David Ball.

International Association of Hydrogeologists

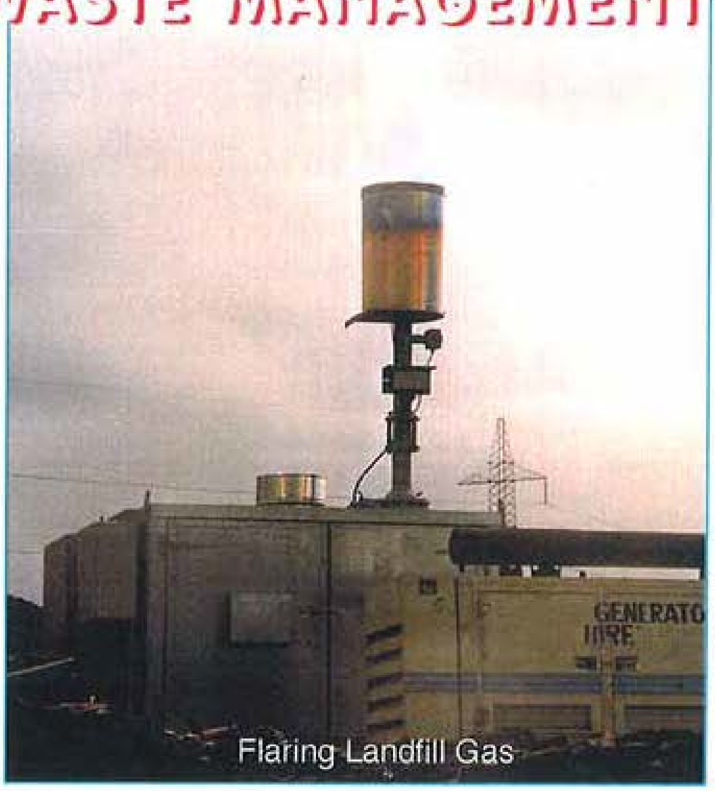
(Irish Group)

Modern Landfill

Old Mine Workings

HYDROGEOLOGY & WASTE MANAGEMENT

Agricultural Slurry



Flaring Landfill Gas

PROCEEDINGS OF THE PORTLAOISE SEMINAR

21st - 22nd April 1998

INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS

IRISH GROUP

HYDROGEOLOGY
&
WASTE
MANAGEMENT

PROCEEDINGS

OF THE

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*This paper was not available at the time of print and is therefore not bound in these Proceedings. The paper was distributed separately during the seminar.

FOREWORD

Groundwater is a valuable resource and its quality is vitally important to every household and enterprise which relies on a source of groundwater for its water supply. In Ireland groundwater of potable quality is still the norm rather than the exception and, although some of our European neighbours may still share this experience, the industrial revolution, intensive farming practices, and increasing population pressures have inevitably taken their toll. Here in Ireland we are now placing the same pressures on our groundwater resources and the increasing number of media reports related to groundwater quality problems bear witness to a general deterioration in the quality of our groundwater resources.

Poor waste disposal and handling practices are often to be linked with this deterioration. Landfill, the landspreading of agricultural wastes, the disposal of mining wastes, sewage discharges to ground, and illegal dumping are all activities which have the potential to render groundwater quality below acceptable standards unless effective waste management practices are implemented. Waste management practice must consequently incorporate the principle of groundwater protection as one of its prime tenets.

Besides recognising the often obvious potential for short term and localised impacts on groundwater quality, waste management plans must also reflect the long term implications of poor practice and adhere to the 'duty of care' principle when waste is disposed of via contractors. These considerations have resulted in the promotion of what is termed as 'cradle to grave philosophy' when dealing with the disposal of waste.

It is against this background that the theme of *Hydrogeology & Waste Management* has been chosen for the 18th Annual IAH (Irish Group) Groundwater Seminar. Selected experts covering the areas perceived to be of the most relevance in these fields have generously volunteered to give of their time for this two day seminar.

The programme of papers to be presented will deal with those issues associated with waste and waste management which are perceived to be of primary importance for groundwater in Ireland. Examples of the some of issues to be dealt with are as follows:

- landfill
- licensing
- landspreading
- waste management
- environmental health
- sewage
- mining

The IAH (Irish Group) is also continuing with its series of regional lectures. This year the *Regional Lecture* deals with Flow Modelling in the Karst Terrain of South Galway.

The IAH Irish Group take this opportunity to welcome you to its 18th Annual Groundwater Seminar and urge your participation in both formal and informal discussions during the two days.

Paper No. 1. ***Keynote Lecture:***

A Conceptual Design for Sustainable Landfill.

Nick Walker, Cleanaway Limited.

Get leachate analyses for different types of waste.

A CONCEPTUAL DESIGN FOR SUSTAINABLE LANDFILL

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Landfill will inevitably remain a fundamental component of waste management strategies of industrial nations. However, in the UK, it is Government policy to promote landfill practices that are more sustainable than at present and which will ensure stabilisation within one generation by introducing the bioreactor approach. A proposal for a sustainable high rate flushing bioreactor (HRFB) landfill is presented based on the hydrogeological and polluting properties of waste. The key components of the design are the use of pulverised wastes, maintenance of a deep saturated zone (to wet all wastes and to maintain high permeabilities) and the flushing of between 5 and 7.5 m^3 of liquid per tonne of waste in the landfill.

1. INTRODUCTION

The UK government policy (DoE, 1995a) recognises that landfill will remain a fundamental component of its sustainable waste management strategy for the foreseeable future. This is not surprising as it is inconceivable that any solid waste management policy in an industrial nation, no matter how much pre-treatment, reuse and recycling it incorporates, could avoid the final disposal to land of some solid fraction. Landfills are placed at the bottom of the hierarchy of waste management options promulgated in "Towards sustainability" (EC, 1992) the European Commission's 5th Programme of Policy and Action in relation to the environment and sustainable development, produced in response to Agenda 21 of the Rio de Janeiro Earth Summit on sustainable development. The primary emphasis of the hierarchy is on the prevention or reduction of wastes, followed by promotion of recycling and reuse, and then by optimisation of final disposal methods for waste which is not re-used. The UK Government policy is to promote landfill practices that are more sustainable than at present and which will ensure stabilisation within one generation by introducing the bioreactor approach.

A frequently used definition of sustainable development, taken from the Brundtland report "Our Common Future" is "development which meets the needs of the present without compromising the ability of future generations to meet their own needs". Put more simply, a sustainable development is one that will not pass problems onto future generations. In the context of landfill, the problems mainly relate to the polluting potential of the wastes. It follows that the high technology 'dry tomb' containment landfills of today are inherently non-sustainable (owing to practically indefinite maintenance and monitoring requirements) and it can therefore be appreciated why landfill is placed at the bottom of the waste hierarchy. If landfill can be made more sustainable and remain more cost effective in comparison with other waste management options, there could be significant benefits to society.

Whilst it is European environmental policy to encourage sustainability (EC, 1992), European legislation on waste management does not explicitly promote sustainable development as an over-riding objective . For example, in the proposal for a Council Directive on the landfill of waste (EC, 1997) sustainability is not specifically mentioned. It does include vague requirements for pre-treatment of waste before landfill and long range targets for the reduction of biodegradable municipal waste going to landfill. However, there is little recognition that significant pollution potential will remain in landfills for very protracted periods even if the reduction targets are met. The standards to be required for the engineering of landfills will lead to no environmental improvements in the UK where the amended Waste (Framework) Directive (EC, 1991) has been fully implemented for several years. Indeed, the requirement for the separation and concentration of hazardous wastes in hazardous waste sites could be considered contrary to the principles of sustainable development because (a) the pollution load may be more difficult to control and remove than say with co-disposal sites which will no longer be allowed and (b) such hazardous sites will be "few and far between" leading to unnecessary transportation not complying with the proximity principle.

In the UK, a sustainable landfill is defined (DoE, 1995b) as one which is brought to a stable non-polluting state 30 to 50 years after the cessation of landfilling activities. Gronow (1996) interpreted this to mean that a sustainable landfill would be in equilibrium with its surrounding environment and there could be confidence that no future maintenance or monitoring of the wastes would be required. This does not mean that the wastes would need to be 100% degraded or that any leachate released would have to be at drinking water quality. However, it does mean that the majority of the waste's pollution load would need to be removed within the timescale.

There are two possible strategies for achieving this:

1) pre-treatment of the waste to remove the majority of the pollution load prior to landfill or
2) *in situ* treatment of the wastes to remove the pollution load by operating the landfill as a bioreactor with a high rate of flushing.

It should be noted that many existing waste pre-treatment processes do not adequately remove the polluting potential of the residues which are ultimately landfilled. Wastes which have been pre-treated to the final storage quality required in Germany (Technical Instruction on Municipal Waste (TA Siedlungsabfall – 'TASI') 1993) would still require to be flushed, to meet the criteria for sustainable landfill. As an example, the assignment value for ammonium within Class II (non-inert) landfills is 200 mg/l, and it is recognised (e.g. Stegmann 1997) that these sites still require high standards of engineering and long term aftercare. The requirement for flushing would also apply to the concentrated inorganic pollutants contained in the ash from MSW incinerators.

This paper will concentrate on the design considerations for the *in situ* treatment option, although some of the factors discussed will also be applicable to the final methods of control of the residual pollution load in the pre-treated waste strategy.

2. SUSTAINABLE LANDFILL – CONTROLLING FACTORS

2.1 THE NATURE AND MAGNITUDE OF THE POLLUTION LOAD OF PUTRESCIBLE WASTES

In considering the polluting potential of (putrescible) wastes, a distinction can be made between the following:

1) the degradable organic carbon content of the waste
2) releasable nitrogen
3) inorganic ions

By definition, putrescible wastes contain a large proportion of degradable carbon. In the case of household wastes (MSW) the mass of degradable and releasable carbon has been estimated in laboratory scale experiments (Beaven & Walker, 1997) to be up to 185 kg per dry tonne of refuse (~130 kg/t_{wet} for a water content of 30% by wet weight). This compares with a total carbon content of 358 kg/t_{dry}.. The theoretical maximum gas yield of MSW is calculated as 370m^3/tonne, with more realistic estimates for achievable yields in the field of approximately 200m^3 /tonne (Barlaz & Ham, 1990). Assuming that the gas produced predominantly contains CH_4 and CO_2, then the total gas yield contains a mass of carbon between 107 and 198 kg/t_{wet}.

The mass of nitrogen which can be released from MSW has been estimated by Beaven & Walker (1997) to be up to 2.7kg/t_{dry} (~1.9kg/t_{wet}), compared with a total nitrogen content of 10kg/t_{dry}. Other work has indicated a range of releasable nitrogen of up to 3.9 kg/t_{dry}. (Table 1).

Whether the inorganic ion content of wastes is considered as having a polluting potential will largely depend on site location. For example, chloride is likely to have a much larger polluting potential in inland landfills adjacent to relatively small freshwater water courses, than in landfills located near to the coast. The mass of releasable chloride determined by Beaven & Walker (1997) was approximately 2.5kg/t_{dry}.

Table 1 Releasable nitrogen content of refuse determined in laboratory scale experiments

Reference	Releasable N per tonne of refuse kg/t	Units/Comments	Waste Stablilised
Knox & Gronow, 1995	1.3	Wet weight 2 year old MSW	No
Ehrig & Scheelhaase, 1993	1.6	Wet weight	?
Burton & Watson-Craik, 1997	~3.9	Dry weight 1-2 month old refuse Total N content ~ 4%	?
Heyer and Stegmann, 1995	1.8	Wet weight 8 year old MSW	?
Heyer and Stegmann, 1995	0.7	Wet weight 13 year old MSW	?
Brinkmann, et al 1995	2	Dry weight Milled MSW	No
Beaven & Walker, 1997	2.7	Dry weight Shredded MSW	Probably

2.2 COMPLETION CRITERIA

In the UK, the "completion condition" for a landfill is when the condition of the land is unlikely to cause pollution of the environment or harm to human health (DoE, 1993). This is translated into requirements for leachate quality related to drinking water standards (assuming a degree of dilution), though a site specific risk assessment is advised in preference to prescriptive standards.

2.3 MECHANISMS FOR THE RELEASE AND REMOVAL OF THE POLLUTION LOAD OF MSW

To reduce the pollution load of solid wastes in a landfill, the pollutants must first be transformed into a gaseous or liquid phase for subsequent removal by gas and leachate extraction. Within the context of a sustainable landfill, the removal of the load should be achieved within the timescale associated with one generation. It has been demonstrated (e.g. Beaven & Walker, 1997) that methanogenic gas production is required to remove the majority of the degradable organic carbon of MSW. To some extent, methanogenesis and biodegradation are also likely to be required to release nitrogen from the solid into the liquid phase. Techniques to accelerate gas production in landfills are relatively well understood. These include the use of shredded refuse, raising the water content of wastes and the introduction of buffering capacity (e.g. Campbell, 1997; Knox, 1996a). This paper, therefore, concentrates on the issues relating to the pollution load which has to be removed from landfills through leachate flushing and extraction.

Leachate flushing is required to remove the pollution load associated with nitrogen, other inorganic ions (such as chloride) and the residual fraction of organic carbon not removed by landfill gas production. The volume of liquid required to flush pollutants from a landfill is a subject which requires considerable further research. Assessments to date (e.g. Belevi & Baccini, 1989; Knox, 1990 & 1996b; Walker 1993) have been based on a flushing model which assumes that landfills operate as continuously mixed reactors. In this 'washout' model, any fluid which is introduced into the landfill is assumed to mix instantaneously with the 'bed volume' (the reservoir of water or leachate) existing in the site. Where clean water is introduced, uniform mixing and dilution of the leachate is assumed. The reduction in leachate concentration is related to the number of bed volumes of water which have been passed through the landfill. The passage of 4.6 bed volumes of fluid is required to reduce leachate concentrations by two orders of magnitude (e.g. from 1,000 to 10 mg/l). Knox (1996b) suggests that the behaviour of landfills correlates reasonably well with the complete mixing model and the theory therefore forms a useful starting point from which to make predictions about how a site will behave. However, it should be recognised that the theory only applies to conservative parameters where, during washout, there is no net addition to, or removal from, solution.

The number of bed volumes removed can be translated into the volume of fluid required to flush a unit mass of refuse. Estimates for the volume of water required to flush the nitrogen polluting load from waste range from 5 to 7.5 m^3 per $tonne_{wet}$ of waste (Beaven, 1996; Beaven and Walker, 1997).

3. PROBLEMS IN DESIGNING A HIGH RATE FLUSHING BIOREACTOR LANDFILL

The successful development of a HRFB landfill requires a number of potential problems to be overcome.

3.1 WATER SUPPLY

The volumes of water required to flush a landfill may be large. For example, a 1 million tonne MSW landfill could require between 5 and 7.5 million m^3 of water to remove its pollution load. If the load were removed over a 50 year time period, then the daily volume of water required would be between 274 and 411 m^3. Though the quality of the flushing water would not have to be high, the availability of this quantity poses obvious problems and has implications for the siting of landfills.

3.2 INTRODUCTION, RECIRCULATION AND COLLECTION OF FLUSHING FLUID

Assuming that every m^3 of refuse in a landfill of depth d must be flushed through by 5 m^3 of liquid, then the volume of leachate to be removed per unit area of landfill (m^3/m^2) = 5 d (m^3) per m^2. If the flushing is to be achieved over a period of 50 years, the required flushing rate is given as:-

$$\text{Flushing rate (m/a)} = \frac{\text{volume to be removed } (m^3/m^2)}{50 \text{ (years)}}$$

$$\text{Flushing rate (m/a)} = \frac{5 \times \text{depth of landfill}}{50} = \frac{\text{depth of landfill}}{10}$$

Therefore, a 30 metre deep landfill would require a flushing rate of 3 m/annum and a 60 metre deep site a rate of 6 m/annum for 50 years. The flushing rate for a 60 metre deep landfill would be increased to approximately 10 m/annum if either the flushing period were reduced to 30 years, or if the volume of fluid required were increased to 7.5 m^3/tonne.

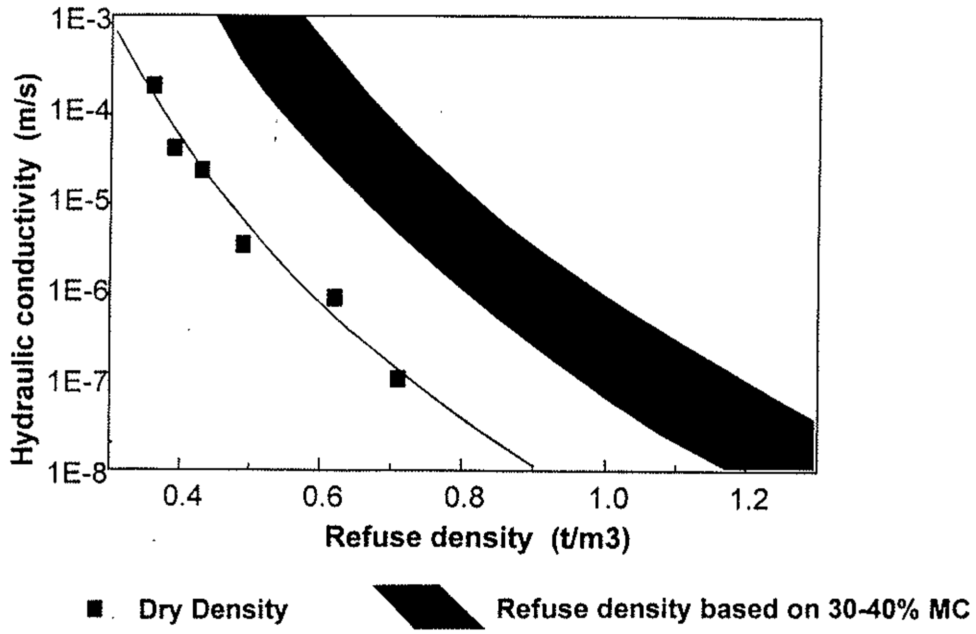
Whereas the required rate of infiltration increases with the depth of landfill, the permeability of the waste tends to decrease with increasing depth of burial. This means that as landfill depths increase and higher flushing rates are required, it becomes progressively more difficult to recirculate fluid through the site. Results of work undertaken by Beaven & Powrie (1995), who used a large scale compression cell to measure the hydraulic conductivity of wastes at various applied stresses and equivalent depths of burial, are reproduced in Table 2. The equivalent depths of burial have been calculated assuming zero pore water pressure. The relationship between waste density and hydraulic conductivity is shown in Figure 1. Assuming that the wastes have not been pre-compacted both the density and the hydraulic conductivity may be related to the vertical effective stress (calculated in accordance with conventional soil mechanics principles by subtracting the pressure of the liquid in the pores from the applied or total stress due to overburden).

Table 2 Hydraulic conductivity of wastes

Applied Street kPa	Equivalent landfill depth (approx) metres	Refuse Type				
		Crude MSW DM2 m/s	Crude MSW DM3 m/s	Pulverised MSW PV1 m/s	Pulverised MSW PV2 m/s	Aged MSW AG1 m/s
Initial	-	1.7×10^{-4}	nd	2×10^{-4}	nd	nd
40	5	nd	3.5×10^{-5}	3.6×10^{-5}	nd	1.4×10^{-4}
87	10	nd	2×10^{-5}	7×10^{-6}	nd	3.5×10^{-5}
165	19	nd	3×10^{-6}	2×10^{-6}	nd	6×10^{-6}
322	33	nd	8×10^{-7}	1×10^{-7}	nd	5.5×10^{-7}
600	57	nd	1×10^{-7}	3.5×10^{-9}	1×10^{-9}	3×10^{-8}

The minimum hydraulic conductivity (assuming unit hydraulic gradient within the landfill) required to achieve flushing rates of 3 m/annum is approximately 1×10^{-7} m/s. This corresponds, according to Figure 1, to a dry density of crude MSW of no more than 0.7 t/m^3, and a wet density (assuming a moisture content of 30 - 40 %) between about 1.0 and 1.2 t/m^3. A flushing rate of 10 m/annum would require wastes to have a hydraulic conductivity of 3×10^{-7} m/s with a dry density of approximately 0.65 t/m^3 and a wet density (assuming a moisture content of 30 - 40 %) in the range 0.9 - 1.1 t/m^3. These densities are comparable with the wet densities of 0.8 t/m^3 to 1.1 t/m^3 achieved by typical compaction plant (Beaven & Powrie, 1996), indicating that the over use of compactors at the tipping face may result in difficulties in achieving the necessary flushing rates through the site.

Figure 1 Hydraulic conductivity vs density for MSW



The data presented in Table 2 indicate that the hydraulic conductivity of crude MSW buried to a depth of approximately 60 metres will approach 1×10^{-7} m/s irrespective of the initial degree of compaction achieved at the tipping face. The evidence also suggests that the hydraulic conductivity of pulverised waste (advocated as a means of accelerating rates of degradation) is lower at a given depth than ordinary crude refuse and reduces to below 1×10^{-7} m/s at depths in excess of about 30 m.

Little work has been carried out on the hydraulic conductivity of unsaturated wastes, although most leachate recirculation undertaken at present involves downward flow through the unsaturated zone to the leachate table. Particulate materials (i.e. soils) can have an unsaturated hydraulic conductivity at least an order of magnitude less than that of the same material when saturated (e.g. Bouwer, 1978). It can therefore be surmised that rates of leachate recirculation and flushing will be lower in unsaturated wastes than in saturated wastes. Also, gas production might interfere with the flow of liquid both above and below the nominal saturation level in the landfill.

A further problem relating to the flushing of wastes is how to achieve a uniform distribution of flow within the landfill. Barriers to flow, caused for example by low permeability daily cover, would lead to perched leachate tables and the shielding of underlying wastes from the recirculating liquid. Conversely, the inclusion of more permeable materials, such as hard-core roads, in the landfill will lead to the development of preferential flow routes.

The final major potential problem relating to the flushing of the waste is the longevity of the liquid injection and leachate extraction systems. A deterioration in the performance of such systems due to clogging, for example, could restrict the ability to achieve the required flushing rates. The microbial clogging of leachate drainage systems has been investigated by Paksy et al (in press) and Powrie et al (1997).

3.3 LEACHATE QUALITY AND FINAL TREATMENT/ DISPOSAL OF FLUSHED LEACHATE

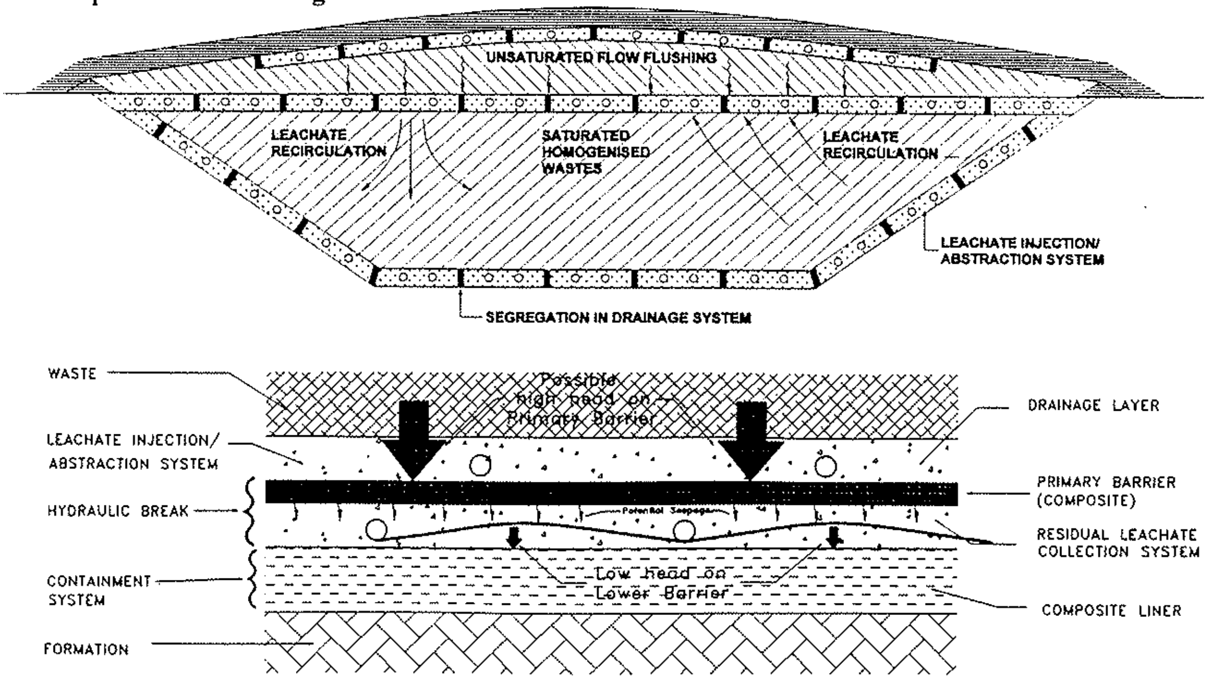
The requirement to move large volumes of fluid around the landfill may result in the development of a large reservoir of high strength acid leachate. This could inhibit methanogenesis and the release of the organic compounds into liquid and gas. Leachate treatment plants are generally designed to nitrify methanogenic leachates and are inappropriate for treating high strength acid leachates. Therefore, techniques to prevent the development of acid leachates would have to be an essential part of any HRFB landfill design.

Assuming that methanogenic conditions and leachates have been established, there are still problems with potential loading rates on any treatment plant over the lifetime of the landfill. If the landfill is flushed at a constant rate, then the pollution load on the treatment plant would reduce over time, meaning that a plant sized to treat the initial load would rapidly become oversized and inefficient. The application of a constant load to a treatment plant might necessitate a continually increasing infiltration rate, eventually leading to rates far higher than those considered in section 3.2. The disposal of flushed leachate to the local environment could pose problems, even after treatment, due to a potential lack of dilution for inorganic ions in the receiving waters.

4. DESIGN SOLUTIONS FOR AN HRFB LANDFILL

A proposed design for an HRFB landfill, which addresses many of the issues outlined above, is shown in Figure 2. A highly controlled environment would be engineered to contain an active processing unit for waste breakdown products in the solid, liquid and gaseous phases. A containment system designed to accommodate saturated conditions and high heads in the waste would likely comprise double composite liners representing a much higher standard of engineering than stipulated in the draft Landfill Directive. Different types of waste would be mixed and homogenised to create a uniform waste mass to accelerate degradation and encourage the even distribution of circulated liquids. Landfilled wastes would be rapidly saturated to provide control over the bulk hydraulic conductivity of the landfill leading to more efficient flushing. Injection, collection and recirculation systems would allow water or treated leachate to be introduced and distributed at infiltration rates equivalent to between 3 and 10 metres/annum. A total of between 5 and 7.5 m^3 of leachate would be flushed from the landfill per tonne of refuse over a timespan of 30 - 50 years. Specific elements of this design concept are discussed in more detail below.

Fig. 2 Proposed HRFB design



4.1 PRE-TREATMENT OF WASTES

Waste pulverisation is considered to be a necessary part of the HRFB design. It has two main benefits. First, pulverisation would reduce the average particle size in the waste and increase the surface area exposed to microbiological breakdown, thereby accelerating rates of degradation. Secondly, the more homogeneous waste mass would help to create relatively uniform hydrogeological characteristics. This would aid the even distribution of circulating liquids so that all parts of the waste would be flushed.

4.2 REMOVAL OF BARRIERS AND PREFERENTIAL FLOW ROUTES

To allow uniform distribution of flow, potential barriers and also preferential flow routes need to be removed. The use of low permeability daily cover would not be permitted in a HRFB landfill: alternative covers would be used, such as hessian sheets. Similarly, any operational practice which resulted in the development of preferential flow routes would have to be changed. For example, the more permeable material used in site roads would need to be removed. Wastes which have an intrinsically low hydraulic conductivity, and which cannot be processed or mixed to create a material with a bulk hydraulic conductivity greater than 1×10^{-7} m/s, may have to be separately treated or disposed of.

4.3 LIMITATIONS ON THE DEPTH OF LANDFILL

Pulverisation of the waste to create a uniform waste mass has the disadvantage of lowering the refuse hydraulic conductivity for a given depth of landfill. However, the bulk hydraulic conductivities of the refuse must be maintained above 1×10^{-7} m/s and preferably between 1×10^{-5} and 1×10^{-6} m/s in order to achieve flushing rates compatible with sustainable development. Limiting the depth of landfill to less than approximately 30 metres would be beneficial in maintaining hydraulic conductivities above 1×10^{-7} m/s and would limit the need for flushing rates in excess of approximately 3 m/annum. Deeper landfills could be successfully operated if other measures were taken to control the permeability of the wastes (e.g. see section 4.4).

4.4 SATURATION OF WASTES

An essential part of the HRFB design is the operation of the site with a deep saturated zone. Hydraulic breaks could be engineered into the double composite liner system to prevent high leachate heads being transmitted to the basal containment liner (Figure 2).

A major advantage of maintaining the waste in a saturated condition is that the positive pore liquid pressures will reduce the effective stresses. This will result in lower waste densities and higher permeabilities than in the case of an unsaturated landfill, in which the pore fluid pressures are near zero. The leachate table within the landfill would need to be raised as the depth of waste is increased, leaving only the top few metres unsaturated to allow the collection of landfill gas. It is recognised that in a saturated landfill in which flow is downwards through the waste, the hydraulic gradient will increase with depth as the permeability decreases, so that the Darcy seepage velocity (specific discharge) remains constant and equal to the infiltration rate. The experimental results of Beaven & Powrie (1995) suggest that it should be possible to landfill uncompacted, sorted and pulverised wastes to depths in excess of 30 m while maintaining an initial (undegraded waste) hydraulic conductivity of greater than 10^{-7} m/s as long as the wastes are kept saturated.

The maintenance of saturated conditions would achieve complete wetting of the waste, irrespective of its hydrogeological properties. The effect of variations in hydraulic conductivity on the efficiency of flushing should also be reduced, due to diffusion through interconnected pores. Nevertheless, problems of slope stability and leachate containment might militate against the maintenance of saturated conditions at sites where the waste rises above the level of the surrounding ground. Flushing of wastes above the surrounding ground level may have to be undertaken with a separate recirculation system based on vertical unsaturated flow.

4.5 REGULATION OF LEACHATE QUALITY AND ACCELERATION OF REFUSE DEGRADATION/ GAS PRODUCTION

A potential problem with establishing saturated conditions shortly after waste deposition is the risk of generating high strength acid leachate which then inhibits methanogenesis. Possible ways of preventing this from occurring might include:-

1) seeding of the landfill with methanogenic waste (Stegmann, 1995) and/or with wastes with buffering capacity.
2) introducing methanogenic leachate from another source (e.g. an adjacent cell)
3) introducing the liquid into the landfill via the basal collection system to create upward flow within the landfill. This would push a front of methanogenic leachate from older wastes upwards into the younger more acidogenic zones.

It may also be necessary to investigate the effects of the depth of the saturated zone on gas release once methanogenic conditions have become established. Solutions may be required to allow the release and removal of gas from the saturated zone, possibly involving the de-gassing of super saturated leachate. Otherwise the use of saturated HRFB landfills may be restricted to shallow sites

4.6 SOURCE OF FLUSHING FLUID

There is no reason why the flushing of the wastes with 5 to 7.5 m^3 of liquid per tonne needs to be undertaken with fresh water of drinking quality. If an external source of water is to be used then direct abstractions from (possibly poor quality) surface or ground waters would be acceptable. The use of effluents from sewage works would also be acceptable and a sensible use of water resources.

A proportion of the 5 to 7.5 m^3 per tonne of liquid required could come from treated leachate. Leachate treatment plants are in operation which can both nitrify and then denitrify (e.g. Robinson et al 1995). This treated leachate could then be reintroduced into the landfill to flush further nitrogen from the site. Alternatively, Knox & Gronow (1995) demonstrated in a pilot scale study that denitrification of a nitrified leachate could be supported by the residual carbon content of young waste within a landfill, without inhibiting methanogenesis. This process would avoid the need for an external source of carbon required in conventional denitrifying plants. The reintroduction of leachate into the site is likely to be beneficial in terms of maintaining levels of trace nutrients and reintroducing methanogens, both of which may encourage further degradation.

