LEACHATE

TREATMENT OPTIONS

FOR CLOSED

LANDFILL'S

Dissertation presented in part fulfilment of HETAC requirements for the Degree of Master of Science in Environmental Protection

By

Dearn Mc Clintock
Institute of Technology, Sligo.
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Supervised by
Dr Billy Fitzgerald

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Abstract

For many years up until the 1990's uncontrolled disposal of Municipal Solids Waste at various communal dumps occurred in Ireland. Since the establishment of the Environmental Protection Agency and the implementation of the European Union Landfill Directive these dumps have either been closed or regulated.

This study sets out the legal and operational implications for local authorities to satisfactorily rehabilitate such disused landfill.

Some local authorities have been left with a legacy of rehabilitating such landfills. The landfill site at Muckish in Co. Donegal is an example of one such site. It is a 2.6 ha site, which closed in 2001 with 56,000 tonnes of disposed MSW. It is partially capped but has no containment system for leachate or any means of treating this leachate.

This study examines the various options available in Ireland, Europe or the United States of America for the treatment of such a leachate. Having established that the use of constructed wetlands was a method successfully used to treat farm wastewaters in Ireland and also used to treat landfill leachates in Europe and the USA this study compares constructed wetlands against other options such as trucking the waste off-site to a wastewater treatment facility or the on-site treatment using a variety of technologies. One such alternative on-site technology is a purpose built filtration system used to treat landfill leachate at another Donegal County Council landfill at Drumaboden. This system has shown a degree of success in treating landfill leachate from this closed landfill and as such it became a basis of comparison for constructed wetlands.

While there are case studies that could be drawn upon to show the ability of constructed wetlands to treat landfill leachate the US EPA have published cautionary comments as to the ability of constructed wetlands to be successful in treating high strength effluents, particularly those which a high nitrogen levels such as encountered in landfill leachates.

The conclusion to the study was that while off-site treatment is a solution the use of a filtration system certainly is a viable alternative. The use of a constructed wetland should be considered however, but its ability to be successful must be measured against the lack of verifiable data to satisfactorily treat landfill leachate.

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Introduction.

Background

The presence of disused and closed landfills in the Irish landscape is a legacy that has been left in many cases for rehabilitation by today's generation of local authorities. Many of the old landfills sites in Ireland did not have containment systems for collecting landfill leachate and the consequence of this was that it seeped into the soil and ultimately could pollute both ground and surface waters. One such closed landfill with surface waters being polluted by landfill leachate is at Muckish outside Falcarragh in Co. Donegal. The effects of this were to reduce the Q rating of the receiving waters in the Ray River from Q5 to Q3 as stated in the EPA Intermin report on Biological Water Quality Report.

The Environmental Protection Agency is obliged to prosecute Local Authorities if they are found to be responsible for pollution from landfills under their control. In this case Donegal County Council were found guilty at Letterkenny District Court on the 24th January 2002 of a number of charges relating to operation of Muckish landfill brought by the EPA. Local Authorities will also have obligations to rehabilitate and restore landfills that have been licensed by the EPA prior to 2001, as required under the Landfill Directive.

Aims of the Dissertation

This dissertation examines the best practice for treating the leachate from old landfills, the leachate from which is a serious potential pollutant threat to surface and ground waters in County Donegal.

Looking to the future, such best practice can be considered for the treatment of leachate from proposed new landfills in remote areas. It is proposed to identify the best option for the treatment of landfill leachate from the site at Muckish and a proposed landfill in Meenabol. The practices used in another closed Co. Donegal landfill site at Drumaboden are compared with other treatment options in Ireland, Europe and the United States of America.

Objectives of the Dissertation.

The aim of this dissertation is to examine the potential ability of constructed wetlands to treat landfill leachate and extrapolate from the evidence available whether such a system would be suitable in the situation that prevails in Co. Donegal.

These aims will be achieved by:

1. Identifying the characteristics of landfill leachate, its composition and the factors that influence its quantity and quality.
2. Examining published best practice and also by communicating with authorities or persons responsible for on-site treatment of leachate.
3. Comparing and contrasting the performance and costs of the various treatment options that might be suitable in the treatment of landfill leachate, while looking at specific working model in Co. Donegal and elsewhere as a benchmark in order to make a comparison.
4. Determining whether a constructed wetland is a viable option in treating landfill leachate in the specific locations in Co. Donegal and if not to propose viable alternatives

CHAPTER 2
LITERATURE REVIEW

2.1. Introduction

At present in Ireland the landfilling of domestic and commercial waste is the most common method of disposal accounting for 90% (EPA, 2000). While the extent to which Ireland is dependant upon landfill is very high, other countries are also dependant upon the use of landfill, for example in the USA their EPA estimated that by the year 2000 49% of their municipal solid waste (MSW) will be landfilled, (Qasim and Chiang,1993).

The use of landfilling of waste is a debate that is on-going in Ireland and the rest of the world and while steps are taken to minimise and recycle waste by various methods, in Ireland legislation has been enacted to prohibit various recyclable fractions from landfill, as provided by the Waste Management (Packaging) Regulations, 2003. This regulation introduced under the Waste Management Act, 1996 is in response to the commitment of the Irish state to the EU Directive, whereby 50% of packaging waste should be recovered by the year 2005.

In September 1998 the Irish Government set targets in the *Changing Our Ways* policy statement that over the next 15 years, there will be a diversion of 50% of overall household waste from landfill, a minimum of a 65% reduction in biodegradable wastes from landfill and the recycling of 35% of MSW. Specifically with regard to the numbers of landfills the target was set at an integrated network of 20 state of the art facilities incorporating energy recovery and high standards of environmental protection. In the *National Waste Database Interim Report 2002,* published by the EPA, they state that the packaging waste recovery rate was 33%, while municipal waste recycling rate was 20.7%, and the household recycling rate is 9.3%. In general, the rate of increase in the production of MSW has slowed down, while volume of municipal waste being landfilling has decreased.

Alternative methods for the disposal of the remaining MSW have been suggested, including thermal options such as incineration.

In Ireland it is inevitable that a proportion of waste will still have to be landfilled. The consequence of this is that as the waste decomposes in the ground, it will have the capacity to pollute both ground and surface waters through the leachate produced. Where leachate can be collected various treatment methods can be applied either on-site or off-site. These can include conventional biological wastewater treatment using activated sludge, fixed film biological treatment/filtration and constructed wetlands. Consequently it will be necessary to describe in the literary review what is a landfill leachate, how it is normally generated, controlled and treated as well as describing the alternative methods suggested.

2.2 Legislative Background

The following is a list of the legislation relevant to the management of landfills and the discharge of waters that occurs as a result of its activities.

1. Council Directive 1999/31/EC, of 26 April 1999 on the landfill of waste.
2. Council Decision of establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 and Annex II of Directive 1999/31/EC.
3. Council Directive 80/68/EEC on the protection of groundwater against pollution caused by certain dangerous substances.
4. The Local Government (Water Pollution) Acts, 1977 and 1990.
5. The Environmental Protection Agency Act, 1992
6. The Waste Management Act, 1996.
7. Planning and Development Act, 2000.
8. Waste Management (Planning) Regulations, 1997.
9. Waste Management (Register) Regulations, 1997.
10. The Environmental Protection Agency (Licensing) Regulations, 1994.
11. The Protection of the Environment Act, 2003
12. Waste Management (Licensing) Regulations 2004

The general implications of the above legislation are that all landfills must operate under a waste licence issued by the EPA. Under the conditions of this licence, a landfill will only be allowed to handle wastes that are specific to that licence; in general the landfills are categorised as inert, non-hazardous and hazardous landfills.

The developer and the EPA must agree the criteria for the design, construction and operation of a landfill. The EPA ensures compliance with the licence conditions.
The operation of the landfill must ensure that any discharge, particularly in the case of leachate, does not pollute any receiving waters.
Other legislation that has an impact on the decision-making regarding the siting, management or operation of a landfill include:

1. Ramsar Convention, The Convention on Wetlands of International Importance especially as Waterfowl Habitat.
2. The Birds Directive, Council Directive 79/409/EC, on the conservation of wild birds.
3. The Habitats Directive, Council Directive 92/43/EC on the conservation of natural habitats and of wolf fauna and flora.
4. The Wildlife Act, 1976
5. The Wildlife (Amendment) Act, 2000.
6. The European Communities (Natural Habitats) Regulations, 1997.
7. The Planning and Development Act, 2000.
8. Environmental Impact Assessment Regulations.

In the list of legislation above the decision making process as to where a landfill can be located can be regulated. Of particular concern are ecologically sensitive areas, which are protected and this can be done primarily by the use of an environmental impact study, which will be necessary under any planning application for a new development, as well as a waste licence application to the EPA.
More specifically ecologically sensitive areas can be protected by The European Communities (Natural Habitats) Regulation, 1997 which provide for the designation of Special Areas of Conservation (SAC's) and for the protection measures that apply to Special Protection Area's (SPA's) and SAC's. According to Gerald Clabby in 'Wetlands of Ireland' (2003) the Wildlife (Amendment) Act, 2000 provides the legal basis for the establishment and protection of a national network of sites known as National Heritage Area's (NHAs). This Act provides a system for designing NHAs and regulating activities within them, and also for the restoration of sites, which have been damaged illegally. In February 2003, 81 sites were in the process of becoming designated. NHAs will become the basis for nature conservation designation in Ireland

and many sites with other important designations (such as SACs, SPAs, National Parks and Nature Reserves) will also be designated NHAs. This will have a very significant impact on the choice of location of future landfills.

Waste operations, including collection and transportation are regulated under the following legislation:

1. Council Directive, 75/442/EEC of 15 July 1975, as amended by Council Directive 91/156/EEC of 18 March 1991.
2. The Waste Management Act, 1996.
3. Waste Management (Permit) Regulations, 1998
4. Waste Management (Transfrontier Shipment of Waste) Regulations, 1998
5. Waste Management (Collection Permit) Regulations, 2001.
6. Waste Management (Packaging) Regulations, 2003.

The above legislation controls what waste can be disposed of in various landfills as well as ensuring how it can be transported across international frontiers.

Legislation also regulates who is authorised to collect waste and how such operations are regulated to ensure compliance. In particular the collection and transportation of waste this will enforced by the local authorities under sections 32, 34 and 39 of the Waste Management Act, 1996. Section 32 provides for the general obligations of a holder of a waste in that they cannot cause environmental pollution and they can only transfer control of that waste to an authorised person. Section 34 allows for local authorities to regulate the collection of waste by issuing waste collection permits; these permit holders become authorised persons to collect waste. Finally, waste must be taken to facilities that comply with section 39 of the 1996 Act; a local authority can permit a facility under certain conditions, mainly if it a non-disposal facility of less than 5000 tonnes per annum. All other waste disposal activities require a waste licence issued by the EPA.

Regulations issued under the relevant sections of the1996 Act are used to ensure that waste permits and waste collection permits are issued and enforced by the local authorities. These regulations are the Waste Management (Permit) Regulations 1998 and the Waste Management (Collection Permit) Regulations, 2001 respectively. The enforcement of the regulations allows the local authorities to monitor and control the activities of waste operations and give a degree of assurance that such opertions are carried out in a manner that does not cause environmental pollution. To allow for the same degree of assurance that waste arising in the Irish Republic is not moved out of the state and cause pollution in other countries the Waste Management (Transfrontier Shipment of Waste) Regulations 1998 transpose the Council Regulation 259/93 Transfrontier Shipment of waste into Irish legislation. This regulation controls the movement of waste and subject to conditions prohibits the movement of certain wastes for disposal.

Finally, legislation is used to ensure that Ireland fulfils its obligations in the Landfill Directive 99/31/EC under article 5 (2) as to the targets that must be obtained to reduce the amount of biodegradable waste, specifically the Waste Management (Packaging) Regulations, 2003 forbid the land filling of specified packaging wastes.
Under article 5 of the Landfill Directive each Member State should oversee a national strategy on Biodegradable Waste, which will set out measures to progressivelly, divert biodegradable municipal waste away from landfill with agreed targets over a 15 year period ending July 2016. Using 1995 as the base year these targets are as follows:

1. A reduction to 65% of the biodegradable municipal waste by weight by 2006
2. A reduction to 50% of the biodegradable municipal waste by weight by 2009
3. A reduction to 35% of the biodegradable municipal waste by weight by 2016.

In Ireland the targets that were set took into account the position of Irelands past performance in prevention and recovery of waste. Ireland was allowed derogation and the progress towards these Irish targets is included in the following table.

Table 2.1 Progress towards national targets for the management of municipal waste

(Source EPA National Waste Database Report 2001)

Target **(Set in 1998, to be achieved by 2013)**	**Position in 2001**
A diversion of 50% of overall household waste away from landfill	In 1998, 3.2% of household waste was recovered. In 2001, 5.6% was recovered.
A minimum 65% reduction in biodegradable wastes consigned to landfill [on a phased basis to meet the requirements of the Landfill Directive 1999/31/EC]	In 1998, 1,039,195 tonnes of organic waste (excluding wood) were landfilled. In 2001, 1,250,048 tonnes (excluding wood) were landfilled; a quantitative increase of 20.3% between 1998 and 2001
Recycling of 35% of municipal waste	In 1998, 9% of municipal waste was recovered. In 2001, 13.3% was recovered.

2.3 Municipal Solids Waste (MSW)

Under the Waste Management Act,1996 MSW is defined as household waste as well as commercial and other waste, which because of its nature and composition is similar to household waste, and waste from commercial or industrial sources similar to household waste. There was a total of 74,071,634 tonnes of waste produced in Ireland in 2001. According to the EPA National Waste Database Report 2001 municipal waste represents 4 %, or 2,704,035 tonnes, while the largest fraction of wastes is agricultural wastes at 76 % or 56,687,400 tonnes.

Table 2.2 National Waste Generation, principal sources in 2001

(Source EPA National Waste Database Report 2001)

Waste Type	Percentage
Agricultural Waste	76
Manufacturing Waste	6
Construction and Demolition Waste	5
Mining and quarrying	5
Municipal Waste	4
Dredge spoils	2
Others	2

Table 2.3 Composition of household and commercial wastes

(Source EPA National Waste Database Report 2001)

Product type	Weight component, % Household	Weight component, % Commerical
Paper and cardboard	22	49
Glass	4	7
Metals	4	3
Plastics	12	10
Textiles	4	1
Organics	32	21
Other	22	9
Total	100	100

In Ireland an approximate figure of 1 tonne of waste per household per year is generated or 1.2 – 2 kg per capita per day has been used as a guide.
Typical moisture content is 15 to 40%
Typical density depends upon compaction with 150kg/m^3 when un-compacted, while 235 – 350 kg/m^3 for compacted waste.
Typical energy levels for collected MSW are 9.890 kJ/kg (Qasim and Chiang 1994).

Collection, transportation, storage and disposal of MSW, typically this involves the collection of MSW by truck with the transport and delivery to a waste transfer station or disposal point. According to Qasim and Chiang (1994), collection of solid waste typically consumes 60 – 80 % of the solid waste budget of a community therefore any improvement in the collection system can reduce overall costs.
Where distances of greater than 30 miles are involved for the transportation of collected waste, Henry and Heinke (1989) suggest that waste transfer stations should be used to reduce inefficiencies. Waste transfer stations are established whereby the collected MSW is off loaded and bulked up onto a larger more efficient vehicle. This bulking up maybe by compaction, baling and reloading the waste onto a purpose built trailer or alternatively by using a bucket loader or grab to fill a 20 tonne bulker trailer. Whatever method is used to carry out this function the purpose is to take the slow and fuel- inefficient compactor off the road and replace it with a vehicle that will carry out the round trip more efficiently.

The enforcement of waste transfer stations in Ireland is in the first instance by either a waste license or certificate of registration by the EPA if more than 5000 tonnes of disposable waste or more than 1000 m^3 of compostable waste are being handled annually or there is hazardous waste being handled or is being operated by a local authority.
Otherwise under the Waste Management Act 1996 and the Waste Management (Permit) Regulations, 1997 the local authorities may grant a waste permit.

Commercial businesses in Ireland are legally obliged to segregate their packaging waste streams at source. The collection and segregation of these fractions of MSW is facilitated by the introduction of kerbside collection and civic amenity sites. Incineration of MSW and recovery of energy both in the form of electricity and space heating can be achieved by direct incineration, pyrolysis and refuse derived fuels. Disposal by landfilling is at the end of the process of recycling, recovery and reuse when there is inevitably a residual fraction that must be dealt with. This fraction may consist of an inorganic ash from the incineration process, or it maybe a putrisible organic fraction. When the organic fraction is buried in the ground at the right moisture levels, it will produce a leachate that must be treated.

2.4 Factors influencing landfill leachate production and composition

As MSW decomposes in the ground the water that passes through it becomes highly contaminated. This product is called leachate, which was shown by Walker (1969) and Kelly (1976) that many contaminants released from a landfill and allowed to migrate could pollute surface and ground waters. Landfill leachate is essentially a high-strength wastewater, and according to Qasim and Chiang (1994) is characterised by low pH, high biochemical oxygen demand (BOD), chemical oxygen demand (COD), and by the presence of toxic chemicals.

These contaminants may include organic fractions that will deplete the oxygen levels of the waters contaminated and also toxic elements that will be harmful to humans, animals and plant life.

The constituents of leachate are highly variable and can be affected by the following factors as discussed by Lu *et al.* (1981, 1984, 1985), Chiang and Dewalle (1976) and Chiang (1977).

Processed Refuse whether the landfilled waste is shredded or baled prior to burial will effect the concentration of pollutants released. Experiments have been conducted by Fungaroli and Steiner (1979), Kemper and Smith (1981) with shredded waste and baled waste. These experiments have shown that shredded waste produced a more concentrated leachate early in the life time of the landfill but stabilized more quickly than in the case of baled refuse. The baled waste produced a more dilute leachate while taking longer to stabilize, in overall terms the total cumulative mass of pollutant removal per kg of solid waste will be the same regardless of waste processing Lu *et al*. (1984)

Depth of Refuse to which the waste is buried has been shown by Qasim and Burchinal (1970*a* and 1970*b*) to affect the concentration of the leachate, with deeper fills producing a more concentrated leachate. However, deeper fills require more water to reach saturation, require longer time for decomposition and distribute the bulk of the extracted material over a longer period of time.

