leachate. Placement of the contaminated substrate material in a secure section of the landfill will ensure it remains trapped permanently within the site.

Future research must address the permanent disposal options, longterm management of the wetland and provide solutions for variations in the standard of treatment caused by factors such as senescence, temperature variations and leachate character fluctuation including pH and loadings.
6.0 REFERENCES


Appendix No. 1

Wetland Design Option 1
Diamond Configuration

primer tank

closed loop system - leachate returns to collection pond

Phragmites
6m

Typha
6m

Phragmites
6m

Phragmites
6m

Phragmites
6m

30m

Shallow sand filter bed planted with rye or torpedo grass

Slope < 5%

Crushed rock dispersal area

Beds 1 - 5:
Area 6m x 6m
Depth 0.5m
Volume 18m³
Appendix 2

Wetland Design Option 2
Circular Configuration

Beds 1-5: Circular gravel treatment beds planted with macrophytes with spray dispersal draining to sump that feeds subsequent treatment beds.

Area: 28.2m²
Depth: 0.5 m
Volume: 14.1 m³
Appendix 3: Red Hill Refuse Disposal Site Statistics

1) Location and Geologic Information
The sanitary landfill site operated by the Regional Council is located on Lot 11 Toodyay Road, Red Hill. The site is part of a high plateau which has gently undulating topography with elevations of between 260 metres and 285 metres. The land is underlain by granite bedrock and granite outcrops occur on the site. On most of the site the rock has weathered to white, red and yellow gritty, kaolinitic clay to depths of 20 to 25 metres. The top three to five metres of the clay sequence is heavily laterised and this latteritic gravel has been quarried from the site.

2) Operation
The site is operated by excavation of overburden and clay which is placed in separate stockpiles. The excavation is then provided with separate systems to isolate overland water, groundwater and leachate. The overland water and groundwater is diverted to siltation ponds while the leachate is collected into leachate ponds. The ground floor and side slopes of the excavation are reworked into an impermeable lining of clay.

3) Main features of the Red Hill Refuse Site Operation
- Capacity of landfill - 4,000,000 cubic metres
- Annual Solid Waste Tonnages (noted below)

<table>
<thead>
<tr>
<th>Year</th>
<th>Eastern Regional Councils</th>
<th>Western Regional Councils</th>
<th>Totals tonnes yr⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>1991</td>
<td>82,000</td>
<td>20,000</td>
<td>102,000</td>
</tr>
<tr>
<td>1996</td>
<td>98,000</td>
<td>23,000</td>
<td>121,000</td>
</tr>
</tbody>
</table>

142
4) Weighbridge

The site has a weighbridge and weighbridge office equipped with computerised weighing, recording and invoicing facilities.

5) Site Equipment

The major items of equipment on-site are:
- CAT 826C landfill compactor equipped with Caron wheels and Caron double U blade.
- CAT 816B landfill compactor - used as backup machine and for hire out purposes
- Gallion grader
- CAT 963 Traxcavator
- Mitsubishi Tip Truck
- International 9,000 litre water truck

6) Site Personnel

The site is staffed by:
- Site manager
- Weighbridge and Administration Clerk
- Three site plant and machinery operators
- Casual staff for litter cleanup etc as required

7) Dust, Odour and Litter Control

Dust control is achieved through a water truck being available full-time on the site. Odour control is achieved by continuous operation involving careful control of compaction and cover materials. Litter control is achieved through the use of perimeter fences and paper fences which are removed continually adjacent to the current site of operation. In addition these three aspects are given further control by confirming the size of the operating face.

8) Fire Control

Fire control is achieved through the Fire Control Management Plan careful on-site operations.
9) **Compaction**

The Regional Council has recently purchased a Caterpillar 826C compactor equipped with Caron Wheel and Caron double U blade with the intention of increasing compaction densities from the 850 - 900kg per cubic metre achieved by the previously used CAT 816B landfill compactor to levels in the region of 1050 - 1000 kg per cubic metre. It is anticipated that this will have the affect of increasing the landfill site life in the vicinity of 20% creating immediate saving in that less landfill space will need to be provided annually and long term savings by the preservation of the cheaper option of landfill disposal.

9) **Daily Cover**

Compacted waste is covered daily using stockpiled clay or overburden depending upon weather conditions.

10) **Final Cover**

Currently it is anticipated that a final cover of 300-450 mm of clay will be required depending upon the final vegetation type required. In addition a growing medium depending on the type of final vegetation will also be required.