The limiting factor on the extent to which treated leachate could be used to flush contaminants from the site is likely to relate to the build up of inorganic ions in the recirculating leachate. Too high a concentration of ions may adversely affect the microbiology of the landfill, the ability to treat the leachate and the ability to discharge the treated leachate to the surrounding environment. The inorganic ions would, in any case, need to be diluted from leachate within the site to achieve the completion condition.

4.7 DESIGN AND OPERATION OF LEACHATE COLLECTION AND DISTRIBUTION SYSTEM

The purpose of leachate collection and distribution systems is to facilitate the even flushing of the wastes at the rates required for the landfill to be sustainable. This task is easier in a landfill with a deep saturated zone (section 4.4).

The flushing rates required for all but the shallowest of sites mean that the leachate collection system will have to be a drainage blanket. It is considered that leachate pumping wells are not adequate as they are too inefficient and would need to be very closely spaced to extract the required volumes of leachate.

There is, however, more scope for a variety of designs for systems to introduce leachate or liquid into the landfill. Perhaps the simplest method for achieving the even distribution of water or leachate at the top of a landfill is irrigation by rain-guns. This type of system requires that the surface of the site is relatively flat and not restored, but may be particularly suitable for flushing wastes above ground level (section 4.4.). Designs for the introduction and even distribution of water or leachate into the body of a site are likely to be based on injection trenches installed, either within a blanket of high permeability drainage aggregate, or on top of an engineered semi-confining layer, which will allow the lateral spread of leachate in combination with controlled downward seepage.

The possible benefit of an upward flow of leachate has already been mentioned (section 4.5). To allow a landfill to be operated with either upward or downward flow, the injection system (nominally at the top of the site) should be of an equal specification to the basal collection (/injection) system. Both systems should be based on a number of discrete drainage zones which can be isolated and operated independently of any other zone. This would help prevent short circuiting of leachate around the drainage system and allow control over the flushing mechanism such that hydraulic gradients could be set up in virtually any direction. This would require more pumping chambers or more complicated pipework systems than would normally be implemented at a landfill, but it would provide considerably more control over the flushing process. It would also allow a small part of the site to be flushed initially, with other parts being brought into service later on. This would help solve the problem of unequal loading on the treatment plant and allow leachate from established cells to be used to encourage methanogenesis in more recently placed refuse.

Finally, any leachate drainage or injection system located within the body of the waste is likely to suffer from differential settlement. By separating the system into discrete zones with multiple injection/abstraction points the long term integrity of the overall system is more likely to be preserved. Risks of clogging could be minimised by using aggregates with a large grain size, 20-40mm or above (Paksy et al, in press).

4.8 RESTORATION, SETTLEMENT AND PLANNING ISSUES

Sites operated on the HRFB principle will be more intensively active, but for a shorter period of time than conventional landfills, after the initial landfilling phase has been completed. Accelerated rates of settlement, together with an overriding requirement to operate and maintain systems to flush the waste, mean that final restoration, for example to a high quality agricultural afteruse, probably could not be achieved until near the end of the stabilisation period when the majority of degradation, settlement and flushing has taken place and

1) the desired final landform has been created by re-filling areas of the site, where there has been excessive or uneven settlement, with stabilised wastes (e.g. from an adjacent cell or site) and
2) there is confidence that any further settlement will be small enough not to damage the final restoration and will not alter the final landform in any significant way.

In conventional landfill sites the same amount of settlement is likely to occur as in the HRFB landfill but over a much longer period of time. The consequential problems are therefore prolonged (into a timescale measured in centuries), with ongoing maintenance and the possible need to refill low areas of the site, requiring the removal and replacement of the restoration. The HRFB landfill shortens the period over which settlement problems occur. The land is returned to a permanent and beneficial afteruse and the full amount of potential airspace is realised in a shortened timescale.

4.9 SITE LOCATION

The operation of the HRFB landfill with a large saturated zone means that there would be large volumes of potentially polluting liquid (leachate) in storage, circulation or in treatment. Together with the transfer of a significant proportion of the solid waste's polluting potential into leachate, this creates a greater hazard to the hydrological environment than conventional landfill. Although the high quality engineering of the basal liner would minimise the risk of pollution, it would still be unwise to place such landfills in sensitive environmental locations, such as upon aquifers. These active processing units would require access to significant volumes of processing liquid and ultimately the ability to dispose of large volumes of treated leachates. Therefore, sites should also be located in areas where there is an adequate supply of (possibly low grade) water and a receiving environment which can accept and dilute the residual inorganic ion concentrations of the treated leachate. Non-sensitive locations, such as on low permeability strata, near to large rivers or coastal waters would be ideal for HRFB landfills.

5. CONCLUSIONS.

Future waste management strategies in industrial nations will continue to require landfill whether for crude wastes, for the waste streams of recycling plants or for the residues of pre-treatment processes such as composting or incineration. There is a requirement to make the disposal of these types of wastes more sustainable by moving away from the 'dry tomb' approach. Landfills could be made more sustainable by removing their polluting potential in a shorter period of time (30 to 50 years) by operating them as high rate flushing bioreactors (HRFB).

To optimise the performance of a HRFB landfill, saturated conditions are required to encourage wetting and degradation. High rates of flushing are required to remove pollutants that will otherwise not be removed by gas production. It is estimated that between 5 and 7.5 m^3 of liquid are required to flush every tonne of waste. The required flushing rate (expressed as an infiltration rate) increases by 1m/a for every 10m increase in depth of landfill, to be continued over a period of at least 30 years.

To achieve the required flushing rates, the waste in the landfill must have a bulk hydraulic conductivity greater than 1×10^{-7} m/s and must also have fairly uniform hydrogeological properties to allow the even distribution of flow. To achieve these conditions, suitable wastes must be pulverised and mixed together and unsuitable (low permeability) wastes segregated and dealt with elsewhere. Relatively low in-place waste densities must be maintained, by limiting pre-compaction at the tipping face and by developing saturated conditions in the site.

The siting of HRFB landfills is important, as there must be a source of (possibly low quality) water to be used for the flushing, and adequate dilution in the receiving environment for the disposal of the inorganic ion concentration of treated leachate. Although the leachate head on the basal liner can be kept small by pumping from the basal leachate drainage layer and by the incorporation of hydraulic breaks in the composite liner system, this type of landfill should not be located directly on sensitive aquifers.

DISCLAIMER

The views expressed in this paper are those of the authors and do not necessarily represent those of any organisation.

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Paper No. 2.

Dealing with Uncertainty in the Assessment of Groundwater Impacts from Landfills: The LandSim Model.

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Dealing With Uncertainty In The Assessment Of Groundwater Impacts From Landfills: The LandSim Model

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Abstract

The LandSim model is a probabilistic performance assessment model for predicting the impacts of landfill development on groundwater. The model was developed on behalf of the Environment Agency in the UK and can be used as a tool for optimising design and undertaking impact assessments. The model can deal with a large range of liner configurations (including no liner) and incorporates the concepts of uncertainty throughout. It essentially provides a formalised method of assessing liner leakage and contaminant migration, and the impact of leachate on groundwater.

Introduction

If we remove uncertainty from all that we do there would never be a need to make a decision (or at least agonise over one). If you know that a certain set of numbers would win the lottery on the Saturday night draw you would not have to make a decision as to which numbers to select (assuming that you wanted to win). Equally, if you are a regulator and knew that the development of a landfill would cause groundwater pollution resulting in the contamination of drinking water supplies then the decision not to permit its development would be straight forward. Invariably, decisions are usually made in the absence of perfect information, resulting in some difficulty in making well informed and justifiable decisions.

Essentially, if one is to undertake an environmental risk assessment it is common practice to break the problem down into a series of compartments, normally, "source-pathway-target". In the case of the development of a new landfill each of these components contain an element of uncertainty.

This paper will address the elements of uncertainty in each component of the risk assessment and present a formalised approach to dealing with the uncertainty such that informed decisions can be made more easily.

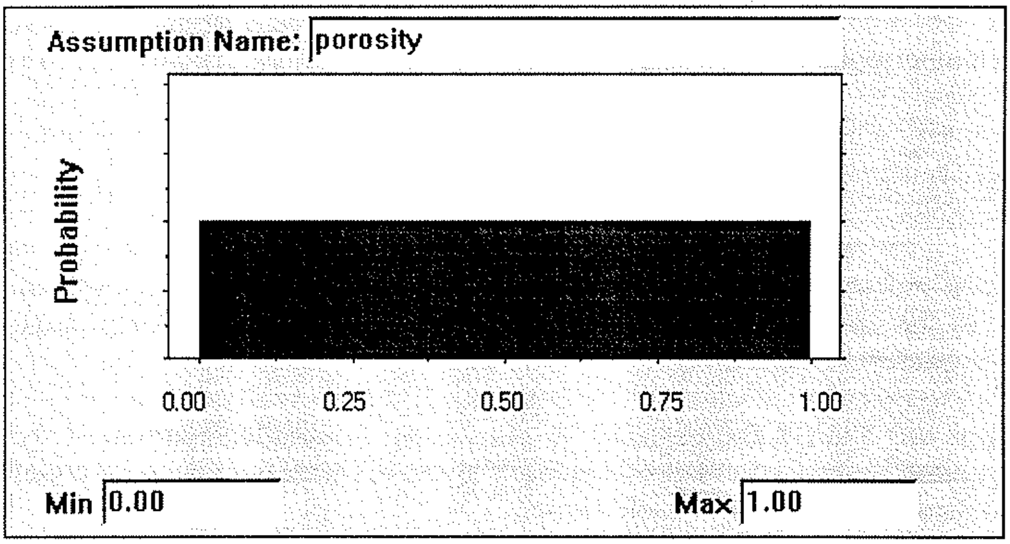
Types of Uncertainty

For some elements of uncertainty it is possible to use experiment or statistical analyses to define uncertainty. For others it is not possible. For example it is possible to conduct experiments or undertake mathematical calculation to define the probability of shaking 3 consecutive sixes from three rolls of a dice. However, there is no experimental basis for calculating, say, the value of the Dow Jones Index three years from today. To overcome this problem we could use Bayesian Probability Theory. This allows the statistical encoding of judgement into a probability distribution. It relies on a relatively simple concept that can be defined (for most risks) as:

$$Risk = \sum_{hazard} chance.outcome$$

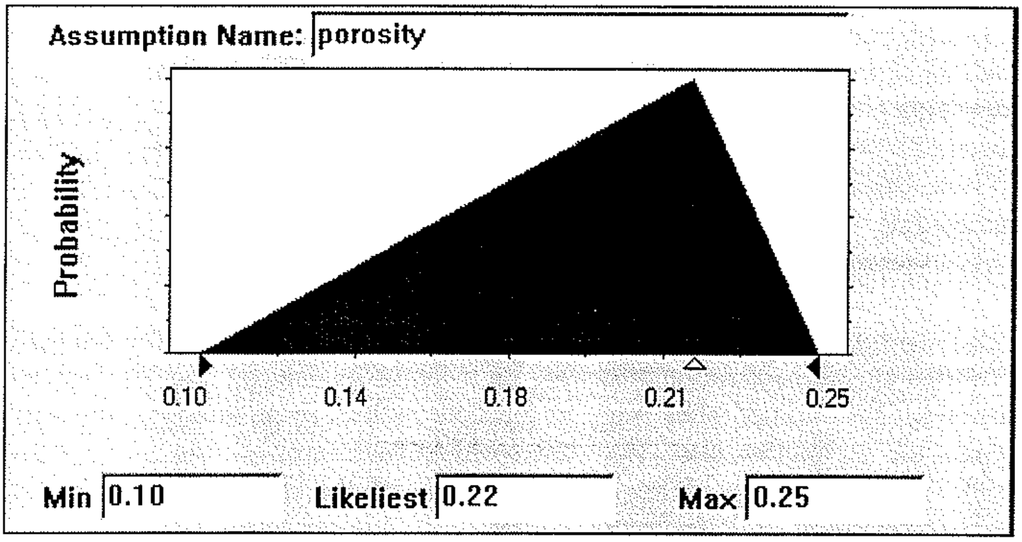
Note that risk equals the product of chance and outcome summed over all hazards. Chance will be a probability between 0 (ie zero chance) and 1 (definite). Risk is therefore expressed in the same units as outcome. Therefore if we wish to examine groundwater concentrations of a certain contaminant (ie the outcome), then the risk will be measured in the same units along with a confidence level.

In the absence of hard fact or statistical data, the probabilities (chance) of an outcome need to be defined using an alternative method. For this we can use a probability density function (PDF). In its simplest form this could be a uniform distribution. Such a PDF simply requires the definition of a credible minimum and credible maximum value. For example, porosity values (determined as a fraction) cannot be less that 0 and cannot be greater than 1 for a typical geological stratum. In the absence of any data a uniform PDF with these limit values could be used (ie uniform (0,1)). However, as we begin to collect data regarding the site and we know which geological strata we are likely to encounter, then our level of uncertainty is reduced and a revised PDF can be generated. Furthermore, the shape of the distribution might not be the same if we wish to skew the distribution towards a value that we consider is more likely. The following figures illustrate common PDFs used in this type of analyses.



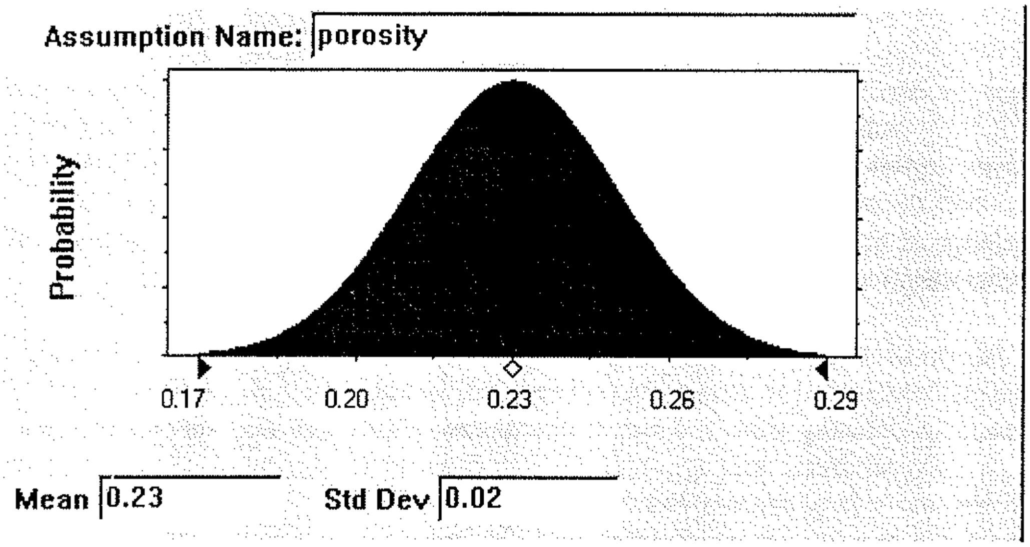
Uniform PDF

Figure 1 Uniform PDF for <u>Total</u> Uncertainty with Respect to Porosity Values



Triangular PDF

Figure 2 Triangular PDF for Porosity With <u>Increased Constraint (Knowledge)</u>



Normal PDF

Figure 3 Normal Distribution PDF Based on <u>Statistical Analyses</u>

The use of each type of distribution is partly dictated by the shape of the actual distribution of the data and also by the degree of knowledge. As more data is accumulated then the degree of uncertainty associated with the parameter becomes less. However, natural variation of a parameter may still require that it is defined as a PDF rather than a single number.

Uncertainty in Defining the Source Term

For a landfill site that is due to accept a wide, but typical, range of wastes (e.g. municipal, commercial and non-hazardous industrial wastes) there will be uncertainty with respect to the following elements:

- leachate strength
- leakage rate
- leachate production rate

The combination of leachate strength and leakage rate essentially define the source term. The volume of leachate production will be a factor in determining the leakage rate in some cases. For unlined sites with no leachate collection or removal, the leakage rate will equal the production rate. For sites with leachate removal, the leachate production rate will control the head of leachate at the base of the site which will be one of the controlling factors in the leakage calculation.

Leachate strength (or the concentration of individual components) can be measured within existing sites and therefore described (as a PDF). Care must be exercised to ensure that the leachate from various parts of the site has been measured. For new sites it is not possible to "measure" leachate strength and therefore reliance on data from other sites accepting similar waste and operated in a similar manner must be made (Robinson 1995).

It must be realised that leachate strength is time dependant and will reduce with time. The factors that will effect the washout time for contaminants are infiltration, field capacity, initial concentration, and the depth of waste (Knox 1995 and Walker 1993). There will be uncertainty associated with each of the parameters due either to lack of knowledge or variation. Figure 4 and 5 indicate the type of relationship between time and concentration with the only variable between them being infiltration rate.

The degree of leakage will be a function of the head of leachate and the engineering barrier(s) constructed to minimise leakage. Different liner types will require a different approach in defining how leakage is calculated, and where the uncertainty lies. For a clay or mineral liner, leakage will be proportional to the head of leachate, the liner thickness, the area of the site and the hydraulic conductivity of the liner. Uncertainty will exist for the precise values of leachate head (which will relate to the efficiency of the drainage system and the infiltration), and the hydraulic conductivity of the liner, but less so for the liner thickness. The degree of variation of the hydraulic conductivity can be determined from laboratory experiments.

Leakage from a site lined with a geomembrane becomes a little more complex. Most geomembranes are essentially impermeable (less that $10^{-15}ms^{-1}$). However, there is now a recognition that membrane liners do leak. Leakage is through defects in the liner caused either by incorrect installation (e.g. bad welds) and by puncture defects caused during placement of the drainage layer and infilling with waste. It is at this point that uncertainty becomes an essential item in the modelling methodology.

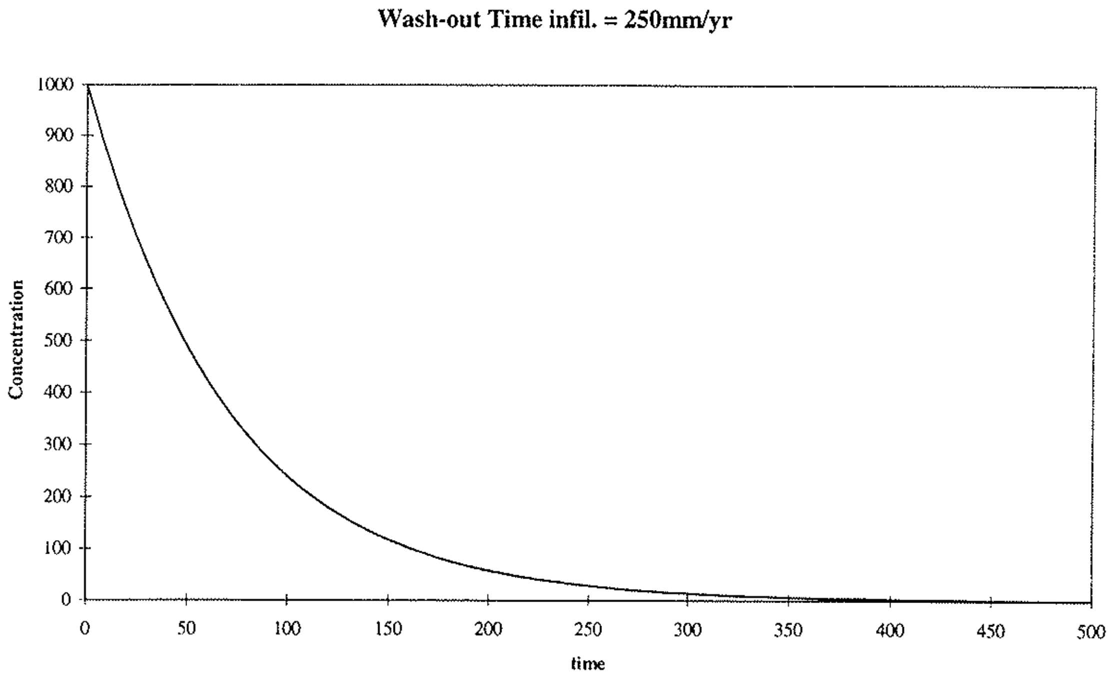


Figure 4 Flushing Time with an Infiltration value of 250mm/yr

It is not possible to define in precise terms the frequency, size distribution and location of all defects within the geomembrane at the time of assessment. Even with advances in geophysical techniques for leak location surveys, it is not possible to predict where new defects will occur, as it is only possible to measure the location of existing penetrations.

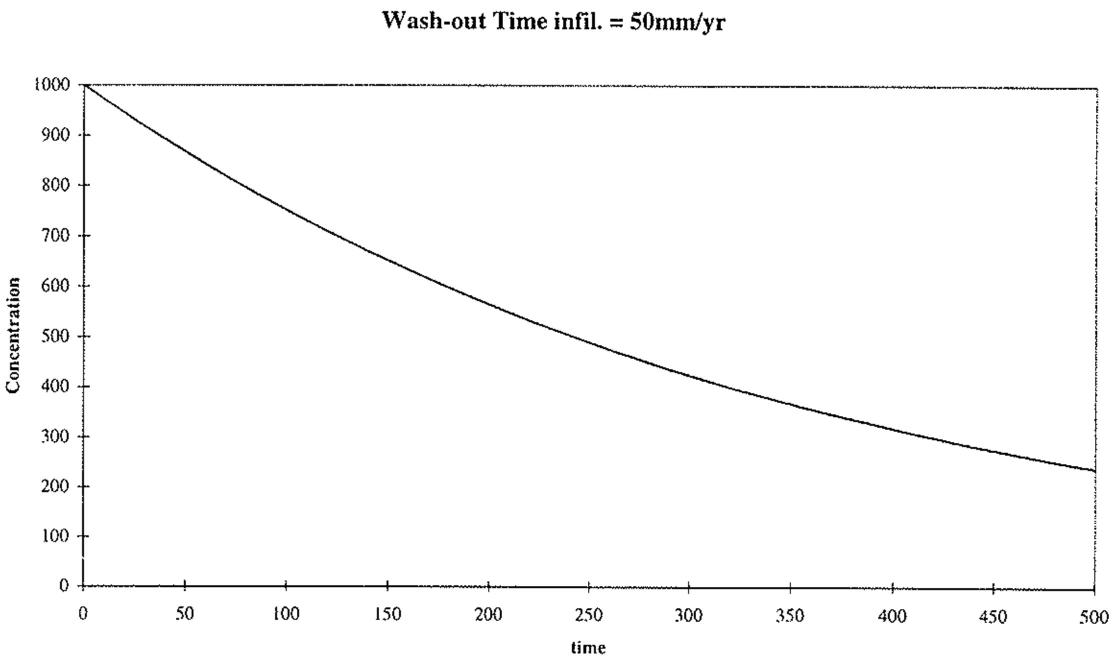


Figure 5 Flushing Time with an Infiltration value of 50mm/yr

The description of the geomembrane defects must therefore be based entirely on statistical data from research. The method of calculating leakage is adapted from the methodologies of Giroud and Bonaparte 1989. Golder Associates have conducted such a research contract as part of the development of the LandSim model so that modelling of both geomembrane and composite lined sites can be undertaken. The distribution of defect numbers and sizes is given in Table 1 below (adapted from Hall 1992).

	Frequency Range		
	Minimum	Most Likely	Maximum
Defect size range			
Pin Holes (0-10mm^2)	0	25	25
Holes (10-100mm^2)	0	5	5
Tears (100-10,000mm^2)	0	0.1	2

Table 1 Defect Frequencies for Geomembrane Liners (defects per hectare)

It should be noted that the frequencies above have been skewed towards the maximum in the first two size ranges to add a degree of conservatism to the assessment.

It should also be noted that the location of the defects is also an unknown (and will remain so unless defects are incorporated into the liner by design!). The location of defects is also an important factor as a defect near the sump will most likely be flooded by leachate to a greater depth than one remote from the sump at the extremes of the leachate drainage system.

Uncertainty in the Geological Pathways

Leachate moving away from the liner system (or the landfill in the case of an unlined site) will undergo attenuation in the unsaturated zone, and (possibly) significant dilution within a aquifer. In the absence of a high quality hydrogeological study, the degree of uncertainty associated with the factors controlling these features can be great. Study and site investigation will reduce uncertainty but some elements of uncertainty and natural variation will remain. It is therefore preferable to continue to use uncertainty in the definition of all elements of the geological pathway(s).

Initial flow of leachate from a landfill site will be through the unsaturated zone (assuming there is one). Within this zone important sorption reactions such as cation exchange or retardation can occur. It is possible to measure cation exchange capacity in the laboratory but some degree of natural variation is to be expected. However, research has shown that the reaction efficiency for the exchange of ammonia is not 100%, and indeed more likely to be in the range 20 - 40% (Cartwright et al 1977).

With a knowledge of the leakage rate, depth of the unsaturated zone, and the moisture content, it is possible to estimate the unsaturated travel time. This may be important for a number of reasons. Firstly, some contaminants have a well defined biological degradation half-life and knowledge of the actual travel time will assist in determining the fate of such contaminants. Secondly, it is useful to know how long it is likely to be before contaminants reach the aquifer when designing the site's monitoring programme. Thirdly, if more than one phase of landfilling has or is likely to occur, then the impact of each phase may be separated in both time and space. The issue of whether the impacts need to be considered as cumulative impacts or, individual impacts separated in time, can be assessed. Finally, very long travel times can result in significant additional dilution due to the effects of dispersion within the unsaturated zone. For example, a clay sequence can readily result in travel times in excess of 500 years (which may be beyond the leachate generating life of the site).

In some cases the aquifer can be regarded as the target of the risk assessment and the end point of the calculations. In other cases a specific compliance point (e.g. a monitoring well) or abstraction well may be the target. If the aquifer is the target, then the analyst must decide whether or not to include dilution as part of the assessment. If the target is a point laterally displaced from the point below the site then the hydraulic properties of the aquifer are important. The aquifer hydraulic conductivity, hydraulic gradient and mixing zone thickness will control the amount of dilution available, and knowledge of the porosity will, in conjunction with the above, allow travel times to be assessed.

The LandSim Model

The LandSim model has been developed to allow the assessment of landfill sites by using Monte Carlo techniques. Essentially, the model allows the description of each of the parameters as a PDF rather than a single value. Figure 6 shows a schematic of the structure of the model.

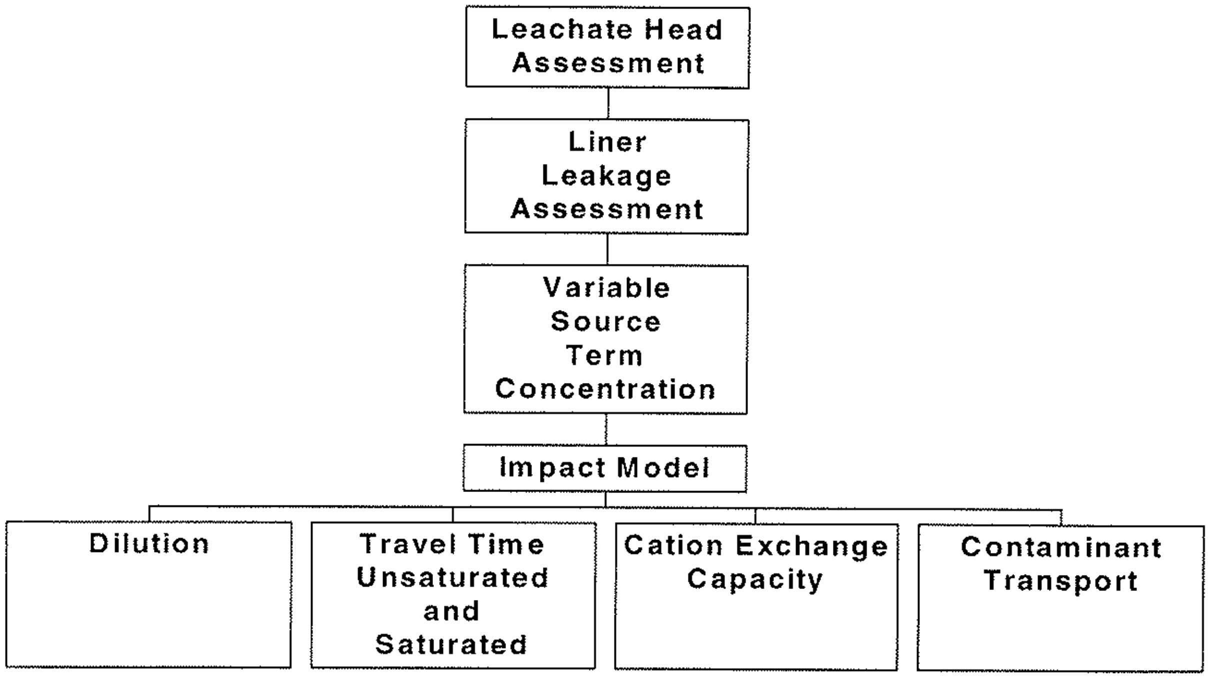


Figure 6 Model Schematic

The model can be used to optimise the performance of the liner and leachate drainage system prior to considering the likely impacts. One of the model outputs is the range of leachate heads that will be generated within the landfill (assuming leachate is being removed from the sump(s)). Such an assessment is a useful check as it is common to specify a maximum leachate head within a lined site. With the large variety of leachate drainage configurations it is useful to have a method that allows a check to be made that the allowable leachate head is actually achievable based on the design and specification of the drainage blanket.

Slight modifications to the landfill geometry and leachate collection system can make large differences to the calculated leachate heads and thus to the leakage rate predicted for a site. For example, increasing the basal drainage angle from a flat surface to one with a 1.5 degree slope can reduce he leachate head by 25% in a cell that is 200 m across.

Once the analyst has completed the optimisation (or, in the case of a regulator encoded the proposed design), they can begin to assess the likely impact of the site on groundwater. It must be accepted that because the model accepts PDFs rather than discrete values, and then applies Monte Carlo methods of analyses, the results are displayed as frequency charts or cumulative curves. Figure 7 shows a typical output graph representing the degree of uncertainty with respect to leakage from a landfill site lined with a composite liner.

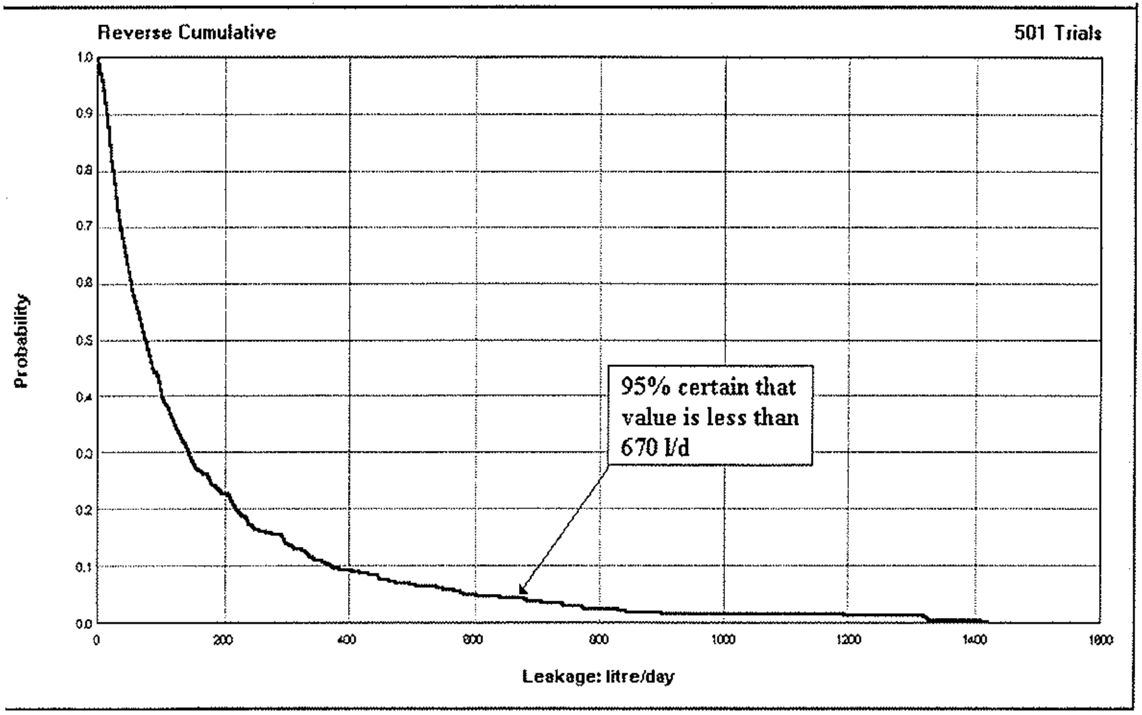


Figure 7 Typical Reverse Cumulative Curve Indicating the Range of Leakage Values

From Figure 7 it can be seen that the possible leakage values cover a wide range. This is simply a result of the inherent uncertainty associated with leakage through this type of liner. Part of the uncertainty results from variations in the leachate head within the site, part from the number, size and distribution of defects, and part due to the natural variations of clay permeability forming the lower element of the liner system.