Co-disposal of MSW with sewage sludge, municipal wastewater treatment sludges, or septage, has been researched by Stone (1974), Emcon Associates (1974), Pohland (1975), Lu *et al.*(1984), and Levine and Rear (1989). From this research it has been determined that the co-disposal of septage and MSW has a significant effect upon the generation and quality of leachate. The effect of additional moisture, microbial seeding and extra nutrients increases leachate generation and waste stabilization. The chemical composition and BOD levels of the leachate will be changed, but the most significant increase will be in the nitrate and enteric pathogen levels.

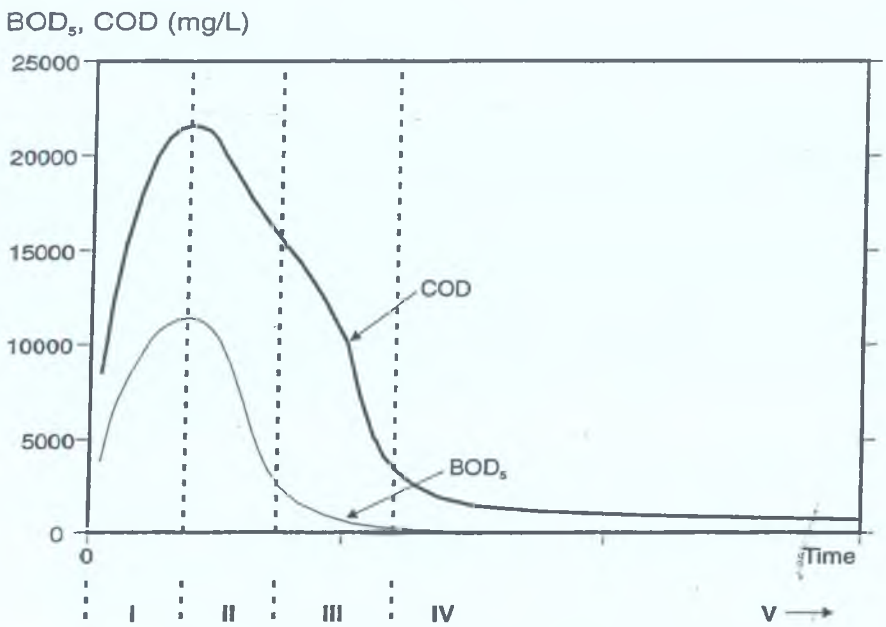
Co-disposal with hazardous wastes from research carried out by Pohland *et al.* (1990) it would indicate that co-disposal of hazardous and MSW will effect the stabilization of the landfill, particularly in the case of heavy metal hazardous wastes. Organic hazardous wastes may have some minor influence on the landfill stabilization making them more resistant to attenuation.

The management of the landfill with regard to gas and leachate control will influence the containment, collection, utilization and recycling of leachate which will in turn influence the mobilization and release of the organic or inorganic hazardous wastes from the landfill via the leachate.

Co-disposal with sorbtive wastes where sorbitive materials such as ash both incinerator and fly, kiln dust or limestone when mixed with MSW will effect the quality of the leachate. This is based on research carried out by Liskowitz *et al.* (1976), Fuller (1978) and Chen and Eichenberger (1993). The results of these studies have shown that there is a reduction in the mobility of many of the hazardous constituents of leachate. The cause of this can be due to adsorption and sorption of metallic ions, formation of less soluble calcium and carbonate compounds or an increase in pH resulting in the precipitation of metals.

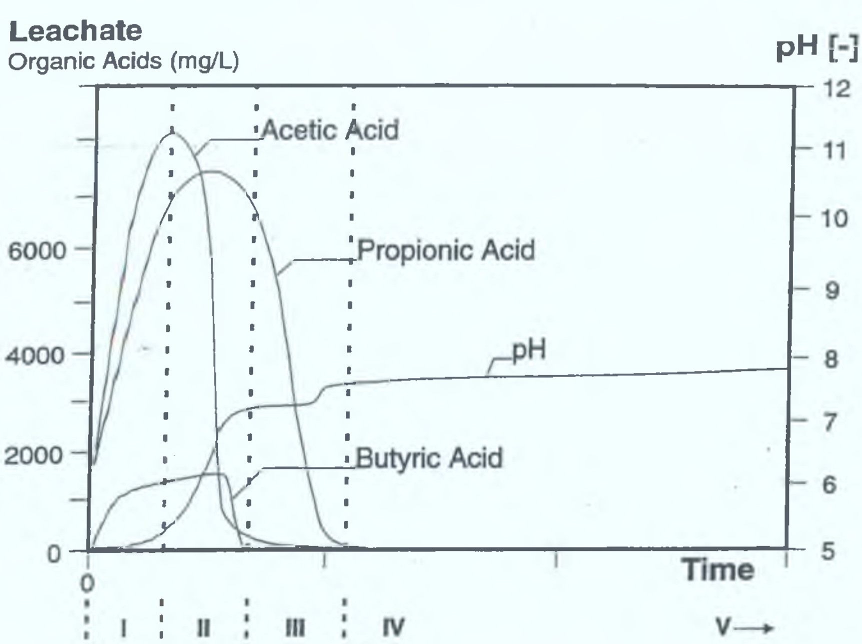
Age of fill according to Qasim and Chiang (1994) variation of leachate quality with age is to be expected, because organic matter will continue to undergo stabilization. It should be noted that the release of constituents from solid waste is obviously governed by the decomposition processes and the rate of water infiltrating through the fill. Age is a convenient means of extraction of pollutants from the refuse bed in that as the time goes by the potentially pollutants buried in the ground are removed by the decomposition process and eventually leave behind an inert material. As a result, many studies describe leachate quality as a function of time. (Figures 2.1 and 2.2Qasim and Chiang 1994).

Figure 2.1 Leachate BOD and COD reduction over time (Quasim and Chiang, 1994)



I- Hydrolysis and Acidification Phase
II- Initial Methane Generation Phase
III- Stable Methane Phase
IV- Slow-down Phase
V- Final Storage Phase

Figure 2.2 Change of concentrations of organic acids in leachate with time (Quasim and Chiang, 1994)



I- Hydrolysis and Acidification Phase
II- Initial Methane Generation Phase
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V- Final Storage Phase

Lu *et al.* (1984) have conducted research into the constituents of leachate and how their concentration and composition are affected with age. They have concluded that pollutant concentrations tend to peak within the first 2 –3 years, followed by gradual decline, particularly in regard to BOD, COD, TOC and microbiological population.(Figure 2.1)

Other constituents such as iron, zinc, phosphate, chloride, sodium, copper, organic nitrogen, total solids and suspended solids exhibit steady decreases in concentration over 3 to 5 years.

Heavy metal concentration may fluctuate because of the effects of precipitation, dissolution, adsorption, absorption and complexation mechanisms.

With regard to organic compounds, research has shown that they can be divided into 3 divisions, first fatty acids of low molecular weight, which may be up to 90% of soluble organic carbon in unstabilized landfills, such acids may be acetic, propionic and butyric. Secondly, humic carbohydrate-like substances of intermediate molecular weight. Finally fulvic-like substances of intermediate molecular weight, the second largest fraction and may consist of carboxyl and aromatic hydroxyl groups. An increase in the proportion of fulvic-like substances indicates that the landfill is aging. (Figure 2.2.)

Pathogenic bacteria and viruses have been detected in fresh leachate, however these populations are deactivated with age, due to an increase in adverse conditions within the landfill, such as increased temperature and persistently low pH.

Factors influencing changes in leachate decompostion and subsequent methanogenises

Water Balance refers not only the fact that the composition of leachate varies but so also does its volume, which according to Bromley *et a.l* (1986), is directly related to the amount of water entering the site. The establishment of the correct amount of water entering the waste is critical as for the process of decomposition of the waste to occur which will result in methanogensis, water must be present. However too much water enrtering the site will result in too high a volume of leachate leaving the site which will require extra hydraulic loading capacity in the final treatment.
Some of this will be from precipitation falling directly on the site, or surface or ground water that has not been deflected. Some will be in the domestic or commercial waste when it arrives. For co-disposal, extra water may come from liquid wastes. Water normally leaves the site as leachate, or through evapotranspiration. Evapotranspiration is the moisture levels that are reduced either by plant life take up for their growth. Transpiration occurs when plants growing on the capped landfill site take up moisture, while evaporation is due to the movement of moisture into the surrounding atmosphere. If the balance between inflow and outflow is not controlled correctly, some overflow may occur.Water balance can be influenced by the capacity of refuse to retain water, which is ability of the refuse to be absorbent and adsorbent.

Field capacity is the amount of water, which a site will absorb before leachate, appears at the base, this is directly linked to the level of compaction of the waste. The field capacity is reduced with increasing density, 12% if refuse is compacted to 0.7 tonnes per m^3 reducing to 8% when compacted to 0.95 tonnes per m^3. The more compact a landfill site is, the slower it will absorb moisture; consequently the slower it will be to begin the process of decomposition.

Decomposition of landfill wastes according to Qasim and Chiang (1994) is where the solid wastes undergo a number of simultaneous biological, physical and chemical changes. The water moves through the fill and carries with it extractable chemicals. The decomposition, stabilization and extraction of pollutants from a landfill depend upon several factors: composition of the wastes, degree of compaction, amount of moisture present, presence of inhibiting materials, rate of water movement and temperature. This leads on to the decomposition of landfill wastes, which is dependant upon microbiological processes at work in the fill.

In modern landfills the largest fraction is paper and cardboard followed by putrescibles (Bromley *et al.* 1986). These components contain a high proportion of carbohydrates in the form of cellulose, which are broken down to inorganic acids and gas. The necessary conditions for the process of decomposition to begin is when the landfill reaches its field capacity, which is when it becomes saturated with a constant water input.

Research by Qasim (1965), Qasim and Burchinal (1970a), Brunner and Keller (1972), Pfeffer (1992) and Tchobangoglous *et al.* (1993) reports that decomposition occurs in a number of stages, as shown in Figures 2.1 and 2.2; however the presence or absence of oxygen is critical to these phases occurring. The presence of oxygen in turn is interlinked with the height of the water levels in the landfill, as the ability of microbological process to work is in turn influenced by moisture levels. In landfills that have excessively high levels of water the process of methanogenises is inhibited, thereby slowing down the decomposition process.

Aerobic decomposition predominates initially, identified as hydrolysis and acidification phase in Figures 2.1 and 2.2. This is short lived as available oxygen is quickly used up, this phase is characterised by the generation of a large amount of heat and the leachate from this period will be expected to dissolve highly soluble salts such as NaCl. Anaerobic decomposition caused by facultative anaerobes now occurs, and produces large amounts of volatile fatty acids such as acetic acid as well as carbon dioxide. This will lower the pH of the leachate, which will help to solubize inorganic materials as well as producing a leachate with a high COD.

The beginning of the anaerobic phase can be identified as the initial methane generation phase in Figures 2.1 and 2.2. At this point a population of methane producing bacteria is built up, as described by Gaudy and Gaudy (1988). The volatile fatty acids produced by the facultative anaerobes along with other organic matter are converted to methane, which is known as methanogenises and reduces the levels of volatile fatty acids, thereby increasing the pH to neutral, which is also a requirement for methane production. At near neutral pH fewer inorganic materials are solubilized and conductivity falls although some materials continue to solubilize as the decomposition process continues.

Anaerobic decomposition slow down as the landfill ages and the rate of bacterial decomposition decreases due to substrate depletion. Identified as stable methane phase and slow-down phase in Figures 2.1 and 2.2. The landfill may re-establish aerobic conditions as oxygenated water infiltrates, however the decomposition of the landfill may continue for may years to come, as long as some organic material is available for microbial activity (Qasim and Chiang 1994). This results in the final storage phase in Figures 2.1 and 2.2.

The rate of decomposition depends upon many factors including water movement, pH, temperature, degree of compaction, age of fill, and composition of solid wastes. As degradation occurs, the volume of the original solid waste is reduced, in effect allowing greater penetration by rain in some cases. In particular, decomposition under relatively dry conditions stops, and materials can remain unaltered for decades. Research on landfill leachates has been conducted by Stanforth *et al.* (1979) who have produced generalised degradation curves. These curves express pH, oxygen, carbon dioxide, methane, acetic acid, solubilized salts, and redox potential. (Figures 2.1 and 2.2).

2.5 Leachate Attenuation.

Most landfills were designed and built with the idea that the leachate would be attenuated (purified) by the natural soil beneath the landfill: thus groundwater contamination would not arise (Qasim and Chiang 1994). Most soils do have attenuation properties and are capable of purifying leachate to a certain degree. However, in recent years, studies have shown that even a small landfill can adversely impact the groundwater quality, if sites are not properly selected and landfills are not properly designed (Bagchi 1990; Kelly 1976).

Attenuation process,

Qasim and Chiang (1994) state that attenuation is a physical, chemical, and/or biological reaction or transformation that causes a temporary or permanent decrease in the concentrations of many contaminants of waste in a fixed time and distance travelled.

Soils provide a medium that allows the above interactions to occur and the components of soil, which are a heterogenous, polydispersed system of solid, liquid and gasesous mixture in varying proportions. These properties and components of soil allow a series of complex biological activities to occur simultaneously. The soil constituents control the rate and dominance of the reactions over one another. According to Qasim and Chiang (1994) the constituents and their levels vary with parent material, time, climate, topography and vegetation. Soil properties most useful in predicting the mobility of waste constituents are texture (clay content) and particle size distribution, content of hydrous oxides (Fe, Mn, and Al), type and content of organic matter, cation exchange capacity and soil pH.

The attenuation of leachate from landfill occurs in two stages, which are as it flows through the unsaturated zone and as it flows through the groundwater aquifer.

Attenuation mechanisms.

The natural attenuation mechanisms may be categorised as physical, chemical or biological.

Physical

Filtration is removal mainly by straining action, although other mechanisms such as impaction, interception and orthokinetic (Divinny *et al.* 1990)

flocculation also attribute removal by filtration. As particles accumulate in the pores, the permeability of the soil will decrease. The extent of attenuation achieved through filtration is difficult to estimate.

Diffusion and dispersion are two mechanisms by which leachate is diluted by the aquifer. Molecular diffusion is caused by concentration gradient of contaminants, resulting in a constituent moving from a high concentration to a low one as shown by Mang *et al.*(1978) regarding leachate flow rates in low concentrations in soil solution, suggesting that diffusion may be a significant migration mechanism.

Hydrodynamic dispersion is a result of variations in pore velocities within the soil. It is effective in attenuating the maximum constituent concentration rather than the total quantity of the constituent in a pulse or slug of leachate. Dispersion can occur in both longitudinal and transverse directions. The relative importance of both diffusion and dispersion has been extensively studied by Perkins and Johnson (1963).

Dilution reduces the concentration of leachate due to mixing with groundwater. The ratio of contaminant dilution is proportional to the solution flux of both leachate and groundwater. Chloride, nitrate, hardness and sulfate found in municipal landfill leachate are not attenuated by soil. These constituents are attenuated only by dilution (Bagchi, 1990).

Sorption, physical sorption is a function of van der Waals forces and hydrodynamic and electrokinetic properties of soil particles. Only a small portion of the reaction of trace contaminants in soil/water solutions can be defined as physical adsorption. Bacterial and virus removal is by physical adsorption mechanism (Gilbert, 1976).

Chemical

Precipitation/dissolution are important reactions that control concentration levels and limit the total amount of contaminants in leachate when leaching through soils. The contaminant levels are usually governed by the solubilities of the solid. In particular, precipitation dissolution reactions are important for migration of trace metals. The attenuation effects on metals is greatly controlled by the pH of the system, at high pH insoluble metal hydroxides are formed.

Adsorption is a process by which molecules adhere to the surface of individual clay particles. Desorption is the opposite of adsorption, in which the molecules leave the surface. Both processes are dependent upon the pH of the environment and the nature of the soil and waste contaminants. It is often the most common mechanism associated with the attenuation of trace contaminants. Adsorption will also cause a decrease in the total dissolved solids in leachate. The adsorption capacity of a soil is determined experimentally. Clay minerals, hydrated aluminium, iron and manganese oxides and organics adsorb constituents in the leachate.

Because of pH-dependent charge characteristics, soils may exhibit sorption, and cation exchange simultaneously. Complexation, or chelation is the formation of inorganic-organic complexes.

Ion exchange can also occur as clays have the property to exchange ions of one type with ions of another type. The total capacity of soils to exchange cations is called the cation exchange capacity (CEC). The CEC of any particular soil is affected by the kind and quantity of clay mineral, organic content and by the pH of the soil. In general, the silicate secondary minerals in soils hold a permanent negative charge. Therefore, the cation exchange property arises from the need to balance the negative charge of clay to maintain neutrality. To accomplish this, the positive ions in the soil solution become associated with the negative charge in the exchange complex (Lu *et al.* 1985). These charges are mobile and readily exchange with other cations in the soil solution to maintain chemical equilibrium. The exchange capacity of a soil system generally depends upon (a) particle size, (b) organic content, and (c) pH. Soils containing smaller grains offer larger surface area and larger available exchange sites. Organic contents improve the exchange capacity and the cation exchange increases with increasing soil pH.

In nature, removal mechanisms of trace elements by ion exchange is not significant because other cations (calcium, magnesium, sodium and potassium), being in higher concentrations in the leachates, utilize most of the active sites. The removal of trace metals by soils occurs simultaneously by adsorption, complexation and ion exchange: therefore these mechanisms are generally grouped together.

Redox reaction are oxidation and reduction reactions that generally affect the solubilities of the contaminants. Iron and manganese in the oxidized state are less soluble.

Microbiological mechanisms

Biological decomposition of the organic component of leachate takes place in the subsoil structure. The microbial activity may be aerobic or anaerobic depending upon the availability of molecular oxygen. The biochemical reactions are complex. Under aerobic conditions the carbonaceous organic matter, ammonia, sulfide, phosphorous, iron and manganese are converted to carbon dioxide, nitrate, sulfate, phosphate and oxidized states of iron and manganese respectively. Under anaerobic conditions, the carbonaceous matter is decomposed to produce organic acids, carbon dioxide, methane, and many other complex organic compounds. Denitrification and reduction of metals are other biochemical reactions of anaerobic activity.