11) **Surface Water Control**

One of the main advantages of the site is its location with respect to surface water (and groundwater catchments) it is located high on the catchment and runoff is generated entirely by incident rainfall, there is no through flow from adjacent and upstream catchment areas. Run off control includes the use of drains at the base of slopes, retention basins and perimeter drains. Comprehensive use is made of run off control structures during operation of successive stages to preclude the intrusion of groundwater into the operational site which would subsequently increase leachate control problems.
12) Groundwater Control

Groundwater is precluded from intruding into the landfill zone through a series of trenches and sub-surface drains which intercepts any groundwater flow.

13) Leachate Control

Formation of leachate is minimised by good compaction and coverage of wastes, grading of the final surface of the site, provision of an impenetrable capping layer, elimination of any ponding at the working face, operating a small working face, as well as the diversion of all surface and groundwaters through or around the site.

A leachate collection system has been incorporated with a network of perforated pipes and filters collecting leachate from the bottom of the excavated cells and taking it to a retention basin. Leachate is prevented from mixing with and contaminating freshwater and/or groundwater.

The leachate is disposed of by recirculating to the working face and allowing it to infiltrate into the fill. A significant proportion is also lost by evaporation from the working face and allowing it to infiltrate into the fill and retention basin.

14) Landfill Gas Control

The escape of landfill gas from the site is controlled by a final cover of 350-450 mm clay material. However the major control will be facilitated through an infrastructure which will extract gas from the site be used for generation of electricity which will be sold the the State Electricity Commission through the power grid.

Recently a final contract has been exchanged with the company, Landfill Gas Pty Ltd., which has been granted the lease of a small area of land for the
construction of a power station and granted a license to extract gas from the landfill site. This agreement has followed some years of intensive testing and negotiation including the right of the Regional Council to retain the use of methane as might be necessary for on site purposes.

Currently the power generation plant is being installed and it is anticipated that power generation will commence in June 1993. It is anticipated the power generated will range from two to six megawatts depending upon the state of development of the site, The Regional Council will receive a royalty based on the amount of power generated.

15) Final Landform, Cover and Surface Material
The final landform of the site has been designed following a topographic slope analysis of the area to project a shape which will relate strongly to the landforms in the vicinity of the landfill site. The composition of the final cover and surface material is currently being investigated.

16) End Use
The objective of rehabilitation is to blend the finished landfill contours into the existing landscape. The process of rehabilitation will be directed toward a grass/shrub cover that is relatively easy to maintain.

Concepts relating to the provision of recreational facilities, parkland and inspection of landfill gas facilities have been investigated and will be further considered as the site develops and a detailed plan for a final site is prepared. At this stage it is the E.M.R.C’s intention that the completed site will be available for public recreational use, and it will be finished to a standard that enhances the appearance and utility of the area, bearing in mind that it will probably eventually be adjoining a National Highway.
Appendix 4 Yosemite Creek Sampling Results

<table>
<thead>
<tr>
<th>Location</th>
<th>pH</th>
<th>Temp °C</th>
<th>Cond mscm⁻¹</th>
<th>Turbidity ntu</th>
<th>NH₄⁺ mg l⁻¹</th>
<th>PO₄⁻ mg l⁻¹</th>
<th>NO₃ mg l⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Upstream</td>
<td>6.9</td>
<td>8.4</td>
<td>0.11</td>
<td>6.2</td>
<td>&lt;0.5</td>
<td>0.04</td>
<td>0.5</td>
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<tr>
<td>Downstream</td>
<td>7.1</td>
<td>7.2</td>
<td>0.05</td>
<td>4.9</td>
<td>&lt;0.5</td>
<td>0.06</td>
<td>2.1</td>
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Appendix 5 Correlation Calculations
### Appendix 5 Correlation Calculations

#### CORRELATION OF TEMPERATURE V. COLIFORMS

<table>
<thead>
<tr>
<th><strong>x</strong></th>
<th><strong>x - x</strong></th>
<th><strong>(x - x)^2</strong></th>
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<tr>
<td>10.2</td>
<td>-2.62</td>
<td>6.86</td>
</tr>
<tr>
<td>10.5</td>
<td>-2.32</td>
<td>5.38</td>
</tr>
<tr>
<td>10.2</td>
<td>-2.62</td>
<td>6.86</td>
</tr>
<tr>
<td>11.5</td>
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<td>23</td>
<td>10.18</td>
<td>103.6</td>
</tr>
<tr>
<td>20</td>
<td>7.18</td>
<td>51.5</td>
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<tr>
<td>14</td>
<td>1.18</td>
<td>1.39</td>
</tr>
<tr>
<td>7</td>
<td>-5.82</td>
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<td>14.5</td>
</tr>
<tr>
<td><strong>115.4</strong></td>
<td></td>
<td><strong>225.63</strong></td>
</tr>
<tr>
<td><strong>x = 72.82</strong></td>
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</table>