Within the LandSim model, all of these results of possible leakage values (i.e. the result from each realisation - typically 500-1000) are passed through to the next part of the model. The leakage rate is combined with variations of leachate quality which are, themselves, reducing with time as a result of flushing of the waste.

The most important results are, however, the predicted concentrations of contaminants in the underlying aquifer. These need to be considered at different time periods. During early years, concentrations may be small, or the probability of break through small simply because of the time taken for contaminants to reach the target. At later times concentrations will decline as a result in of the reduction of concentrations in the leachate leaking from the site. In addition, different contaminants will be retarded at different rates, and some (e.g. ammonia) may never reach the target if there is sufficient attenuation capacity within the unsaturated zone. Typical results of contaminant concentrations in the aquifer are shown in Figure 8.

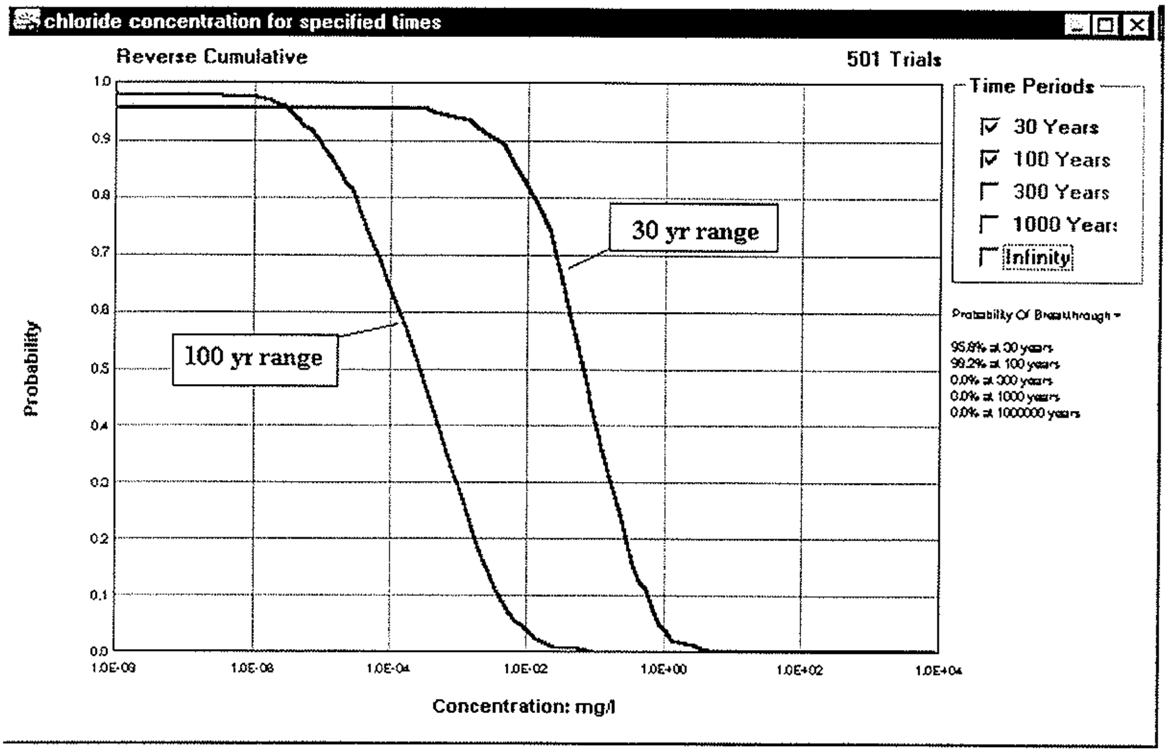


Figure 8 Results showing Range of Chloride Concentrations at two time periods

It is worth noting that in the example shown above, the 30 year profile has higher concentrations but a slightly lower probability of contaminant break through.

It is normal to assess the degree of impact at either the 90 or 95 percentile confidence limit. If the input parameters have not been skewed then this would give an impact level that would be achieved or bettered 9 in 10 or 19 in 20 times respectively. One of the real benefits, over and above simply having a system like LandSim to model these impacts, is that the decision maker can see the entire range of possible outcomes. This includes whether or not the "tail" beyond the 90 percentile is short (perhaps like Figure 8) or long (perhaps like Figure 7).

Model Validation

Prior to the release of the model a series of validation studies were completed. A series of landfill sites with the full range of liner, drainage blanket and geological regimes all with extensive time series monitoring data would have been ideal for the purposes of the validation study. In reality, few sites, if any, were found where there was sufficient data to perform a complete validation study. The approach adopted therefore was to study factors such as leachate head, unsaturated travel time, attenuation, groundwater travel time, and source term reduction at specific sites where these factors are dominant. Few if any sites have a data set that will actually allow an assessment of the overall leakage rate (except for unlined where leakage equals recharge). It was not, therefore, possible to calibrate or verify any of the leakage models directly. If, however, the model can predict the development of a contaminant plume from a landfill where the contaminants have undergone leakage, unsaturated travel and attenuation, mixing and dispersion within the aquifer and further retardation, then confidence in the model is developed indirectly and some indication that the leakage modules work, is gained.

In total, seven sites have been used to perform a validation and the results of these studies have been reported (LandSim Landfill Performance by Monte Carlo Method, 1996). In addition the modelling of extensions to existing facilities has often necessitated the 'calibration' of the 'old' site prior to the predictive stage of modelling.

In general, when modelling for the purposes of validation and calibration one would regard a reasonable fit with the data if the output indicated a match in the 30 - 70 percentile range. During many modelling studies of relatively simple sites (i.e. known leachate strength and simple hydrogeology) a match in the 40 - 60 percentile range is often achieved.

Findings from these studies have shown the following typically apply:

- at the 95 percentile the model is conservative
- the model is not particularly sensitive to landfill cell geometry
- the aquifer hydraulic properties will have a major impact on the degree of dilution essentially because estimates often span orders of magnitude
- errors in estimating the mixing zone thickness are likely to be less significant than other hydraulic parameters
- a combination of large ranges in leachate strength (orders of magnitude) and aquifer properties (orders of magnitude) result in a wide range of output values.

Extended Model Usage

The LandSim model can also be used as an investigative tool to assess the effects of operating landfills in different ways.

For example, it has been used to calculate the impact of a landfill with different post closure options. These include varying the type of cap, the recirculation of leachate, and exploring the operation of sites as flushing bioreactors. In the latter case there are a number of issues that might need to be addressed. Firstly, can the leachate collection and recovery system be relied upon to extract leachate from a landfill with infiltration rates in excess of 2000mm/yr? If so, what degree of leachate head build up will there be and what is the impact on liner leakage? Further, if the source term reduction resulting from operations of this type is included, what is the ultimate effect on the ability of the natural systems to cope with the perceived increase in contaminant loading?

From work undertaken by the author (Hall, 1997) the conclusion is that the operation of sites as flushing bioreactors places less stress on the aquifer system over the life of the site, and that the perception that such sites should only be placed in the most secure geological environments is misplaced. One of the prime reasons for this is that many sites have lined bases and capping systems that allow some degree of leakage. The limited quantity of infiltration that actually occurs results in a very slow (>250 years) degradation of the waste mass. During this period the proportion of leakage as a percentage of total infiltration will approach 90% for clay lined sites and 20-40% for composite lined sites. In addition, the leachate concentration remains high for a considerable period. The flushing bioreactor site, on the other hand, may leak at a higher overall rate, but less as a proportion of the total leachate production over the life of the site. Equally, the leachate concentration is likely to decline relatively quickly (<50 years). As a result, the total loading of contaminants such as ammonia, measured over the life of the site will be considerably less (upwards of a factor of 100) (Hall, 1997).

Model Limitations

LandSim is no different to any model in that it relies on good quality data in order to provide a good quality result. In addition there are some situations that LandSim is not an appropriate model. For sites that lie beneath the water table the leakage part of the model will not function correctly, and neither will the contaminant transport model for the unsaturated zone (as none is present).

LandSim also assumes a simple geological/hydrogeological regime. Sites with complex geology and/or multi-level aquifers or flow that does not conform to Darcian flow should not be modelled with LandSim.

LandSim may also be inappropriate for a highly sensitivity site as it would be difficult for the model to assimilate all the relevant data. This category includes sites adjacent to major public supply wells (which may be totally inappropriate for many other reasons - and LandSim does not include a radial flow model). Equally, in areas of extreme ecological importance when dealing with trace organics where their behaviour in the environment is not well understood LandSim may not be the most appropriate model. Finally, the model should not replace common sense.

Conclusion

The LandSim model was originally developed between 1993 and 1996, and has been in use and commercially available since September 1996. The model has been used at various levels and for various purposes associated with the design, planning and permitting of landfill sites both by developers and regulators. The model provides one of the few readily useable means of determining liner leakage and combining leakage with an overall groundwater quality risk assessment.

Further development of the model is planned in order to enable modelling of multiple sites (i.e. where a single site has cells with different designs), to account for background water quality (not currently included) and to provide some additional liner systems (for those who just have to have a double composite liner!).

Extended use of the model has shown that while some of the modelling approaches are relatively unsophisticated (e.g. unsaturated flow), the model can provide a good prediction of the likely impacts.

From the authors perspective, use and dissemination of the model has removed many of the poorly founded prejudicial views held regarding the performance of different liner materials and configurations. It also provides a common framework for developers and regulators alike to review new landfill designs. Furthermore, it steers developers towards the collection of appropriate site investigation data (something that, in the advent of the "contained" landfill, many seemed to ignore).

Disclaimer

The opinions expressed in this paper are entirely those of the author and not necessarily those of any of the funding organisations or participants in the development of the LandSim model.

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Paper No. 3.

Integrated Pollution Control. Licensed Industrial Activities - Waste Management.

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Integrated Pollution Control Licensed Industrial Activities - Waste Management (..... and Groundwater !)

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Introduction

Waste and waste disposal practices represent and will continue to represent a significant pollution risk potential for the nations aquifers.

The advent of Integrated Pollution Control (IPC) licensing in Ireland has had a significant positive impact on the knowledge and information available in respect of waste emissions from IPC licensed industries. This information has been used to influence the determination of BATNEEC for an industry, and in the formulation of licence conditions, particularly in respect of implementing groundwater pollution prevention technologies and monitoring requirements.

This paper will give the reader an appreciation of what waste is controlled under IPC, how it is controlled, and the future of waste management under IPC.

The First Schedule of the EPA Act and waste emissions

The first Schedule to the EPA Act 1992 specifies a range of classes of industrial activities which are to be controlled under IPC. These classes are further divided into 61 categories. It is anticipated that some 800 industrial activities will eventually be brought under IPC control. When one examines the National waste arisings statistics and the influence/involvement of IPC licensing on waste management in Ireland, it comes as a surprise to many the extent of control on these arisings that is exercised via IPC.

Projecting forward one year National waste arisings may be in the order of 44 million tonnes per annum. Approximately 29% of this is would be of a non-agricultural origin. The IPC licensing system includes activities in both sectors. In the agricultural sector IPC controls many intensive farming activities such as pig and poultry installations which account for c.7.2% (2.2Mtpa) of agricultural waste produced. Of the non-agricultural waste generated (c.13.5Mtpa) approximately 71% is industrial derived. Of this 71% or c.9.6Mtpa, it is estimated some 60% or (5.8Mtpa) comes under the IPC licensing system. In total then IPC controlls some 8Mt of annual waste production.

Figure 1 summarises the main waste producing sectors in the First Schedule to the EPA Act. The mining and mineral processing (including Aughinish Alumina) sector accounts for 52% of all waste produced by IPC industries, followed by the pig production sector at c.28%.

The amount of hazardous waste produced by IPC licensed facilities is estimated at 180,000tpa. This is a very small proportion of all waste produced by IPC controlled industries. However the interesting observation that can be made is that this small proportion of IPC controlled waste (<2.3%) originates substantially from one IPC sector, the chemical/manufacturing sector, where it represents c.26% of waste production in this group.

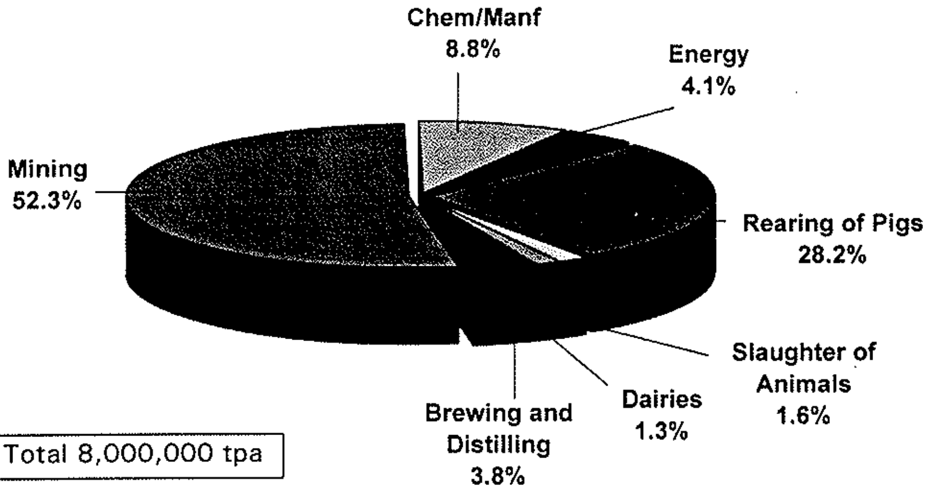


Figure 1: Main waste producing sectors in IPC

It is worth comparing the amount of waste disposed of by First Schedule companies to that disposed of by all the Local Authorities in the country. Figure 2 takes only six industrial operations coverd by IPC legislation. These alone produce and dispose of on-site annually more waste than all municipal waste disposed of by the Local Authorities.

	Footprint (ha)	Capacity (Mm^3)	tpa
Tara	180	23	1M
Galmoy	35	1.6	230,000
Lisheen	70	4	500,000
Aughinish Alumina	104	13	600,000
Irish Sugar M & C	-	-	205,000
Total Municipal Waste	-	-	2.5 M

Figure 2: On-site landfills

Controllable waste is that proportion of the National waste production that is or can be regulated under current legislation. With the exception of pig and poultry waste it would exclude all other agricultural waste. It would also generally exclude sand and gravel and stone quarry spoil heaps. It is estimated that c.15.7Mtpa National annual waste production and disposal is controllable. Of this amount over 51% is produced and disposed/treated on-site by IPC facilities.

Waste Management Routes

Records available would indicate that of the c.180,000tpa hazardous waste produced by IPC controlled industries nearly 60% is recovered on the site of production for re-use, or treated on-site. Of that sent off site for disposal less than 2% ends up in an Irish landfill. The rest is either recovered, treated or exported. There is a deal of vagueness in these figures, however one of the successes of the IPC licensing process has been the correct classification, categorisation and accounting for hazardous waste in IPC facilities. Another feature of the new regulatory control has been the diversion of hazardous waste away from municipal landfills.

Referring again to the main waste producing sectors of the First Schedule to the EPA Act (Figure 1) there are a number of fundamental observations that can be made in respect of the data:

(a) >96% of waste produced by first schedule industries is disposed of on-to or into land.

(b) Approximately 35% is landspread, and 61% is disposed of to landfill

(c) In the region of 7.7Mtpa of waste is disposed of into or over an aquifer.

The land-take involved in the various waste management options employed by IPC licensable companies also produces interesting observations:

- Landspreading of waste from IPC industries will capture c.400,000ha (not far off 1M acres)
- Under landfill the three mines (Tara, Galmoy and Lisheen) and Aughinish's Red Mud Stack have a combined foot-print of 389ha (~960 acres).

It is understandable then, having regard to these figures, why the Agency focuses heavily on groundwater protection for certain industrial sector's: the landspreading CoP being a case in point. Also the Agency stipulated BATNEEC requirements in respect of groundwater protection, for Lisheen mine tailings facility and for the extension to the Aughinish Alumina landfill has cost these industries an additional (approximately) £3M and £1.2M respectively over-and-above their initial groundwater protection proposals.

Environmental Management Programmes and Waste Management in IPC

The IPC licence and the Environmental Management Programme element of the licences requires licensees to aspire to the environmentally friendly end of the waste management hierarchy (Avoid < Minimize < Recycle < Treat < Energy Recovery < Dispose).

Section 5 of the EPA Act refers to BATNEEC as *'the use of best available technology not entailing excessive costs to prevent, or eliminate, or where that is not practicable, to limit, abate or reduce an emission from an activity.'*

Waste reduction is (via Section 83(3)) a legal imperative in considering the grant of an IPC licence.

The initial thrust of waste management under IPC was, and is, to get licensees to carry out proper characterization of their waste and implement waste accounting procedures. It had been recognized that many licensees had never properly characterized their waste (hazardous/non-hazardous), nor had they maintained proper record of waste production. In many cases there were very poor records of who took waste from the licensees and where it ended up. There has been significant improvements in these areas since the introduction of IPC licensing.

The IPC licence conditions require the recording and annual reporting of detailed information on waste produced. Viz.:

(i) The names of the agent and transporter (from site) of the waste.

(ii) The ultimate destination of the waste.

(iii) The name of the persons responsible for the ultimate disposal/recovery of the waste.

(iv) Written confirmation of the acceptance and disposal/recovery of any hazardous waste consignments sent off-site.

(v) The results of any waste analyses required in Schedules to the licence.

(vi) The tonnages and European Waste Catalogue (EWC) code of the various waste classes defined in the Schedules to the licence sent off-site for disposal/recovery.

(vii) Details of any rejected consignments.

(viii) The tonnages and European Waste Catalogue (EWC) code of the various waste classes defined in the Schedules to the licence recovered, or disposed of, on-site.

This information is required to be maintained and form part of the waste management section of the Annual Environmental Report (AER). The latter being a key document in assessing a company's performance in every environmental aspect of its operation. The following discussion sets out how some of this information is to be presented in the Waste Management section of the AER.

Annual Waste Arisings

From the information required (in the licence) to be recorded, Table 1 (Hazardous Waste) and Table 2 (Non-hazardous Waste) is completed for the AER.

The Waste Material should be as detailed in the appropriate schedules to the licence. The source or sources of each waste material must be identified and the appropriate tonnage assigned. The licensee should detail any on-site treatment (e.g., crushing, shredding, mixing, concentration, neutralization, solidification, distillation, etc.,) applied to the waste.

Under the Waste Management Option heading of the tables the licensee must record the method and amount of waste recovered for re-use directly or indirectly within the on-site processes. Recovery is defined as *'any activity carried out for the purposes of reclaiming, recycling or re-using in whole or in part the waste'.*

Appendix A details the thirteen classes of waste recovery activities presented in the Fourth Schedule to the Waste Management Act, 1996. In the case of any off-site recovery options sourced the information required is similar.

In the remaining columns of the Waste Management Option sections of Tables 1 and 2 the licensee must detail the tonnage and disposal method employed for either on- or off-site disposal of the waste. Appendix B to this document details the various recognised waste disposal activities presented in the Third Schedule to the Waste Management Act.

Licensed activities with on-site disposal operations will be required to submit a separate comprehensive and detailed report section (in the AER) on the disposal operation. In the case of on-site landfills the Agency has published specific guidelines which detail the annual monitoring and reporting requirements ('Landfill Monitoring Manual', 1995).

In each case where there is an on-site disposal operation the licence conditions will detail the annual reporting requirements for that operation. The licensees must have regard to the requirements of these conditions when preparing their AER.

The licensee should in cases of waste sent off-site for recovery/disposal complete Table 3 for Hazardous Waste and Table 4 for Non-Hazardous Waste. In the case of hazardous wastes this information must be tabulated for each consignment. For all other wastes monthly totals (or such smaller interval as the licensee may prefer) for each waste class should be presented.

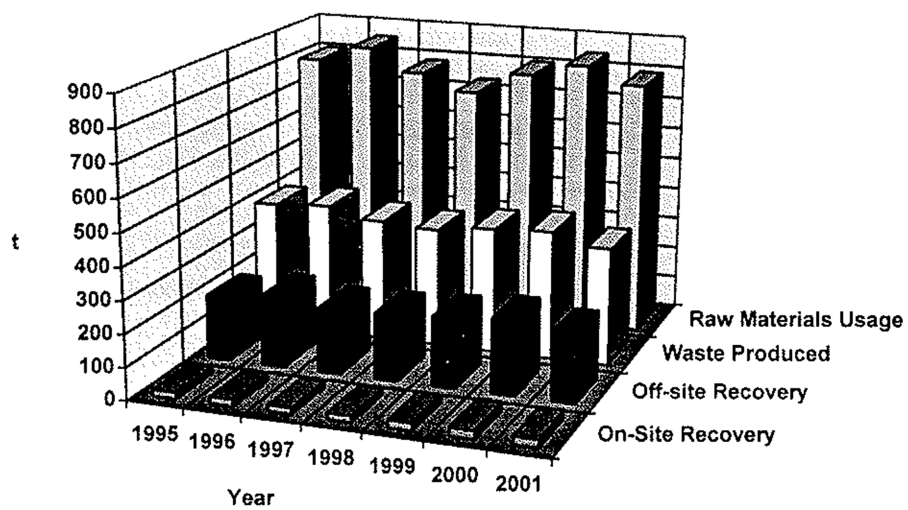
Waste Analysis

The licensee may as part of their licence be required to periodically submit to chemical analysis, and/or other tests as may be proscribed, certain wastes generated on the site. The results of this monitoring should be included in the AER with relevant reference ID's so as to identify the waste consignment tested and when testing was done.

Waste Minimization Index

One of the principal objectives of the IPC licensing system is to secure from licensees annual improvements in waste minimization. There are of course certain exceptions to this. For example, in the case of metal mining the more waste produced may be as a result of more efficient extraction of metal from the ore, and thus the higher the value of the product. However for the majority of licensees a waste minimization programme is an essential component of the Waste Management Plan for the site. From the information presented in Tables 1 and 2 it is possible to generate the following type of plot. The proportion of waste materials recovered from the amount produced is readily apparent (Figure 3).

Figure 3: Key Annual Waste Management Data



In this plot the annual raw materials usage is included. The relationship of raw materials usage to waste produced is an important one in assessing the environmental performance of a licensee in respect of waste. To this end it will be useful to the Agency and the licensee to apply an annual index to the amount of waste produced and its relationship to raw materials usage.

The waste minimization element of the Waste Management Plan must include a performance index - a sort of eco-productivity index - for all the wastes produced on-site. The Waste Minimization Index (WaMI) introduced here looks (with the exception of canteen and office waste) at all waste produced on a site, from reject product to packaging waste to waste solvents and is Gross of any recovery operations.

The **Gross WaMI** is calculated as follows;

$$\text{Gross WaMI} = \{ \text{Waste Produced (t)} \div \text{Raw Materials (t)} \} \times 100$$

Note: 1. Excludes energy and fuels, canteen and office supplies.

It is desirable to have the index decreasing over a number of years. A plot of the index will give an indication of the effectiveness or otherwise of any procedures or actions taken by the licensee to reduce waste generation associated with on-site processes, i.e. waste minimization.

This index can be adjusted to factor in any recovery operations applied to the waste, giving two further plotable indices which are important in the overall appreciation of the licensees efforts at achieving an environmentally responsible waste management policy for the licensed activity.

The first of these allows for any on-site recovery of waste for reuse on-site, viz.;

$$\left. \frac{\text{Waste Produced (t) - Amount Recovered on-site (t)}}{\text{Raw Materials (t)}} \right| \times 100$$

This index could be termed the **Nett of Process WaMI,** or Nett WaMI, as it is nett of any recovery achieved on-site for re-use on-site. Here too a decreasing index is desirable. Indeed it is preferable to have this index lower and decreasing at a greater rate than the Gross WaMI, which would indicate management success at improving on-site recovery of waste.

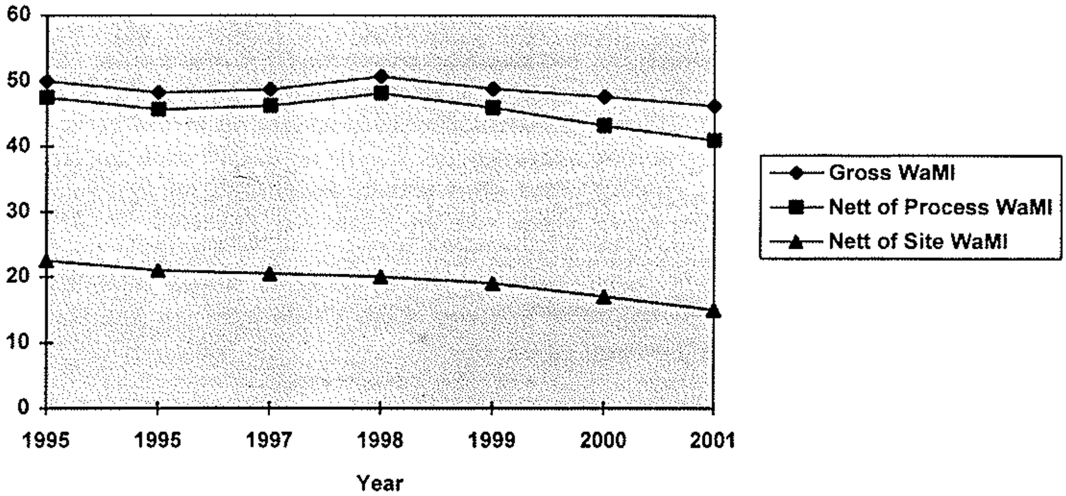
The last index is used to measure the licensees ability to source environmentally beneficial off-site recovery options for their waste rather than defaulting to the often easy disposal option. Examples would be landspreading for agricultural benefit, waste oil recovery, drum recycling, etc. This index could be termed the **Nett of Site WaMI,** or Nett-Nett WaMI. This index is calculated as follows;

$$\left. \frac{\text{Waste Produced (t) - Amount Recovered on-site (t) - Amount Recovered off-site (t)}}{\text{Raw Materials (t)}} \right| \times 100$$

Again it is desirable that this index would be lower than the Nett WaMI, and for it to be decreasing at a greater rate than that for either the Nett or Gross WaMI.

Taking the data presented in Figure 3 the following graphical representation could be used to assist presentation of the three waste management indices.

Figure 4: Waste Management Indices



National Waste Database

The National Waste Database (NWD) is a national framework established by the Agency to record and report waste management statistics for the country. The Agency is of the opinion that that responsibility for compiling and reporting information on waste rests with those responsible for its production and management (Local Authorities, industry, waste contractors and recovery operators). The NWD employs a variety of indicators by which waste management data from the industrial/manufacturing sector can be represented. These indicators are calculated by reference to a number of factors (number of employees, recovery rate, % hazardous waste to non-hazardous, etc.). The NWD also employs the EU Statistical Programmes Committees (NACE) protocol on nomenclature and coding for economic activities (industrial sectors) within the community [Council Regulation (EEC) 3037/90 and Commission Regulation (EEC) 761/93].

To permit maintenance of the NWD licensees are required to complete Table 5 and submit it as part of their AER.

Table 5 National Waste Database Data Report Sheet

Industrial Sector NACE[1] Code (two letters and max. four numbers)							
Reporting Period							
Number of Employees (for reporting period)							
Total tonnage of Waste Produced							
Hazardous							
Non-Hazardous							
Total tonnage of Waste Recovered							
Hazardous							
Non-Hazardous							

[1] Commission Regulation (EEC) 761/93

Concluding Remarks

It is certain that the quality of data yielded by regulated industrial activities in their AER's in respect of waste emissions will improve substantially in the future. This will in turn improve the quality of the National Waste Arisings statistics. From this data it will be possible to compare and contrast different licensees operating in the same sector to see which are not achieving sector averages for waste reduction and the consequent impact on groundwater. In coming years once proper accounting of waste production and disposal has been established there will be increased pressure on licensees to minimise at source and to vigorously pursue recovery options. However having regard to the main waste producing sector, mining, the concept of waste minimisation is a difficult one. A mine operator strives to improve metal yields from ore and if successful will produce more waste per tonne of raw material.

There are three broad categories of groundwater risk associated with waste disposal. Landspread waste generally presents a short term risk to groundwater (accumulative aspects aside). Municipal and general industrial waste landfills represent a medium to long term risk. However mines waste facilities like that at Galmoy and Lisheen will represent a groundwater risk in perpetuity.

Therefore, it is essential that in respect of waste management at IPC facilities, where waste is placed into or above aquifers, that a high degree of protection, caution, vigilance and monitoring is maintained if the quality of our national groundwater resource is to be assured, and our current waste management practices considered sustainable.

Table 1 Annual Waste Arisings - Hazardous Waste

					Waste Management Option							
Waste Material	**EWC Code**	**Source**	**t**	**On-site treatment**	On-Site Recovery		Off-Site Recovery		On-Site Disposal		Off-Site Disposal	
					Method	t	Method	t	Method	t	Method	t

Table 2 Annual Waste Arisings - Non-Hazardous Waste

					Waste Management Option							
Waste Material	**EWC Code**	**Source**	**t**	**On-site treatment**	On-Site Recovery		Off-Site Recovery		On-Site Disposal		Off-Site Disposal	
					Method	t	Method	t	Method	t	Method	t

Table 3 Hazardous Waste sent off-site for Recovery/Disposal

Consignment Note Number	Date of dispatch	Waste Material	EWC Code	t	Description & Nature of Waste	1. Broker 2. Haulage Contractor	Recovery Contractor	Disposal Contractor

Note: Any Consignments rejected must be recorded as a * in the first column and details presented in an attachment.

Table 4 Non-Hazardous Waste sent off-site for Recovery/Disposal

Reporting Period	Waste Material	EWC Code	t	Description & Nature of Waste	1. Broker 2. Haulage Contractor	Recovery Contractor	Disposal Contractor

Note: Any Consignments rejected must be recorded as a * in the first column and details presented in an attachment.

APPENDIX A

Waste Recovery Activities

1. Solvent Reclamation or Regeneration.

2. Recycling or reclamation of organic substances which are not used as solvents.

3. Recycling or reclamation of metals and metal compounds.

4. Recycling or reclamation of other inorganic materials.

5. Regeneration of acids or bases.

6. Recovery of components used for pollution abatement.

7. Recovery of components from catalysts.

8. Oil re-refining and other re-uses of oil.

9. Use of any waste principally as a fuel or other means to generate energy.

10. Spreading of any waste on land with a consequential benefit for an agricultural activity or ecological system, including composting and other biological transformation processes.

11. Use of waste obtained from any activity referred to in a preceding paragraph of this Appendix.

12. Exchange of waste for submission to any activity referred to in a preceding paragraph of this Appendix.

13. Storage of waste intended for submission to any activity referred to in a preceding paragraph of this Appendix, other than temporary storage, pending collection, on the premises where such waste is produced.