The movement of nitrogen between the atmosphere and plant life, which require it as a basic nutrient to exist, depend upon microbial activity. This microbial activity is also used in the attenuation process in the soil of nitrogen compounds from landfill leachate. This process begins once decomposition begins and nitrogen is returned to the soil mostly in the form of amino acids. In well-drained oxygen-rich soils, these amino acids are rapidly converted to ammonium (NH_4^+). This process, which is the first of two steps in what is known as mineralisation, is referred to as ammonification. The second step of mineralisation is the conversion on ammonium to nitrite (NO_2^-) by specialised microorganisms of the genus *Nitrosomonas*, followed by the conversion of nitrite to nitrate (NO_3^-) by microorganisms of the genus *Nitrobacter*. This second step is referred to as nitrification. Nitrification requires oxygen and a neutral to alkaline pH.

When upon flooding of a soil, oxygen availability becomes limited, the conversation of ammonium to nitrite by *Nitrosomanas* becomes inhibited, and ammonium starts to accumulate in the soil. Most wet soils have a thin upper layer of oxidised soil with sufficient oxygen supply to sustain nitrification. Any ammonium diffusing from the deeper, chemically reduced layers into the upper oxidised layer or the oxidised rhizosphere of the plants (the region in the plant roots that contain the genus *Nitrosomonas*) will also be oxidised to nitrate. But most nitrate diffusing down into the chemically reduced layer will not be converted to ammonium, but instead nitrous oxide (N_2O) or elemental nitrogen (N_2). Because nitrous oxide and elemental nitrogen are gases, they are lost to the atmosphere. This process is known as denitrification.

In general, the microbiological activity causes immobilization by conversion of organics and inorganics into cellular mass and by precipitation of inorganics. It may also cause mobilization of organics by solubilization of metals by reduction reactions and release under acidic conditions (carbonic and other organic acids).

Migration Trends of Contaminants.

The migration trends of contaminants from landfills depend upon the characteristics of the soil, the characteristics of the leachate, and the environmental conditions and activity in the fill, these environmental characteristics and their effects upon chemical constituents are described below.

Many reactions in leachate are governed by the redox potential and pH. Landfill leachate is generally acidic, because of the accumulation of organic acids during the early life of the fill. Some neutralization may occur due to the dissolution of calcium carbonate and other minerals in the soil column. Redox potential and pH also affect solubilization or precipitation of iron, manganese and other metals, sulphur and phosphorus as well as the conversion of nitrogen, and other reactions.

The migration of organic matter in the soil/water system is greatly influenced by microbiological activity, surface sorption, and chelation. Microbiological decomposition of organic matter in lechate and soil is a significant attenuating mechanism.

Alkalinity in leachate is due to carbonates, biocarbonates, silicates, borates, ammonia, organic bases, sulfides and phosphates (Lu *et al.* 1985). Alkalinity in the soil is affected mainly by dissolution and precipitation of metal carbonates.

The major ions in leachate are sodium, potassium, calcium, magnesium, chloride and sulfate. The attenuation of these ions depends upon solubilities and ion exchange. Dilution within the aquifer is also a major cause of concentration reduction of these ions.

Nitrogen and phosphorous are macronutrients and nitrogen may exist as organic, ammonia, nitrite, or nitrate nitrogen. The transformation of nitrogen is dependent upon micro-organisms, pH, and redox potential. Micro-organisms play an important role in the conversions and attenuations. Other mechanisms are adsorption, ion exchange, and complexation. Nitrate ions are relatively mobile and are not retained by the ion exchange process. Phosphorous compounds in the soil/water environment undergo complex physical, chemical and microbiological transformations. The attenuation mechanisms of phosphorus compounds are microbial uptake, precipitation, complexation, solubilization and sorption. The solubility of phosphate in leachate depends upon pH and alkalinity.

The movement of trace elements in the soil/water environment is extremely complex. Major mechanisms that influence the mobility of trace metals are: (1)precipitation/solubilization, (2) sorption, (3) ion exchange, (4)complexation/chelation, and (5) dilution. Each metal behaves differently in the soil/water environment. The governing environmental factors that influence the mobility of metals are pH, redox potential, microbiological activity and soil chemistry. Most metals attenuate well in clayey soils; non-metals are not attenuated well.

Chlorinated hydrocarbons and pesticides are attenuated mainly by sorption. The adsorption and attenuation of chlorinated hydrocarbons and pesticides increase with an increase in clay content.

Virus survival in soil depends upon the pH, temperature, moisture content, nutrients and antagonism (Keswick and Gerba 1980). Viruses survive longer in soil than at the surface of the soil. The specific factors that control their travel distances are soil composition, pH, soluble organics, and leachate quantity. Yates *et al.* (1992) presented models of virus transport in unsaturated soil.

2.6 Leachate collection

. Modern landfills according to Qasim and Chian (1994) are designed so that leachate collection systems are used to limit the migration of potential leachate and thereby protect the groundwater from contamination. A collection system is also used to remove the leachate for treatment and disposal. Liners are installed along the bottom and on the sides of a landfill to reduce the migration of leachate to groundwater beneath the site, as well as laterally. It should be noted that this impervious liner also minimises in the infiltration of groundwater to the landfill site. The liner might be constructed of a compacted clay or mixed material, a prefabricated synthetic material, or a combination of the two. Synthetic liners, although essentially impermeable under ideal conditions, often leak under field conditions. Therefore, synthetic liners are sometimes placed over clay liners for additional safety.

Landfill liners must be constructed of materials that have appropriate chemical properties and strength and are a sufficient thickness to prevent failure from internal or external pressures. The liner must also rest on a foundation or base capable of providing support and resistance to settlement or bucking. Liners in general operate in two ways; (1) they impede the flow of pollutants and pollutant carriers, and (2) they absorb or attenuate suspended or dissolved pollutants. The absorptive or attenuative capacity of a liner depends on its chemical composition and its mass. Liners can be classified in a variety of ways such as construction methods, physical properties, permeability, composition and type of service.

Most liners incorporate flow-control and filtration mechanisms, but to different degrees. Membrane liners are the most impermeable, but have little adsorptive capacity. Soils have a larger adsorptive capacity, but can be more permeable. However, greater thickness of the soil liner will have lower potential for movement of pollutants through it. Due to their availability, soils normally are considered as the first alternative for landfill liners. Synthetic liners use materials constructed or fabricated by man, and include soils and clays of low permeability, either available at the site or brought to the site and compacted with additives, to further reduce permeability and increase strength.

Cheremisinoff *et al.* (1979), Lu *et al.*(1984), Haxo *et al.* (1985), Loehr (1987) , Matrecon (1980, 1988), U.S. Congress (1989), Bagchi (1990), Goldman *et al.* (1990) and Tchobangoglous *et al.* (1993) provided discussions on various types of liners and installations.

The classification of liners include natural soil and clay systems, due to their availability these should be considered as the first alternative for a waste confinement liner. The native material must be evaluated and should be used first. If the result of such analysis is negative, the soil from other sources must be evaluated for treatment, remoulding and compaction to increase strength and reduce permeability. Bentonite is often used for sub-grade cover in areas where compatibility or soil teats show that the proper application will lower the permeability to the desired level. Common application systems include spreading, mixing and compacting. Bentonite is an extremely absorbant, porous clay, which holds liquid and becomes impermeable. In general, clay liners are more permeable to water than synthetic liners. Engineered soils, however, are less permeable than un-compacted soils. The permeability of natural soil liners to organic chemicals is variable. It depends on the characteristics and the concentration of the chemicals, degree of compaction, and other engineering properties of the soil. Compacted clay liners can adsorb much of the organic pollutants in leachate, however, little is known about the adsorptive capacity of chemical solvents (Bingemer and Crutzen, 1987). Soil liners can become desiccated by some solvents, which are insoluble in water. Such solvents are for example xylene and carbon tetrachloride, and this desiccation may cause water to migrate out to the soil. When desiccation occurs, the soil may shrink and channelling of the soil may form pathways through which liquids can flow. Daniels (1988) indicated that the majority of the flow through clay liners, take place through micro cracks created by desiccation or improper placement and compaction rather than by permeation through micro-pores of the clay liner. Clay liners if properly selected and engineered have the mechanical properties to achieve permeability.

Admixed liners can be comprised of asphalt concrete, soil cement, or soil asphalt. Sprayed on linings can be applied as air blown asphalt, membranes of emulsified asphalt, urethane modified asphalt, or rubber and plastic latexes.
Soil sealants is where the permeability of some soils, soil cement and other surfaces can be reduced significantly by the application of various chemicals or latexes.
Polymeric flexible membrane liners are membrane or plastic liners which include butyl rubber, chlorinated polyethylene (CPE), chlorosulfonated polyethylene (CSPE), elasticised polyofin (ELPO), epichlorohydrin rubber (CO and ECO), ethylene propylene diene mono rubber (EPDM), neoprene, polyethylene, polyvinyl chloride (PVC), and thermoplastic elastomer.
A composite liner is where natural and synthetic liners are utilized together as a means of isolating the leachate within the fill to protect the soil and groundwater below. This combination provides higher protection than individual liners because each liner component has individual resistance properties.

The concern of liner selection and performance is their ability to maintain integrity and impermeability over the lifetime of the landfill. Subsurface water monitoring, leachate collection, and/or clay liners commonly are included in the design and construction of a waste landfill when polymeric membrane liners are used.
To effectively serve the purpose of containing a leachate, a liner must possess a number of physical properties such as:

1) High tensile strength, flexibility, elongation without failure,
2) Ability to resist abrasion, puncture, chemical degradation by leachate,
3) Good weatherability, manufacturer's guarantee for long life,
4) Immunity to bacterial and fungal attack,
5) Colour: black to resist UV light,
6) Minimum thickness 0.5 mm,
7) Uniform composition of membrane, free of defects,
8) Ability to withstand temperature variation and ambient conditions,
9) Easy installation,
10) Economic.

The natural soil containment system often provides higher permeability, but, with sufficient depth, may have the capacity to attenuate the contaminants. Synthetic liners have been under intensive evaluation to establish their structural strengths, chemical reactions with different wastes and physical properties.

In all cases the main purpose of the liner construction and installation is to impede the downward movement of the liquid into the underlying, undisturbed soil. Whether this is done by compaction, depth and densification of clay liners or composites using clay or by the underlying of a sub-base for synthetic liners followed by installation of the liner, great care must be taken to ensure that leaks or breaks do not occur.

Leachate collection and removal systems, as reviewed by Qasim and Chiang (1994), Bagchi (1990) and Haxo *et al.* (1985), is by perforated plastic or PVC pipes 10 to 15 cm in diameter along with drainage layers and blankets, header pipes and sumps. The pipes are placed in drainage layers that are filled with sand and gravel, which are placed on to the liner. It is vital to efficiently remove the leachate as the hydrostatic pressure of the leachate above the liner can rupture it. The recommended maximum height of the leachate upon the liner is 30 cm. The base of the landfill is sloped in order to ensure leachate flow to the collection sumps.

Under the liner there may be a drainage layer to prevent groundwater from building up pressure under the liner. This drainage layer also helps to detect leachate leaks, if the liner should happen to rupture.

Maintenance of the leachate collection system is essential to ensure the free flow of leachate, this is done mainly by inspection and monitoring the flow rates at manholes and analysing the records of leachate production.

2.7 Leachate Treatment

As previously mentioned, landfill leachate is essentially a high strength wastewater, high in BOD and COD and also contains toxic chemicals. To further complicate matters, the leachate make up is inconsistent both in flow and composition, both seasonally and throughout its lifetime. The treatment system for the leachate will have to reflect the complicated nature of concentration and flow of the leachate. Qasim and Chain (1994) identify this and point out that neither conventional biological waste treatment nor chemical treatment processes separately achieve high removal efficiency over the whole life of the landfill.

Factors associated with the treatment of landfill leachate are listed below (Qasim and Chaing 1994)

1) The high strength of waste and magnitude of pollution potential dictates the selection and use of reliable treatment processes.
2) The changes encountered from landfill to landfill are such that waste treatment techniques applicable at one site may not be directly transferable to other locations. It may be necessary that each instance be separately engineered for proper treatment.
3) The source of leachate is primarily percolated water that may be seasonal, depending on hydrologic and climatic factors.
4) The chemical nature of the solid wastes accepted at a landfill has a marked effect on the composition of the leachate.
5) The fluctuations in the leachate quantity and quality, which occur over both short and long time intervals, must be considered in the treatment plant design. The processes designed to efficiently treat the leachates from a young landfill should be modified in the future to treat the leachate adequately as the landfill ages, or effluent standards change.

The leachate treatment systems could be classified as physical, chemical or biologically wastewater treatment (WWT) facilities. Some WWT facilities are a composite of these treatment options.

Physical and chemical treatment.

The following can be considered as physical options.

a) Equalisation, flow and mass loadings are balanced,
b) Screening of suspended or floating debris,
c) Flocculation by gentle stirring.
d) Sedimentation, settable solids and flocs removed by gravity.
e) Flotation, solids removed by fine air bubbles.
f) Air stripping, stripping tower removes ammonia, other gases and VOC's.
g) Filtration, filter beds or micro-screens.
h) Membrane processes, ultra-filtration, reverse osmosis and electro-dialysis.
i) Natural evaporation, drying beds.

The following can be considered to be chemical options:

a) Coagulation, chemicals including alum, ferrous and polymer mixtures destabilizes colloidal particles.
b) Precipitation, chemical reactions reduce the solubility of contaminants.
c) Gas transfer, removal of gases.
d) Chemical oxidation, removes organics, H_2S, ferrous and other metal ions.
e) Disinfection, destruction of pathogens.
f) Ion exchange, themoval of inorganics such as ammonia and demineralisation.
g) Carbon adsorption, used to remove BOD, COD, toxic and refractory organics, heavy metals.

Biological Processes.

In the past, harnessing the ability of microbial activity to utilize the potential pollutants present in the effluent for their own needs has treated wastewaters. In a controlled environment organic matter and macronutrients such as nitrogen and phosphate have been removed to acceptable levels. These can be in aerobic and anaerobic conditions depending on the type of pollutant or the desired final product. In an aerobic environment a constant oxygen supply to the microbial pollution can remove organic matter and produce CO_2 and water, while at the same time allowing for the nitrification of ammonia. In an anaerobic environment the removal of an organic pollutant along with the denitrification of nitrogen source can result in methane being produced, which can then be utilized as a fuel source.

The implication of the availability of different possibilities is that depending on the circumstances, a purpose built WWT facility can be built to treat landfill leachates. The final outcome can then be predicted in the initial design to remove the pollution potential from the captured landfill leachate before it is released to its receiving water. The following can be considered to be aerobic WWT processes:

a) Suspended growth, where wastewater containing BOD, solids, and nutrients are mixed with a large population of active micro-organisms suspended in an aeration basin including: activated sludge, nitrification aerated lagoon, sequencing batch reactor.
b) Attached growth, the population of the active micro-organisms is supported over solid media. The solid media may be rocks or synthetic media which include trickling filters and Rotating Biological Contactor (RBC)
c) Combined suspended and attached growth, such a system will have micro-organisms in suspension and attached to a solid media; this process is effective at removing BOD, total suspended solids and also achieves nitrification.

Anaerobic processes are where microorganisms are cultivated in the absence of oxygen. The complex organics are solubilized and stabilized. Carbon dioxide, methane and other organic compounds are the end products. The following can be considered to be anaerobic processes:

a) Suspended growth, the waste is mixed with biological solids in a digester and the contents are commonly stirred and heated to an optimum temperature.

b) Conventional suspended growth is where high organic strength or sludge is stabilized in a digester, the digesters are standard rate, high rate, one-stage or two-stage.

c) Contact process suspended growth is where the waste is digested in a completely mixed anaerobic reactor. The digested solids are settled in a clarifier and returned to the digester.

d) Up-flow Anaerobic Sludge Blanket (UASB), waste enters the bottom and flows upward through a blanket of biologically formed granules or solids. Denitrification can be achieved where nitrite and nitrate are reduced to gaseous nitrogen in an anaerobic environment.

An anaerobic attached filter is where the reactor is filled with solid media and the waste flows upward. Medium-strength wastes are treated in a relatively short hydraulic retention time. An expanded bed or fluidised bed is where the reactor is filled with media such as sand, coal and gravel. The influent and recycled effluent are pumped from the bottom. The bed is kept in an expanded condition.

Rotating bio-disks or circular disks are mounted on a central shaft and rotated while completely submerged in an enclosed housing. Biofilm grows over the disks and stabilizes the organic wastes.

Aerobic-anaerobic stabilization ponds, these stabilization ponds are earthen basins with an impervious liner. The basins may be aerobic, facultative or anaerobic depending, on the depth and strength of wastes. Source of oxygen is by natural aeration.

Land treatment can be used where the waste is applied over land to utilize plants growth. This is a method used to deal with septic tank sludges and wastewater treatment sludges. The soil matrix and natural phenomena treat waste by a combination of physical, chemical and biological means. The methods of land application are slow-rate irrigation, rapid infiltration-percolation, and over-land flow.

As previously mentioned instead of the microbial population being suspended in a liquid medium it can be attached to a solid one. The organic matter is stabilized as the waste comes in contact with the attached growth. This can be operated in an aerobic or anaerobic environment or a combination of both within the one system with an initial area that is aerobic followed by an anaerobic level thereafter. Examples of such systems include such filtration systems where compacted peat acts as an attachment media over which an effluent flows. The microbial pollution suitable to oxygen levels is present at different levels at the incoming effluent, which is then treated as it flows down through the system. Such a system operates under the brand name of 'Puraflo' and is supplied by 'Bord na Mona'.

There are a number of WWT systems that utilize the principle of suspended or attached medias but in smaller modular forms and are generally referred to by their brand names, such as 'Puraflo', 'Biocycle' and 'Biofitler'. These systems act as a means of removing pollutants from effluents in a low cost and low technology manner .

The Puraflo system is essentially a filtration system that operates in an aerobic environment. It incorporates a sump, a pump, and a number of bio-fibrous media containing modules. According to the Puraflo IAB certification the effluent from the septic tank is evenly distributed over the surface of the bio-fibrous media and percolates through the media before emerging as a treated liquid at the base of the unit.

Treated waste water quality will produce BOD (mg/l) <15, TSS (mg/l) <15, NH_3-N <5, Nitrate-N (mg/l) 20, Total Coliforms >99.9% elimination and Faceal Coliform >99.9% elimination with pathogenic bacteria absent. An adequate percolation area in conjunction with correct sizing of the system to the influent along with an adequate electricity power source is necessary to maintain the above performance figures.