<table>
<thead>
<tr>
<th><strong>x</strong></th>
<th><strong>x^2</strong></th>
<th><strong>xy</strong></th>
<th><strong>y</strong></th>
<th><strong>y^2</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>8.5</td>
<td>72.25</td>
<td>7.650</td>
<td>900</td>
<td>810.00</td>
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<tr>
<td>7.8</td>
<td>60.84</td>
<td>15.6</td>
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<td>4</td>
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<tr>
<td>16</td>
<td>256</td>
<td>432.00</td>
<td>2.7x10^4</td>
<td>7.29x10^4</td>
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<tr>
<td>28</td>
<td>625</td>
<td>675.00</td>
<td>2.7x10^4</td>
<td>7.29x10^4</td>
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<td>16</td>
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<td>6</td>
<td>36</td>
<td>4.800</td>
<td>8x10(^2)</td>
<td>640.000</td>
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<td><strong>x = 13.21</strong></td>
<td></td>
<td><strong>1.128.585.6</strong></td>
<td><strong>y = 9.378.66</strong></td>
<td><strong>1.459.774.904</strong></td>
</tr>
</tbody>
</table>

\[ r = \frac{1}{n} \Sigma xy - xy \div SxSy \]

\[ Sx = \sqrt{\Sigma x^2 - x^2 + n} \]

\[ Sy = \sqrt{\Sigma y^2 - y^2 + n} \]

\[ r = \frac{1}{6} \times 1128585.6 - 13.21 \times 9378.66 + (6.54 \times 12463.40) \]

\[ r = 0.84 \]

This result demonstrates a strong correlation between f. coliforms and temperature.
Appendix 6

NORTH KATOOMBA LANDFILL SITE WETLAND DESIGN STATISTICS
RECTANGULAR CONFIGURATION

A: Total land area required: 20.5 x 20m = 410m²

B: Area of macrophyte beds 1 - 5:
Area of a single bed: 10 x 4m = 40m²
Area of 5 beds: 40m² x 5 = 200m²

C: Volume of macrophyte beds 1 - 5:
Volume of a single bed: 4 x 10 x 0.5 = 20m³
Volume of 5 beds: 20m³ x 5 = 100m³

D: Capacity of beds 1 - 5: (note: this may vary according to the type and size of substrate used, however these figures may be assumed to present a reasonable estimate - figures are based on allowing a 50% reduction in volume for the substrate, ie. 500l m⁻³⁻¹).
Capacity of a single bed = 20 + 2 x 1,000 = 10,000l
Capacity of five beds: = 10,000 x 5 = 50,000l

E: Dimensions of shallow grass filter bed: (No.6)
Area: 10 x 20m = 200m²
Volume: 10 x 20 x 0.2 = 40m³
Capacity: 40 + 2 x 1,000 = 20,000l

F: Retention period: (based on 1990 flow estimates)
Wet flow volume: 8,640l d⁻¹
Retention periods in beds 1 - 5 combined 50,000 ÷ 8,640 = 5.7 (days)
Dry flow volume: 2,880 l d⁻¹
Retention period in beds 1 - 5 combined 50,000 ÷ 2,880 = 17.4 (days)
Shallow grass filter bed: (6)
Wet flow: 20,000 ÷ 8,640 = 2.3 (days)
Dry flow: 20,000 ÷ 2,880 = 6.9 (days)

G: Total Retention period in system:
Wet flow: 5.7 + 2.3 = 8 days
Dry flow: 17.4 + 6.9 = 24.3 days
DESIGNING A CONSTRUCTED WETLAND TO TREAT LANDFILL LEACHATE

by Jennifer E. Scott
B.App. Science (Env. Health)

A thesis submitted in partial fulfilment of the requirements for the degree of Master of Science (Hons)

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AUTHORS STATEMENT

I declare that the work presented in this thesis is, to the best of my knowledge and belief, original, except where acknowledged in the text, and that the material has not been submitted, either in part or whole, for a degree at any other institution.

Jennifer E Scott

1st November, 1994