APPENDIX B

Waste Disposal Activities

1. Deposit on, in or under land.

2. Land treatment, including biodegradation of liquid or sludge discards in soils.

3. Deep injection of the soil, including injection of pumpable discards into wells, salt domes or naturally occurring repositories.

4. Surface impoundment, including placement of liquid or sludge discards into pits, ponds or lagoons.

5. Specially engineered landfill, including placement into lined discreet cells which are capped and isolated from one another and the environment.

6. Biological treatment not referred to elsewhere in this Appendix which results in final compounds or mixtures which are disposed of by means of any activity referred to in this Appendix.

7. Physico-chemical treatment not referred to elsewhere in this Appendix which results in final compounds or mixtures which are disposed of by means of any activity referred to in this Appendix.

8. Incineration on land or at sea.

9. Permanent storage, including emplacement of containers in a mine.

10. Release of waste into a water body (including a seabed insertion).

11. Blending or mixture prior to submission to any activity referred to in this Appendix.

12. Repackaging prior to submission to any activity referred to in this Appendix.

13. Storage prior to submission to any activity referred to in this Appendix, other than temporary storage, pending collection, on the premises where the waste concerned is produced.

Paper No. 4.

Hydrogeology of Magheramourne Quarry and Northern Ireland Landfill Legislation.

Peter Bennett, Hydrogeological & Environmental Services Limited.

HYDROGEOLOGY OF MAGHERAMORNE QUARRY AND NI LANDFILL LEGISLATION

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ABSTRACT

Magheramorne is a disused basalt and chalk quarry near Larne, with a void space of almost 20 million m³, which has been refused planning permission as a waste disposal landfill on environmental grounds. The investigation and assessment illustrate how the waste legislation process is managed in Northern Ireland.

INTRODUCTION

New landfills in Northern Ireland currently require:-

- planning permission - from Planning Service of DoE, requiring a Environmental Assessment
- a discharge consent - from Environment and Heritage Service of DoE under the Water Act, 1972 (soon to be replaced and improved) requiring hydrogeological investigation
- a site licence - from the District Council under the Pollution Control and Local Government Order, 1978

The site licence cannot be issued unless planning permission and discharge consent are granted so the latter two become the contentious issues. Almost all landfill developments for domestic and/or industrial wastes are large projects because of the high capital costs of the modern engineered approach to containment, particularly lining.

The majority of planning applications for landfill therefore tend to attract widespread objections so they end up being referred to a Public Enquiry where the applicant, regulators, and objectors present their proposals, arguments and criticisms before an inspector, usual assisted by a technical assessor, who conduct the enquiry, adjudicate and report their findings to the Planning Appeals Commission which can accept or reject the inspector's recommendation. The Minister then has a final say.

In the case of the Magheramorne application the inspector recommended refusal, the Planning Appeals Commission recommended granting permission, and the Minister turned it down, mentioning inter alia the precautionary principle.

This talk uses the Magheramorne case history to illustrate the process, mainly from the hydrogeological point of view, with the technical details condensed as follows.

BACKGROUND

Magheramorne quarry is an enormous, disused limestone quarry (almost 20 million m^3 void space) owned by Blue Circle Industries plc. It is by far the largest quarry pit in Northern Ireland, being 3 to 4 times the size of most of our other large quarries so, if developed for landfill, would be a major regional facility capable of taking not only all of Greater Belfast's wastes but many other surrounding council areas.

The Cretaceous Chalk limestone had been worked for cement manufacture at Magheramorne for well over a century until the late 1970s when escalating oil prices made drying of the clay component of the Portland cement mix uneconomic (the clay was being dredged from Larne Lough and removing 100m of basalt overburden to get at the Chalk wasn't helping either.) Thereafter cement manufacture continued at the plant, using imported raw materials, until around 1990.

In 1985 Blue Circle applied for planning permission etc for domestic waste landfill in the quarry but withdrew after strong local opposition.

In 1994 Blue Circle Waste Management Ltd, a subsidiary company operating large engineered landfills in England, began the process of investigation and application again. Kirk McClure Morton were appointed as the lead consultants, charged with engineering design and coordination of the various lines of environmental assessment, with my own firm involved in the hydrogeological study. During the phase of investigation and preparation of an Environmental Statement Blue Circle Waste Management Ltd was bought over by Haul Waste Disposal Ltd, a subsidiary of a water company plc in SW England, who then financed the project through to Public Enquiry in 1996. Part of that deal allowed for transfer of ownership of Magheramorne Quarry from Blue Circle Industries to be delayed until such time as approval would be granted for landfill. Blue Circle are still owners because a ministerial announcement in July 1997 turned down the development. Reasons mainly concerned threats to bird life on Larne Lough, particularly roseate terns which breed on Swan Island, with the Precautionary Principle cited.

Hydrogeology had been a major and contentious aspect of the scheme, but our interpretations and arguments appear to have been almost entirely accepted by the Planning Commissioners, so groundwater issues were not a factor in the refusal. It is interesting to note that Friends of the Earth used the enquiry to further their campaign against landfill, based on their argument that *all* waste disposal landfills are illegal under the EC Groundwater Directive (80/68/EEC) because they inevitably leak List I substances.

GEOLOGY

The quarry itself is at the eastern edge of the Antrim Plateau, an almost 4000km^2 area of Tertiary basalt lavas up to 800m thick in places.

The basalts are underlain by upper Cretaceous Chalk, usually more than 90% $CaCO_3$ and 30 to 55m thick so extensively quarried. Only the top 25m layer of the Chalk at Magheramorne was suitable for cement manufacture so the basal 8 to 18m of glauconitic Chalk and Greensand were usually left unquarried.

The Greensand is underlain by weak, impermeable Lias Clay, which is a major cause of landslips in Co. Antrim, is only a few metres thick, and rests on fine-grained sediments of Triassic age, the Mercia Mudstone.

HYDROGEOLOGY

The Chalk comprises a minor aquifer in Co. Antrim because, although it is very hard with little intergranular porosity and permeability, it does have significant secondary permeability through fractures and fissures opened by dissolution. Both palaeokarst and recent karst features are seen in the Magheramorne area.

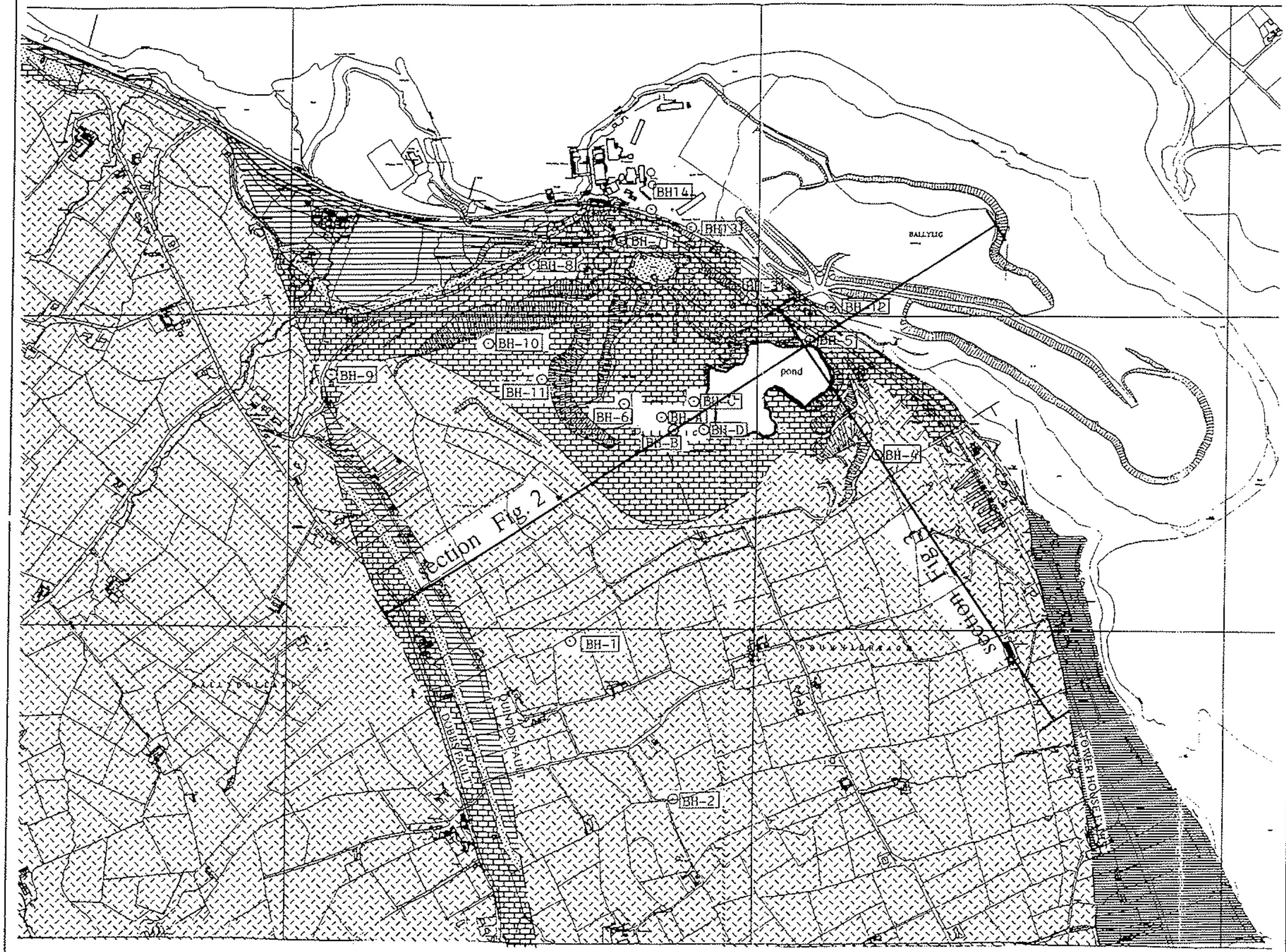
The groundwater in the Chalk is mainly recharged through the overlying basalts which, although layered and of very variable composition, have sufficient fracture permeability to allow some water to percolate down through them into the more open Chalk.

The Lias Clay and Triassic strata are essentially impermeable so groundwater in the Chalk tends to form a gravity spill spring line at the base of the formation where it outcrops. The quantities of water which emanate from the springs are many times too great to be derived solely by recharge over the limited outcrop areas of the Chalk, indicating that most recharge occurs through the basalt cover.

The bottom of the quarry pit is at a level of some 22m *below* Ordnance Datum, where the Greensand was bottomed and the Lias Clay encountered. Until the early 1990s water was being pumped from the pit for use in the cement plant and the pond level lay at 9 to 10m below OD, spring seepages were apparent from bedding planes in the Chalk on the quarry floor adjacent to the flooded area, and the pond remained fresh. Since abstraction of water from the pond stopped the water level has had ample time to fully recover to 1 to 2m above OD, depending on season. There is no surface drainage system from the site, so runoff from within the quarry area accumulates in the ponds, mixing with groundwater inflows (induced by the excavation to flow into its base), and the ponds drain through the Chalk and Greensand into Larne Lough via a number of routes and to various risings below high water mark. Rainstorm events do not cause flooding of the quarry above normal pond levels.

The drainage paths (Fig. 4) have been confirmed by both dye and bacterial tracing experiments, and transit times are only a few hours. Yet no tidal influence is observed on groundwater hydrographs from within the site close to the coast (Figs. 5 & 6).

It remains an enigma how water can drain out of the site so readily yet sea water does not flow back in, even when the hydraulic gradient is reversed. Also the water level in Larne Lough rises to more than 1m above OD (Fig. 6).



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Kirk McClure Morton

Hydrogeological & Environmental Services

KEY

Basalt

Chalk

Hibernian Greensand

Lias Clay

Trias Marl / Mudstone

Fault, tickmark on downthrow side

Borehole location & number

FIG. 1

ORIGINAL SCALE 1:10000

0 100 200 300 400 500m

MAGHERAMORNE LANDFILL PROJECT

SOLID GEOLOGY AND BOREHOLE LOCATION MAP

DIAGRAMMATIC SECTION THROUGH MAGHERAMORNE QUARRY

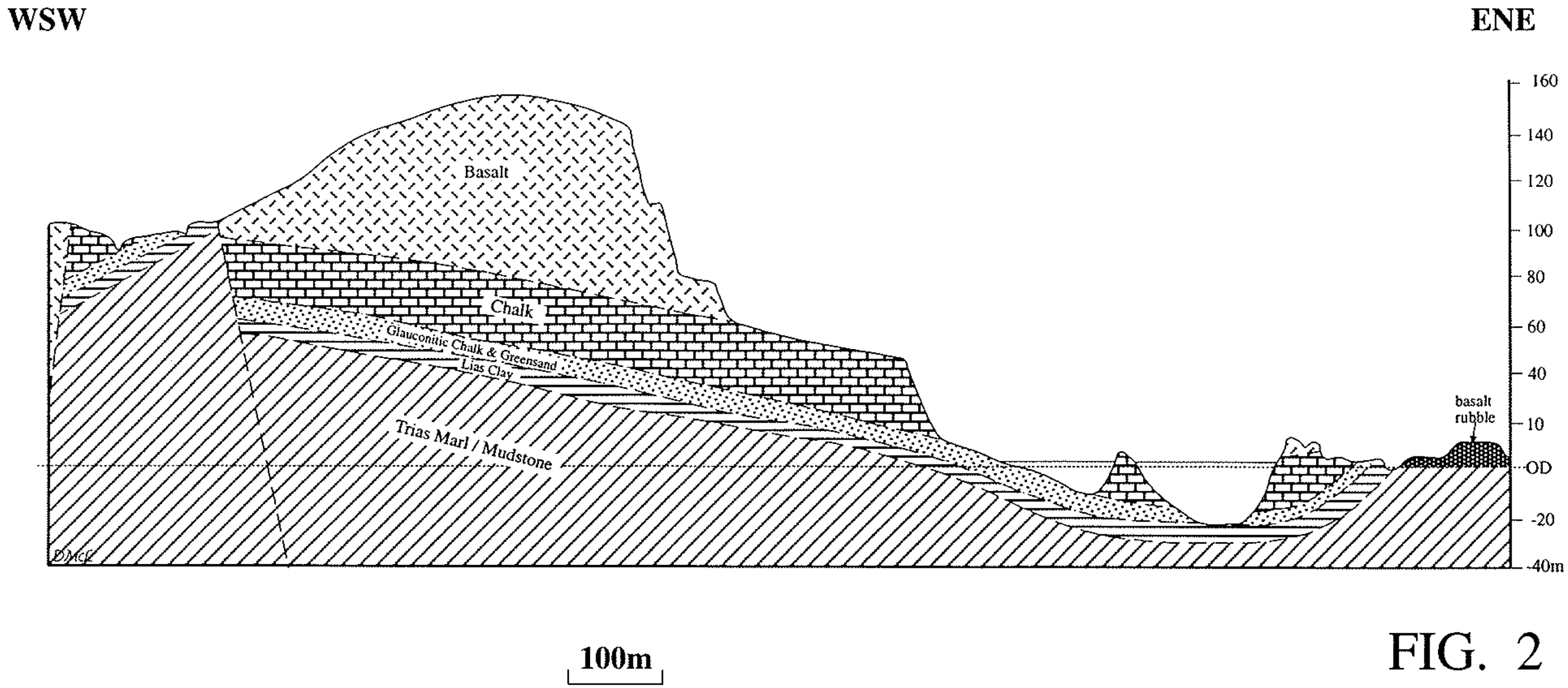

FIG. 2

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Fax. 01232-682195

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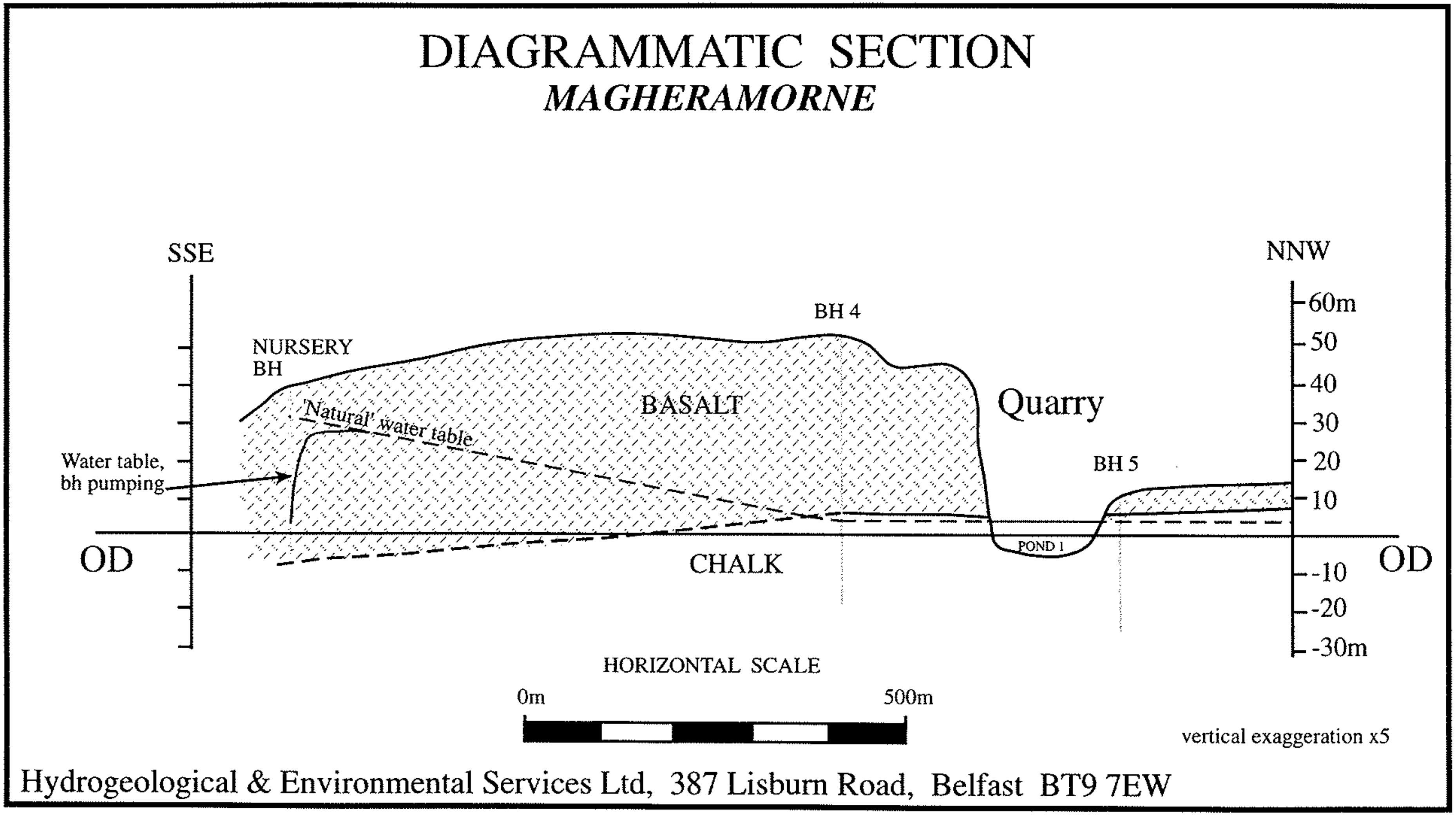

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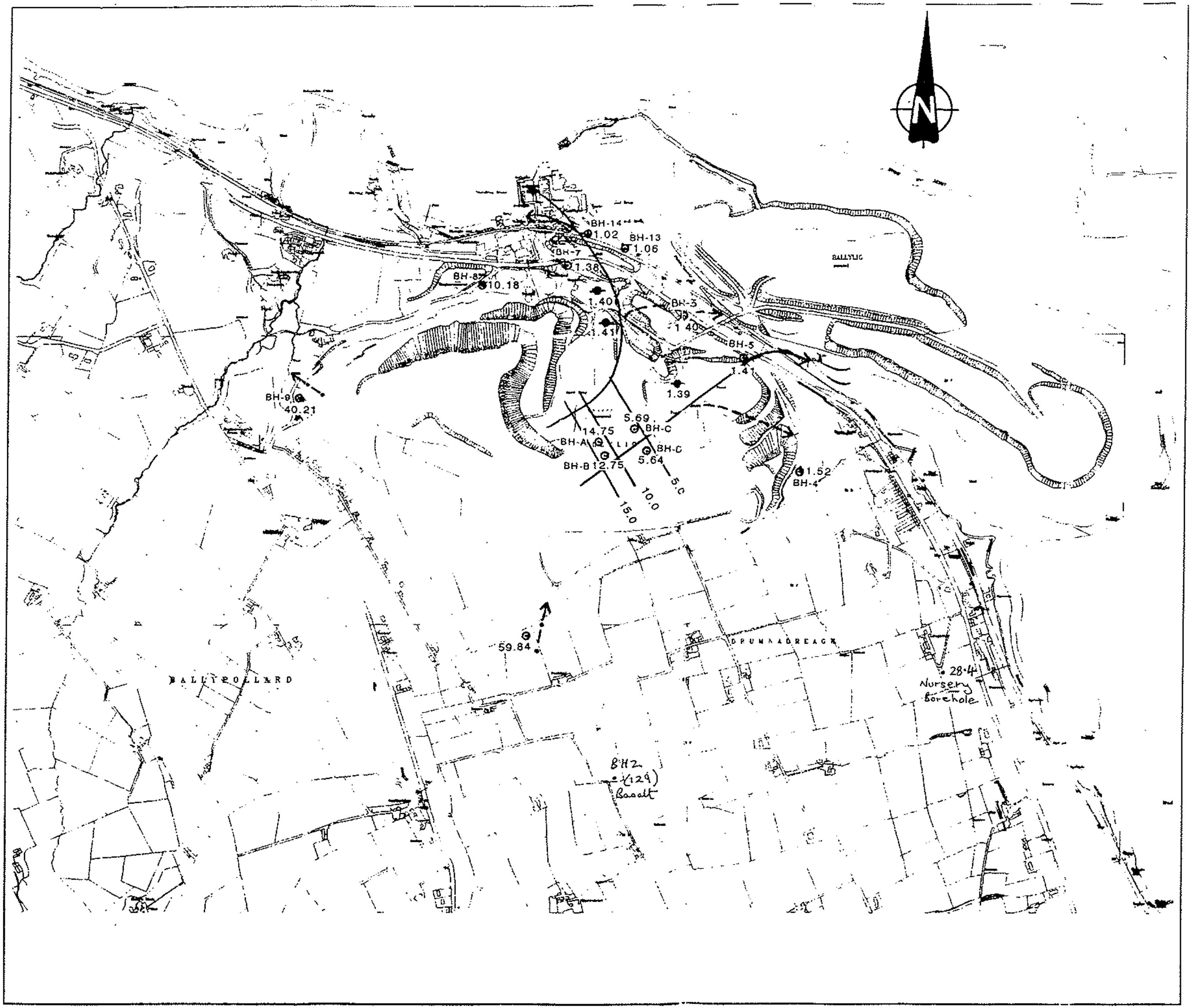
FIG. 3



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Kirk McClure Morton

Hydrogeological & Environmental Servic

→	Main groundwater flow path
- - →	Minor groundwater flow path
-·-→	Probable groundwater flow path
5.0	Groundwater contour (mOD)
BH-3 1.40	Borehole water level (14/10/94) mOD
1.39	Pond water level (14/10/94) mOD

FIG. 4

Note:
All water levels recorded in chall

ORIGINAL SCALE 1:10 000
0 100 200 300 400 500m

MAGHERAMORNE LANDFILL PROJECT

Groundwater Contours Flow Paths and Water Levels

Hydrogeological & Environmental Services Ltd. HYDROLOG Archive Report

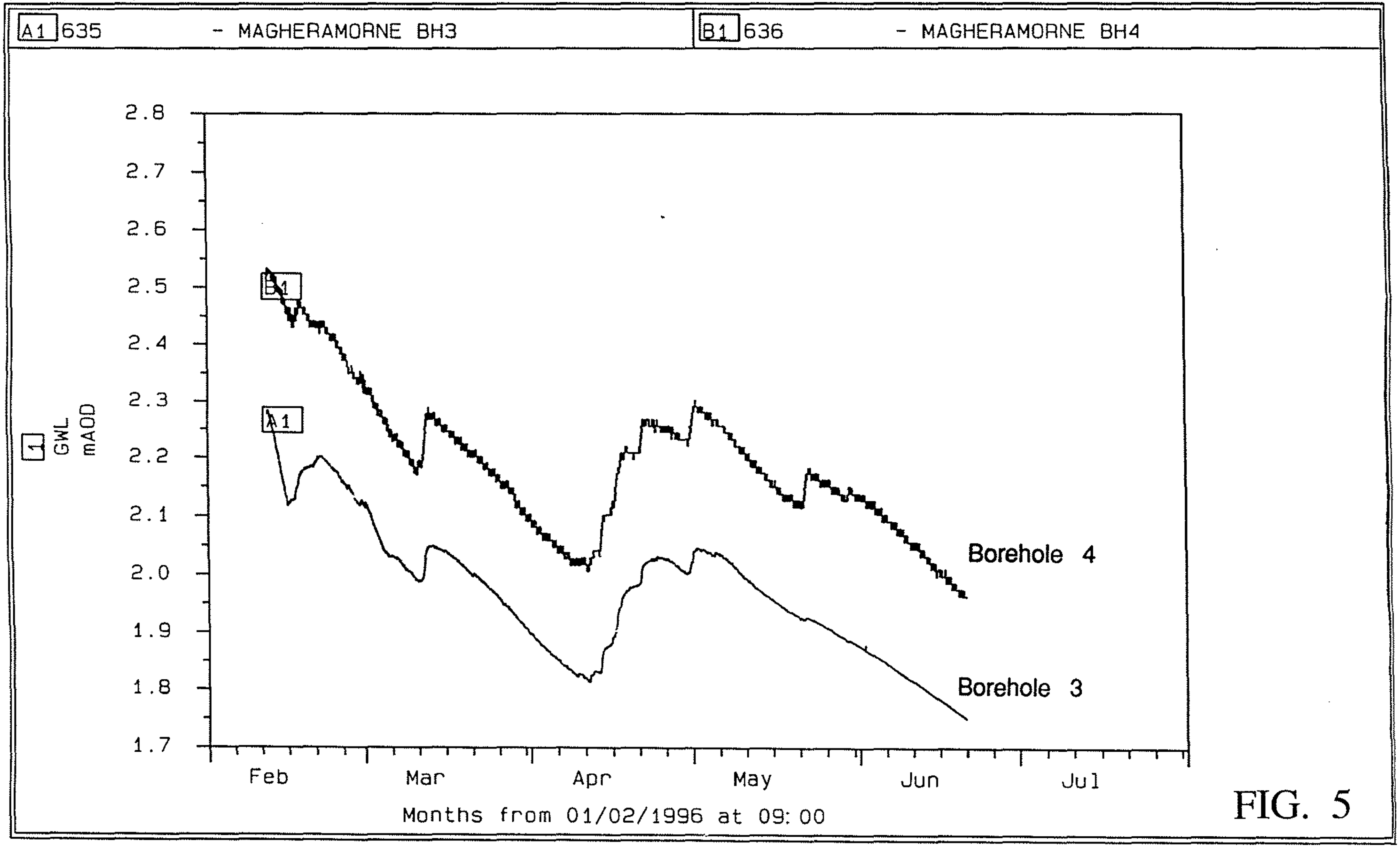
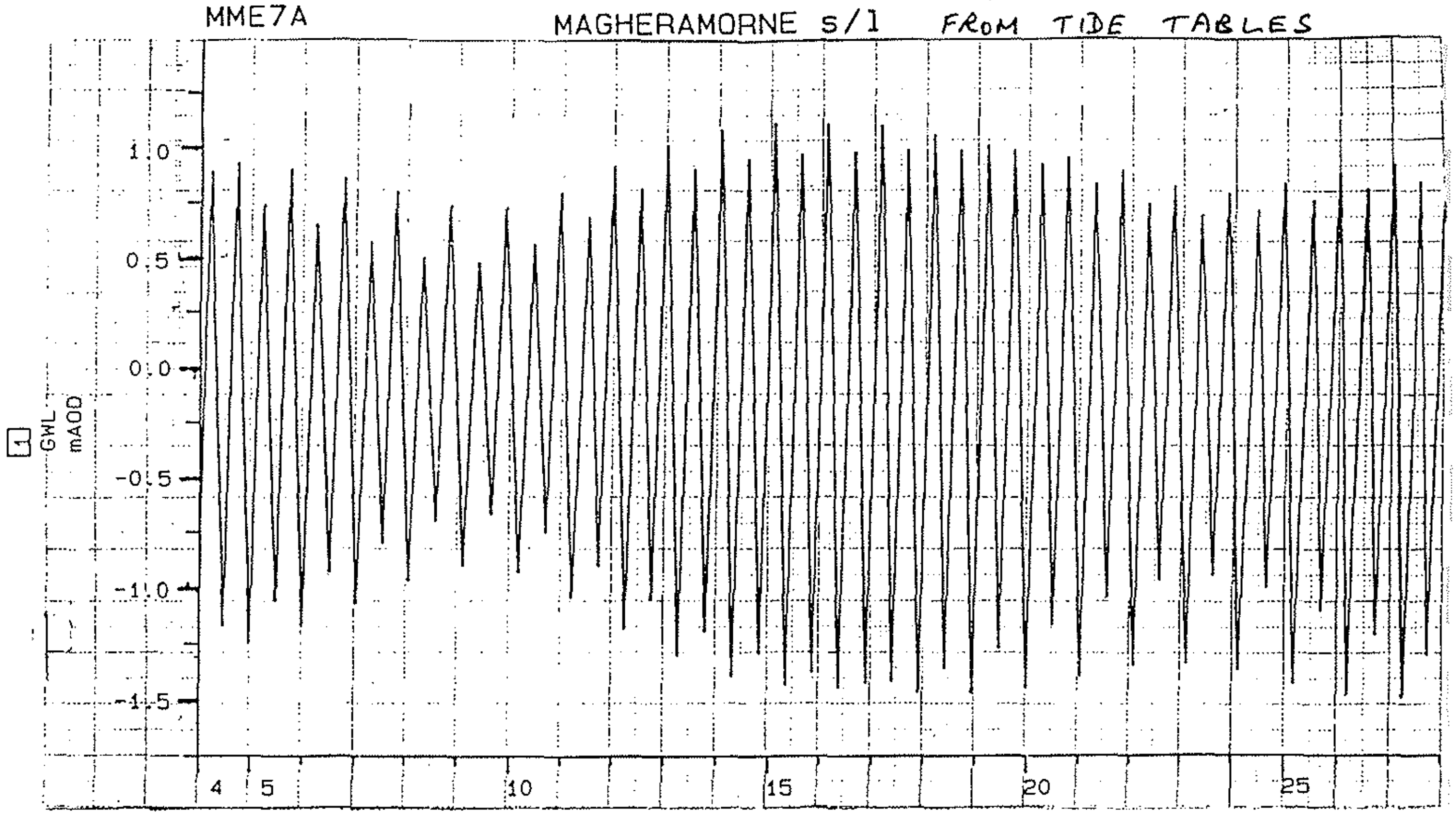


FIG. 5

LARNE LOUGH SEA LEVEL FROM TIDE TABLES



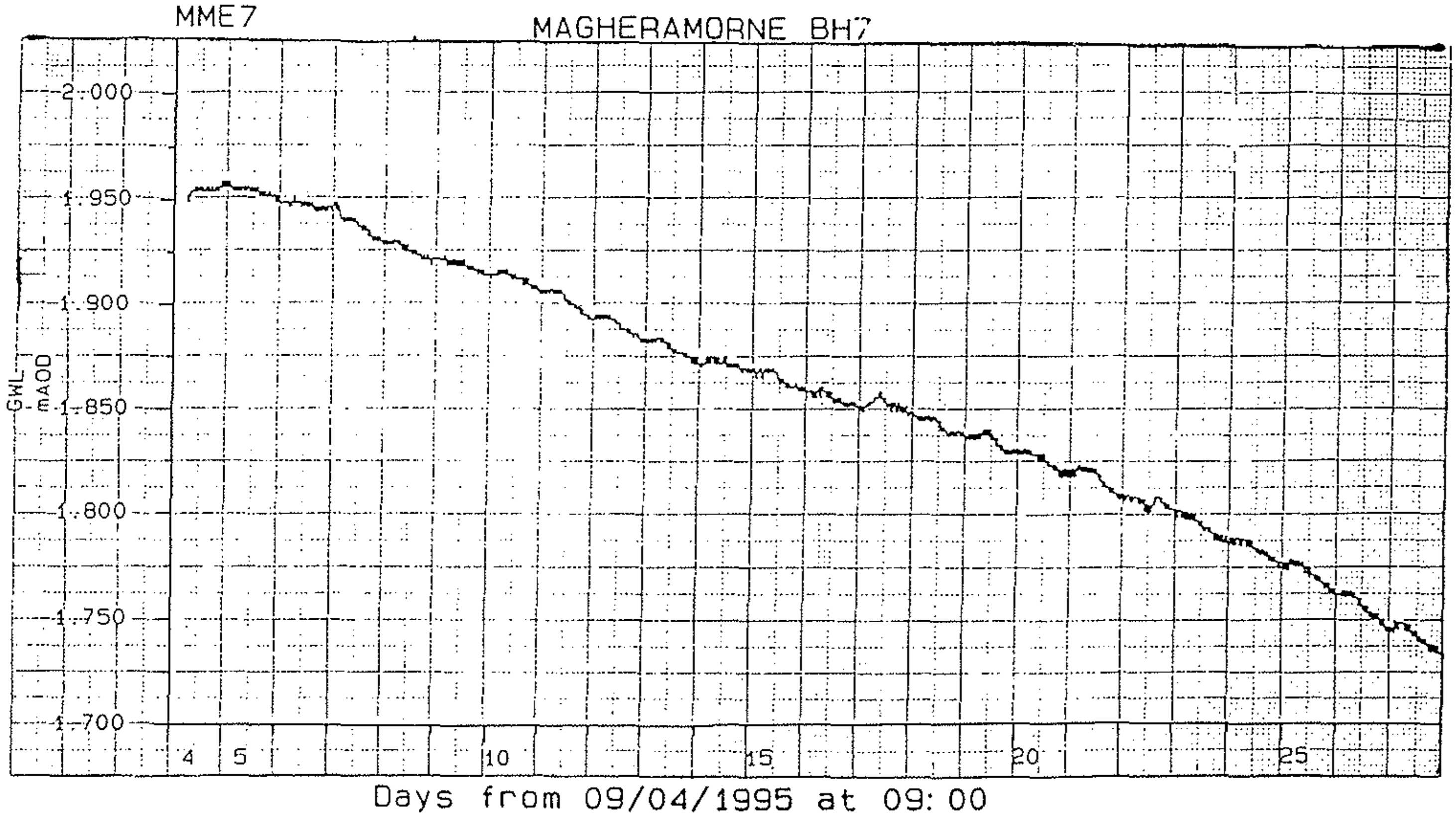


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FIG. 6

Paper No. 5.