In general all of the above involve the hard engineering of the landscape and the provision of electrical services. The final effluent may be treated but there is usually a final residue that must the captured and removed for further processing and disposal. This is usually in the form of sludge. Land treatment is one method by which this waste can be dealt with but this is becoming increasingly unacceptable.

An alternative option that provides for a soft engineering approach is constructed wetlands, and also minimises the creation of sludges

Constructed Wetlands

Operational theory.

The construction of artificial wetlands for the treatment of wastewater has been carried out in Europe for many years and has been discussed by Brix and Schierup (1986), Buchsteed (1987), Cooper and Hobson (1987). According to Trautmann *et al.* (1988) a bed is excavated and lined with an impermeable layer, filled with gravel or soil, and planted with wetland plants. Usually reeds (*Phragmites* sp.) are used, but rushes (*Scirpus* sp.), cattails (*Typha* sp.) and sedges (*Carex* sp.) also are common. The US EPA Manual on Constructed Wetlands Treatment of Municipal Wastewater (2000) identify the common types of constructed wetlands as free water surface (FWS), which are like natural wetlands and have standing water and vegetated submerged bed (VSB), which has little or no clear water.

Although the capability of the wetlands to treat waste water is widely recognised, its ability to treat municipal wastewaters needs to be compared against what the US EPA (2000) consider to be the common misconceptions about the constructed wetlands. They state four basic misconceptions:

1. Wetland design has been well-characterised by published design equations; the fact that wetlands are complex biological systems makes it very difficult to design a wetland based on previously established data, as such data derives from smaller wetlands and can not be relied upon to extrapolate for larger more complex systems. In effect, greater study of larger systems must be undertaken before this statement is proven.

2. Constructed wetlands have aerobic, as well as anaerobic treatment zones; is probably the most common misconception. The emergent plants are suitable to anaerobic conditions as they can move oxygen to their roots, this does not mean however that they are able to substantially aerate their surrounding soil. Field experience and research have shown that the small amount of oxygen leaked from plant roots is insignificant, compared to the oxygen demand of municipal wastewater applied at practical loading rates.

3. Constructed wetlands can remove significant amounts of nitrogen: this misconception is associated with the previous problem of the aerobic ability of the wetlands and is of particular interest to the treatment of landfill wastewaters where the nitrogen levels in the form of ammonia are high. While the plant up-take of available nitrogen can account for 20% the US EPA state that the rest is expected to be removed by nitrification and denitrification. It is only in the FWS wetlands, where there is sufficient open water, that nitrogen removal will occur. Constructed wetlands can be designed to remove nitrogen if sufficient aerobic (open water) and anaerobic (vegetated) zones are provided. Otherwise, constructed wetlands should be used in conjunction with other aerobic treatment processes that can nitrify to remove nitrogen.

4.Constructed wetlands can remove significant amounts of phosphorous: has not been proven and as the plant growth can only utilize a certain amount of phosphorous during the growing season, when growth stops then the up-take will also cease. The soil matrix has the ability to absorb some of the phosphorous, but when it becomes saturated, this up-take will also cease.

It is important to realise at the outset that while constructed wetlands have shown the ability to be very effective at removing water borne pollutants, their ability to treat strong wastewaters such as landfill leachates has to be considered against the above limitations of the system.

Theoretically, wastewater flows horizontally through the root zone, where plant roots supply oxygen and channels for wastewater flow. Solids are aerobically decomposed in the layer of plant litter at the soil surface. Pathogens are filtered out of the wastewater by the soil. Wetland plants, although essential to the treatment process, are not thought to play a significant role in removing organics or nutrients from wastewater. Rather, their root structure theoretically maintains or increases soil hydraulic conductivity and supplies oxygen to soil microorganisms. In this environment, soil microorganisms can oxidize organic matter and nitrify ammonia nitrogen.

Phosphorous is theoretically removed through oxidation to phosphate, which precipitates and absorbs to soil particles. Heavy metal removal has received little attention to date because most existing systems treat domestic wastewaters with low concentrations, but removal by plant uptake and adsorption has been documented by Gersberg *et al.* (1984).

In Slovenia research conducted by Urbanc-Bercic (1994) puts the use of this technology in context. A pilot-scale constructed reed-bed system was established to investigate the potential of this low-technology approach to the treatment of contaminated municipal waste dump leachate which is recognised to be a serious problem in Slovenia.

The $600m^2$ gravel media reed-bed, planted with *Phragmites australis* has been in operation since October 1990, receiving a mean daily leachate flow of $26m^3$. Although the percent removals for BOD_5, COD and SS were only 32%, 36% and 73%, respectively, this was adequate to achieve an acceptable effluent quality in terms of discharge requirements. This occurred despite widely fluctuating influent characteristics, the presence of organic toxins in the leachate, and operational and environmental problems including poor hydraulics of the gravel media and poor establishment of reeds in the first year.

While the US EPA warn against commonly held misconceptions regarding the ability of constructed wetlands other authors have found wetlands to be effective removers of leachate contaminants, such as been researched by Martin and Moshiri (1994). In this case leachate containing septage had removal percentages of BOD_5 97.7%, TOC 94.6%, TPO_4 69.5%, TKN 79% and NH_3 98.3% giving the conclusion that constructed wetlands can be a cost-effective and efficient approach to the treatment of landfill leachate with high TOC, BOD, nitrogen, and phosphorous.

A publication by Horne (1995) looks at the ability of wetlands to not only remove ammonia but also to remove nitrate by denitification and it is claimed that the nitrate removal rate of between 200 to 5,000 mg N m^{-2} d^{-1} can be achieved with initial nitrate values of 2 to14 mgl^{-1}. These rates are 1-2 orders magnitude greater than occur in most natural lake, estuarine or wetlands sediments and can be mostly attributed to denitrification rather than growth of rooted plants.

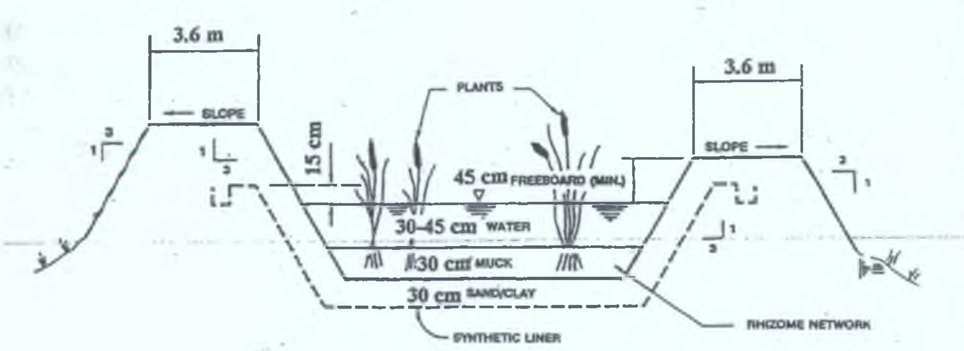
2.8 Construction and Management of constructed wetlands for the treatment of landfill leachate.

While the US EPA has classified reed-beds or constructed wetlands into free water surface and vegetated submerged bed, they can also be classified as either horizontal-flow beds or vertical-flow beds.

The basic principle of operation is the same, in that as the contaminated water passes through the reed bed, the plants root structures and rhizomes utilize the nutrients in the water. There are large populations of bacteria present in the rhizosphere, while the leaves and stems of the plants ensure the system is kept aerobic. The principle is that as the rhizomes grow both horizontally and vertically, they help to keep the bed open for water flow.

This provides for an environment that allows for the successful reduction in the BOD levels and specially constructed systems allows for the reduction of COD particularly Ammonia and Phosphates.

According to Cooper (2001), the ability of the system to successfully remove ammonia and phosphate depends upon its construction, specifically with regard to ammonia suitable oxidation is required, while a suitable substrate is required to remove phosphates. Figure 2.3



Cross-sectional detail of a typical constructed wetland cell.

Figure 2.3 (Johnson *et al.* 1999).

Characteristics of Horizontal-Flow systems are good BOD and TSS removal, but are oxygen limited, they can typically remove 80 – 90% BOD and 80 – 90 % TSS with up to 40% Total N removal and 25% Total P removal.

They have limited nitrification but can achieve denitrification under anoxic/anaerobic conditions and they are analogous to a semi-anaerobic pond.

Characteristics of vertical-flow systems are that they have an intermittent feed over the bed surface, which allows for air to be trapped in the bed. They are very aerobic hence good nitrification (as well as BOD removal). However they are not as good for TSS removal and are analogous to a rustic biological filter.

Other designs of constructed wetlands include surface flow systems and sub-surface flow systems, as shown by (Otte 2003). In the sub-surface flow system water is forced to flow below the surface by maintaining a water level below the surface of the substrate. In such systems, the inflow is often also located below the surface of the substrate.

According to Otte (2003), the constructed wetlands in Ireland are typically of these two categories.

The surface flow system is constructed of a relatively dense substrate of low hydraulic conductivity with shallow water depth and emergent plant types. Microbial growth and purification occurs predominately in the water and upper sediment zone.

The sub-surface flow system consists of a substrate with relatively high hydraulic conductivity, which supports root attachment but also makes subsurface flow of water through the substrate possible. Purification occurs predominately in the substrate.

Both systems rely on some primary treatment, such as septic tanks or macerators to prevent excessive build up of solids at the system inlet.

Integrated Constructed Wetlands (ICW) have been developed by Harrington (2002); the difference between the ICWs and others is that they use engineered soil as their impermeable layer, instead of a liner and the surface area is larger.

ICWs are described as free water surface flow systems consisting of a series of lagoons or ponds across which influents flow (Harrington 2002). The bottoms and sides of these are made virtually impervious, generally through use of in-situ soils, to prevent the seepage of contaminants to groundwater. The initial receiving pond serves as a mixing, diluting and balancing area for the various influents. Subsequent ponds, usually 3 to 4, and often more, are sequentially arranged to maximise the distance over which the influent must travel, and are ideally designed to allow for a maximum retention time.

The basic structure consists of a minimum of 4 ponds with deep and shallow areas to facilitate emergent and aquatic plant growth, using trout or other salmonoids as a bio-indicator of water quality.

Maximum vegetation cover with semi aquatic plant species is established through the use of plants from suitable nurseries. The ponds are generally shallow, 10 – 30 cm deep, with deeper sections where the vegetation is sparser.

Pond surface area is calculated on total peak influent, the design population equivalent, generally in the order of 20 m^2 per PE, and the precipitation levels for the area.

The design of the ICWs have considered the following issues:

Phosphorous removal is generally the most limiting factor in fresh-water ecosystems, an over abundance of which results in eutrophication of surface waters. The ability of wetlands to remove phosphorous is a key performance criteria. This ability to capture and retain phosphorous is dependant on the plant density and the soil properties, but principally on the available wetland area and the consequent residence time within the wetland.

Precipitation is an important factor in determining pond size. Another important issue regarding residence time is the influence of the emergent plants, which create a resistance to the flow of the waters through the ponds, thereby increasing retention time and improving phosphorous removal.

Infiltration to groundwater is limited by the use of the shallow ponds and associated low hydraulic pressures. Harrington (2002) points out that research by Purcell *et al.*(2001;2002) would indicate that the presence of organic matter in the soil and the accumulating detritus/necromass further decreases water infiltration. The use of emergent plant species and the effects of the necessary soil type and depth ensure that wetlands provide a denitrifying role in a treatment context.

Plant Functions.

According to Warren and Scott (2000) and Bonadonna *et al.* (2002), the macrophytic vegetation used in ICW design essentially performs a variety of functions; its primary function is the support of biofilms (slime layers) which carry out the principal cleansing functions of the wetlands; it also facilitates the sorption of nutrients, and acts as a filter medium, and through the use of appropriate emergent vegetation can control odours and pathogens. While the vegetation has the ability to filter suspended solids, it also increases hydraulic resistance, thus increasing residence time. The appropriate choice of plant species and the density at which they are planted are important in the overall functioning of the wetland. Generally, emergent species such as sedges, rushes, grasses etc. that are rooted in the wetlands soil and which grow through the water column are most effective, though floating and submerged plants also perform useful functions. The common reed *Phragmites australis* is a minor species in the ICWs, other species that are used include rushes (*Scirpus* sp.), cattails (*Typha* sp.) sedges (*Carex* sp.), Yellow Flag (*Iris* sp.), common rushes (*Junus* sp.)and pondweed (*Elodea*).

Multi-stage and Free Water Surface.

The number of ponds and the sequential processing and cleansing of contained dirty water in each pond ensures that there is segregation between differing degrees of contamination. This consequently facilitates the concentrated management of ammonium, which is of particular relevance to the welfare and growth of plants. It also enhances the potential for overall habitat diversity, due to differing plant densities and the relative areas of open water at each stage.

Landscape fit.

According to Stiener (1991), consideration of how the necessary wetland area is accommodated on site and generally in the location is a strategic issue in the ICW design approach. Site assessment provides the necessary information with regard to: the actual size of the area required, the overall topography, adjacent structures and the general landscape into which the wetland structure will be placed.

Biodiversity and Habitat Restoration.

Constructed wetlands provide an opportunity to reverse the decline in wetland areas from the Irish countryside. The initial waters that would create eutrophication are progressively reduced in their polluting potential and in the process provide habitants for biodiversity. A typical layout is shown in Figure 2.4

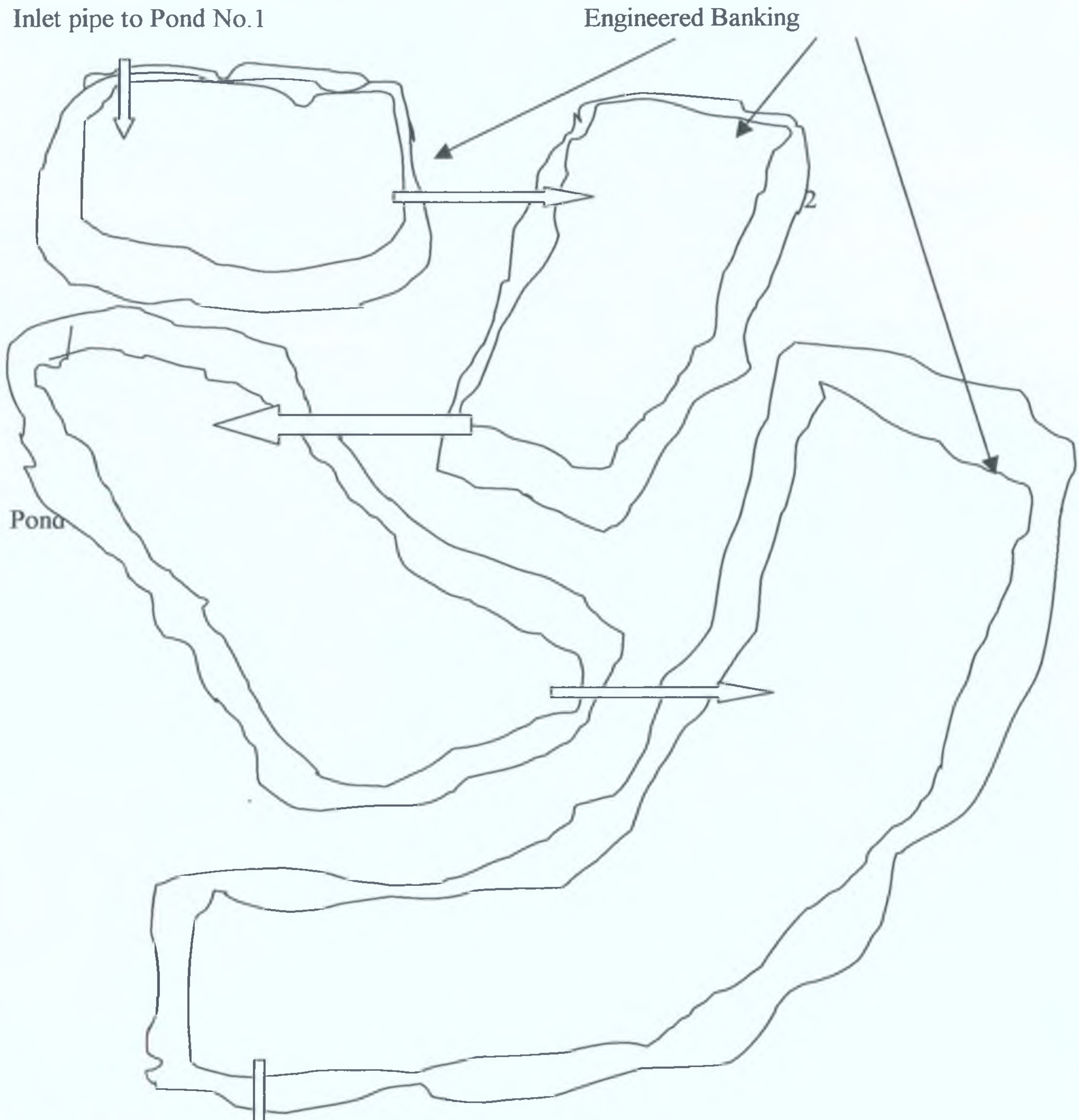


Figure 2.4 A typical layout of an ICW

Source National Hydrology Seminar 2002.

2.9 Management of Municipal Solid Wastes in County Donegal

In the Waste Management Plan for Donegal County Council published in October 2000, it was stated that out of a population of 130,000 from 26,000 homes, 38,000 tonnes of MSW was produced. This waste was deposited in 5 landfills, one of which, Churchtown was only accepting inert waste; the others were Ballinacarrick, Balbane, Muckish and Glenalla which accepted non-hazardous wastes. The wastes arising in the county were categorised under the classifications identified in the Waste Management (Planning) Regulations 1997 as follows:

WASTE TYPE	QUANTITY
Household Collected	25,000 tonnes
Household Delivered	130 tonnes
Other household (uncollected)	6,250 tonnes
Commercial	11,000 tonnes
Industrial Sludge's [tonnes dry solids]	3,342 tonnes
Industrial	4,000 tonnes
Construction/Demolition Waste	1,900 tonnes
Ash/Incineration Residue	N.D.
Contaminated Soil	N.D.
Spent Mushroom Compost	17,160 tonnes
Litter/Street Sweepings	2,000 tonnes
Water Treatment Sludge (m^3)	47,430 m^3
Wastewater Treatment Sludge (m^3)	49,590 m^3
Mining and Quarrying	N.D.
Healthcare	124 tonnes
Agricultural (m^3)	1,453,451 (m^3)

Source: Donegal County Council Waste Management Plan 2000

Table 2.4 Quantities of Waste Arising Within County Donegal, 1998.