Landfills: Assessing and Managing the Long Term Risk and Liabilities.

Geoff Parker, K T. Cullen & Co. Limited.

Leachate 100yrs+
life span of liners and drainage layers

Remote v proximate site location.

LANDFILLS: ASSESSING AND MANAGING THE LONG TERM RISK AND LIABILITIES

Geoff Parker B.Sc.(Eng.), M.E.Sc., M.I.E.I.

Leachate treatment problems on remote sites – i.e. perpetuity

ABSTRACT

This paper provides an overview of risk assessment in the context of landfills. The paper also provides some suggestions with respect to assessing these risks and the liabilities attached. Lastly, the paper provides some concluding remarks on my suggestions for some applied research in relation to landfill assessments in Ireland.

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1. INTRODUCTION

With the advent of licensing of all landfill sites in this country by the Environmental Protection Agency (EPA) starting last year as described in Carty, 1998, consultants, local authorities and private operators are now acutely aware of the need to gather much more detailed information on the environment and the proposed design and operating procedures of the landfill sites than was previously required in the regulatory regime related to planning permissions. It is obvious that the EPA has drawn upon an existing body of experience available within its own resources and from abroad in particular the UK and USA, to develop a framework and guidelines for providing the information necessary to assess the risk of existing and proposed landfill sites. Fortunately, I have had the benefit of 15 years consulting experience in Canada therefore I can say that many of the issues that are cropping up here in Ireland have been the subjects addressed by legislators, researchers and consulting scientists and engineers over the last 20 years in North America.

2. LANDFILLING IN IRELAND

Landfilling of waste in Ireland has been carried out in variety of ways in a broad range of geological/hydrogeological and hydrological settings. In general, the environments which were generally used in Ireland for waste disposal were:

- bogs e.g. Carrowbrowne, County Galway
- estuaries e.g. Balleally north County Dublin
- sand and gravel deposits e.g. Silliot Hill County Kildare
- river valleys e.g. Friarstown in South County Dublin and Kinsale Road, Co. Cork
- green fields underlain by boulder clay e.g. Ballyogan, south County Dublin
- rock quarries e.g. Drogheda

Because most of the landfills in the country apart from the few in large urban centres have been on a relatively small scale (See EPA, 1996), the impacts on the environment have possibly not extended far from the landfill boundaries. However, until very recently there has been very little monitoring of groundwater and surface water quality and gas levels around many of the landfill sites in the country. So the real impacts and therefore risks and liabilities are still not known in many cases.

With the passing of the Waste Management Act in 1996, the existence of a draft EU Directive on the Landfill of Waste and the Landfill Manuals which the EPA are currently preparing and publishing, landfilling in Ireland must and will be upgraded to be consistent the modern practices that have been followed in other countries in Europe, the UK and North America. The number of landfills in the country will decrease because of the high cost of the new risk management controls. The shift to larger landfills has a negative spin off effect; the insignificant impacts experienced with smaller landfills may be magnified with a larger facility. To mitigate the increased magnitude, duration or frequency of the impacts associated with larger landfills the level of environmental controls must be increased.

In order to determine the nature of the environmental controls an assessment of the hazards associated with the specific wastes must be made together with a detailed investigation and description of the physical setting in particular the geological, hydrogeological, surface water and air regimes. The pathways for water and airborne pollutants from the landfill should also be established from an understanding of the physical environment and any planned and/or existing engineered environmental pollution control features. It is only after establishing the hazards and pathways associated with an existing or proposed landfill that a realistic assessment can be made of the risks and liabilities.

3. RISK ASSESSMENT

The framework for assessing the risks and liabilities of landfilling is therefore like the risk assessment methodology for contaminants or contaminated land. The frameworks suggested by CCME, 1992, Derham, 1996 and, Harget and Miller, 1996 can be drawn upon to develop a framework for establishing the risk associated with the closed, existing and proposed landfills in this Country.

Basically the risk assessment methodology involves (see attached figure):

- Hazard identification and quantification
- Pathway description
- Receptor identification
- Exposure frequency
- Toxicity Assessment
- Health Risk Assessment

A determination of risk can be made qualitatively or quantitatively based upon the data/information assembled in relation to each of the items listed above. Quantitative risk assessments involving toxicity and health risk assessment require detailed scientific data for a range of potential contaminants that humans an/or fauna may be exposed to. These assessments may be very costly and are usually not warranted in most cases and in particular if the goal of the exercise is to determine the relative risk of one landfill site versus another. In cases where there is a civil action pending a quantitiative risk assessment involving a limited number of contaminants and receptors may be warranted

The methodology outlined by Derham, 1996 involves a qualitative assessment of the risk using a matrix scoring system that relates to the hazard, pathway and receptor. This system would require a certain level of information on the site conditions and also leachate and gas levels. It would be a useful screening tool for assessing closed and existing landfill sites and prioritising sites for further investigation.

The methodology described by Harget et al, 1996 involves categorising closed landfill sites in the Birmingham area based on quantitative information concerning the site's characteristics , adjoining land uses and gas levels in the landfill. The categories used for the sites relate to the assessed risk of the site and its level of priority for remediation.

The system presented by the Canadian Council of Ministers for the Environment in 1992 was developed for the purpose of classifying, on a national level contaminated sites across Canada. The purpose of the system was to provide scientific and technical assistance in the identification of sites which may be considered high, medium and low risk. The system classifies contaminated sites into general categories of concern in a systematic and rationale manner according to their current and potential adverse impact on the human health and the environment. The system presented by the CCME uses an additive numerical method that assigns scores to a number of site characteristics or factors. The system is not designed to provide a quantitative risk assessment, but rather it is a tool to screen sites with respect to the need for further action.

4. BASIC INFORMATION REQUIREMENTS FOR ASSESSING RISKS

The essential information for semi - quantitative assessment of the risk associated with landfills is as follows:

Identification of Hazard

- Waste quantities- total, annual, daily; desired site life
- Waste types - characterisation
- Potential leachate composition - leaching tests and literature search e.g. EPA waste licence applications
- Potential quantity and composition of landfill gas - laboratory analysis or theoretical calculations or rules of thumb
- Boundaries and shape of the site

Description of Pathway

- Hydrogeological /Hydrological conditions
- Geotechnical conditions
- Climatic conditions - rainfall , evapotranspiration, wind direction and speed
- Ground surface topography
- Landfill design details

Receptor Identification

- Location of housing , industry and recreational/amenity areas
- Adjoining infrastructure and land uses and human activities
- Structural design of buildings
- Ecological conditions i.e. flora and fauna

A key consideration is the type of waste being disposed which has a direct bearing on the nature of leachate and gas and the consequences of these on the environment. I personally believe considerably more work needs to be undertaken with respect to leachate and gas from landfills in Ireland.

5. HAZARDS

5.1 LEACHATE

Leachate can be generated by percolating rainfall, wastes deposited beneath the water table, liquid wastes and by the liquid element of waste degradation. Leachate is a highly polluting liquid whose composition is determined by the nature of the wastes deposited at a landfill. The volume of leachate generated varies with the range of measures used to limit the ingress of rainfall, surface water and groundwater. In general, landfill leachate from municipal solid waste(MSW) landfill sites has high B.O.D. and C.O.D. levels and high concentrations of ammonia, chloride, sodium and potassium. Iron and manganese tend to be high due to anaerobic conditions at the base of the site. The oxygen component of the nitrate ($NO3$) and sulphate ($SO4$) are often low and sulphide compounds are often high. Landfills also contain a wide range of organic compounds. As indicated in Howard et al, 1996, leachate from MSW landfills in the Toronto. Canada area contain volatile, semi volatile and chlorinated organic compounds that are at concentrations that could represent a real threat to ground and water and surface water quality.

For example if toluene is present in the leachate from a 10 ha landfill which is unlined is capped with soil only and is resting on sand deposits, approximately 1.3 kg of toluene could be escaping into groundwater each year. (Assuming 300 mm/year/m2 leachate generation and toluene at a concentration of 0.39 mg/l)

Leachate generation continues as long as water passes through the waste column and as long as there is leachable material left in the waste column. It is generally accepted that the concentrations of dissolvable organic compounds, anions and cations reach a peak in the landfill leachate sometime after the waste has reached its field capacity. With time it is also acknowledged that the concentration of a particular constituent in the leachate will diminish with time due to flushing of the leachate out through the base of the landfill or via leachate collection and pumping systems. Landfills have a potential to contaminate groundwater for hundreds of years. The question therefore arises; How long will the current generation of leachate collection and lining systems perform as intended ?. A model was developed by Rowe, 1991 to determine the length of time a landfill may pose a threat to the environment. An estimate of the contaminating life span of a landfill with different infiltration rates is shown on the attached Figure based on Rowe's model.

Because of the wide range of wastes being disposed in the Country leachate characterisation studies should be carried out to establish the level of hazard attached to the various waste types. In this regard, the available data from all existing landfills should be collated and reviewed to determine the peak concentrations of potential contaminants. Further research will be required in Ireland to develop leachate signatures for the different types of landfills. The range of parameters that leachate is being tested for should be critically reviewed as analyses for synthetic organic compounds in the leachate from landfills are not often undertaken in Ireland.

5.2 LANDFILL GAS

Wherever biodegradable material is deposited in landfill sites, microbial activity will generate landfill gas, which is a mixture of flammable, toxic and asphyxiating gases. The quantity of landfill gas depends on the mass of biodegradable material deposited and the age of the waste.

The composition of the gas varies according to the type and phase of breakdown which is occurring within the site at any specific time. Initially carbon dioxide predominates, through significant quantities of hydrogen are also evolved. Methane (about 65%) and carbon dioxide (about 35%) are the major constituents of the gas which evolves during the usually predominant anaerobic phase of waste breakdown. However, studies have shown that there are a wide range of organic compounds that are emitted to the atmosphere including vinyl chloride, toluene and benzene from landfills. The paper by Morris et al would suggest that a 900 tonne per day MSW landfill would produce about 3.2 tonnes per year of volatile toxic emissions.

The major constituents of landfill gas are colourless and odourless although they are normally found mixed with other gases, some of which give rise to odour. It is usually saturated with moisture, and is corrosive. The density of landfill gas is dependent on the relative proportions of its major components, but is usually about the same as air.

The onset and rate of degradation processes in the wastes vary both within and between sites. The evolution of significant quantities of methane may take from three months to more than a year to start and can continue for well in excess of 20 years. There are many factors which influence gas evolution including the physical dimensions of the site, the types of waste and their input rates, moisture content, landfill pH, temperature and waste density, together with site operational practice.

Gas pressure within a landfill is dependent on the gas evolution rate, the permeability of the fill, and the permeability of the surrounding strata; it can be varied by changes in the level of the leachate in the site.

Differences in atmospheric pressure will affect the pressure differential between the site and the atmosphere which in turn affects gas emissions from the site.

6. PATHWAYS

6.1 LEACHATE MIGRATION

Leachate will migrate from the base of the landfill downwards towards the water table and migrate away from the site in the direction of groundwater flow. The impact of the leachate on groundwater will depend on;

- the volume of leachate generated.
- the depth of the unsaturated zone below the base of the landfill.
- the permeability and attenuation capacity of the geological strata.
- the nature of the permeability (inter granular/fissure).
- the hydraulic gradient.

Where there is a significant unsaturated zone beneath the landfill the impact can be minimised as the migrating leachate undergoes attenuation in this oxygen rich environment. This is particularly the case when the landfill overlies overburden which displays inter granular flow permeability. Whereas the leachate plume may only be detectable up to several hundred metres away where inter granular permeability predominates, the plume could extend some 2-5 kms where karst limestone conditions exist.

Where non engineered landfills were located on wet land the impact of leachate migration was usually limited to the nearby drainage features. In this case, small streams discharging from the site tend to be grossly polluted and estuaries can display deterioration in water quality.

The primary migration pathway for leachate is through the landfill base. The quantity of leachate through the base will depend on the landfill design and the hydrogeological conditions. The direction and rate of migration will depend on the hydrogeological conditions. The concentration of the contaminant along the migration pathway will depend on the physical, chemical and biological processes/conditions that exist in the geological media through which the groundwater/dissolved contaminant is migrating. Because of the complexity of the geological settings in which landfills have been and will be placed in Ireland, (apart form possibly estuarine sites) the migration pathways are probably the least understood parameters in the overall assessment of the risk of the landfill. Many tens of thousands of pounds and 6 months to a year of time are required to acquire a basic understanding of the potential pathways in many areas in the country.

Leakage calculations may be carried out using the equations provided in the paper by Giroud, 1992 to obtain a comparative estimate of leakage through a composite liner which has a water table at the underside of the mineral layer. Leakage calculations need to be carried out for the various options taking into account a variable height of leachate, and the hydraulic gradient across the liner and in the underlying hydrogeological conditions. There may be a tendency in the engineering profession to ignore the underlying hydrogeological conditions which could possibly lead to overestimates or underestimates of the rate or quantity of contaminant movement out of the landfill because in practice the gradients are less than or greater than 1 which is the value assumed in some of the leakage equations by Giroud, 1992. There may be cases where the hydraulic gradients beneath the landfill are actually upward. On the other hand there could be a leachate head greater than 1 m and therefore a hydraulic gradient greater than 1 might arise in some landfills. These situations may arise if there is an operational need to store large leachate quantities within the landfill itself or because the leachate collection piping system fails due to clogging. It is therefore vital to examine a worst case scenario with respect to leachate head. Therefore, I would recommend use of the following equation by Giroud for leakage calculations through circular holes:

$Q = 0.21\ iavg\ \ a^{0.1}\ hw^{0.9}\ \ ks^{0.74}$

6.2 GAS MIGRATION

Gas may move in any direction within the wastes. Lateral gas movement will be encouraged by low permeability compacted layers whilst vertical gas movement will occur around gas and leachate wells particularly if they have been surrounded by hard-core. Gas may also move vertically through compacted clay covers (albeit at low rates) and at the sides of the site at the interface between the wastes and surrounding strata and escape via settlement cracks.

Gas migration from the site can occur in several ways. It will move through permeable strata or peremable horizons in otherwise lower permeabilty strata or for considerable distances along faults, fissures or cavities in the strata. It can pass along man-made features such as mine shafts, roadways, sewers, or dissolve in leachate or groundwater and subsequently be released some distance from the site boundary.

Migration pathways are affected by surface capping which in turn may be sealed by heavy rainfall, ice or snow, by changes in the permeability of the waste as it settles and decomposes or by subsequent disturbance of the site. Gas movement also alters with variations in gas and atmospheric pressure. Emission of landfill gas may be detectable by smell, sound, or by the presence of bubbles in surface water. Vegetation may also be adversely affected by landfill gas in the ground resulting in bare patches or brown foliage and subsequent die back.

7. LANDFILL DESIGN AND RISK MANAGEMENT

7.1 DESIGN CONCEPTS

As discussed above the risks associated with landfills for different types of wastes will be different. Clearly, it can be said the level of environmental protection provided in the design of a landfill for builders rubble would be different than a facility to contain solid or liquid hazardous waste. The risk of the former facility should be significantly less.

Similarly the risks associated with specific hydrogeological settings and design concepts will be different for example the risks associated with the design concepts depicted in the attached Figures will decrease with increasing level of engineered and natural containment.

The design concept chosen will depend on the maximum degree of risk acceptable to the public and the regulations. If zero risk to groundwater is an objective, as dictated by the downstream users of groundwater and also as required by the EU Groundwater Directive and Irish regulations, the design concept will have to resemble the Hydraulic Trap concept illustrated in the accompanying Figure .

If a small degree of risk is acceptable then indirect discharge or leakage may be permissible to a certain value as long as there is not a risk to human health or ecosystems etc. The Waste Management Act Section 41(2)(b) gives the authority to the EPA to specify the concentration of a pollutant in an environmental medium or a discharge rate which shall not be exceeded. Even if leakage is permitted, engineered or hydrogeological containment will be a feature of the design to reduce leakage and risk to a practical minimum. Although clearly the European Commission draft directive on Landfills does allow for site specific assessments and a site specific non prescriptive design approach which I would favour. Therefore, in some cases a landfill design without a HDPE liner at the base of the landfill might be acceptable.

7.2 LEACHATE RISK MANAGEMENT

Landfills are lined to manage the risks associated with leachate and gas. There are several possibilities with respect to liner design. The modern landfill will generally have a liner that includes a geomembrane layer and normally mineral soil layer with a minimum thickness of 1 m. Variations may include a bentonite enhanced soil or a geocomposite clay layer. Double lining with HDPE sheets with an intervening leakage detection layer may also be installed in some geological settings. Leakage calculations need to be performed and the degree of impact on groundwater beneath and downgradient of the site should be determined. The appropriate design will depend on the maximum allowable leakage which in turn depends on the site specific hydrogeological conditions.

The unfortunate consequence of the philosophy of containment of leachate is the need to collect and treat leachate indefinitely. Thus as a society we will be passing on to future generations a financial liability which would otherwise did not arise with the former dilute and disperse landfill sites. If we turn the clock forward ten years and imagine that all of the active landfill sites in the country are lined and the tonnage of MSW being landfilled is in the order of 10,000,000 tonnes per year and this is spread over a depth of 5 m I estimate that there will be something in the order of 600,000 m3 of leachate collected and requiring treatment in the country.

Leachate treatment may involve transporting collected leachate by tanker to a municipal waste water treatment plant if the landfill is close to an urban area. Alternatively, a pipeline may be used. However, these methods may be cost prohibitive for a site that is located in a remote/rural area. In such isolated areas on site treatment will be required that will include as a minimum an equalisation tank, and air stripping to reduce ammonia, BOD and COD levels. Further treatment using reed beds or physio-chemical processes may also be required in order to reduce the constituents in the effluent to acceptable levels. It is unlikely that direct discharge of leachate and its subsequent dilution alone in a surface water body will be acceptable in the current regulatory regime that includes BATNEEC.

The risk associated with leachate may also be managed by operating a bioreactor and also installing a specific capping system. There are two approaches in relation to landfill cap design. One is to minimise infiltration immediately after landfilling ceases. This approach effectively decrease the level of risk in the short term and pushes the risk off to the future if the impermeable capping system fails or degrades. The other is to allow rainwater to infiltrate the waste body under controlled conditions to allow waste biodegradation and flushing and collection of contaminants in order to decrease the contaminating life span of the landfill to perhaps decades which is the proven service life of liners and leachate collection systems. Since the service life of some engineered systems is short it is important to design and build redundancy into leachate collection systems and also provide a means for long term maintenance to ensure the longest service life possible.

7.3 LANDFILL GAS RISK MANAGEMENT

Landfill gas in modern practice will generally be collected actively or vented passively. Purpose built systems of pipe installed through the waste body would be effective for gas collection and also recirculation of leachate. If the gas is collected it would normally be flared or burned to generate electricity.

Measures to control the migration of landfill gas from a landfill site are a critical element in risk management. The nature of the surrounding geological conditions., the proximity of houses and the nature of adjoining land use dictate the degree to which engineered barriers or pumping systems will be required to close down a pathway and minimise risk.

My experience suggests that methane can be found at concentrations exceeding 50% in unsaturated sand and gravel deposits at a distance of 100 metres from the landfill whereas where the landfill is within or resting on boulder clay the methane levels is undetected at 10 m away. The general rule of thumb for unlined landfills in sand and gravel deposits is that it is possible that landfill gas may be at a level of concern that would trigger some remedial action at a distance that is 10 times the depth of the waste below the original ground surface. The city of Birmingham risk assessment approach would landfills which have housing 50 metres away in a high risk category and requiring action. In the same model the maximum distance of concern was 250 m from the landfill. It is clear from this discussion that buffers around a landfill site are a key tool in gas risk management.

7.4. LANDFILL MONITORING

Landfills are monitored in order to confirm that risks are being managed and the maximum accepted release levels are not being exceeded.

An integrated programme must be developed involving:-

- leachate
- groundwater including under liner sampling points
- surface water
- gas/air

Because the risks associated with a landfill could exist for decades to hundreds of years the monitoring and after care programmes must also be continued for the same length of time. Even modern landfills will leave a legacy of potential risk to groundwater and surface water for several generations beyond the end of their operating life.

8. CONCLUSIONS

The conclusions of the paper are a summary of some of the work that I see as being required in order to quantify the risk associated with closed, existing and future landfills in the country.

The work that might be carried out as publicly or privately funded applied research projects is listed below.

- preparing an inventory of landfill sites including closed sites and a protocol for assessment of the risk of these sites to the environment and humans
- carrying out a study to characterise leachate across country to determine if synthetic organic compounds are a significant constituent and hazard associated with Irish landfills
- carrying out a risk assessment of all Irish landfills larger than 20,000 tonnes per year and any significant closed sites
- a study to develop a master plan for treatment all the leachate that will be collected in the new generation of landfills
- determining the real leakage rate through the Irish composite liners and capping systems proposed by the EPA
- preparing a national protocol for the testing and verification of the permeability of compacted clay liners

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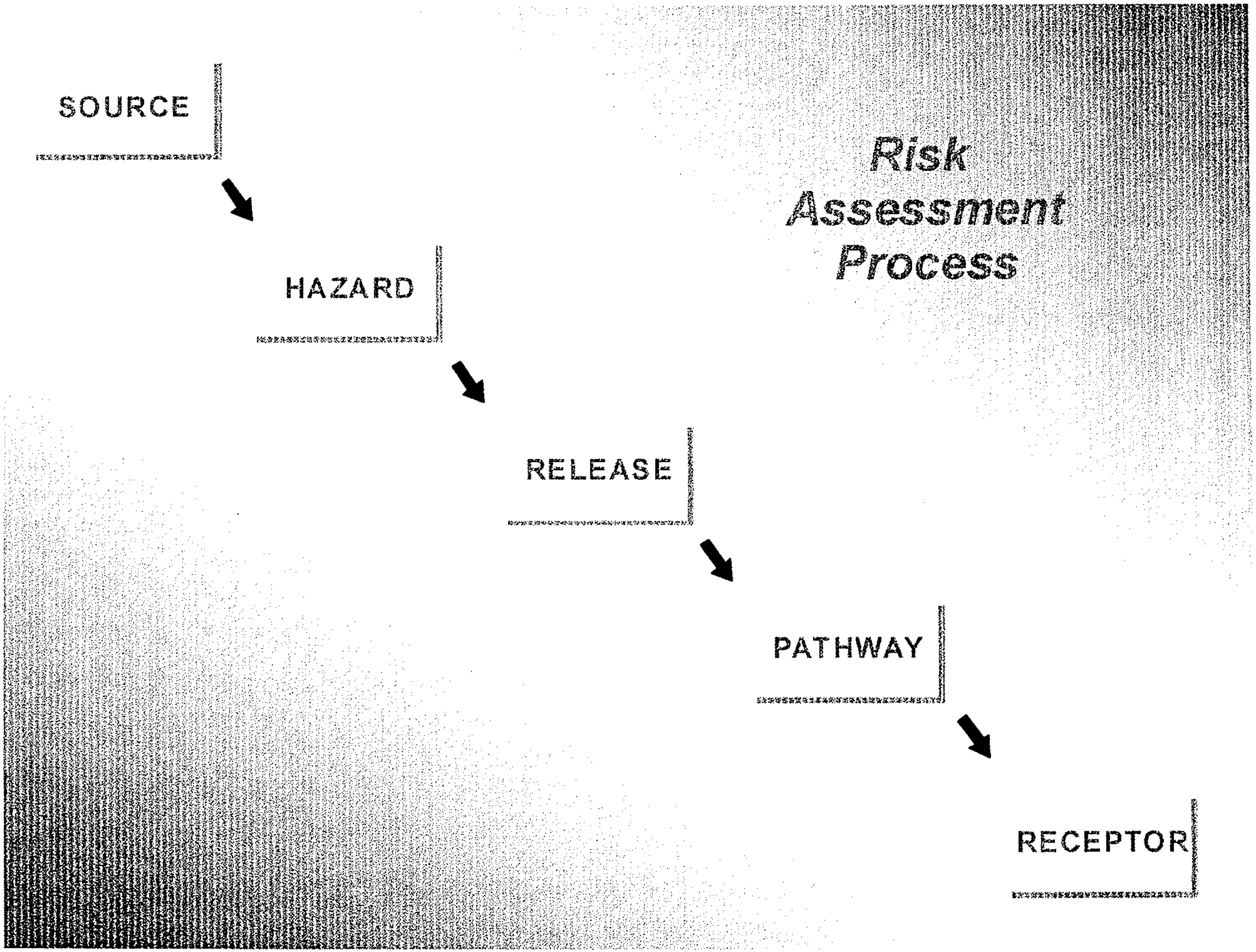
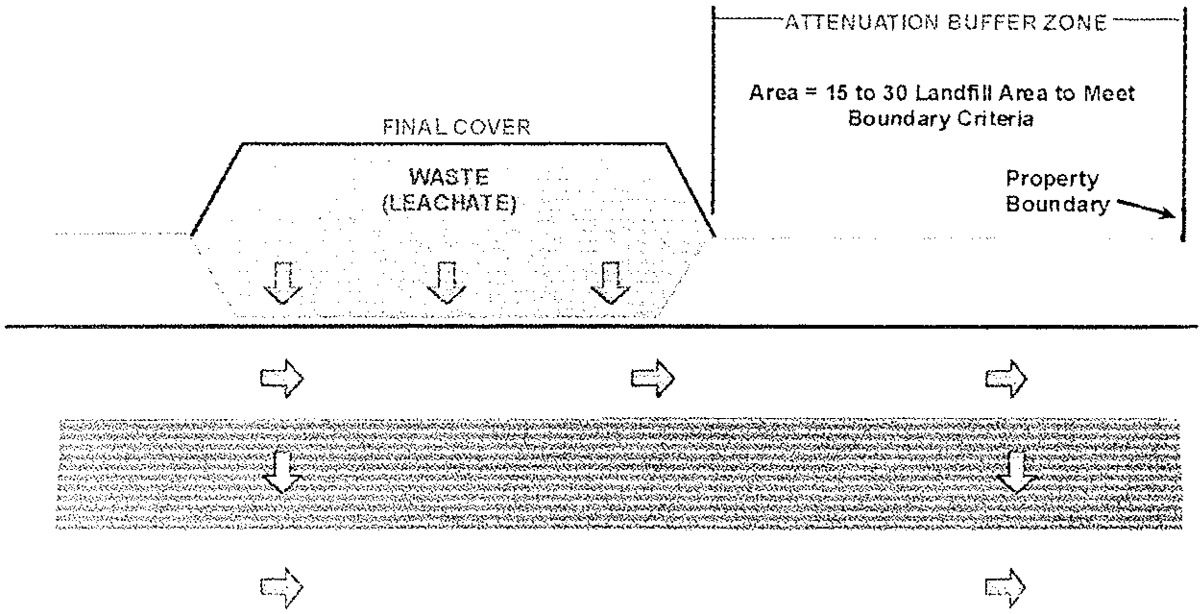
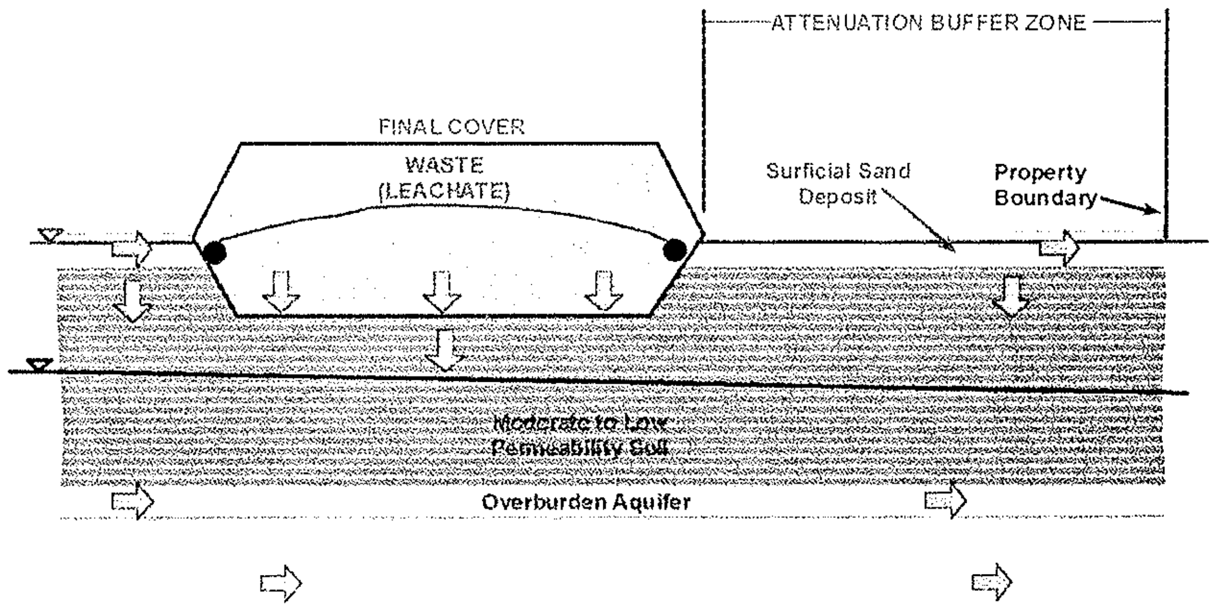

Risk
Assessment
Process
SOURCE
HAZARD
RELEASE
PATHWAY
RECEPTOR

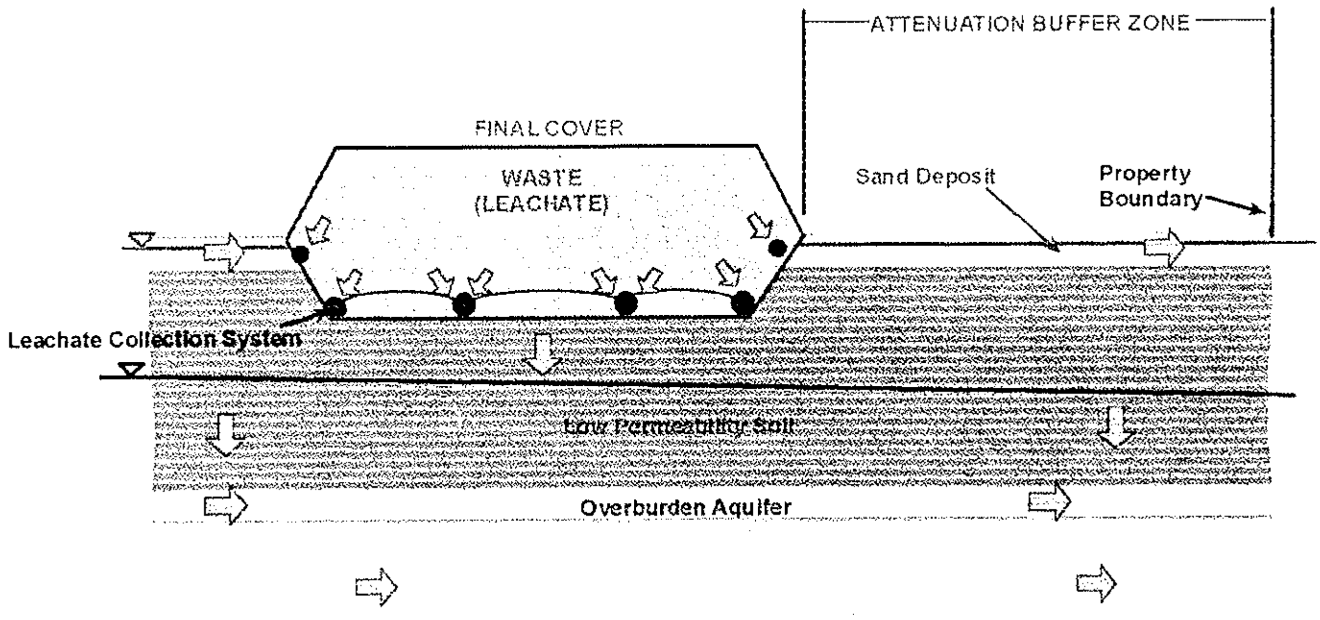
Natural Attenuation Landfill Concept in Sand & Gravel Deposits



Natural Attenuation Landfill Concept in Moderate to Low Permeability Soils



Leachate Collection & Geologic Containment Landfill Concept



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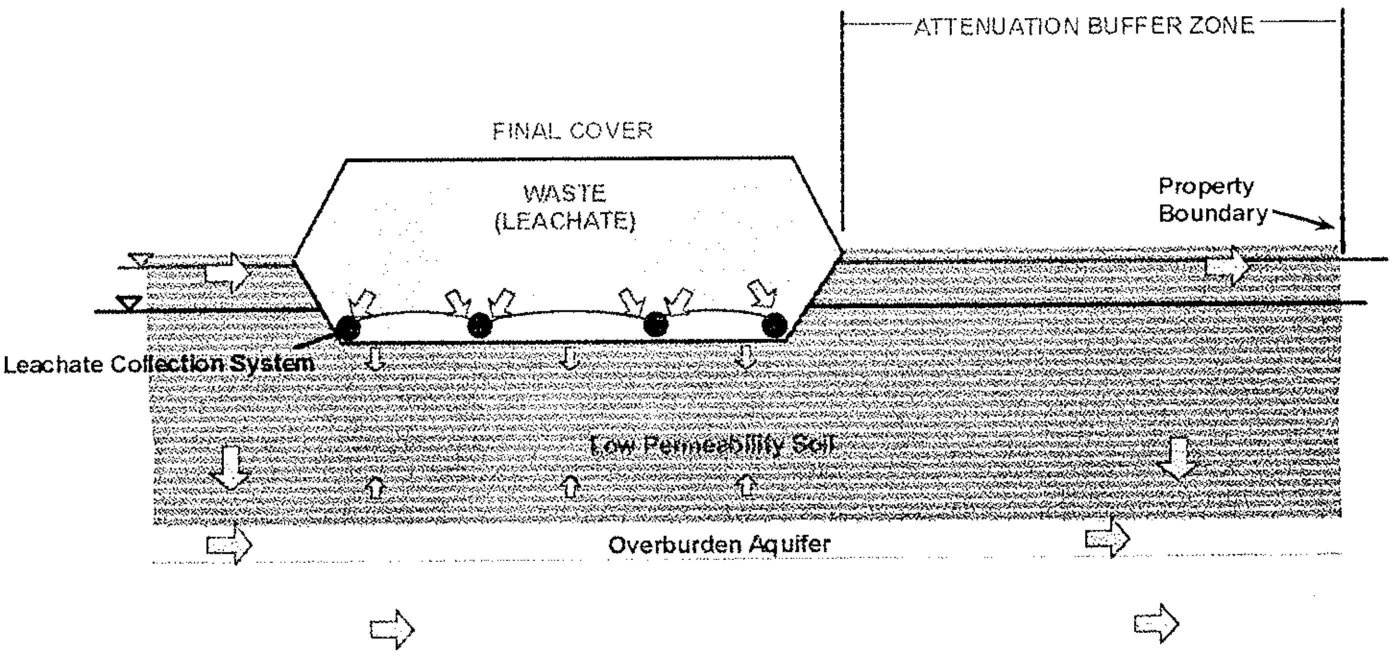
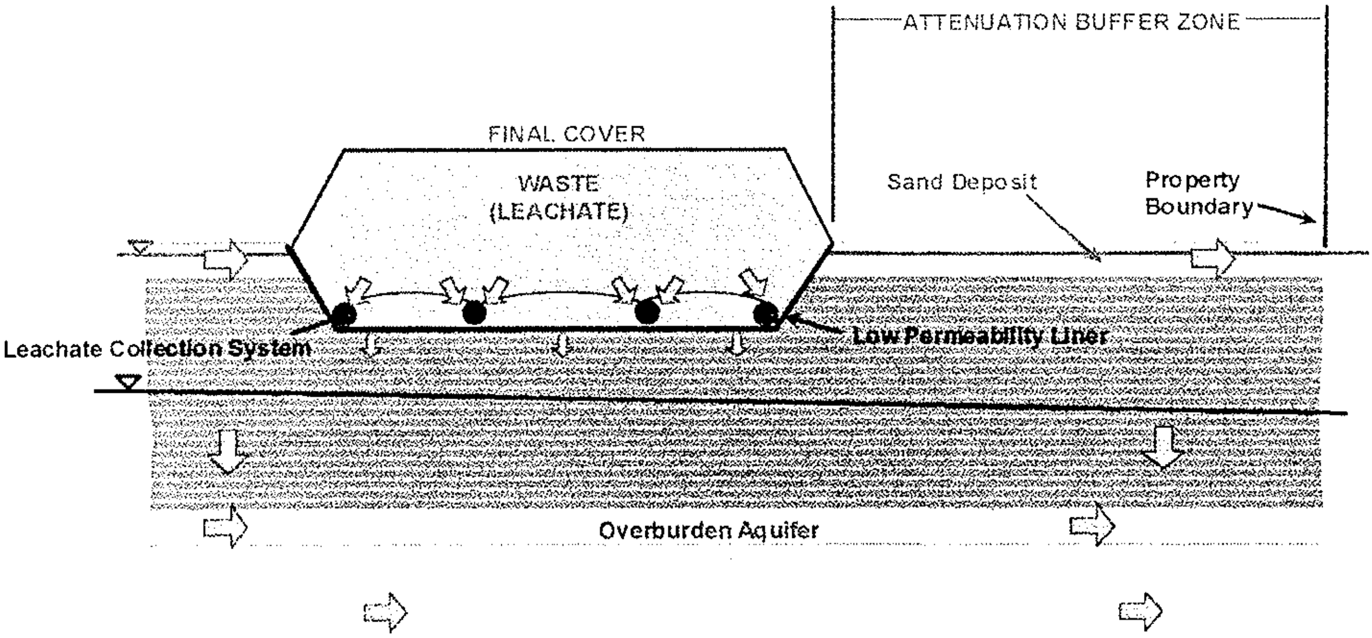


Fully Engineered Landfill Concept



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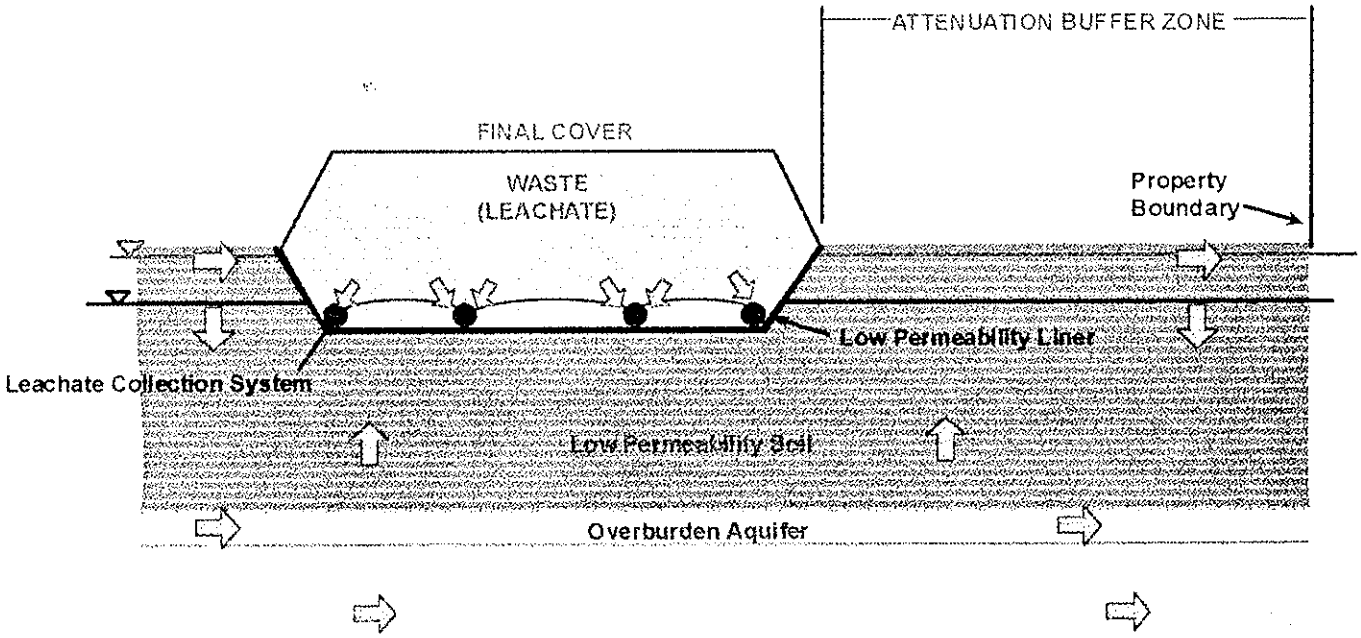
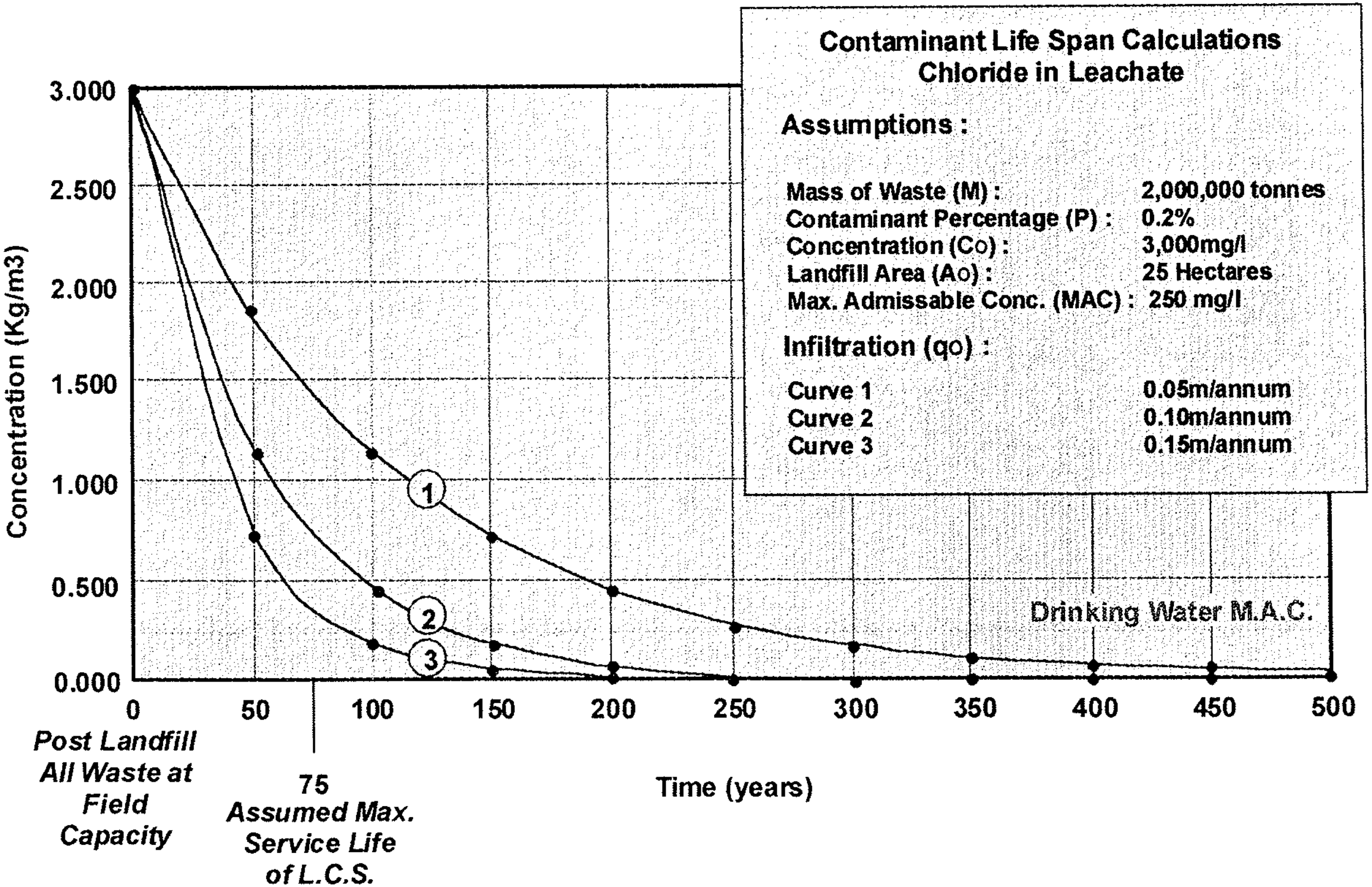


Effect of Infiltration / Leachate Removal Rates on Chloride Concentration Decay Curves - 2 Million Tonne. Landfill on 25 Hectares



Paper No. 6.

Licensing of Waste Recovery and Disposal Activities.

Gerry Carty, Environmental Protection Agency.

LICENCING OF WASTE RECOVERY AND DISPOSAL ACTIVITIES

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Paper presented to the International Association of Hydrogeologists (Irish Branch) 18th annual seminar on 22nd April 1998.

ABSTRACT

The Waste Management Act 1996 provides for the integrated licensing of waste recovery and disposal activities. The licensing system was introduced in April 97 with the EPA as the national licensing authority. The first six applications were received by the Agency by 1st May 97 and to date a total of 34 licence applications have been received. Applications for prescribed activities must be made to the Agency by specified dates and to date these have been for waste disposal activities such as landfills and transfer stations. By March 99 a licence application will be submitted for all existing local authority landfills. As none of these activities have been licensed or permitted previously, and the majority are unlined sites, significant improvements are required to achieve the standards set out in the legislation. One of the most significant issues is the hydrogeology of the site and the reduction or elimination of emissions to the soil and groundwater. A licence can only be issued if the applicant demonstrates, to the satisfaction of the Agency, that operation of the activity in accordance with the licence conditions, will not result in environmental pollution.

INTRODUCTION

The Fifth EU Action Programme on the Environment, *Towards Sustainability*, identified waste management as one of the priority issues to be tackled within the Community. Certain policies and targets were set including:

- a requirement for waste management plans in Member States;

- stabilisation of quantities of municipal waste generated at EU average 1985 level (330 kg/capita/annum);
- no export outside EU for final disposal of municipal waste and hazardous waste (amber and red list wastes);
- EU-wide infrastructure for safe collection, separation and disposal of hazardous waste;
- recycling/re-use rates for all consumed paper, glass and metals to reach an average of at least 50% by the year 2000;
- creation of a market for recycled materials; and
- considerable reduction in dioxin emissions.

The actions to be taken to meet these targets included:

- landfill Directive to be made operational;
- packaging Directive to be made operational;
- cleaner technologies and product design to be encouraged;
- policies on priority waste streams to be developed;
- prohibition on landfilling of certain wastes;
- development of reliable waste statistics;
- system of liability to be put in place;
- economic incentives and instruments to be applied; and
- standards for dioxin emissions from municipal waste incineration.

A review of the Fifth Action Programme showed a certain amount of progress in areas such as waste management planning, packaging and priority waste streams (EEA, 1995). However, it is clear that in general terms progress is slow. More specifically, it is clear that the target set for municipal waste generation, a key indicator, will not be met and that, indeed, per capita generation of municipal waste is likely to increase in the next five years.

In 1996 the European Commission completed a review of the Community Strategy for Waste Management. The Commission confirmed that the hierarchy of principles established in 1989, namely the prevention of waste shall remain the first priority, followed by recovery and finally by the safe disposal of waste. It also stated that the landfilling of waste should be seen as the last and least best solution. However it does recognize that, in particular cases that landfill is the only reasonable form of waste disposal. The pre-treatment and sorting of waste prior to landfilling is advocated and the landfilling of only non recoverable waste and inert waste in the mid term is suggested. It is also recommended that EU countries become self sufficient with a view to avoiding shipments for disposal between member states.

A revised proposal for a landfill directive was published in December 1997. This proposal develops some of the principles contained in the review referred to above. Limits are proposed on the landfilling of biodegradable waste, the pre-treatment of waste is proposed, the cost of landfilling should include all costs involved in setting up and operating the facility as well as closure and aftercare costs for a period of at least 30 years. For existing sites a conditioning plan must be presented within 3 years and implemented within 5 years. A much higher priority must be given to the collection and treatment of landfill gas. The distances from residential and recreation areas must be considered. It will also require the preparation of a national strategy for implementation of the reduction of biodegradable municipal waste going to landfills.

NATIONAL WASTE ARISINGS

National waste arisings for 1995 were estimated to be 42,260,757 tonnes. Of this, approximately 31,000,000 tonnes originated from agricultural sources, mainly animal manure's. The municipal and industrial sectors are therefore estimated to have produced over eleven million tonnes of waste in 1995 as set out in Table 1 below.

Table 1: Total Non-Agricultural Waste Arisings in Ireland (1995)

Waste Category	Quantities Arising	
	tonnes/annum	**(%)**
Municipal (including recycled materials)	1,848,232	16.4
Other Wastes collected by or on behalf of local authorities	953,189	8.5
Industrial (non-hazardous)	7,410,982	65.8
Industrial (hazardous)	243,754	2.2
Healthcare Wastes	20,000	0.2
Dredge Spoils	784,600	7.0
TOTAL	**11,260,757**	**100**

RECOVERY FACILITIES

Waste recovery infrastructure consists of a growing number of urban and rural bring schemes and one collect system (Kerbside Dublin) operating in three areas in west and north west Dublin. There are also civic amenity sites mostly on existing landfills where the public can deposit recyclable materials.

DISPOSAL FACILITIES

Questionnaires completed by local authorities in 1995 indicate that there were 118 active landfill sites in Ireland. Local authorities operated 87 of these landfill sites, while the other 31 sites were privately run. Of the latter, one was operated by a contractor on behalf of a local authority; the remaining 30 were mainly private industrial landfills. While information was obtained for the majority of landfill sites, only limited information was provided for some of the privately operated sites.

The majority of landfill sites are relatively small with 58% accepting less than 15,000 tonnes per annum. A further 34% accept up to 50,000 tonnes per annum with only 7% accepting greater than 50,000 tonnes per annum.

The availability of accurate information is the key to good planning and it is hoped that the National Waste Database will contribute increasingly accurate information on waste management in Ireland in the coming years. It should be stressed, however, that the accuracy of any database is only as good as the information provided to it. There is still much room for improvement in this area. Accurate tracking of waste arisings, recovery rates and disposal rates will require more sophisticated measurement tools than are currently used with the widespread absence of weighbridges at landfill sites a case in point.

LICENSING UNDER THE WASTE MANAGEMENT ACT 1996

Regulations implementing an integrated licensing system for waste management facilities were introduced in April 97. Commencing with new and existing landfills, facilities for waste recovery and disposal require a licence from dates specified in the regulations.

The first applications were received by the Agency by the 1st May 97 and the next tranche which included existing landfills accepting more than 40,000 tonnes per annum were lodged with the EPA by 1st October 97. Applications for landfills accepting 20,000 to 40,000 tonnes per annum were lodged with the EPA before 1st March 98. Applications for existing hazardous waste activities and transfer stations with a capacity in excess of 25,000 tonnes per annum are to be submitted before 1st May 98. Applications for remaining local authority landfills are due before 1st October 98 and 1st March 99. In addition all new landfills and hazardous waste activities require a licence prior to commencement.

The EPA has published a guide to implementation and enforcement in Ireland titled 'Waste Management Licensing' and this is available from our Dublin office for those who wish to know more about the licensing process.

To date a total of 34 applications have been submitted. Two of these have been withdrawn by the applicant. All remaining applications are at various stages of processing and decisions on a number of applications are likely to be made within the next month. Where local authorities are required to submit applications they have in all cases been submitted by the prescribed date.

As none of the local authority activities were licensed or permitted in the past; preparation of the application, collection of monitoring information and management of the activity have been the primary areas on which attention has been focused.

In processing applications additional information has been sought in all cases as, in general, insufficient monitoring or investigation information has been provided with the initial application. It is also worth noting that where investigations and/or studies have been undertaken quality control of the information collection and reporting has in many cases been poor. Where this has occurred additional expenditure is needlessly incurred by the applicant. Closer monitoring and supervision of work undertaken by advisors and contractors would reduce costs significantly. As a significant number of applications for existing activities, mainly landfills, have to be submitted and applicants become more familiar with the legislation an improvement in the quality of applications is anticipated.

The Regulations require that specific information is submitted before an application is to be considered a valid application. In processing an application the EPA must check that the requirements set out in the Regulations are satisfied. Where the information in the application is incomplete or inconsistent requests for additional information and/or clarifications are issued. Delays in responding to these requests and the need to undertake further investigations where the initial investigations were inadequate have delayed the issuing of decisions by the EPA.

All information and correspondence in relation to an application is available for inspection by any interested party. The Regulations provide for certain information to be considered as confidential information by the EPA but to date no such requests have been received in respect of the applications on hand.

BATNEEC

This is an acronym used to describe *'best available technology not entailing excessive cost'*. It is a requirement of the Waste Management Act that it is implemented to prevent or eliminate, or where that is not practicable, to limit, abate or reduce an emission from an activity. A licence shall not be granted unless BATNEEC is used. BATNEEC is concerned with emissions from an activity. It should therefore be **Best** at preventing pollution and **Available** in the sense that it is procurable by the operator of the activity. **Technology** includes management techniques, training and maintenance. **Neec** sets out the balance to be achieved between environmental benefit and financial cost.

In defining BATNEEC the Act differentiates between new and established (existing) activities. Account is taken of the state of technical knowledge, the requirements of environmental protection and the risk of environmental pollution. In addition for existing activities the nature, extent and effect of emissions, the nature and age of the facility and the costs incurred in improving or replacing facilities must be taken into account. In the identification of BATNEEC, emphasis is placed on pollution prevention techniques and in particular management of the activity.

What must be considered in defining BATNEEC for existing activities is what is actually occurring. Effectively the fulcrum between BAT and NEEC is located in contrasting places. For new activities the emphasis is much more in the direction of BAT. For existing activities it is closer to the NEEC end of the concept and is highly dependent on a site by site and case by case assessment. Where monitoring returns indicate that an emission is causing environmental pollution remedial measures will be required as a licence cannot be issued if this continues.

Further details of what the EPA considers as BATNEEC is contained in a paper presented to the Institution of Engineers of Ireland (IEI,1998).

An issue which must also be considered because of the number of unlined landfills for which licence applications are being submitted is the need for **hydrogeological isolation**. With the exception of a handful of lined sites constructed in the past ten years all of our landfills are unlined. Hence, where emissions are causing environmental pollution, investigations need to be undertaken to determine on a site by site basis the measures which are required to control and eliminate the emissions. This is an information intensive process where the site is investigated; groundwater flows, permeability, and solute transport are modelled, and design alternatives are examined in terms of effectiveness, reliability, robustness and cost. A solution which can be constructed and maintained is required if a licence is to be granted.

QUALITY MANAGEMENT

Our experience to date with specialist hydrogeological reports and investigations submitted with licence applications indicates a need for improved quality control on the factual information generated and interpretation of this information. In some instances reports are submitted on groundwater and geological issues without adequate checking. The need to seek further information and possibly have further investigation work carried out to clarify deficient information is an expensive and undesirable situation. Neither does it lead to any increase in public confidence but rather the reverse - it increases public concerns and is wasteful of public resources. The introduction of a quality management systems approach in this area would lead to significant improvements.

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Paper No. 7.

Landspreading of Organic Wastes and Groundwater Protection.

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Environmental Protection Agency.

LANDSPREADING OF ORGANIC WASTES AND GROUNDWATER PROTECTION

Donal Daly, Geological Survey of Ireland
Vera Power and Margaret Keegan, Environmental Protection Agency

ABSTRACT

A legislative framework comprising the Water Pollution Act, Environmental Protection Agency Act and the Waste Management Act is in place in Ireland to regulate the landspreading of a range of organic wastes. Currently, an estimated 240,000 ha of land is required for disposal of waste under Integrated Pollution Control (IPC) licensing and in the future the Waste Management Act will further extend the control of landspreading. One of the roles of licensing is to ensure that landspreading is carried out in a suitable manner such that it will not cause environmental pollution.

In identifying suitable lands for landspreading, a risk and risk management approach should be taken. A Groundwater Protection Scheme provides the framework for such an approach. It considers the hazards, which are in this case certain microbes, some inorganic elements and some metals; the pathway element, which is the vulnerability of groundwater whereby the soil and subsoil act as a protecting layer; the targets at risk, which are wells and aquifers; and the potential consequences. This is followed by a response to the risk.

The Groundwater Protection Scheme consists of a series of maps, including vulnerability, aquifer and groundwater protection zone maps, and groundwater protection response matrices for different activities. The vulnerability map comprises zonations based on type and thickness of subsoil, and indicates the likelihood of groundwater becoming contaminated. The consequence of contamination depends on the value of the groundwater and is represented by the aquifer categories, ranging from regionally important to poor aquifers, and groundwater sources (such as local authority public supplies). These two sets of maps are integrated to form groundwater protection zones, which are associated with a groundwater protection response matrix. The response matrix outlines the acceptability of the activity within the zones and describes conditions that may be required to ensure good environmental practices are followed. It provides a structured framework for assessing the suitability of lands for landspreading to ensure groundwater protection.

INTRODUCTION

The concept of groundwater contamination risk is proposed as a framework for decision-making on landspreading of organic wastes. The different elements of risk and risk management are considered in turn: i) the hazard posed by landspreading; ii) the natural vulnerability of the groundwater; iii) the value of the groundwater resource; and iv) the use of land-use planning controls and preventative measures. All four elements are encompassed in the national groundwater protection scheme (to be published end 1998). This framework is presented in the context of legislative control of landspreading under the Waste Management and Environmental Protection Agency Acts.

Landspreading is regulated in certain circumstances under; the Environmental Protection Agency (EPA) Act 1992, the Waste Management Act 1996 and the Water Pollution Act 1990. The groundwater protection response for landspreading of organic wastes can be applied within this legislative framework. The EPA Act requires the licensing of industrial activities and their associated landspreading above certain threshold levels. Examples are piggeries with a threshold of greater than

3,000 units on mineral soils and greater than 1,000 units on gley soils. Those activities that fall below the threshold set down in the Act will not require a licence from the Agency but will be subject to the control of the local authority.

There are also provisions for licensing landspreading operations under the Waste Management Act 1996. However, there are currently a number of exclusions under section 51 of the 1996 Act for the recovery of sludges and agricultural wastes. For example a waste licence shall not be required for the spreading of (i) sludges from water and waste water treatment plants run by local authorities, (ii) blood of animal or poultry origin, (iii) faecal matter of animal or poultry origin in the form of manure or slurry, or (iv) such other natural agricultural waste as may be prescribed.

Waste recovery is defined in the 1996 Act and includes landspreading. To be so defined the spreading activity must demonstrate benefit to agriculture or ecological improvement. As of yet no dates have been prescribed for licensing of recovery activities and hence, those activities which satisfy these requirements do not need a licence.

In cases where a landspreading activity does not involve a beneficial use for the land it is considered to be a disposal activity and therefore will require a waste licence under the Third Schedule of the Waste Management Act.

The Water Pollution Act 1990 was amended by the Waste Management Act 1996 to allow a local authority to require a nutrient management plan to be prepared for the purposes of preventing, eliminating or minimising the entry of polluting matter to waters. This includes landspreading activities. Such plans should be sought by local authorities where it is considered that significant environmental impacts could occur.

LANDSPREADING UNDER IPC LICENSING

A number of activities currently undergoing Integrated Pollution Control licensing (IPC) produce wastes which are suitable for landspreading. Such industries include dairy processing, slaughtering, rendering, brewing, and pig and poultry production.

Landspreading of wastes involves a return of nutrients to the soil which replaces the application of artificial fertilisers. When suitable wastes are landspread in an appropriate manner it does not present a significant threat of environmental pollution. One of the roles of the licensing process is to ensure that landspreading is carried out in such a manner. This is assured by a range of mechanisms:

1. minimising the volumes requiring landspreading (e.g. reducing wash water volumes),
2. diverting unsuitable wastes from being landspread (e.g. blood),
3. reducing the pathogen content (e.g. by storing waste for an adequate period to ensure pathogen die off),
4. applying wastes on appropriate land (e.g. to minimise surface or groundwater contamination)
5. applying waste in accordance with an appropriate nutrient management plan,
6. ensuring wastes are applied in a controlled manner, at appropriate times, and with regard to suitable buffers,
7. maintaining records of land application, and
8. monitoring.

While all the factors listed above will contribute to reducing the risk of significant environmental pollution from landspreading activities, the focus of this paper is to present a structural framework for assessment of how lands suitable for spreading may be sourced and identified to ensure ground water protection.