With regard to the future implications of the above waste and the disposal facilities available in 2000, the Waste Management Plan for Donegal County Council identified the need for a long-term secure landfill capacity. It was estimated in the Plan that waste going for disposal in landfill would increase from 38,000 tonnes per annum in the year 2000 by a rate of between 3 and 4.5 % per annum to an estimated probable total of 49,486 by 2005 and 95,769 in 2020.

Therefore, for the County to achieve its targets for the diversion in household waste from landfill, a 65% reduction of biodegradable municipal waste from landfill and a 35% recycling of MSW within 15 years of 1995, a combination of recycling and the

use of a waste-to-energy option would need to be employed, with a suitable landfill for the final ash/residue fraction. These reduction targets to be achieved by 2010 are in line with government policy set out in 'Changing our Ways' and have been incorporated into Irish legislation from the Landfill Directive 99/31/EC.
However, the Plan acknowledged that it was not a financially cost efficient option for Co. Donegal to go alone in the implementation of a waste to energy (WTE) option, and indeed it was proved to be politically impossible for incineration to be considered either. The best option to realise the recycling/WTE/landfill solution was as part of a cross-border alliance between Co. Donegal and the neighbouring local authorities in Northern Ireland.
The future landfill capacity in Co. Donegal is shown in Table 2.5 the following table.

Landfill	**Total capacity (tonnes)**	**Amount of void space filled January 2000 (tonnes)**	**Void space remaining (tonnes)**
Ballinacarrick	331,250	231,250	100,000
Balbane	140,000	123,750	16,250
Muckish	70,000	53,125	16,875
Glenalla	19,100	16,100	3,000
TOTAL	560,350	424,225	136,125

Source: Donegal County Council Waste Management Plan 2000

Table 2.5 Estimated Values for County Donegal's Future Disposal Capacity.

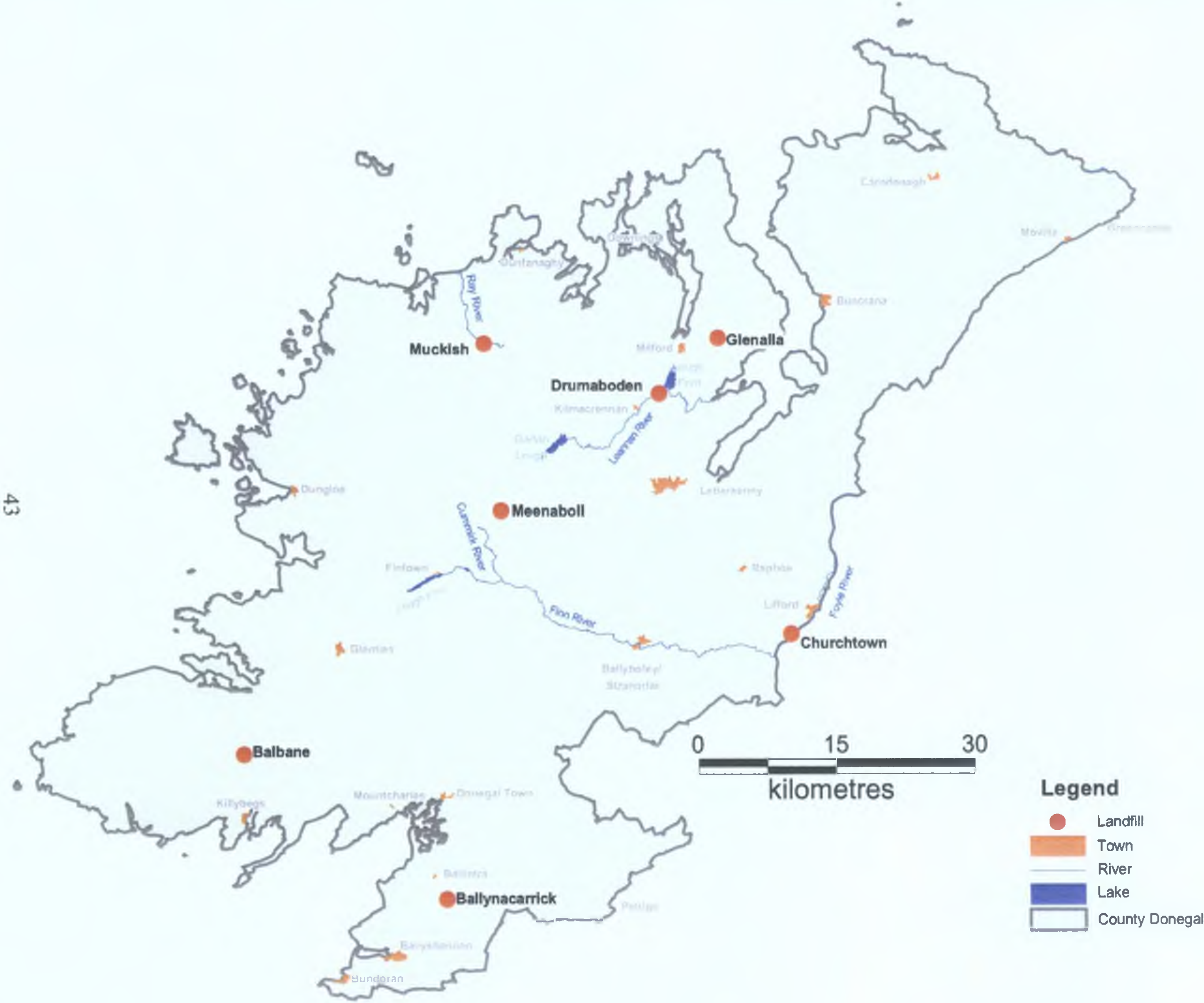
Based on the void space, it can be seen that, at disposal rates of 38,000 tonnes per year, there was only 3.2 years left.

The Plan then identified three short-term scenarios:

Scenario A – Application to the EPA to grant landfill licences to all 4 non-hazardous landfills

Scenario B – Application to the EPA for a landfill licence to Ballinacarrick only

Scenario C - Reviews all licence applications with the Environmental Protection Agency.



Muckish
Glenalla
Drumaboden
Meenaboll
Churchtown
Balbane
Ballynacarrick
Ray River
Leannan River
Finn River
Foyle River
Milford
Dungloe
Finntown
Lifford
Donegal Town
Mountcharles
Killybegs
Bundoran
0
15
30
kilometres
Legend
Landfill
Town
River
Lake
County Donegal

MAP 1

Donegal County Council decided to apply for non-hazardous waste licences for Muckish, Glenalla, Ballinacarrick and Balbane. Map 1 shows the locations of the sites and their proximity to the larger urban areas, rivers and lakes. A commitment was given to keep these landfills operational and by applying to the EPA for waste licences Donegal County Council, was obliged to maintain them in accordance with the Landfill Directive. If Donegal County Council had not applied for a waste licence for these sites prior to 2001 and had closed then they would have had no legal obligation to maintain.

Operational waste licences were not granted for Muckish, Glenalla, Churchtown and Drumaboden. Instead the waste licenses granted were for the closure and restoration of the sites. Therefore Donegal County Council were committed to fulfilling their legal obligations to ensure that these facilities were closed in a manner that did not cause environmental pollution. This involved a capping and leachate treatment systems to be installed at these closed sites, which had to be funded by Donegal County Council from their own revenue streams.

EPA successfully prosecuted Donegal County Council in the Letterkenny Circuit Court on the 24th January 2002 regarding the operation of Muckish landfill site, specifically regarding operating a site without a waste licence after the proscribed date and subsequent discharges from the site.

The disposal of waste in Donegal was to be restricted to Ballinacarrick and Balbane, with Balbane due to close by January 2004. Ballinacarrick was to be upgraded with leachate treatment and gas flaring to be installed.

Future development of landfill facilities included an extension in Ballinacarrick and a new green field location at Meenaboll. Planning permission and waste licence applications were put in place and at this point in time no decision has been made on the outcome of these applications. Ballinacarrick is due to close in its existing format by the end of 2004.

All of the above plans relied upon 38,000 tones of deposited waste entering Donegal Landfills with a gate fee that would fund these planned developments, what actually has taken place since 2000 was completely different.

It transpired that when a close monitoring of weights of waste entering Donegal landfills occurred the quantities of waste being landfilled started to decrease. This was attributed to the substantial in disposal costs from £40 punts per tonne in 2000 to €100 per tonne in 2003 and €125 per tonne in 2004. It was then discovered that the waste was being illegally trans-frontier shipped out of the jurisdiction into Northern Ireland. There may have been some reduction as a result of a home composting and bring bank recycling initiatives, but for whatever reason waste deposited in Donegal County Council Landfills in 2003 totalled 20,000 tonnes. So in three years instead of an increase in landfill deposition a reduction of 47% has occurred. This created a crisis in the incoming revenue streams for Donegal County Council as the privatised waste collectors also began to incur significant arrears in their landfill gate fees.By early 2004 the outstanding arrears would be enough to fund the entire remediation works at Ballinacarrick.

The illegal transfrontier shipment of MSW was stopped by November 2003 and Donegal County Council could begin to get a more precise picture on the actual volume of waste arsing in the county. Waste collectors who at significant gate fee arrears where informed that they would be barred from Ballinacarrick if they did not address their financial problems. A determined policy of prosecution for illegal dumping and fly tipping by refuse collectors was initiated by Donegal County Council which along mobile checkpoints to remove unpermitted waste collectors from the road served to ensure compliance with transfrontier shipment regulations.

This allowed Donegal County Council to have some confidence in the amount of waste arising in the county and to ensure that the waste went to a properly managed landfill, the income from which would be used to rehabilitate the old redundant landfills, while at the same time building new facilities for the future.

Description of Muchish Landfill, Co. Donegal

Muckish closed in 2001 with 53,125 tonnes of waste landfilled (Map 2) at grid reference N 197710 E 427309 (Site plan 1). The EPA instructed Donegal County Council they have a legal obligation to rehabilitate and restore this landfill as they had continued to operate it after the proscribed closure date of 2001. The licence that was granted was for the closure of the site and no further waste could be deposited on the site. At present on this site there is no means of treating the leachate that is produced. It is estimated by Donegal County Council that this site produces 18m^3 of leachate per day with an ammonia level of 55mg/l. (Appendix E).

Plate 2.1 Ponding of leachate at Muckish Landfill.

Plate 2.2 Seepage of leachate from Muckish landfill into receiving waters.

The most serious environmental implication for the ongoing situation at Muckish is a further reduction in the water quality of the Ray River. This leads Donegal County Council into a situation whereby it faces possible prosecution by the EPA, particularly as the new Protection of the Environment Act, 2003 allows the Office of Environmental Enforcement in the EPA to take Local Authorities to court even when

they do not have the funds available to carry out the necessary works to limit or prevent such pollution.
Another implication of the failure of local authorities to prevent or limit pollution under their control is when new licences are applied for the EPA may refuse to grant them until such remedial work on existing sites is carried out to their satisfaction.

The options available to Donegal Council at Muckish are to either treat the waste off site by using an activated sludge system at one of its wastewater treatment plants. This will involve trucking the leachate to the nearest facility at Letterkenny some 15 miles away. Whether this facility has sufficient capacity left to treat such extra loading is doubtful. The cost of treating wastewater in the 2004 annual budget was put at €0.505 per m^3. Which is estimated at $18m^3 * 0.505 =$ €9.09 per day.
Alternatively on-site treatment using either a constructed wetland or using a filtration system similar to the one being used in Drumaboden.

Drumaboden Landfill, Co. Donegal.
The above facility is situated at grid reference N 216761 E 421844, (Map 3, Site plan 2).
The facility closed in 2001 at which time it contained 50,000 tonnes of MSW.
The surface area of the site is 3.2 ha.
The filtation system is 5 meters in diameter, is 3 meters high and is contained in a circular corrugated steel tank. (Plate 2.3 and 2.4). The filtration system, 'Puraflo' consists of peat and heather and was supplied by Bord na Mona Ltd.
The leachate is fed across the surface of the system by in built sprinklers and flows downward through the peat medium until it discharges from the system.
The landfill is an uncapped, unlined facility the leachate from which is collected and directed though the 'Puraflo' system. The average daily flow of leachate is 4.33 m^3.
The leachate enters the receiving waters from a non-point source by dispersement through the adjoining land. The receiving waters are the River Lennon and Lough Fern, both of which are salmonoid.

Plate 2.3 Drumaboden Puraflo system.

Plate 2.4 Interior of Drumaboden Puraflo system.

The filtration system was installed in 2001. The results of the chemical analysis of the influent and effluent indicate the system is capable of reducing BOD, TSS and Ammonia, successfully. (Appendix D). Typically there is an 80% reduction in BOD with a final effluent of 2.5 mg/l much lower that the 25mg/l permissible level for discharge. The TSS was reduced by 61% to 13mg/l, which is lower than the 35 mg/l permissible. The reduction in levels of ammonia by 80% to 9.5 mg/l however results in an increase in the levels of nitrates.

Total oxidised nitrogen in the effluent at 10.9 mg/l represents an increase of 90% but is lower than the 15mg/l permissible level.

Phosphate levels are reduced to 60% of the influent and the effluent at 0.4 mg/l is within the 2mg/l permissible level.

Consequently the overall effect of this system is that it does not present a risk of euthrophication of surface waters due to nutrient enrichment.

It can be concluded that this system is one, which can be used successfully on a redundant landfill to attenuate the level of pollution from the landfill leachate on site.

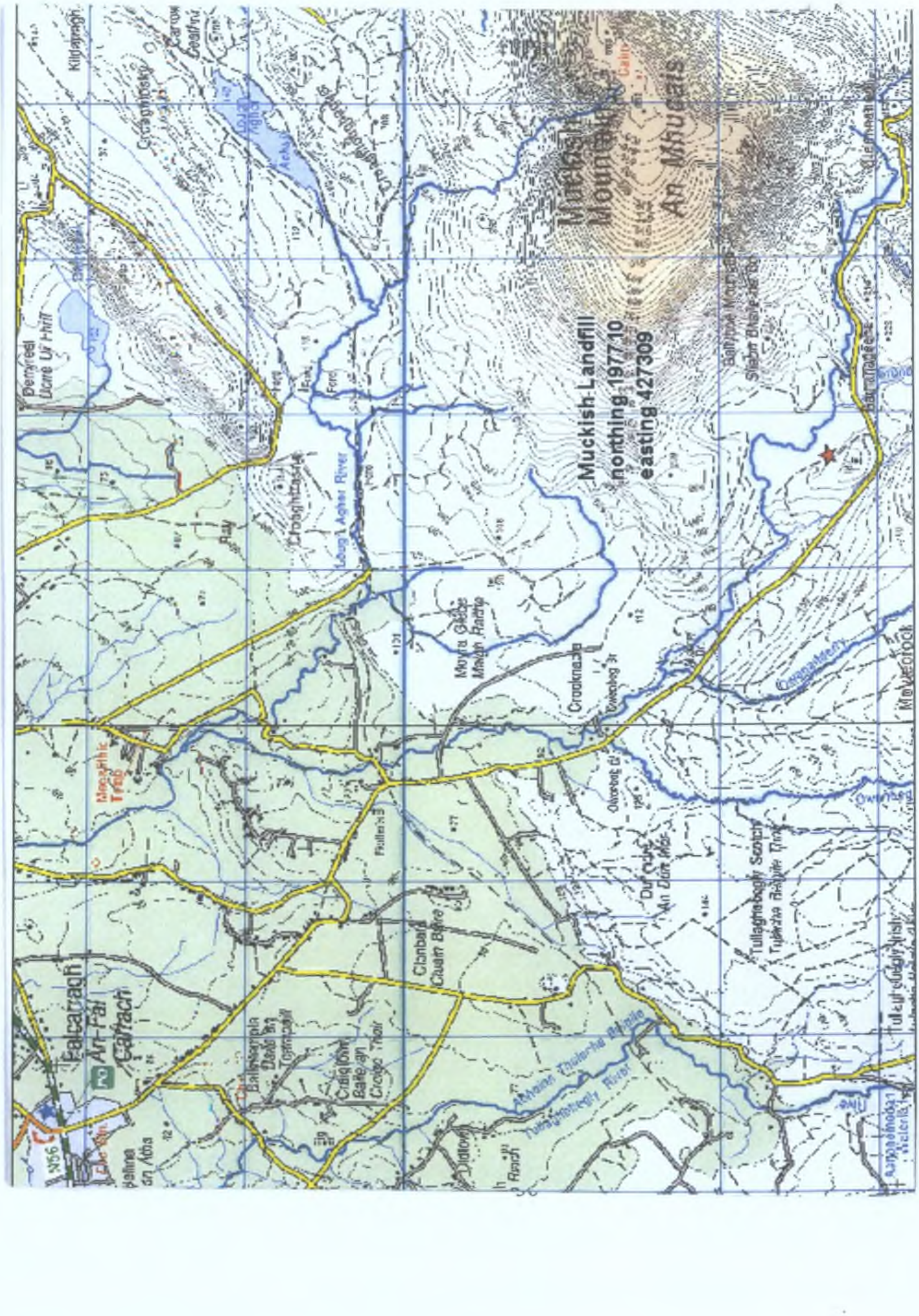
The disadvantages of this system are it is a large unsightly visually obtrusive feature. It has an electrical power demand to drive the onsite pumps and monitoring equipment, it requires maintenance and has ongoing running costs.

Proposed new landfill at Meenabol

This is a green field site situated centrally in the county some 15 miles from Letterkenny at grid reference N 199645 E 409041 (Maps number 4 and 5).

A constructed wetland has been proposed in the applications for the planning permission and in the waste licence to the EPA. However this constructed wetland will not be used to treat leachate but only to allow storm water from site return to receiving waters. This is because the waters in this location are considered to be pristine. Consequently leachate produced on site will be tankered off for treatment at a Donegal County Council waste water treatment facility.