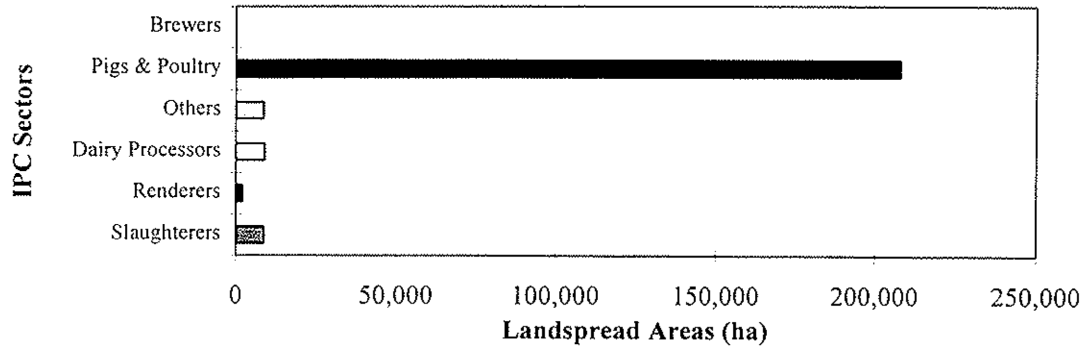


Figure 1 Landspread Area required by individual IPC Sectors

An estimate has been made of the total area of land that will be required for disposal of waste via landspreading under IPC (Figure 1). This estimate (a total of ~240,000 ha) is based on details of lands currently submitted as part of the IPC application process by the slaughtering, rendering and dairy processing industries. For the slaughtering and dairy processing industries it is likely that the figures presented are an underestimate of what will ultimately be required. An estimate for the pig and poultry sectors is made on the basis of the total number of animals and their waste produced that will be covered under IPC. In terms of the breweries no estimate will be available until after applications have been submitted from this industry. A number of 'other' industries with significant landspreading activities have also been included.

In terms of overall land utilisation this represents ~4% of the utilisable agricultural area. In regions with high volumes of intensive animal production and food processing this proportion will be considerably higher. When the Integrated Pollution Prevention and Control Directive (IPPC) is introduced in Irish law it is likely that this will further extend the industries brought in under IPC particularly in terms of the food processing industries, but most notably in terms of poultry production. As a result, implementation of IPPC will significantly extend the amount of controlled landspreading. Control of activities under the Waste Management Act will further extend this control.

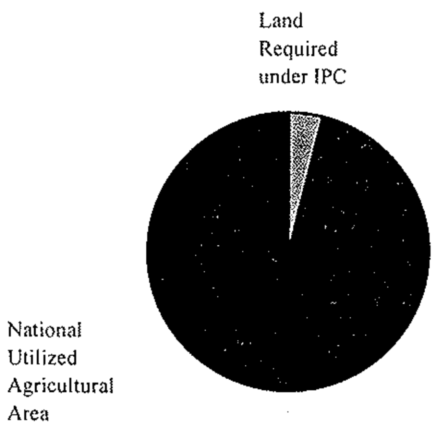
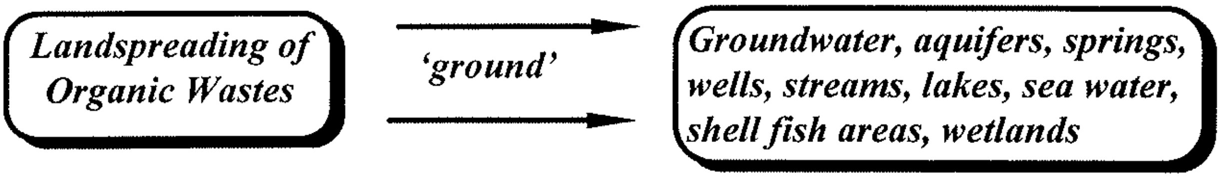


Figure 2 Proportion of Utilised Agricultural Area that will be required for landspreading from the IPC sectors.

THE RISK AND RISK MANAGEMENT APPROACH – A FRAMEWORK FOR DECISION-MAKING

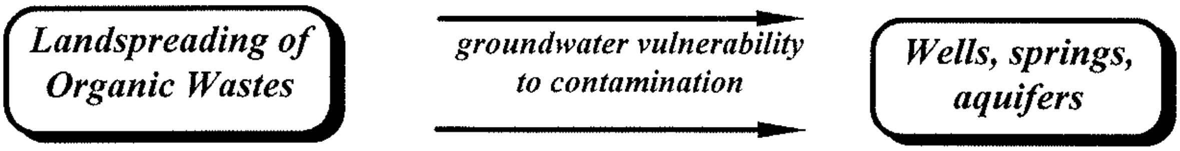
The conventional *source-pathway-target* model for environmental management can be applied to groundwater risk management and to the consideration of landspreading.

The potential ***source*** of contamination, called a **hazard,** is **landspreading of organic waste**. The ***pathway*** can be underground in geological materials, overground as surface runoff or both; the underground is the pathway of interest in this paper. There are several potential ***targets;*** those of concern in this paper are groundwater in aquifers and groundwater sources – wells and springs.



The risk to groundwater from landspreading activities is influenced by the following factors:

- the hazard, which depends on the chemical and microbiological content of the waste, and the loading rate, method and timing of application;
- the likelihood of contamination, which depends on the groundwater vulnerability and the proximity to groundwater sources;
- the consequences of a contamination event, which depend on the value of the groundwater resource as indicated by the presence of important groundwater sources and the aquifer category.



In the case of landspreading of organic wastes, **risk management** is based on consideration of the *hazard*, assessment of the potential *pathways*, determination of the *target/s* at risk and potential consequences, followed by a response to the risk. This response includes the assessment and selection of solutions and the implementation of measures to prevent or minimise the probability and consequences of a contamination event.

LANDSPREADING OF ORGANIC WASTES : A HAZARD FOR GROUNDWATER

GENERAL

Organic wastes, by their nature, are composed mainly of the nutrients nitrogen (N), phosphorus (P) and potassium (K). Untreated wastes, such as manures and slurries from piggeries, are likely to contain faecal bacteria, viruses, protozoa (such as Cryptospiridium) and helminthic parasites. Some wastes may contain metals such as copper which must be considered in developing nutrient management plans.

NITRATE

Nitrogen in the form of nitrate is one of the common contaminants identified in groundwater world-wide. It is highly mobile and easily leached from the rooting zone. Nitrates in groundwater have posed less problems to date in Ireland than in most other countries with intensive agriculture. However, an assessment of draft EPA reports on nitrate levels (Wright, 1997) has shown that a significant number of public supply sources in eastern, south-eastern and southern counties have mean nitrate levels greater than 25 mg/l. Agricultural activities, whether from yard or field losses, are a source of nitrate in these areas.

PHOSPHORUS

Phosphorus poses less of a threat to groundwater than nitrate because it is relatively immobile in topsoil and subsoil. However, in areas with thin topsoils and subsoils, application of organic wastes could potentially result in the contamination of groundwater as a result of bypassing the matrix through preferential flowpaths (macropores).

MICROBIAL POLLUTANTS

The greatest threat posed to groundwater and human health by landspreading is from faecal bacteria, viruses and Cryptospiridium. There may be a tendency to under-estimate the impact of these in Ireland. The number of annual waterborne illnesses in the US is estimated between 1 and 15 million cases (Macler, 1995). While a direct comparison between Ireland and the US may not be possible, it can be argued that waterborne illnesses should be an issue of concern in Ireland.

The presence of *E. coli* in water is taken as an indicator of the possible presence of other pathogenic microbes, which have a greater potential for health impacts than *E. coli.* However, a particular strain of faecal bacteria, *E. coli 0157*, which has an infective dose of 10-50 organisms (Ball, 1997), has increased concerns with regard to such microbial contamination. The drinking water standard requires zero *E. coli* per 100 ml in a water supply.

There are more than 100 types of viral pathogens associated with groundwater (Macler, 1995). The most common impact is likely to be gastro-enteritis, which can result in diarrhoea. Certain types of viruses can live for up to 170 days in groundwater. As they are smaller than bacteria and can survive for longer, they may pose a greater threat to groundwater than bacteria. Unfortunately, they are difficult to detect in water and the absence of faecal indicator bacteria in standard 100 ml samples of water does not guarantee the absence of viruses (Berg and Metcalf, 1978).

Cryptosporidium has emerged as a common cause of gastro-enteritis in recent years (Ball, 1997). Cryptosporidium oocysts are hardy and can persist for longer than *E. coli.* They cause infection at low levels (probably <10 viable oocysts), are present in farm animals and are resistant to disinfectants such as chlorine at current levels in treated water supplies. Consequently, they pose a threat to groundwater and human health.

IRON AND MANGANESE

Effluent from wastes can cause deoxygenation in the ground which can result in dissolution of Fe and Mn from soil, subsoil and bedrock. This can occur from the spreading of excessive quantities of slurries and from high BOD effluents such as milk wastes and blood. However, high Fe and Mn concentrations in groundwater can also be due to natural conditions.

HYDROGEOLOGICAL FACTORS INFLUENCING CONTAMINANT ATTENUATION AND GROUNDWATER VULNERABILITY

SOILS

Soil (topsoil) is potentially a cost-effective medium for treating and attenuating contaminants. Firstly, growing crops can use the nutrients (N, P, and K) in organic wastes, and secondly bacteria, protozoa, helminthic parasites and viruses can be removed by filtration, predation, competition for food, and exposure to other adverse conditions. However, excessive quantities of nutrients (i.e. not matching crop requirements during the growing season or landspreading when crop growth is minimal) can lead to leaching, particularly of nitrate. Also, bypassing of the topsoil (through macropores) by nutrients and pathogens can occur under certain conditions. If bypass flow is not taken into account, it may result in an underestimation of the threat posed to groundwater and human health by landspreading.

A substantial volume of research material, summarised by Abu-Ashour et al. (1994), Beven and Germann (1982), Thomas and Phillips (1979) and White (1985), indicates that it is an over-simplification to: (a) treat topsoils as a homogeneous medium conforming to Darcian principles of water flow, and (b) assume that the only recharge process is displacement (by piston-like flow) of all resident water and solutes by incoming water and solutes. The presence of macropores (e.g. pores formed by soil fauna (mainly earthworms), plant roots, weathering cracks and natural soil pipes) can greatly decrease the time taken for solutes and micro-organisms to migrate through the topsoil relative to predictions based on Darcian principles. As a consequence of preferential flow in macropores, gravitational flow of water can occur in soils that are well below 'field capacity' and water can move rapidly through unsaturated soil ahead of the wetting front in the soil matrix. Fleming et al. (1990) found rapid movement of faecal bacteria to tile drains and the presence of faecal bacteria at 0.5 m depth in underlying soil, following landspreading of liquid manure; they concluded that macropore flow was the most likely transport mechanism. Hallberg (1989), in a review of pesticide pollution of groundwater in the humid US, concluded that preferential flow may be making soils more susceptible to leaching of pesticides (and other land-applied compounds) to groundwater than evaluations based on traditional concepts of Darcian flux would suggest.

The role of preferential flow is likely to be less in Ireland than in some of the countries where they are shown to have a significant impact because Irish soils are relatively young (less than 16,000 years since the Ice Age ended) and climate variations (particularly temperature) are less than, for instance, the US and continental Europe. While little research has been carried out on this in Ireland, there is sufficient evidence to show that it is also an issue that is relevant to contaminant movement and groundwater protection here:

- Ryan and Noonan (1995) showed the role of macropores in soils at Johnstown Castle with a field tracing experiment where dye, applied at the surface, moved to almost 0.9 m bgl in thirteen days.
- In their review of the soil associations in Ireland, Gardiner and Radford (1980) describe the presence of roots below 0.5 m in most soils and below 1.0 m in some soils. According to Gleeson (1998), grass roots may reach depths of 2.0 m during dry summers.
- Ball (1993) has reported summer cracks in fields of pasture land down to 2.0 m. In a situation with 3-5 m soil and subsoil over bedrock, and a water table 4.0 m bgl, he found a rise in the water table of 1-3 cm and a sudden increase in faecal bacteria in the groundwater following 24 hours of intense isolated showers in June and August. He concluded that summer rainfall can bypass the soils moisture deficit and recharge the groundwater system through macropores.
- Field observations by geologists and hydrogeologists in the GSI and consulting firms report the presence of preferential flowpaths as deep as 2.0 m bgl.
- GSI automatic water level monitoring shows recharge occurring during the summer months after heavy rainfall in certain circumstances.

Preferential flowpaths are likely to exist in virtually all soils down to about 0.4-0.5 m bgl. Below this their presence will reduce with depth, depending largely on the soil and subsoil texture. In some situations, particularly where (a) the bedrock is shallow and permeable, (b) the soil and subsoil have a high clay/silt content, and (c) the underlying subsoil is sand/gravel, they may be present to depths greater than 1.0 m.

SUBSOILS

Subsoils, defined as the sediments between the topsoil and bedrock, have an intergranular permeability and act as a protecting, filtering layer over aquifers by both physical and chemical/biochemical means. Fine grained sediments such as clayey till, lacustrine clays and peats have a low permeability and consequently can act as a hindrance to the vertical movement of contaminants. In areas where these sediments are present, surface water is more at risk than groundwater as most if not all the contaminants cannot migrate downwards and can only move laterally. Even if the permeability is sufficiently high to allow slow intergranular movement of contaminants, for instance in sandy tills, the sediments can strain out and absorb bacteria and viruses. In contrast, high permeability deposits - sands and gravels - allow easy access of pollutants to the water table although they also provide opportunities for dispersion of the pollutants among the pore spaces and dilution in the groundwater. Sorption, ion-exchange and precipitation are vital chemical processes in attenuating pollutants. The cation-exchange capacity of subsoils depends on the clay and/or organic content. Clays and peats can attenuate bacteria, viruses and chemical pollutants such as potassium and ammonium whereas clean sand and gravel has little effect. In general, the higher the clay content and the lower the permeability, the greater the protection of groundwater from pollution. Obviously, the thicker the subsoils the greater the attenuation. However, macropores are likely to be present in places immediately beneath the topsoil.

BEDROCK .

The bedrock in Ireland is not taken to be a significant factor in attenuating pollutants or in defining vulnerability. This is because once contaminants enter the bedrock there is little attenuation other than by dilution (usually relatively limited) where groundwater is reached, owing to the fissure permeability that characterises Irish bedrock

Groundwater flow velocities in bedrock aquifers are generally high (typically 0.5-5 m/d). Exceptionally high velocities (>1000m/d) can occur in karst aquifers. Even in bedrock classed as poor aquifers, flow rates are often high in the upper, fractured and weathered zone (usually 1-5 m thick). Therefore, if groundwater in bedrock becomes contaminated, nearby wells can readily be affected.

ZERO FLUX PLANE

When evaporation exceeds rainfall, the water in the soil/subsoil profile moves upwards from deeper layers towards the root zone and the soil surface. Below these deeper layers, water moves downwards towards the water table. Dividing the zone of upward and downward-moving water is a 'plane' where the hydraulic gradient and moisture flux are zero – this is called the zero flux plane (ZFP), (Cooper et al. (1990). This plane moves upwards and downwards depending on meteorological conditions. In Britain, the zero flux plane can reach depths of over 5.0 m bgl (Wellings and Bell, 1980). In Ireland, Hosty and Mulqueen (1996) showed that a ZFP developed in early May at a midland site in 1992 and moved downwards to a maximum depth of 1.5 m as the summer progressed. Based on a modelling study, they concluded that the ZFP could reach 1.75 m in a dry summer. A ZFP can help maintain nutrients and pathogens in the upper layers of soil and subsoil. However, a zero flux plane will not be able to develop in bedrock in Ireland. Therefore, a minimum depth of 1.75-2.0 of soil and subsoil can help in reducing leaching and movement of contaminants into groundwater.

VULNERABILITY MAPS

'Vulnerability' represents the ease with which groundwater may be contaminated – it is a measure of the likelihood of contamination occurring. The vulnerability of the groundwater to contamination is taken to depend mainly on the thickness and permeability of the subsoils which lie between the point of release of contaminants and the groundwater in the bedrock or a sand/gravel aquifer. Where this subsoil is absent or thin, or where it is very permeable, the groundwater is most vulnerable. Where the subsoil is thick and has a low permeability, the groundwater is least vulnerable. Karst features indicate high permeabilities and rapid recharge. Consequently, groundwater in the vicinity of karst features is 'extremely' vulnerable. The vulnerability of groundwater is assessed, under GSI guidelines (Daly and Warren, 1998), according to four classes: extreme **(E)**, high **(H)**, moderate **(M)** and low **(L)**.

The guidelines used to produce vulnerability maps are given in Table 1.

Table 1 Vulnerability Mapping Guidelines

Vulnerability Rating	Hydrogeological Requirements				
	Subsoil Permeability (Type) and Thickness			Unsaturated Zone	Recharge Type/karst feature
	high permeability (*sand/gravel*)	moderate permeability (e.g. *sandy till*)	low permeability (e.g. *clayey till, clay, peat*)	(*sand/gravel* aquifers <u>only</u>)	
Extreme (E)	0 - 3.0 m	0 - 3.0 m	0 - 3.0 m	0 - 3.0 m	point (<30 m radius)
High (H)	>3.0 m	3.0 - 10.0 m	3.0 - 5.0 m	>3.0 m	diffuse
Moderate (M)	N/A	>10.0 m	5.0 - 10.0	N/A	diffuse
Low (L)	N/A	N/A	>10.0 m	N/A	diffuse

Notes: i) N/A = not applicable.
ii) Precise permeability values cannot be given at present.

(from Daly and Warren, 1998)

GROUNDWATER SOURCES AND RESOURCES – HYDROGEOLOGICAL ELEMENTS OF RISK

The **consequences of groundwater contamination**, which is one of the elements of the risk posed by landspreading, depends on the **value** of the groundwater. This value is defined by the groundwater in the following areas, which are listed in decreasing order of importance:

i) **Catchment Area of Public Supply Sources**
   a) Inner Protection Area (**SI**), designed to give microbial protection.
   b) Outer Protection Area (**SO**), encompassing the source catchment area or zone of contribution (ZOC).

ii) **Regionally Important (R) Aquifers**
   These are subdivided into '**Rg**' (sand/gravel aquifers), '**Rk**' ('karst' limestone aquifers) and '**Rf**' (aquifers with fissure flow).

iii) **Locally Important (L) Aquifers**
   These are subdivided into '**Lm**' (Generally Moderately Productive), '**Ll**' (Productive only in Local Zones) and '**Lg**' (Sand/Gravel).

iv) **Poor (P) Aquifers**
   These are sub-divided into '**Pl**' (Generally Unproductive except for Local Zones) and '**Pu**' (Generally Unproductive).

Landspreading poses a greater risk to the environment and to groundwater if it is located over a regionally important aquifer rather than a poor aquifer and the greatest risk is where landspreading occurs in the catchment area of a public or major industrial groundwater supply.

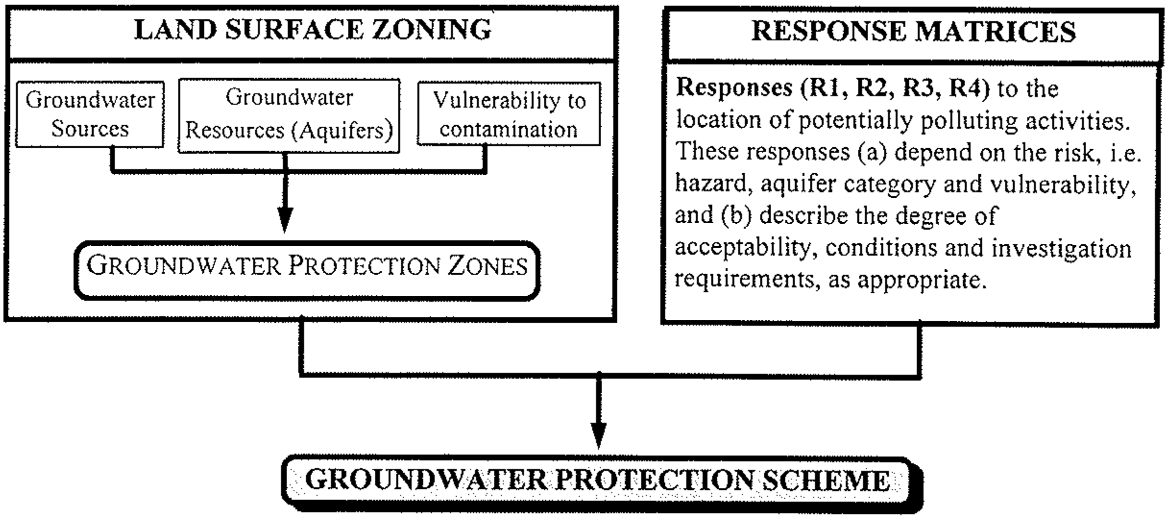
THE GROUNDWATER PROTECTION SCHEME – AN AID IN RISK ASSESSMENT AND MANAGEMENT

HOW A GROUNDWATER PROTECTION SCHEME WORKS

There are **two main components** which are integrated to give the groundwater protection scheme:

- **Land surface zoning.**
- **Response matrices for potentially polluting activities.**

Summary of Components of Groundwater Protection Scheme



Land surface zoning provides the general framework for a groundwater protection scheme. The outcome is a map, which divides any chosen area into a number of groundwater protection zones according to the degree of protection required.

There are three main hydrogeological elements to land surface zoning:

- Division of the entire land surface according to the **vulnerability** of the underlying groundwater to contamination. This requires production of a vulnerability map showing the four vulnerability categories.
- Delineation of **areas surrounding groundwater sources** (usually public supply sources); these are termed source protection areas.
- Delineation of areas according to the value of the groundwater resources or **aquifer category**; these are termed resource protection areas.

These three elements are integrated together to give maps showing **groundwater protection zones;** source protection zones and resource protection zones.

The location and management of potentially polluting activities in each groundwater protection zone is by means of **groundwater protection responses** for each activity or group of activities. By consulting a **Response Matrix**, it can be seen (a) whether such a development is likely to be acceptable on that site, (b) what kind of further investigations may be necessary to reach a final decision, and (c) what planning or licensing conditions may be necessary for that development. The groundwater protection responses are a means of ensuring that good environmental practices are followed.

While the zonation maps and groundwater protection responses are different components, they are incorporated together and closely interlinked in the scheme.

GROUNDWATER PROTECTION ZONES

The matrix in Table 2 below gives the result of integrating the three elements of land surface zoning (vulnerability categories, source protection areas and resource protection areas) – a possible total of 8 source protection zones and 24 resource protection zones. In practice this is achieved by superimposing the vulnerability map on source protection area maps and on the aquifer map, thereby producing a map of the groundwater protection zones. Each zone is represented by a code, e.g. **Rf/M**, which represents areas of <u>regionally important fissured</u> aquifers where the groundwater is <u>moderately</u> vulnerable to contamination. Where maps showing groundwater protection zones are unavailable at present, the GSI can give a provisional aquifer category designation for any area and the vulnerability can be ascertained by site investigations. Therefore the position of any site within the matrix below can be obtained.

Table 2. Matrix of Groundwater Protection Zones

VULNERABILITY RATING	SOURCE PROTECTION		RESOURCE PROTECTION Aquifer Category						
			Regionally Imp.		Locally Imp.		Poor Aquifers		
	Inner	*Outer*	*Rk*	*Rf/Rg*	*Lm/Lg*	*Ll*	*Pl*	*Pu*	
Extreme (E)	Sl/E	SO/E	Rk/E	Rf/E	Lm/E	Ll/E	Pl/E	Pu/E	↓
High (H)	Sl/H	SO/H	Rk/H	Rf/H	Lm/H	Ll/H	Pl/H	Pu/H	↓
Moderate (M)	Sl/M	SO/M	Rk/M	Rf/M	Lm/M	Ll/M	Pl/M	Pu/M	↓
Low (L)	Sl/L	SO/L	Rk/L	Rf/L	Lm/L	Ll/L	Pl/L	Pu/L	↓
	→	→	→	→	→	→	→	→	→

(Arrows (→ ↓) indicate directions of decreasing risk)

DRAFT GROUNDWATER PROTECTION RESPONSES FOR LANDSPREADING

Table 3 is the draft Response Matrix for landspreading (EPA/GSI, 1997), and this is followed by the specific responses to the landspreading of organic wastes in each groundwater protection zone.

Table 3 Draft Response Categories for Landspreading Activities

VULNERABILITY RATING	SOURCE PROTECTION		RESOURCE PROTECTION Aquifer Category					
			Regionally Important (R)		*Locally Important (L)*		*Poor Aquifers (P)*	
	Inner	*Outer*	*Rk*	*Rf/Rg*	*Lm/Lg*	*Ll*	*Pl*	*Pu*
Extreme (E)	R4	R4	$R3^2$	$R3^2$	$R3^2$	$R3^2$	$R3^1$	$R3^1$
High (H)	R4	$R2^1$	R1	R1	R1	R1	R1	R1
Moderate (M)	$R3^3$	$R2^1$	R1	R1	R1	R1	R1	R1
Low (L)	$R3^3$	$R2^1$	R1	R1	R1	R1	R1	R1

R1 Acceptable, subject to normal good practice.

$R2^1$ Acceptable subject to a maximum organic Nitrogen load (including that deposited by the grazing animal) not exceeding 170 kg/hectare/yr.

$R3^1$ Not Generally Acceptable, unless there is a consistent minimum thickness of 1 m of soil and subsoil.

$R3^2$ Not Generally Acceptable, unless there is a consistent minimum thickness of 2 m of soil and subsoil.

$R3^3$ Not Generally Acceptable, unless it is shown that there are no alternative areas available and that detailed evidence is provided to show that contamination will not take place. (No spreading will be allowed within a minimum distance of 50 m radius of a well.)

R4 Not Acceptable.

NOTE: These responses assume that there is no known significant groundwater contamination problem in the landspreading area. Should contamination by ***nitrate*** *(or other contaminants) be known of in any particular area, then more stringent responses may be necessary. Monitoring carried out as part of any Local Authority or Agency authorisation will determine whether or not a variation on any of these responses is required.*

In addition to the above, the following notes apply in all cases.

- Where landspreading is permitted, total N (organic and inorganic) load applied shall not exceed Teagasc's nutrient recommendations for growing crops.
- As landspreading is a *"potentially polluting"* diffuse source activity, special attention should be given to private water supplies. No spreading should be allowed within 50 m of wells. The recommended 100 m buffer zone around dwelling houses may incorporate the well buffer zone.
- In **karst** areas farmers must take account of karst features such as swallow holes, caves, streams connected to karst systems, etc. Karst features can occur in any limestone rock type irrespective of aquifer category. Landspreading within 30m of karst features is not permitted.

Site Investigation

The vulnerability category should be verified by site investigations (e.g. trial pits, augers, borings, cuttings etc.). The intrusive investigations should reach sufficient depths to show that the minimum subsoil thickness is present. The density of the sample points is dependent on the risk of contamination of the groundwater. In cases of extreme vulnerability and/or within the source protection zones there should be at least one investigation point per hectare. In all other cases the data points should be at a minimum frequency of one investigation point per 5 hectares. Any of the above site investigation techniques may be used as an investigation point. Data obtained should be logged. The depth to water inflow, if any, should be recorded.

Implementation

The draft code of practice for the protection of groundwater from landspreading is currently being implemented in the assessment of applications for IPC licences. While new activities would be expected to source appropriate lands in response to this code, for existing facilities application of BATNEEC and this code is made on a site by site basis.

SUMMARY AND CONCLUSIONS

1. The Waste Management and EPA Acts provide a legislative framework for the control of landspreading from licensed facilities.
2. The risk and risk management approach is used as a framework for groundwater protection in Ireland and for decision-making on landspreading of organic wastes.
3. The landspreading of organic wastes has the potential to cause contamination of groundwater, particularly by nitrate and microbial contaminants. However, the threat can be minimised when the groundwater protection responses outlined in this paper are followed.
4. The movement of contaminants away from the landspreading areas and the vulnerability of groundwater depend on the properties of the underlying soil, subsoil and bedrock. While the soil can utilise and attenuate nutrients and other contaminants, it may be bypassed in certain circumstances, particularly after heavy rainfall. The subsoil provides the main protection for groundwater. The thickness of subsoil over bedrock aquifers is a critical factor in determining the threat posed to groundwater by landspreading. Little attenuation occurs in the bedrock and flow rates are fast.
5. The groundwater protection zone for any area is obtained by combining vulnerability assessments with the aquifer category or source protection area. This zone gives the hydrogeological setting for the area and summarises the hydrogeological components of risk.
6. The groundwater protection responses take account of the vulnerability, the value of the groundwater (with sources being more valuable than resources and regionally important aquifers more valuable than locally important and so on) and the contaminant loading. They show (a) whether landspreading is likely to be acceptable, (b) what kind of site investigations are required, and (c) what licensing conditions may be necessary. While they cannot guarantee that contamination of groundwater will never occur, they help ensure that the threat posed by landspreading is minimised.

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Paper No. 8.

Landspreading of Organic Wastes: Selected Case Studies.

Richard Church, Minerex Environmental Limited.

Landspreading of Organic Wastes

Selected Case Studies

by Richard Church, Minerex Environmental Ltd.

Abstract

Minerex Environmental Limited has undertaken aquifer vulnerability assessments for a number of landbanks used for landspreading of organic waste from meat processing. The Code of Practice for Landspreading from the EPA and the draft National Groundwater Protection Scheme were used for these assessments. Case studies of two assessments from different hydrogeological settings are presented.

INTRODUCTION

Minerex Environmental Limited (MEL) was commissioned by Anglo Irish Beef Processors (AIBP) Development to undertake aquifer vulnerability assessments for landbanks which could be used in the landspreading of waste arising from meat processing and effluent treatment. In addition to fulfilling the Integrated Pollution Control Licence (IPCL) conditions, AIBP is developing a voluntary 'Waste Disposal Code to Protect the Environment'.

The landbanks were associated with eight meat processing plants located throughout Ireland. The areas covered a wide range of geological, quaternary and hydrogeological environments. Each meat processing plant has between four and fourteen different landbank areas available for landspreading, resulting in some fifty landbanks in total. The number of landbanks per processing plant is dependant on the volume of waste, nutrient requirements of the soil and agricultural usage of the land.

The aquifer vulnerability assessments were undertaken using the code of practice for landspreading operations which was produced by the Geological Survey of Ireland (GSI) and the Environmental Protection Agency (EPA). This provided a framework for the vulnerability assessments allied to the specific polluting activity. An operational draft for implementation has currently been issued by the EPA. The full code of practice is being covered in a separate paper and only a summary table is presented here.

The IPCL application for the landbanks covered a wide range of issues. This paper only deals with those relevant to hydrogeology. The hydrogeological vulnerability assessments assist in the development of a coherent nutrient management programme for each landbank. Each landbank is unique and different management strategies apply.

WASTE TYPES

The landspreading waste arises from the slaughter of animals and sludges from the Waste Water Treatment Plant (WWTP) onsite. The primary sources of waste are organic in nature and consist of:

(i) Sludge's from the wastewater treatment plant: Dissolved Air Flotation sludges and interceptor sludges. These are liquid or semi-solid in form. They are not stored onsite and require immediate disposal.

(ii) Paunch Contents: This is the contents of the stomach of the animal, it consists of undigested food (grass or silage) in the first stomach and partly digested food in the second stomach. The paunch is dried and is fibrous in nature. It is stored offsite on open ground and application normally occurs in spring prior to the sowing of tillage crops or in the autumn after the crop is removed.

(iii) Truckwash and Lairage Slurry Wastes: This waste is similar in nature to farm slurry and results from the slatted lairages (stockade areas for the animals) and the washing out of transportation trucks.

The waste differs from pure agricultural waste in its lower liquid content and lower nutrient content.

NUTRIENT MANAGEMENT PLAN

A nutrient management plan was required by the EPA for each meat processing plant. This determined the following:

- Type and Volumes of waste
- Quality of waste
- Total hectares of spreading
- Land application details
- Rate of Spreading (m^3/ha/yr) and equipment used
- Nutrient requirements of soil for agricultural usage (no. of grass silage cuts, or grazing).
- Aquifer Vulnerability Assessment

AIBP made a submission to the EPA concerning the addition of nutrient-rich wastes to the soils in replacement of artificial fertilisers. These differ slightly from the submissions of other meat processors and in summary this is:

To have a standard application rate for spreading of 10 tonnes per hectare. This often results in application rates lower than the Teagasc recommended amounts. AIBP have taken this precautionary approach as the availability and content of the nutrients, in particular nitrogen and phosphate, vary with waste type. Furthermore, this sustainable landspreading activity will ensure an extended operational life of the landbank.