MAP 2



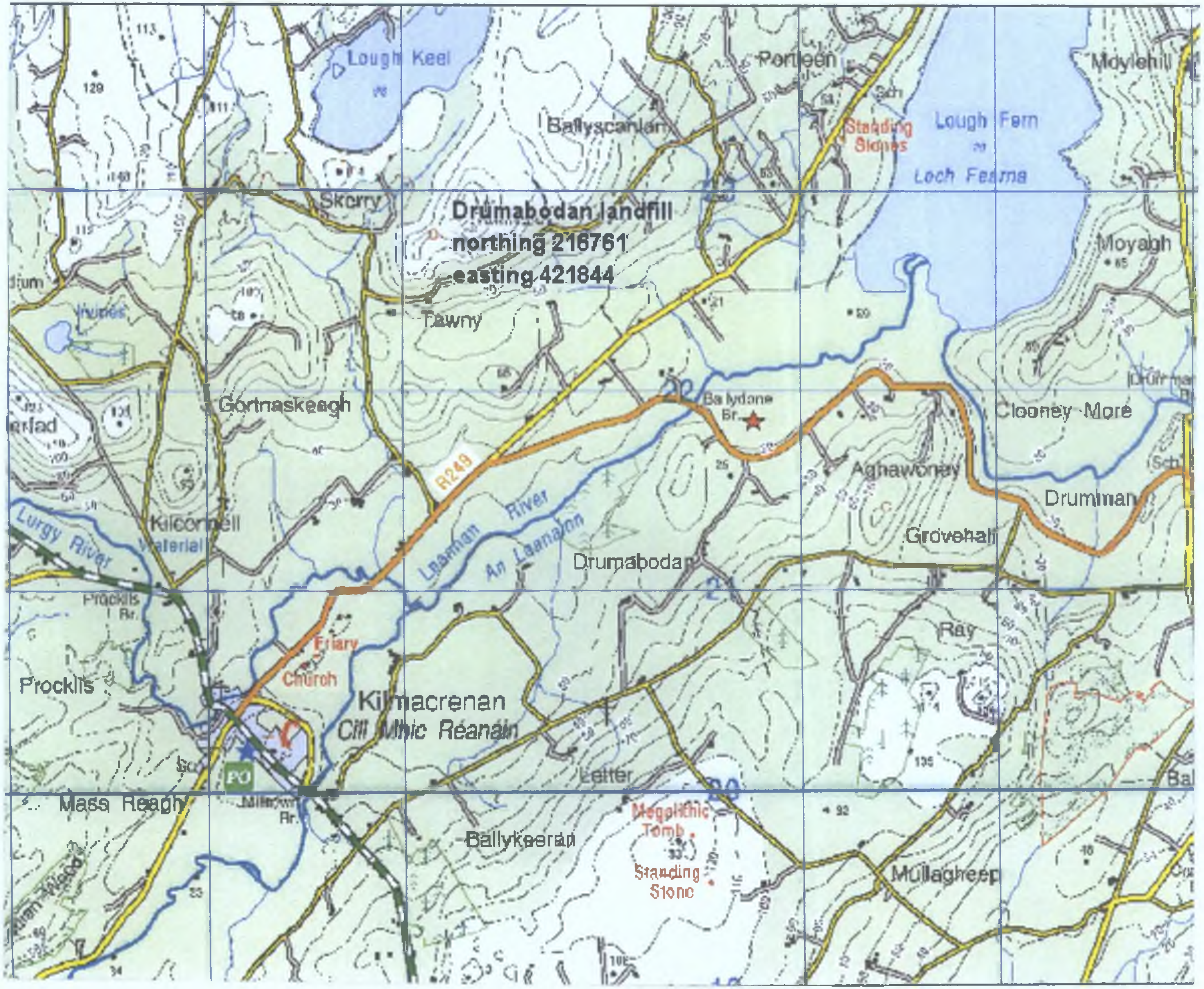
Muckish Landfill
northing 197710
easting 427309
Muckish Mountain
An Mhucais
Falcarragh
An Fál Carrach
Loug Agher River
Croaghnageer
Crocknasle

SITE PLAN 1

Muckish Landfill Site

Surface Area. Approx 26,000m^2.

MAP 3


Drumabodan landfill
northing 216761
easting 421844
Lough Keel
Lough Fern
Loch Fearna
Portleen
Ballyscanlan
Standing Stones
Moylehill
Skerry
Tawny
Moyagh
Gortnaskeagh
Ballydone Br.
Clooney More
R249
Aghawoney
Drumman
Lurgy River
Kilconnell
Waterfall
Leannan River
An Leanainn
Drumabodan
Grovehall
Prockils Br.
Prockils
Friary
Church
Kilmacrenan
Cill Mhic Reanain
Ray
PO
Letter
Mass Reagh
Millown Br.
Ballykeeran
Megalithic Tomb
Standing Stone
Mullagheep

SITE PLAN 2

Drumaboden Landfill Site

Surface Area: 32,000m^2

Flow rate from Puraflo : Flow measurements from the Puraflo system outlet indicated an approximate flow rate of 0.18m^3/hr or 4.33m^3/day.

Legend

Leachate sample	LG
Ground water Sample point	GW

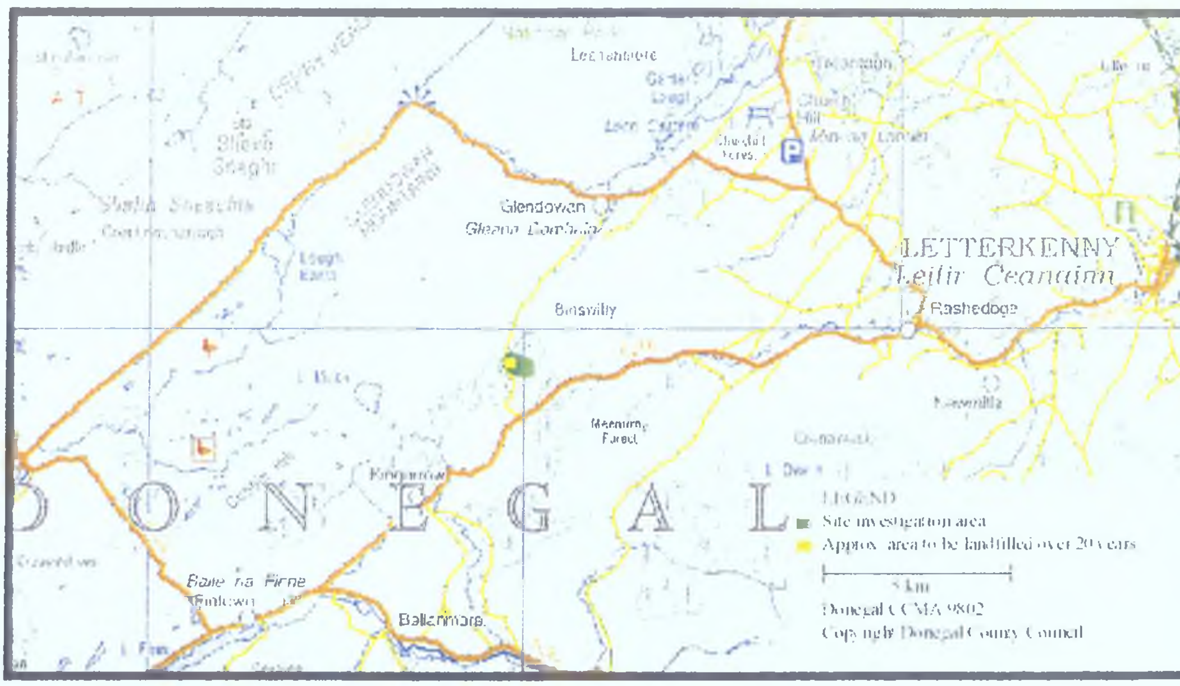
MAP 4


Meenaboll Proposed Landfill
northing 199645
easting 409041
Pollandoo
Lough Muck
Loch Muc
Tievedeevan
Tievedeevan Hill
Fintown

MAP 5

Meenaboll landfill site.

The proposed landfill will cover an area of approx 4.6 ha. This drawing was taken from the information provided during the public consultation.



2.10 Options for the Treatment of Leachate

2.10.1 Filtration Systems

There are a number of modular systems in use to treat domestic effluent that could be considered in the context of treating landfill leachate. The systems which are known by their brand names include 'Puraflo', 'Biocycle', 'Biolfilter'. In all three cases the systems are approved under the Irish Agrement Board Building Product Certification system and have the ability to attenuate organic pollutants down to at least a less than a 20 mg/l BOD and 30 mg/l Suspended Solids and are capable of successfully breaking down ammonia and nitrate to acceptable levels over a 10 year life time.
In one Donegal landfill, Drumaboden, a Puraflo system has been operating and the results of its ability to attenuate the biodegradable pollutants are shown in the Appendix D.

The method of operation for each of the above systems varies, with the Biofilter and the Biocycle being similar and the Puraflo having a different feature.

Puraflo, according to the technical specification and control data published by Puraflo Liquid Effluent Treatment System the method of operation is where the effluent is pumped into the Puraflo module, which contains compressed biofibrous media 2.5 cubic meters of biofibrious media is compressed into 2 cubic meters. Percolating the liquid evenly over the surface of the media treats the influent. Treatment is achieved by a combination of physical, chemical and biological interactions between the pollutants and the biofibrous media.
The expected quality of the treated wastewater is BOD_5 <15 mg/l, Total Suspended Solids <15 mg/l, Ammonia <5 mg/l and Nitrate 20 mg/l.
Each module is capable of handling 3 population equivalents. A population equivalent is considered to be either 200 l/person/day or 60g BOD_5 or 40g TSS/person/day.
The system can be adapted to provide a series of modules depending on the PE of the incoming effluent.

After treatment in the module the effluent is dispersed through a soil percolation area in proportion to the original influent and the soils ability to disperse the hydraulic loading.

'Biocycle', this method relies upon a mini aerobic/anaerobic activated sludge system. Each 'Biocycle' module is designed to treat the population equivalent of 8 persons/day. According to 'Biocycle' Waste Water Treatment System, Certificate number 96/0033, it will produce a final product with the following characteristics BOD_5 <20 mg/l, Suspended solids <30mg/l, Ammonia <10 mg/l N, Nitrate 5 – 10 mg/l N, Total Phosphorus < 5 mg/l P.
The final effluent is dispersed through the soil in an area proportional to the hydraulic loading of the original influent and the soils ability to handle the effluent.

The 'Biofilter' system operates in a similar fashion to the 'Biocycle' in that it is a mini aerobic/anaerobic activated sludge system but it also incorporates a biofilter unit similar to the 'Puraflo' system. According to technical specification and control data of 'Biofilter' Package Sewage Treatment Systems a population equivalent of 14 persons can be treated. The quality of the final effluent is BOD_5 <20 mg/l, Total Suspended Solids<30mg/l, Ammonia as N <5 mg/l, Nitrate as N <23 mg/l, Phosphorous as P < 11 mg/l.
The final effluent is dispersed through the soil in proportion to the soil characteristics and the original influent hydraulic loading.
In the context of treating landfill leachates on site a modular filtration system, such as the Puraflo system is worth considering. The filtration system in Drumaboden has proven to be satisfactory and on this basis it is has merits for the site at Muckish.
The capital costs for the filtration system is predicted by Bord Na Mona to be €35,000 While there are supporting civil works costs which would involve 2 sections of 43m by 4 m and the system will stand 1.2m high. The system could be buried in the ground to reduce its visual impact. Running costs are not expected to be high at these include both maintenance at € 700 and electricity at € 1000 per year.
The problems associated with the installation of such a system in Muckish is providing electricity to the site and the visual impairment of such a structure, neither of which are insurmountable problems.

2.10.2 Constructed Wetlands

According to Mitsch and Gosselink (2000) natural wetlands have inadvertently received and treated wastewater for thousands of years.

This technology is not new, it was used in the military camp at the Curragh in Co.Kildare in 1880 (Otte, 2003) to treat the domestic effluent from the facility. Harrington (2002) has used this technology for the treatment of agricultural effluents on 13 sites in Co.Waterford. This technology has also been used to treat industrial wastewater from the Glanbia cheese facility at Kilmeaden Co.Waterford (Otte, 2003). In the paper delivered by Harrington at the National Hydrology Seminar 2002, the structure and performance of a particular type of constructed wetland was given. In general constructed wetlands can be classified as either surface flow systems or sub-surface flow systems depending on whether the influent enters above or below the water level.

Another option has been operated by Harrington which is integrated constructed wetlands (ICW), which are surface flow systems that have a site specific approach that allows for the widest possible range of ecological conditions. These systems generally rely upon the construction and engineering of the soil to act as a impermeable layer to prevent movement of either the water in the systems into the groundwater or vice versa. A feature of ICW is that they use a much greater footprint to operate, typically it has been recommended by Harrington as 20 m^2 per population equivalent. These systems have shown themselves to able to remove pollutants such as ammonia and phosphate to acceptable levels, as shown in Table 2.7 below.

ICW in Anne Valley, Co. Waterford

Twelve farm ICWs and one sewage treatment wetland have been constructed in the Anne Valley Co. Waterford. While all of the ICWs received individual planning permission, some have not been constructed to the requirements of that permission. In general many of the ICWs have been built undersize.

The monitoring of the ICW sites in the Anne Valley in County Wexford has been carried out under the direction of the project team and implemented under the leadership of Paul Carroll, Waterford County Council. Grab sampling from each wetland has been conducted approximately monthly since August 2001. Since February 2003, the monitoring programme has progressed to flow proportional sampling at selected sites, with continuous flow measurements, weather data collection and groundwater monitoring.

A summary of the results of the grab sampling programme for the twelve farm ICWs for the period August 2001 to August 2003 are presented Table 2.6. Sampling frequency at each ICW, and at three sites along the Anne River, was between 17 and 25 times over the period.

Parameter	Max Influent	Mean Influent	Max Effluent	Mean Effluent	Max Removal	Mean Removal
PH	7.9	7.1	8.75	7.5	N/a	N/a
AMMONIA	399	68.5	2.6	0.49	99.3%	99.3%
NITRATE (Effluents)	N/a	N/a	7.4	1.89	N/a	N/a
MOLYBDATE REACTIVE PHOSPHORUS (MRP)	168	25.3	1.55	0.38	99.1%	98.5%
BOD	10497	1271	67.9	19.5	99.3%	98.5%
COD	21511	2363	116.4	48.8	99.5%	97.9%
TOTAL SUSPENDED SOLIDS	6636	682	81	21.6	98.8%	96.8%
COLIFORM BACTERIA (occasional monitoring)	5000	1000	100	100	98%	90%

Source: Paul Carroll, Waterford County Council.

Table 2.6 Removal percentages for tested parameters on the twelve farm wetlands in the Anne Valley Co. Waterford

All testing referred to in this report was carried out using validated standard methods in Waterford County Council's water and wastewater laboratories. The laboratories participate in the EPA proficiency scheme for chemical tests and are registered as approved laboratories for the chemical and physical tests for which results are presented.
A copy of the full data report received by personal communication from Paul Carroll, Waterford County Council is given in Appendix A.

The Annestown Stream into which the effluent from all of the 12 monitored ICWs eventually flows, has shown an improvement in water quality since the installation of these ICWs. This is demonstrated by the improvement in the EPA quality rating for this stream, and by the recent return of sea trout to the stream after an absence of many years

Constructed Wetland for the treatment of leachate at Arthurstown Landfill

It is the ability of wetlands to remove pollutants from landfill leachate that is of particular interest to this dissertation and in Ireland trials have taken place in Arthurstown Landfill in Kill, Co.Kildare.

This facility receives baled waste from South Dublin County Council and deposits it in an engineered landfill. Gas and leachate are captured. The gas is flared off while the leachate receives preliminary treatment on site in a sequential batch reactor. The effluent from the reactor is then tankered off site for further treatment and disposal at a wastewater treatment plant. Appendix B shows chemical analysis results of the leachate

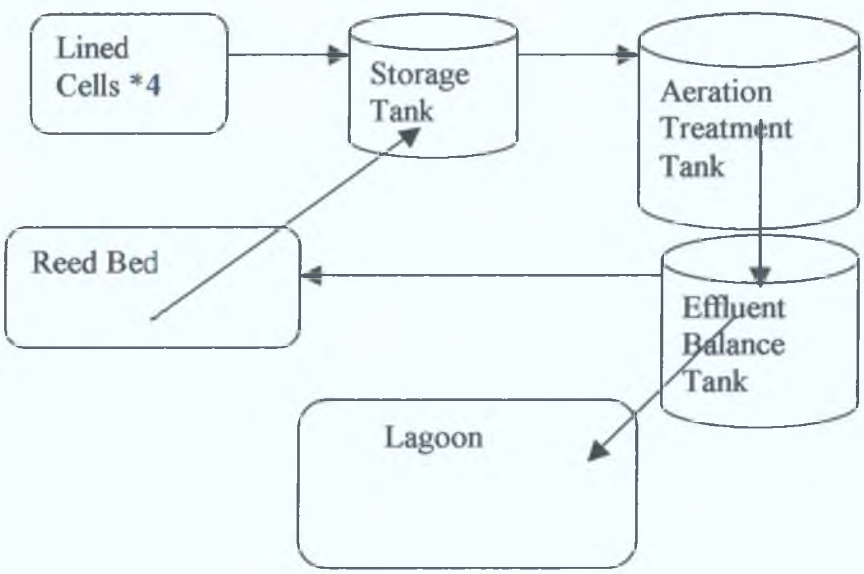


Figure 2.5 Flow diagram of leachate system at Arthurstown.

The implications of this case study are that a pilot scale constructed wetland can successfully treat landfill leachate. However as pre-treatment has already occurred prior to arriving at the wetland its full potential is not evident. In particular the ability of a constructed wetland to remove ammonia and phosphates is not fully explored in this case study.

Constructed Reedbeds for Treatment of Leachate at Slovenj, Gradec, Slovenia.

Elsewhere in Europe, particularly the research carried out in Slovenia by Olga Urbanc-Bercic there is a situation that is very similar to the one facing Donegal County Council at its redundant landfill sites. In Slovenia uncontrolled dumping of municipal solid wastes resulted in a situation whereby the surface and ground waters are being contaminated. A low cost solution was identified as constructed reed beds and a degree of success has resulted in that the available data indicate that most pollutant parameters at the outlet (BOD_5 COD and TSS) fulfil the general demand for effluent concentrations set by the authorities, and the performance of the system is considered satisfactory at this stage. (Appendix C).

Preliminary analysis indicated a flow rate of 26 m^3 per day and the reed-bed was designed to take a daily loading rate of 0.3 l/s or 50 $l/m^2/day$. The wetland consisted of a $600m^2$ gravel media sub-surface wetland with a flow rate of $5x10^{-4}$ m/s,

Performance of the system in its first year is shown in Table 2.7. The unavailability of removal efficiencies of ammonia and phosphates limits this case study.

Parameter	% Removal
BOD_5	32
COD	36
SS	73

Table 2.7 Removal of BOD_5, COD, and SS at the Slovenj, Gradec, Slovenia Constructed Wetlands after year 1.

Constructed Wetlands for the treatment of landfill leachate at Peridido, Escambria County, Florida, USA.

In the USA a study was conducted by Martin and Moshiri (1994) at the Peridido solid waste facility in Escambria County, Florida. Wetlands were used as part of a combination of treatments for leachate and septage. The results represented in Tables 2.8 and 2.9 indicate that constructed wetlands are able to offer a solution for the removal of pollutants from landfill leachate.

Temperature	28°C
.pH (standard units)	8.1
Conductivity (mhos/cm)	1850
Alkalinity	436
Chlorides	150
TDS	4750
BOD_5	240
TKN	526
TOC	395
TPO_4	7.7
NH_3	482
Cr	0.024
Cu	0.061
Fe	3.460
Mn	0.280
Pb	0.020
Ni	0.030
Zn	0.230

Table 2.8 Leachate pre- treatment chemical analysis results (ppm).

Treatment Component	BOD_5	TOC	TPO_4	TKN	NH_3
Leachate	180	395	7.7	526	485
Leachate/Septage	45	525	4.7	22.9	13.3
Compost	101	120	4.3	17.1	21.7
Wetlands 1	23	62	3.4	9.1	4.2
Wetlands 5	5	38	2.5	4.4	0.45
Wetlands 9	4	33	2.0	3.8	<0.1
Sand Filter	4	28	1.6	3.8	<0.1
Removal Percentage	**97.7**	**94.6**	**69.3**	**79.0**	**98.3**

Table 2.9 Leachate post-treatment chemical analysis results (ppm).

This study shows the ability of a constructed wetland to successfully remove ammonia to a high level of efficiency. This system was unable to remove phosphates to a similar level of efficiency.

Constructed Wetland for the Treatment of Leachate at the Chunchula Landfill, Mobile County, Alabama

Johnson *et al.*(1999) carried out research on the use of a constructed wetland to treat landfill leachate. They state that a 1.29 ha surface-flow wetland receives leachate from a capped but unlined landfill. The treated leachate is mixed with storm water and released into a receiving water.The discharged effluent meets established state and federal water quality standards through monthly analysis for BOD_5, COD, pH, TOC, NH_3 and heavy metals including Ar, Cd, Cr, Cu, Pb, Hg, Ni, Ag and Zn.

The authors also state that the wetland was designed to treat both the leachate and leachate contaminated groundwater up to a capacity of 5 X 10^2 g/day of influent but was frequently overloaded by a factor of 20 without any noticeable change in effluent quality shown in Table 2.10 Chemical analysis of influent and effluent results.

	Influent (W-1)	Effluent (W-7)	% Removal
.pH (s.u.)	6.32	6.86	-
TSS	1008	30	97
TDS	1078	396	63
COD	456	45	90
TOC	129	17	87
Cu	0.05	0.024	52
Pb	0.078	0.004	94
Hg	0.0019	0.0019	0
Ni	0.082	0.01	88
Zn	0.08	0.03	62

Table 2.10 Removal efficiencies for parameter (mg/l) Chunchula Landfill Wetlands.

An Integrated System for Landfill Leachate Treatement at Isanti-Chisago, Minnesota.

Loer *et al.*(1999) conducted research at the Isanti-Chisago Sanitary Landfill near Cambridge Minnesota on an integrated solution for treating leachate from an unlined municipal solid waste facility.

This landfill closed in 1992 and leaching of soluble wastes had contaminated the surface and ground waters with toxic organic compounds and heavy metals.

The site consisted of an estimated 315,000 m^3; it was in operation from 1973 to 1992 and covered 8.9 ha.

The selected systems incorporated a series of sedimentation ponds, one with a 6 day residence time for a flow of 600 m^3/day and a depth of 1.2 m. The other is a 0.6 ha free-water surface constructed wetland with a 3 day residence time at 600 m^3/day, average free water depth is 30 cm.

The wetland is lined with a soil covered polyethylene liner.

Additionally a water cascade removes volatile organic compounds (VOC's).

Power supply to the system is from solar and wind energy while the topography is utilised to aid gravity flow.

Treatment occurs as a function of the macrophytes, principally cattails or bulrush *Typha angustifolia* and *T. latifolia*, as distinct from the more usual common reed *Phragmities australis.* Continued treatment is a function of aeration, sorption, biological storage and transformation, and the trapping of solids.

Parameter	Average System Removal Efficiency (%)	Average System Removal Rate (hg/ha/season)
Volatile Organic Compounds	97	0.081
Iron	97	3.8
Zinc	93	0.044
Manganese	91	0.36
Arsenic	89	0.0064
Lead	80	0.00021
Mercury	75	0.000037
Chromium	67	0.001
Cadmium	65	0.00056
Nickel	19	0.013
Copper	-	0.00056

Table 2.11 First-year Removal Characteristics

A Constructed Wetland System for Treatment of Landfill Leachate, Monroe County, New York.

Eckhardt *et al.* (1999) conducted research into a wetland system that was constructed in 1995 to evaluate the treatment of landfill leachate. The system consisted of a surface-flow bed of topsoil and a sub-surface bed of pea-sized gravel. Both contained *Phragmites australis* reeds and were designed for leachate application at a rate of 1.8m^3 per day.

The chemical analysis data demonstrate that removal rates for 14 constituents ranged from 49% to 100%. The removal rates were highest for total phosphorous, BOD and VOCs. Removal rates were lowest for major inorganic ions including copper, nickel, zinc and barium. Outflow from the wetland system met New York State discharge regulations for all constituents except ammonium, phenol, magnesium, nickel, sodium and dissolved oxygen. The results indicate the total iron removal was 98%, and load reduction for most metal species was facilitated by oxygen from rainfall, aeration, plant input, especially the surface flow bed. Total phosphorous removal was 99% and was mainly through the uptake and concentration in plant tissues, especially the rhizomes. Nitrogen removal (mostly as ammonium) was 91% but insufficient to meet the level required by New York State discharge regulations

The authors explain that Ammonium is often the most persistent constituent encountered in leachate remediation. Ammonium concentrations in wastewater discharges are limited by the State to prevent excessive DO demand and toxicity to fish and invertebrates in the receiving waters. Ammonium concentrations from wastewater treatment wetlands in North America have ranged from 0.01 to 23 mg/l. Treatment efficiencies in these systems typically are about 90% for ammonium, as was observed in this study. Nonetheless, the ammonium concentrations in the sub-surface flow outflow in this study were unacceptably high and consistently exceeded the state limit (2 mg/l) by a factor of 10 or more. Knight *et al.* (1993) suggest that the efficiency of ammonium removal declines as inflow concentrations increase or when inflow-loading rates exceed 20kg/ha/day. The loading rate for ammonium in this study was 7.7 kg/ha/day, and inflow concentrations ranged from 14 to 400mg/L. However, mineralization of organic nitrogen, which was not measured but has been inferred to be present in the leachate, likely occurred in the wetland. The mineralization of organic nitrogen to ammonium would effectively increase the loading of ammonium to the wetlands beds and decrease its treatment efficiency.

Nitrification and plant uptake are primary removal mechanisms for the ammonium (Kadlec and Knight, 1996). Some ammonium is nitrified where oxygen enters the system, and a net increase in nitrate load measured between the inflow and sub-surface flow outflow. Nitrite and nitrate were detected in only 1 of 18 inflow samples, but both were detected at concentrations less that 7mg/l in outflow samples while nitrite was in 5 of 18 samples and nitrate was in 11 of 18 samples at the surface flow outflow. Nitrite was in 4 of 18 samples and nitrate was in 7 of 18 samples at the sub-surface flow outflow. Nitrate concentrations exceed nitrite concentrations in all samples, but the difference was greater at the sub-surface flow outflow than at the surface flow outflow. The presence of nitrite in an anaerobic environment indicates some nitrogen removal through denitrification and outgassing. The loss of ammonia through volitilization is assumed to be nearly negligible, however, because dissolved ammonia (gas) concentrations are insignificant at the nearly neutral pH of the leachate. The sum concentration of nitrite and nitrate in outflows was always less that the state limit of 10mg/l.The implications of this case study are that the previous sizing of the constructed wetland to take 20kg/ha/day ammonium may have to be revised in that this system was unable to adequately treat 7.7kg/ha/day.

However this study again confirms the ability of constructed wetlands to treat ammonia and phosphates to a high level of efficiency, particularly surface flow constructed wetlands.

Chapter 3

Options for the Treatment of Leachate at Muckish.

A comparison of Constructed Wetlands and other Treatment Options.

Constructed Wetlands

To determine the size of a constructed wetland the following equations from publications can be used;

In Cooper (2001), a calculation per population equivalent (PE) is given as $0.7m^2$ The estimated PE for Muchish is based on 220 l per day and an average of leachate flow of $18m^3$ gives a PE of 82. At maximum flow rates of $37.3m^3$ per day gives a PE of 169. Therefore the estimated area based on an average PE is 0.7*82 = 57 m^2 or based on a maximum PE of 169 *0.7 = 241 m^2

In Loer *et al.* (1999) the authors cover the issue of the size of their constructed wetland system for treating ground water contaminated with landfill leachate, which is $6000m^2$ for a total of $316m^3$/day. This gives a figure of approximately 18 m^2 per m^3. To extrapolate this figure into the Muckish scenario with a maximum daily flow rate of $37.3m^3$ the expected size of the constructed wetland would be 18 * 37.3 = $671m^2$.

In personal communication with Paul Carroll Waterford County Council he advised that the best practice at present for an integrated wetland was to use a figure of $20m^2$ per P.E. Using this figure with a PE 82 the size of the wetland would be 82 *20 = 1640 m^2.

At maximum flow rates of $37.3m^3$ per day a PE of 169 is given, which would require 169 * 20 = 3380 m^2.

According to Kadlec and Knight (1996) a 14-day retention time is required to reduce nutrients in a surface flow wetland by 90%. The average daily flow in Muckish is $18m^3$ per day, with the maximum flow at $37.3m^3$ per day would give an estimated retention of average flow rate of 18*14 = $252m^3$.

Using the Reed *et al.* (!995) Field test method No. 2 calculations a predicted size would be 687 m^2.

Looking at the various construction size options from the above examples at a maximum flow of $37m^3$ the estimated sizes are within a wide range:

Cooper (2001)	241 m^2
Reed *et a.l* (1995) Field Test Method No.2	686 m^2
Loer (1999)	671m^2
Paul Carroll, personal communication	3380 m^2

Significantly on the Muchish site there is approx 26,000 m^2, of which 3,600 m^2 is presently unused ground at the lower end of the site where the leachate at present is lodged.

Even at the largest projected figure there is sufficient area available to build a constructed wetland which would allow for the removal of nutrients.

Construction costs.

In the publication of Loer *et al* (1999) the authors cover the issue of costs for an integrated constructed wetland for landfill leachates in 1995 and they identified the following costs:

$40,000 Treatability testing

$95,000 Design costs

$550,000 Bidding and Construction costs

Giving total set up costs of $685,000 for a 6000$m^2$ constructed wetland.

This averages out as $114 per m^2

Annual Operation Costs are given as $20000 for the first year, this averages out at $3.3 per m^2

Using the present currency rate of $ 1.2 to the € and a maximum size of 3380m^2 the estimated costs would be:

Construction = € (0.83*114) * 3380 = €319,815, depreciated over 20 years gives an annual cost of €15,990

Annual operation costs = € (0.83 * 3.3) * 3380 = €9257

Total construction, operating and depreciation costs (without interest charges for borrowed capital) = €25247 per annum

Filtration System

These figures could then be compared to the construction of a similarly sized filtration system, these figures were calculated by Alan Wright of Bord na Mona for a Puraflo for 6570 m^3 per annum flow rate with an average daily flow of 18 m^3 and a maximum daily flow of 40 m^3 and influent ammonia level of 55mg/l. The expected effluent ammonia level should be <15mg/l with BOD<20 mg/l and SS < 30mg/l.

Construction costs = € 35,000, depreciated over 20 years = €1750

Annual operation costs € 1700

Total construction, operating and depreciation costs (without in interest charges for borrowed capital) = €3550

Treatment of site by Donegal County Council.

Finally the charge to businesses by Donegal County Council for treating waste water as adopted by the council members in 2004 budget is €0.505per $m^{3.}$

Therefore the cost of collection and transporting this landfill leachate to a Donegal County Council waste water treatment facility would be expected to be 6570 *€0.505 = €3,318 per annum.

However the ability of any of the present Donegal County Council wastewater treatment facilities to successfully treat this waste is of concern as they are already operating at capacity.

Chapter 4.

Conclusions.

This study explained how Donegal County Council has been left with a legacy of dealing with disused landfills, the legal implications of non-compliance and present situation regarding existing operating landfill capacity.

The original purpose of this study was to determine if the leachate from a disused landfill could be successfully treated by a constructed wetland. As part of that study this option should be compared against other alternative treatment options such as tankering the leachate off-site to a wastewater treatment facility or installing a purpose built filtration system.

Local authorities in endeavouring to fulfil their legal environmental obligations also have to ensure that the most cost effective option is used and in coming to a conclusion on which option for the treatment of landfill leachate it cannot be forgotten that other environmental factors have to be considered. Such other factors include the effects of an increase in heavy vehicular traffic to a remote site, the visual impact of power lines on a site where there is no electricity present or the noise implications of providing electricity by a mobile generator. However in a remote site like this Muckish site in Co. Donegal, which has no electricity supply, the greatest single consideration is the environmental impact of the building of a filtration system that will visually impact on the landscape.

It is evident that a compromise will have to be reached to ensure that some form of satisfactory treatment occurs and what is known is that constructed wetlands have successfully treated wastewater effluents from farms in Ireland and have been successful in treating landfill leachates in the United States. Even the most stubborn of landfill leachate constituents, ammonia and phosphates can be removed if the constructed wetland is surface flow and of a sufficient size. This study provides case studies to verify the ability of constructed wetlands to treat landfill leachate. At even at the largest sizing suggested there is adequate space available. Additionally a constructed wetland will be low maintenance with limited power requirements and will become a natural part of the landscape

The problems with constructed wetlands are that they do have a large footprint and the time taken for the macrophytes to become established has to be considered. The costs of construction have to be considered against other options as well as considering the expected decline in leachate strength over time.

The option of installing a purpose built filtration system like the one already in existence at Drumaboden has merit in that the existing one is operating successfully. Significantly it has a smaller footprint that a constructed wetland.

However it is visually obtrusive and while it can be built at a lower level to hide it, it has a power requirement along with maintenance needs not associated with a constructed wetland. This power requirement in itself may also be overcome with on-site generators or solar power generators being able to provide sufficient power to drive the systems pumps.

A final option of tankering the leachate off site is the short-term solution and has already started. It appears to be cost effective and in reality it will involve only one or two tankers loads per day at most. Once again the decreasing strength of the leachate over time will reduce the strengh of the leachate being transported.

The problems with this option are the lack of capacity in the county to successfully treat leachate in any of the existing wastewater treatment facilities and the unknown environmental impact of heavy vehicles operating in a remote environment.

Therefore having examined the problem, the various viable options and their documented success a constructed wetland would appear to be the best environmental option for Co. Donegal over an extended period of time. In the short time the option of trucking the leachate off-site has begun so as to prevent further prosecution and limit the impact of this site to the environment.

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Appendix A

Results from Anne Valley ICW's

The following represents the results and conclusions prepared by Mr Paul Carrol Waterford County Council.

pH

Site	1	2	3	4	5	6	8	9	10	11	12	13
INFLUENT – pH												
Min	4.0	6.6	6.3	6.5	6.7	6.6	6.5	4.9	6.5	3.8	4.9	5.1
Mean	6.4	7.3	7.1	7.4	7.3	7.6	7	7.1	6.9	7.1	7.1	6.8
Std dev	*1.2*	*0.4*	*0.4*	*0.5*	*0.3*	*0.5*	*0.2*	*0.7*	*0.3*	*0.9*	*0.7*	*0.5*
Max	7.6	7.8	7.6	8.9	7.9	8.2	7.4	7.1	7.4	8.9	8.4	7.8
EFFLUENT – pH												
Min	7.0	6.7	7.0	7.1	6.7	6.7	7.0	7.0	6.5	6.9	6.6	6.5
Mean	7.5	7.4	7.5	8	7.4	7.8	7.6	7.7	7.2	7.8	7.1	7.3
Std dev	*0.4*	*0.4*	*0.6*	*0.7*	*0.6*	*0.6*	*0.5*	*0.4*	*0.4*	*0.6*	*0.4*	*0.5*
85% ile	7.6	7.8	7.7	8.9	7.7	8	7.9	7.9	7.6	8.3	7.6	7.8
Max	8.5	8.0	9.4	9.3	9.3	9.7	8.9	8.6	8.0	9.0	8.3	8.0

Effluent pH values are in the mid range and slightly basic. The relatively high maximum values in the effluents are indicative of plant and algal growth within the wetland system.

NITROGEN (Ammonia and Nitrate)

UWW Standards: 15 mg/l Total nitrogen, (10,000 – 100,000 p.e. if discharged to sensitive area)

Site	1	2	3	4	5	6	8	9	10	11	12	13
Influeg,nt - Ammonia mg/l N												
Mean	131	45	40	91	39	33	18	36	21	29	275	64
std dev	*190*	*44*	*29*	*57*	*56*	*34*	*67*	*44*	*42*	*41*	*364*	*227*
Max	651	202	124	217	158	132	302	190	182	188	1473	969
Effluent Ammonia mg/l N												
Mean	0.1	0.3	0.2	2.9	0.5	0.2	0.2	0.3	0.3	0.3	0.2	0.1
Std dev	*0.1*	*0.6*	*0.6*	*4.4*	*0.9*	*0.3*	*0.2*	*0.4*	*0.5*	*0.6*	*0.5*	*0.1*
85% - ile	0.3	0.2	0.3	7.6	1.2	0.3	0.4	0.7	0.6	0.5	0.4	0.1
Max	0.4	2.6	2.2	13.6	3.5	1.2	0.7	1.7	1.9	1.8	2.0	0.4
Effluent – Nitrate mg/l N												
Mean	2.5	4.3	1.3	2.6	2.6	0.9	0.2	1.1	1	3.5	1.7	1
Std dev	*1.8*	*4.1*	*2.0*	*4.0*	*2.6*	*1.4*	*0.6*	*1.4*	*1.2*	*4.3*	*2.1*	*1.5*
85% ile	4.4	8.3	2.1	4.6	5.6	2.3	0.7	2.1	2.6	6.9	3.5	2.5
Max	7.1	12.6	8.2	15.5	6.9	3.8	1.2	4.0	2.8	17.1	5.7	4.4

Nitrate was not measured in the influents, but would be expected to be low due their anoxic nature.

The nitrogen standards for UWW are calculated as an annual mean. TN was not measured in this part of the monitoring programme, but it can be seen that the average combined ammonia and nitrate levels for all effluents are well below the UWW limit of 15 mg/l.

Molybdate Reactive Phosphate (MRP)

UWW Standards: 2 mg/l Total phosphorus, (10,000 – 100,000 p.e. if discharged to sensitive area)

Site	1	2	3	4	5	6	8	9	10	11	12	13
INFLUENT mg/l P												
Mean	86	14.7	19	26	13.5	11	1.5	11.4	6.6	7.8	83	23
Std dev	*227*	*7*	*10*	*13*	*23*	*10*	*1*	*27*	*9*	*10*	*89*	*91*
Max	918	28.3	31.7	60.0	99.3	40.0	5.0	125	42.6	51.0	300.0	375.9
EFFLUENT mg/l P												
Mean	0.1	0.2	1	1.4	0.3	0.1	0.04	0.4	0.1	0.8	0.1	0.05
Std dev	*0.3*	*0.2*	*0.8*	*1.4*	*0.8*	*0.2*	*0.1*	*0.2*	*0.1*	*0.3*	*0.3*	*0.1*
85% ile	0.1	0.4	2.1	3.1	0.2	0.2	0.1	0.7	0.2	1.4	0.2	0.1
Max	1.3	0.7	2.5	4.7	3.0	1.0	0.3	0.9	0.3	2.3	1.3	0.3

The phosphorus concentration in the wetlands effluents is critical, given the role of P in causing eutrophication, and the requirements of the Phosphorus regulations. Studies carried out during the monitoring indicate a 2:1 relationship between TP and MRP, giving an effective limit of 1 mg/l MRP, to comply with UWW standards. 11 of the 12 ICWs had MRP concentrations meeting this UWW standard.

BOD

UWW Standards: 25 mg/l O2 – 85% compliance required(when sampling frequency is 8-16 samples per annum)

50 mg/l O2 – 100% compliance required

Site	1	2	3	4	5	6	8	9	10	11	12	13
INFLUENT mg/l BOD												
Mean	7058	484	443	555	266	224	62	606	69	635	2544	2308
Std Dev	*17081*	*587*	*700*	*406*	*368*	*293*	*63*	*1176*	*54*	*1166*	*2681*	*9301*
Max	57500	2517	3068	1531	1231	1261	242	4500	181	5282	10250	38402
EFFLUENT mg/l BOD												
Mean	12.3	13.6	27.5	38	20	20	13	14	16	24	18	17
Std dev	*9*	*11*	*18*	*26*	*23*	*16*	*10*	*11*	*11*	*23*	*39*	*16*
85% ile	24	24	50	64	44	39	22	30	27	33	24	35
Max	30	46	62	98	74	49	36	38	39	109	176	58

The overall average value for effluent BOD was 19 mg/l, with an 85 percentile overall value of 35 mg/l. Sites 4, 11 and 12 had high maximum values during the period, but relatively low mean and 85 percentile values

COD

UWW Standards: 125 mg/l O2 – 85% compliance required(when sampling frequency is 8-16 samples per annum)

250 mg/l O2 – 100% compliance required

Site	1	2	3	4	5	6	8	9	10	11	12	13
INFLUENT												
Mean	9261	768	1036	1404	903	622	139	952	184	1433	8133	3523
Std dev	*23482*	*724*	*1423*	*1053*	*1621*	*676*	*88*	*1681*	*151*	*2799*	*13866*	*14345*
Max	89000	3600	6000	5000	6720	2857	345	6430	590	13000	63600	61000
EFFLUENT												
Mean	28	38	78	90	43	55	33	42	42	62	40	34
Std dev	*10*	*14*	*21*	*32*	*35*	*30*	*9*	*15*	*32*	*30*	*58*	*18*
85% ile	40	50	98	119	95	70	40	55	65	87	41	53
Max	45	65	126	150	117	130	50	70	122	171	280	70

11 of the 12 ICW sites complied with UWW Standards, which require an 85% compliance value of 125mg/l, and 100% compliance value of 250mg/l.

Suspended Solids

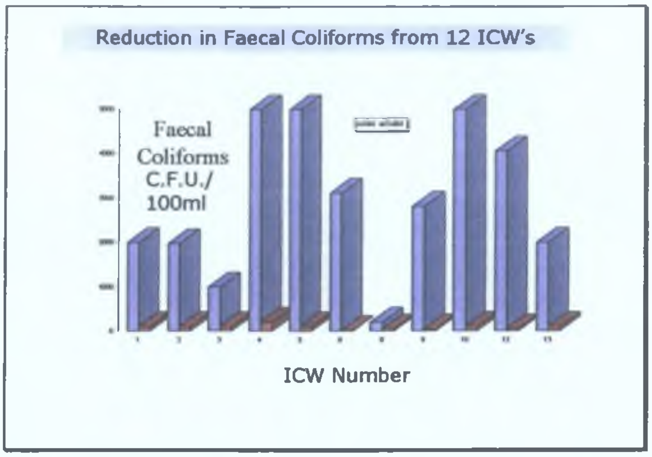
UWW Standards: 35 mg/l – 85% compliance required (when sampling frequency is 8-16 samples per annum)

88 mg/l – 100% compliance required

Site	1	2	3	4	5	6	8	9	10	11	12	13
INFLUENT												
Mean	1168	159	152	1519	133	222	45	585	116	79	3730	276
Std dev	*3277*	*121*	*157*	*3827*	*136*	*506*	*61*	*1154*	*297*	*106*	*9140*	*475*
Max	14000	432	560	17525	425	2220	216	3960	1260	357	37200	1485
EFFLUENT												
Mean	13	26	22	51	13	23	9	14	20	22	25	22
Std dev	*12*	*35*	*18*	*44*	*9*	*26*	*7*	*8*	*21*	*19*	*31*	*23*
85 % ile	22	36	38	82	21	37	16	22	38	48	33	51
Max	53	154	64	190	40	101	26	32	74	60	112	68

The overall average value for effluent suspended solids was 22 mg/l, with an 85 percentile overall value of 37 mg/l. Sites 2, 4, 6 and 12 had high maximum values over the period, but relatively low mean and 85 percentile values.

Faecal Coliforms



The wetlands are capable of achieving very significant reduction in bacterial contamination levels. From the above data, it can be seen that the faecal coliform counts went from a range of 1000 to over 5000 per 100 mls in the influent, to less than 100 per 100 mls in the effluents from each ICW. These results are from one grab sample at each wetland, but project results indicate that such performance is consistent over time and at each site.

Receiving Water Quality

The Annestown Stream, which runs through the Anne Valley, was monitored over the period as part of the project. Water quality results are presented below for the three sampling sites along the length of the river.

Sampling site	Ballyphilip Bridge			Castle (Br W of Dunhill Lodge)			Monument		
Approximate distance from sea (km)	3.5			2			1		
	mean	*median*	*max*	*mean*	*median*	*max*	*mean*	*median*	*max*
Ammonia mg/l N	0.04	0.04	0.09	0.03	0.03	0.07	0.05	0.04	0.13
Nitrate mg/l N	4.3	4.3	5.9	5.0	4.9	7.0	4.4	4.6	6.3
Phosphate mg/l MRP as P	0.03	0.03	0.07	0.02	0.02	0.09	0.03	0.03	0.09
BOD mg/l	2.1	1.3	6.2	2.2	1.8	4.8	2.3	1.9	5.4

Water quality was satisfactory at all sampling stations.

The Biological Q rating, assessed by the EPA in 1999 and 2001, indicates an improvement in water quality at Ballyphilip Bridge– from Q2 to Q3-4 - since the installation of the ICWs in the Anne Valley. The Q rating at Br W of Dunhill Lodge was unchanged (Q 3-4) between surveys. The next biological survey is due to be carried out in 2004

Conclusions

Results to Date

The above results on a variety of farm ICWs, some of which are non-compliant with design and planning requirements, indicate the overall efficiency of the ICW process, and the general robustness of the system. Further analysis of individual ICWs should enable the design team to isolate the particular factors affecting performance at the various sites, and allow the draft ICW Protocol document to be developed.

Receiving Water Quality

As indicated above the Annestown Stream into which the effluent from all of the 12 monitored ICWs eventually flows, has shown an improvement in water quality since the installation of these ICWs. This is demonstrated by the upward movement in the EPA quality rating for this stream, and by the recent return of sea trout to the stream after an absence of many years.

APPENDIX B

RESULTS FROM THE MONITORING OF LEACHATE SAMPLES AT ARTHURSTOWN LANDFILL FACILITY.

Results of Chemical Analysis of Leachate Samples.

Parameter	Av. Cells	Storage Tank	Aeration Tank	Balance Tank	Reed Bed	Lagoon
.pH	7.55	7.7	7.8	8.0	8.4	9.2
Temp (°C)	16.25	16.9	29.2	26.8	9.4	8.6
Conductivity	26,437	27320	17010	17440	14130	1433
Odour	Strong Odour	Strong Odour	No Odour	No Odour	No Odour	No Odour
Visual Inspection	Dull/Black Colour, Frothing Present	Black Colour, SS Present	Brown Colour, SS Present	Brown Colour, SS Present	Red Colour, SS Present	Yellow Colour, SS Present
BOD_5 – TCMP (mg/l)	1590	7000	800	<40	<40	10
COD (mg/l)	5223	11490	9370	858	625	69
Ammonia –N (mg/l) ISE	2100	2250	0.7	0.4	0.4	<0.2

Parameter	Av. Cells	Storage Tank	Aeration Tank	Balance Tank	Reed Bed	Lagoon
Ammonia – N (mg/l) Konelab	2152	2175	0.77	0.43	0.32	<0.02
TSS (mg/l)	162.5	190	11680	64	<5	31
Total Organic Carbon (mg/l)	3204	5630	4319	47	444	27
Chlorine	1627	2366	5004.1	2689.3	2230.8	147.1
Nitrate – N (mg/l)	1.34	1.11	259.3	894.1	778.3	81.4
Nitrite – N (mg/l)	<7.5	0.36	0.02	0.17	0.04	0.52
Total Oxidised Nitrogen (mg/l)	3.2	1.47	259.32	894.18	778.34	81.92
Sodium (mg/l)	2352	2489	3728	2681	3147	274
Potassium (mg/l)	1847	3229	4262	3180	3584	168
Calcium (mg/l)	48	259	1716	173	169	28
Iron (mg/l)	9	24	534	4.5	3.9	266
Copper (mg/l)	21	3	194	13	29	<2

The conclusion on results from Arthurstown reed bed system is that there is that it makes no significant difference, except for a reduction in TSS and an increase in TON. This minimal impact of a reedbed on landfill leachate is to be expected when the reedbed is positioned after the aeration tank. It should also be noted that nitrate levels have also have increased.

Appendix C

Results of chemical analysis from Constructed Reedbeds for Municipal Waste Dump Leachate Treatent at Slovenj, Gradec, Slovenia.

Month	TSS Influent	TSS effluent	COD influent	COD effluent	BOD_5 influent	BOD_5 effluent
November	40	90	110	90		
December			190	160	60	50
January	30	17	250	210	50	60
February	50	16	90	50	60	50
March	55	11	50	47		
April	40	11	180	130	35	75
May	35	10	265	90	75	10
June	50	20	270	110	70	72
July	61	15	265	155	68	40
September	45	5	190	100	70	22
October	55	2	210	110	65	10
November	57	2	215	115	65	45
December	45	5	205	155	125	70
January	35	17	170	170	18	22
February	20	12	190	180	10	10
March	45	15	80	100	60	22
Average	44.2	16.5	183	123	54.7	39.8
Reduction % between influent and effluent	37.4		67.3		72.8	

Appendix D

Drumaboden Leachate Chemical results 2003

	pH	Temp	Electrical cond	Ammon nitrogen	COD	BOD	Dissolved oxygen	Nitrite
Inlet	7.8	18.3	1381.7	49.6	112.4	12.7	1.3	1.3
Outlet	7.8	18.3	1365.1	9.5	141.8	2.5	8.4	3.7
%Change	0.0	0.0	1.2	80.8	20.7	80.3	84.5	64.9
			reduction	reduction	increase	reduction	increase	increase

	Ammon nitrogen	Nitrate	Total oxidised Nitrogen	Phosphate PO4	Suspended Solids	Coliforms total	Coliforms faecal	Depth
Inlet	49.6	3.4	1.1	1.0	33.5	0.0	0.0	0.0
Outlet	9.5	43.3	10.9	0.4	13.0	0.0	0.0	0.0
%Change	80.8	92.1	89.9	60.0	61.2			
	reduction	increase	increase	reduction	reduction			

	Iron	Lead	Magnesium	Manganese	Zinc	Potassium	Sodium	Residue on evapor
Inlet	73.2	47.8	47.0	140.9		54.2	130.3	
Outlet	115.3	5.0	58.5	57.5		62.2	134.5	
%Change	36.5	89.5	19.7	59.2		12.9	3.1	
	increase	reduction	increase	reduction		increase	increase	

	Residue on evapor	Chloride	Sulphate	Mercury	Boron	Flouride
Inlet		121.7	65.5	0.1	0.1	0.1
Outlet		128.7	3.4	0.1	0.1	0.1
%Change		5.4	94.8	0.0	0.0	0.0
		increase	reduction			

Appendix E

Muckish Leachate Chemical Results 2003

All reults in mg/l except were stated

Date	Location	Sample type	Site No	Lab No	pH	Temp	Electrical cond us/cm	Ammon nitrogen	COD	BOD	Dissolved oxygen	Residue on evapor	Calcium
20-Feb-03	Muckish	leachate	L1	644	6.47	12.95	642	58	79	15.8	3.66	nd	170.59
09-Apr-03	Muckish	leachate	L1	1357	6.65	17.74	877	58	5.68	2.95	<0.03	<0.04	<0.01
08-May-03	Muckish	leachate	L1	1733	6.55	17.82	602	25	98	18.2	2.623		
26-Jun-03	Muckish	leachate	L1	2636	7.18	20.87		51	194	1.72	3.78		
14-Jul-03	Muckish	leachate	L1	2830	6.95	20.48	772	83	252	11.44	2.16		
17-Sep-03	Muckish	leachate	L1	3880	6.59	19.91	1110	61		2.87	2.98		
29-Oct-03	Muckish	leachate	L1	4499	6.52	14.57	1210	58	370		10.54		
16-Dec-03	Muckish	leachate	L1	5174	7.18	13.21	1614	46	158	6.08	5.66		
Average					6.8	17.2	975.3	55.0	165.2	8.4	3.9	0.0	85.3

Date	Location	Sample type	Site No	Lab No	Cadmium ug/l	Chromium ug/l	Chloride ug/l	Alkalinity	Copper ug/l	Cyanide	Iron ug/l	Lead	Magnes-ium ug/l
20-Feb-03	Muckish	leachate	L1	644	1.7	4.5	116		13.2		49873	8	41.88
09-Apr-03	Muckish	leachate	L1	1357									
08-May-03	Muckish	leachate	L1	1733			180						
26-Jun-03	Muckish	leachate	L1	2636									
14-Jul-03	Muckish	leachate	L1	2830									
17-Sep-03	Muckish	leachate	L1	3880									
29-Oct-03	Muckish	leachate	L1	4499			64	320					
16-Dec-03	Muckish	leachate	L1	5174									
Average					1.7	4.5	120.0	320.0	13.2	0.0	49873.0	8.0	41.9

Muckish Appendix E (Cont)

Date	Location	Sample type	Site No	Lab No	Magnesium	Manganese	Mercury	Nickel	Potassium	Sodium	Sulphate	Zinc	Phosphate
					ug/l	ug/l	ug/l						
20-Feb-03	Muckish	leachate	L1	644	41.88	1387.5	<0.1		31.55	68.53	<10	68.2	
09-Apr-03	Muckish	leachate	L1	1357									
08-May-03	Muckish	leachate	L1	1733									
26-Jun-03	Muckish	leachate	L1	2636									
14-Jul-03	Muckish	leachate	L1	2830									
17-Sep-03	Muckish	leachate	L1	3880									
29-Oct-03	Muckish	leachate	L1	4499									
16-Dec-03	Muckish	leachate	L1	5174									
Average					41.9	1387.5	0.0	0.0	31.6	68.5	0.0	68.2	1.2

Date	Location	Sample type	Site No	Lab No	Nitrite	Nitrate	Total oxidised	Boron	Flouride	Phenol	Coliforms total	Coliforms faecal	Depth
20-Feb-03	Muckish	leachate	L1	644	<0.03	<0.04	0.9	678.2	<0.1				3.42
09-Apr-03	Muckish	leachate	L1	1357									
08-May-03	Muckish	leachate	L1	1733	<0.03	<0.04	<0.01						4.04
26-Jun-03	Muckish	leachate	L1	2636	0.328	0.442	0.2						
14-Jul-03	Muckish	leachate	L1	2830	<0.03	<0.04	0.17						3.96
17-Sep-03	Muckish	leachate	L1	3880	0.3	1.326	0.4						3.98
29-Oct-03	Muckish	leachate	L1	4499	<0.03	<0.03	<0.04						3.61
16-Dec-03	Muckish	leachate	L1	5174									3.55
Average					0.1	0.3	0.3	678.2	0.0	0.0	0.0	0.0	3.8