Individual farmers can make up the deficit with artificial fertilisers specific to the nutrient requirements of the crop. Farmers then view some benefit from the reduction in requirements of artificial fertilisers whilst maintaining a good productive crop. An additional advantage is that this simplifies application rates for the waste contractor and mistakes are reduced in the application for different landbank areas.

It is recognised by the EPA that landspreading proposals for a number of plants are a current interim measure (EPA, 1998). Individual licenses can be issued for spreading between November and February in exceptional circumstances. The initial storage of waste during periods of non-spreading is currently a problem at some plants due to lack of infrastructure for storage. Meat processing plants have been given a five year period to ensure that facilities onsite exist for four months of storage of organic wastes. In situations where there is a threat of significant environmental pollution this time-frame may be shorter.

HYDROGEOLOGICAL INVESTIGATION TECHNIQUES

A large number of landbanks exist for AIBP and all these required some hydrogeological interpretation. To minimise the work volume and allow time to be focussed on important areas a scheme for the interpretation of each site was developed.

Technical notes were produced to summarise the quaternary and bedrock geology of each landbank, make an assessment of the aquifer vulnerability using the GSI guidelines (GSI, 1995) and make further recommendations.

The landbank areas associated with a meat processing plant were indicated on a 6" Ordnance Survey sheets. A full assessment of the bedrock geology, quaternary geology and hydrogeological data for each landbank was then made at the GSI and relevant data indicated on the sheet. The relevant data used included:

- Bedrock Geology Sheets (either Recent 1:100,000 or Chevron Series).
- Quaternary Mapping data – if available.
- Groundwater Vulnerability and Protection County Maps – if available.
- Well records for the relevant townlands.
- Other relevant reports or studies - if available.

Subsequent to the desk study, if inadequate data to make a correct determination of vulnerability for the sites existed or where the aquifer was indicated as having an EXTREME vulnerability classification, on-site trialpitting was undertaken.

The data was then compiled on maps and applied to the GSI Groundwater Protection Scheme to determine the aquifer status and the EPA Code of Practice for Landspreading. The EPA table of classification is presented below:

Vulnerability Rating	**Source Protection**		**Resource Protection Aquifer Category**					
			Regionally Important		**Locally Important**		**Poor**	
	Inner	***Outer***	***Rk***	***Rf/Rg***	***Lm/Lg***	***Ll***	***Pl***	***Pu***
Extreme (E)	R4	R4	R3²	R3²	R3²	R3²	R3¹	R3¹
High (H)	R4	R2¹	R1	R1	R1	R1	R1	R1
Moderate (M)	R3³	R2¹	R1	R1	R1	R1	R1	R1
Low (L)	R3³	R2¹	R1	R1	R1	R1	R1	R1

R1 Acceptable subject to normal, good practice.

R2¹ Acceptable subject to adherence to the 30m restriction zone around karst features.

R2² Acceptable subject to a maximum organic nitrogen load (including that deposited by a grazing animal not exceeding 170kg/hectare/yr.

R3¹ Not generally acceptable unless there is a consistent minimum thickness of 1m of soil and subsoil.

R3² Not generally acceptable unless there is a consistent minimum thickness of 2m of soil and subsoil.

R3³ Not generally acceptable, unless it can be shown that there are no alternate areas available or that detailed evidence is provided to show that contamination will not take place. No spreading will be allowed within a minimum distance of 50m radius of a well.

R4 Not acceptable.

To demonstrate the process undertaken two meat processing plants have been chosen with contrasting hydrogeological environments for their landbanks.

CASE STUDY 1: AIBP Carrigans, County Donegal.

Six landbanks in total were delineated for this meat processing plant. Site specific information was obtained from the Geological Survey of Ireland and due to the inadequate data available for the quaternary geology further trial-pitting work at the site was undertaken to determine overburden type and thickness. Four landbanks close to the processing plant are presented to indicate the hydrogeological environment. A map of the landbanks is presented in Figure 1.

The area around Carrigans is typically overlain by a thin layer of quaternary till, derived from glacial sediments, which is less than 5 metres thick. This is classified as EXTREME vulnerability under the GSI guidelines. The bedrock geology consists of Dalradian Schists which are generally unproductive. The area would therefore be given an aquifer vulnerability rating of Pu/E under GSI guidelines.

In Detail:

1. The bedrock is Upper Dalradian in age and probably composed of metamorphosed volcanogenic/volcanoclastic sediments and psammites. There are no major faults recorded in the area. The GSI geological map of Donegal (1:100,000) and a report to accompany it is currently under development. A large portion of Donegal is composed of low permeability crystalline and sedimentary rocks, which prevent the infiltration of precipitation. As a result in Donegal there is a high runoff to streams and reduced recharge to the aquifers.

2. Land slopes on average are 6%. In 1993, Local Authorities, Teagasc and the GSI produced draft guidance notes for the 'land application of animal wastes'. The notes suggest a minimum distance of 20m between landspreading and main river channel and lakes and that this figure depends on slope. In general, slopes of less than 6% are not considered of significance. Therefore, current thinking, suggests that there is a no significant risk of pollution of the nearby river from runoff for a 'safety strip' of greater than 20m.

3. At the GSI, there were only a limited number of wells recorded in the area. One drilled in 1971 for the Old Rectory in Carrigans, was drilled 235' deep and only yielded 100gph. In 1973, another well, hand dug, on the townland, was dug to 7' and yielded 300gph. Apparently shale bedrock was encountered at the base of the well. Four wells were drilled at Dunmore House, all dry out in the summer and none are used.

4. The field on the southern side of the railway line is underlain by alluvium. The overburden under two fields to the north of the railway line consists of 'Drift', which could comprise sand, gravel or boulder clay. No overburden thicknesses are recorded in the GSI databases. Therefore, trial-pitting was undertaken in the fields and summaries of the logs are presented below:

Trialpit	Stratigraphy	Total Depth
TP1	Sands and gravels.	2.4. Refusal at bedrock.
TP2	Sandy clay then dirty gravels.	>3.0. Refusal at collapse.
TP3	Clay with boulders	1.2. Refusal at bedrock.
TP17	Sandy clay then boulders	1.4. Refusal at bedrock.
TP18	Sands, gravels, boulders.	2.6. Refusal at bedrock.
TP4	Shaly clays	2.5. Refusal at bedrock.
TP5	Coarse, subrounded gravels in CLAY matrix	3.5. Refusal at bedrock.
TP6	Light orange/brown, sandy CLAY with occasional stones.	>3.5. Limit of excavator.
TP7	Brown CLAY with thin blue/grey CLAY layers.	3.3. Refusal at bedrock.

TP8	Clay, shales with boulders.	1.2. Refusal at bedrock.
TP9	Light grey CLAY with shale fragments increasing in size with depth	1.6. Refusal at bedrock.
TP10	Coarse grained SAND, wet.	1.6. Refusal at bedrock.
TP11	Sandy CLAY then fine gravels.	>2.8. Refusal at collapse.
TP12	Sandy CLAY with gravels	2.0. Refusal at bedrock.
TP13	CLAY with fine gravels and some well rounded boulders	>3.0. Refusal at collapse.
TP14	CLAY with fine gravels and some well rounded cobbles	>2.6. Refusal at collapse.
TP15	Pale grey sandy CLAY	>3.2. Limit of Excavator.
TP16	Reddish CLAY with rounded gravels	1.1. Refusal at bedrock.
TP19	Reddish clayey SAND.	0.7. Refusal at boulders.

Conclusions

The bedrock is considered a poor aquifer and falls into the 'generally unproductive' (Pu) aquifer category.

Landspreading should not occur on the steep land to the west and north of the landbank or within 20m of streams flowing through the landbank.

One portion of the landbank is probably within the zone of contribution to a reported well and therefore landspreading is considered a potential risk to groundwater quality. The local houses are on mains water and the wells are not currently in use. Landspreading should not occur within 30m of the field boundaries close to Dunmore House.

Trial-pitting indicates that a minimum thickness of 1m of subsoil exists across the landbank.

Applying the EPA Code of Practice the landbank areas fall into the R3[1] category. Therefore, with the exception of the areas indicated above, landspreading is acceptable subject to good normal working practice.

CASE STUDY 2: AIBP Rathkeale, County Limerick.

Three landbanks were available for landspreading and are indicated in Figure 2. Site specific information was obtained from the Geological Survey of Ireland and further trial-pitting work at the site was undertaken to determine overburden type and thickness. A county wide aquifer vulnerability assessment has been undertaken for Limerick by Limerick County Council in conjunction with the GSI (GSI, 1995) and this was used in the initial assessment of the sites.

The area around Rathkeale is typically overlain by a thin layer of quaternary till, derived from glacial sediments, which is less than 5 metres thick. This is classified as EXTREME vulnerability under the GSI guidelines. The bedrock geology consists of Waulsortian Limestone, which can be a very productive aquifer and is classified as 'regionally important' under GSI guidelines. The aquifer can be karstified and evidence of karst is present in the local area. This gives the area an aquifer vulnerability rating of Rk/E under the GSI guidelines.

In Detail

1. The overburden consists of limestone tills where present (graded RKc by the GSI), which could comprise sand, gravel or boulder clay. Bedrock is mapped as close to surface with numerous outcrops in the local area. Borehole records indicate a maximum overburden thickness of 4m and this is south of the fields for landspreading. In general, overburden thicknesses are very variable over the area changing from less than one metre to several metres thick in less than a field.

2. The underlying bedrock is the Waulsortian reef limestone. This is a massive blue/grey limestone which is more thinly bedded with shales close to the top of the sequence. Some karstification and dolomitisation is generally present. Muddy shelf limestones are present to the south of the landbanks as indicated on Figure 2.

3. The wells local to the area, where in use, are marked as good with well specific capacities of the order of 20 $m^3/d/m$. However, many of the houses local to the landbanks are on a group water scheme supply.

4. Land slopes are low throughout the area upto a maximum of 5% on some localised ridges. None of the landbanks are within 100m of a watercourse.

5. The Rk/E grading over the whole area by the GSI meant that trialpitting was required for all of the landbank fields. Trialpits were undertaken at locations on Figure 2. Data from these is presented in the table below:

Trialpit	Stratigraphy	Total Depth
TP1	Grey clayey gravel with large boulders. Weathered limestone bedrock, angular boulders, unable to penetrate, smooth karstic weathered surfaces. No watermakes.	1.4. Refusal at bedrock.
TP2	Clayey gravel with large angular limestone boulders. Weathered limestone bedrock, angular boulders, unable to penetrate, smooth karstic weathered surfaces. No watermakes.	0.7. Refusal at bedrock.
TP3	Clay with gravels and boulders – light brown in colour. Blocky, angular, broken limestone – unable to penetrate further. No watermakes.	1.1. Refusal at bedrock.
TP4	Gravelly clay, light brown. More gravels, clayey gravels, well rounded. Weathered bedrock, angular boulders with clay matrix. Blocky limestone bedrock unable to penetrate further. No watermakes.	1.9. Refusal at bedrock
TP5	Weathered limestone bedrock – unable to penetrate further. No watermakes.	0.2. Refusal at bedrock.
TP6	Subsoil – clay with a few gravels – light brown. Weathered limestone bedrock. Unable to penetrate further. No watermakes.	0.9. Refusal at bedrock.
TP7	Clayey gravel with limestone boulders. Weathered limestone bedrock. Unable to penetrate further. No watermakes.	0.8. Refusal at bedrock.
TP8	Clayey gravels with limestone boulders.	0.8. Refusal at bedrock.
TP9	Gravelly clay with boulders. Brown, stiff clay. Boulders in sandy clay. Weathered limestone bedrock. Unable to penetrate further. No watermakes.	2.5. Refusal at bedrock.
TP10	Clayey gravel with rounded clasts. Clayey sand with gravels. Weathered limestone bedrock. Unable to penetrate further. No watermakes.	1.3. Refusal at bedrock.
TP11	Sandy, gravelly clay with rounded limestone clasts. Weathered limestone bedrock, unable to penetrate further. No watermakes.	0.9. Refusal at bedrock.
TP12	Weathered limestone bedrock. Unable to penetrate further. No watermakes.	0.2. Refusal at bedrock.
TP13	Weathered limestone bedrock. Unable to penetrate further. No watermakes.	0.2. Refusal at bedrock.
TP14	Sandy clay, light brown. brown clay, slight sand, stiff. grey clay, stiff, with large well rounded boulders. Weathered limestone bedrock. Unable to penetrate further. Watermake at 1.5m.	1.5. Refusal at bedrock.
TP15	Sandy, brown subsoil. Sandy clay with angular boulders. Weathered bedrock. Slight water seep. Limestone bedrock.	1.0. Refusal at bedrock.
TP16	Brown stiff clay, few gravels. Stiff clay, few boulders. Sandy clay, few well rounded boulders. Slight watermake at 1.0m. Ridged weathered bedrock with clint and gryke form. Unable to penetrate	1.1. Refusal at bedrock.
TP17	Buff coloured, sandy subsoil. Sandy clayey gravels with well rounded boulders, some silts. Gravels increasing with depth, fewer boulders.	2.5. Limit of excavator.
TP18	Clayey gravels with some sand. Gravelly clays with sand, well rounded limestone gravels and boulders. Clay with gravels.	2.3. Limit of excavator.

TP19	Sandy, clayey gravels with well rounded clasts. Sandy gravel. Gravelly clay, grey, damp, stiff with large boulders.	3.2. Limit of excavator.

Conclusions

The bedrock is considered a regionally important aquifer with karstification present and is given the rating Rk.

The trial-pitting has indicated that the overburden generally consists of thin gravelly and sandy clays with few watermakes.

Applying the EPA Code of Practice the landbank areas fall into the $R3^2$ category.

In the northern fields around Ballingarrane the soil – subsoil sequence is not consistently thicker than 2m (as required by the code of practice) and is generally less than 1m in thickness. Evidence of karstification was present in the uncovered upper bedrock at this site. In the southern fields the trialpitting (TP17, TP18 and TP19) have indicated that the soil – subsoil sequence is consistently thicker than 2m.

The EPA IPC for this landbank stated: "All lands where lairage/truck solids waste from the licensed activity is to be landspread must be outlines in the IPC application. Alterations to this landbank are subject to prior written agreement with the Agency. No other waste shall be spread without prior written agreement with the Agency.

In view of the proposed restrictions, AIBP proposes to locate alternative landbanks with more sustainable groundwater vulnerability (as per the EPA Draft Code of Practice for Landspreading, 1997).

CONCLUSIONS

Comprehensive aquifer vulnerability assessments are being undertaken for all landspreading operations. The EPA Code of Practice is being implemented in the assessment of the landbanks. Current assessments are being made on a site specific basis using data available from national databases (GSI, Teagasc) and from fieldwork. A large volume of organic waste results from the meat processing industry. This waste is significantly different to agricultural waste and amended nutrient management plans are being suggested.

ACKNOWLEDGEMENTS

The author would like to thank AIBP Development and in particular John Durkan, Environmental Manager and Michael O'Callaghan in the preparation of this paper.

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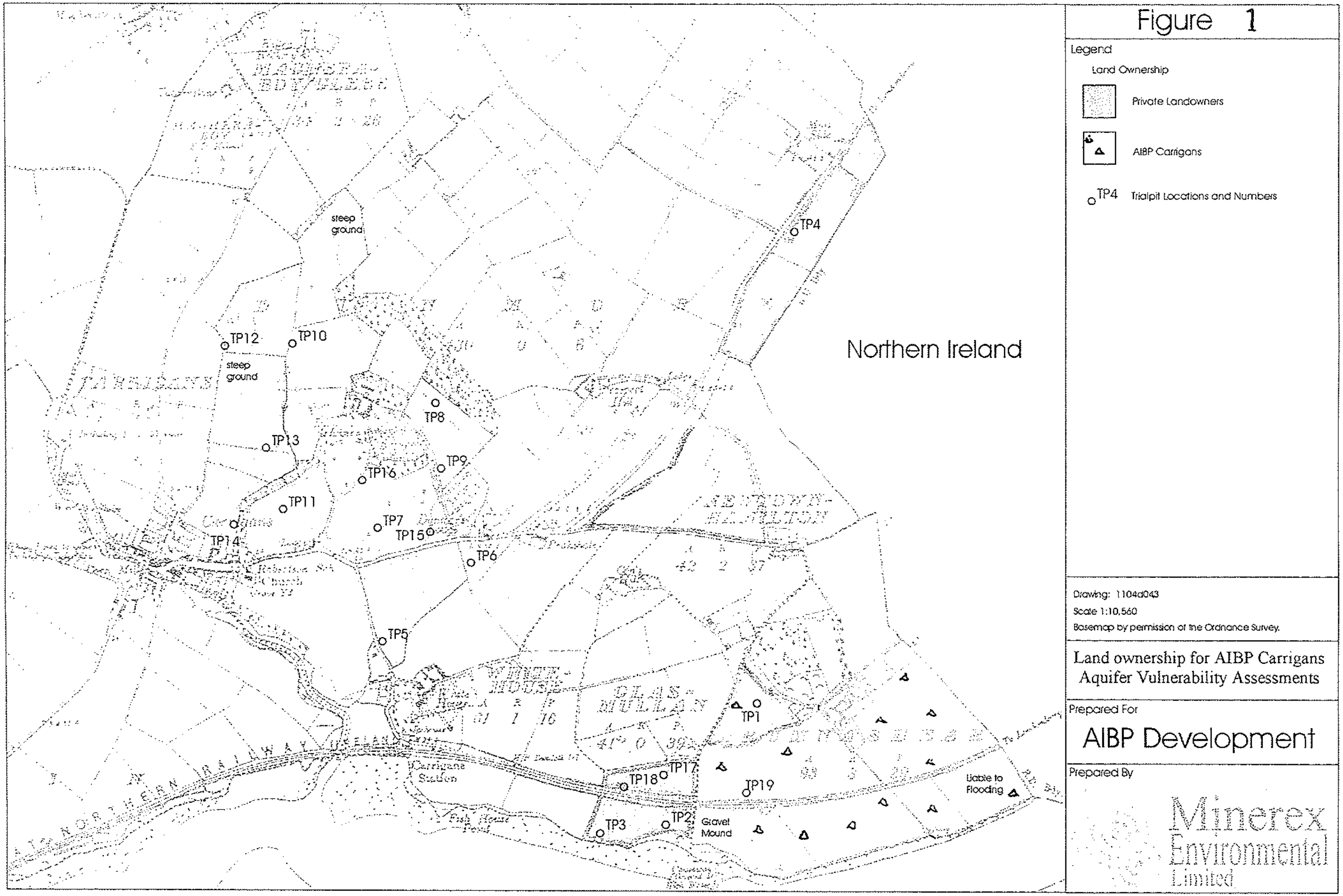
Figure 1
Legend
Land Ownership
Private Landowners
AIBP Carrigans
TP4 Trialpit Locations and Numbers
Northern Ireland
TP1
TP2
TP3
TP4
TP5
TP6
TP7
TP8
TP9
TP10
TP11
TP12
TP13
TP14
TP15
TP16
TP17
TP18
TP19
steep ground
steep ground
Gravel Mound
Liable to Flooding
Drawing: 1104d043
Scale 1:10,560
Basemap by permission of the Ordnance Survey.
Land ownership for AIBP Carrigans
Aquifer Vulnerability Assessments
Prepared For
AIBP Development
Prepared By
Minerex Environmental Limited

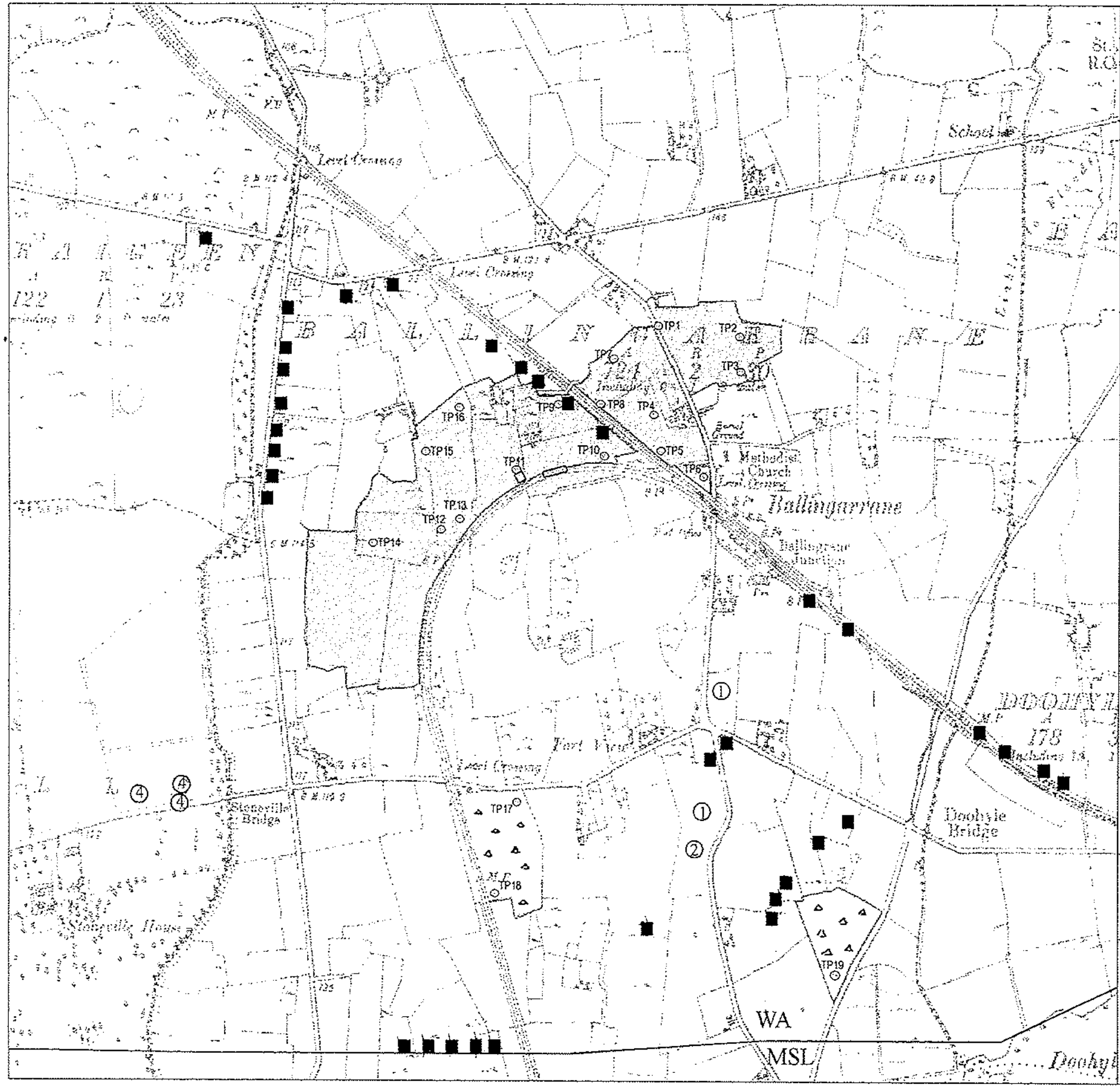

Figure 2

Legend

- Subsoils <2m in depth
- Subsoils >2m in depth
- WA Waulsortian Reef Limestone bedrock
- MSL Muddy Shelf Limestone bedrock
- Bedrock at Surface
- ④ Borehole indicating depth to Bedrock
- TP2○ Trialpit Number and Location
- Area waste is currently stored
- Area proposed for storage

Drawing Ref.: 1104d041
Scale 1:7,500
Data from Groundwater Vulnerability and Protection in County Limerick by Jenny Deakin, GSI, 1994.
Basemap by permission of the Ordnance Survey.

AIBP Rathkeale
Aquifer Vulnerability Assessments

Prepared For

AIBP Development

Prepared By

Minerex Environmental Limited

Paper No. 9.

Waste Management Strategy for the Dublin Region.

P. J. Rudden, M. C. O'Sullivan & Co. Limited.

Waste Management Strategy for the Dublin Region

by P. J. Rudden of
M. C. O'Sullivan & Co. Ltd., Consulting Engineers

ABSTRACT

The paper describes the development of a new Waste Management Strategy for the Dublin Region comprising: Dublin City, Fingal, South Dublin and Dun Laoghaire Rathdown Counties. The strategy framework is outlined in legislative and environmental terms. The study focused on existing landfill disposal, its deficiencies and strengths. Alternative waste management scenarios were examined in technical, environmental and cost terms and the Best Practicable Environmental Option recommended. Institutional, regulatory and financial means of achieving the strategy were determined and recommended. The entire approach was underpinned by a multi-faceted public consultation approach which involved all the main players in the debate – householders, industry, recycling and environmental organisations. The strategy has been adopted by the four Dublin Authorities and a single Regional Waste Management Plan is now under preparation.

1. PURPOSE OF STRATEGY STUDY

In early 1997, the four Dublin Local Authorities *(Dublin Corporation, Fingal County Council, South Dublin County Council and Dun Laoghaire - Rathdown County Council)* commissioned the MCCK Consultancy Group to undertake a strategy study into the future management of all non-hazardous waste in the Dublin Region. The purpose of the study was to find the Best Practicable Environmental Option (BPEO) for the future management of wastes generated in the four Local Authority areas.

2. PROGRAMME

The study commenced in February, 1997 and was completed in December, 1997. The strategy was adopted by the Elected Members of each of the four local authorities at a series of special meetings held in January, 1998. Work is now proceeding to formulate a single Draft Waste Management Plan to go on public display in the summer of 1998.

3. STRATEGY FRAMEWORK

The study was commissioned against the background of the Waste Management Act, 1996 and developing legislation from the European Union. The new Waste Act requires that waste minimisation be given priority attention. If waste generation cannot be avoided, the next priority is to re-use or recycle in preference to disposal. If waste is not re-used or recycled, energy recovery from waste ought to be investigated in advance of disposal. Finally, if waste cannot be recycled or recovered, it should be disposed of in an environmentally safe manner and in accordance with the *"polluter pays principle"*. This latter principle requires that the full cost of waste disposal be borne by the producers of such wastes.

The development of the strategy study was carried out against the background of a developing Capital City and Region with a population slightly in excess of one million people. The requirement for modern waste management practice forms part of an ever improving physical infrastructure which allows residents to live, work and play and where communities can evolve and develop. The provision of infrastructural services should be seen primarily as the creation of a sustainable social and physical environment.

4. AIMS OF THE STRATEGY STUDY

The principal aims of the study, which covers technical, environmental, institutional and financial aspects, are as follows:

- *To determine and assess the current waste management situation in the Dublin Region with regard to the quantity and nature of waste generated by households, the commercial and industrial sectors and other waste streams.*

- *To recommend an integrated waste management strategy for the efficient future management of these wastes in the Region in accordance with current legislation and developing environmental policies, together with a plan of its implementation.*

- *To recommend the most appropriate organisational, regulatory and funding mechanisms in support of the preferred waste management strategy.*

Waste management planning is a very dynamic process, with new and emerging technologies to be considered on an on-going basis. Therefore, the study has considered a long-term horizon of 15-20 years, but with short and medium term objectives and recommendations.

5. DEFINING THE PROBLEM

The current problem with waste disposal is serious in scale together with the difficulty in meeting future legislative requirements, particularly in terms of meeting the latest EU Landfill Directive and EU Packaging Directive The region's population based in the 1996 Census is 1,056,666 contained within some 350,000 households. There is currently 4.3 million tonnes per annum of total waste in the Dublin Region from household, commercial, industrial, agricultural and quarrying sources. The principal problem is with approximately 2.3 million tonnes of household, commercial and non-hazardous industrial waste which is currently landfilled, 50% of which comes from construction / demolition activities. Over 1 million tonnes of this waste is currently landfilled at Balleally in North Dublin which is the largest waste facility in the country.

Table 1 below gives the breakdown of the various waste types by source which constitute the current landfill quantity of 2.3 million tonnes. People generally are surprised to see that household waste – the real problem for landfill – only constitutes some 13% of the quantity currently landfilled. The figures indicate a household waste generation contribution of 325kg per head per annum.

Waste Type	Annual Tonnage (t/annum)	% Total
Household	303,407	13%
Green Garden	61,658	3%
Building Items	18,000	1%
Commercial	308,052	13%
Industrial	408,156	17%
Construction/Demolition	1,223,013	53%
Total	**2,322,286**	**100%**

Table 1: Municipal and Similar Waste Arisings

The Waste Management Act 1996 together with Irish and EU recycling targets cannot be realised with the current system either in terms of capacity or environmental standards. New methods of waste collection, treatment and disposal must be found to solve the two principal problems of:

- *lack of waste recycling and disposal capacity*
- *the need to meet new EU and Irish environmental standards*

6. PUBLIC CONSULTATION

Following advertisement of the commencement of the strategy study in national and local newspapers in early February 1997, submissions were invited from all interested persons or organisations over a two month period. Submissions were received from individuals, groups and environmental organisations and these were taken into account in formulation of the strategy. It was obvious from the public comment that the problem of waste management and litter prevention and control are intrinsically linked in the public mind.

There was also a perception that household waste is the real problem together with a call for more recycling. The reality is that most people in Dublin did not regard waste as a problem at all. This is because their waste is removed efficiently by local authority staff every week. Furthermore, they pay nothing for this service, regardless of the amount of waste produced or left out for disposal. With new waste legislation forcing dramatic changes in our future approach to waste management, existing patterns of disposal can no longer continue.

7. IDENTIFICATION OF DEFICIENCIES/STRENGTHS IN LANDFILL SITUATION

As current waste disposal is to landfill, the following deficiencies and strengths were identified in terms of the regional problem.

- *The combined void space in currently approved landfills in the Dublin Region (including the new Arthurstown facility) equates to approximately 2.5 years of filling at current rates of disposal*

- *The fact that the new Arthurstown landfill can only accept baled waste places a number of significant restrictions on the waste disposal system for the Dublin Region*

- *The full utilisation of Arthurstown landfill is dependent on sufficient baling capacity being available in the Dublin Region*

- *There is an urgent need for the immediate provision of unbaled waste capacity in the Dublin Region*

- *There is an urgent need to divert certain wastes from Balleally landfill to recovery options wherever possible, particularly construction/demolition wastes whose volumes are high and recycling potential are optimum in terms of readily usable products*

However, the following strengths exist in the landfill situation:-

- *Landfill as a technology can cater for all components of the waste stream*

- *The new Arthurstown landfill will provide short-term to medium term relief for baleable waste disposal in the Dublin region to up-to-date EU standards*

- *The number of landfills in the Dublin region will be reduced to two in the short-term, one for baled waste in the South West and the other for unbaled waste in the Fingal area providing two facilities capable of operation to new EPA standards*

- *The local authorities recognise the need for closure and aftercare programmes for the major closed facilities and, are making budgetary provisions in this regard*

The disposal of these very large waste volumes constitutes a major challenge for the Dublin Local Authorities in the immediate future. At present, the majority of household and commercial waste, with a large proportion of construction/demolition waste, is disposed of to Balleally Landfill, which is rapidly nearing capacity. The commissioning of the Arthurstown waste disposal facility for baled waste at the end of 1997 has provided a facility for municipal baled waste to very high standard over the coming years. Nevertheless, there is an urgent need for new waste management initiatives to reduce the waste volumes produced and to divert waste from landfill to the maximum possible extent.

8. ASSESSMENT OF ALTERNATIVE STRATEGIES

Having regard to strategy objectives of meeting Irish and EU recycling targets and reducing our dependence on landfill as a sole means of disposal, a number of waste management scenarios were examined to enable the Best Practical Environmental Option to be chosen.

- *Scenario 1: To meet the mandatory recycling targets and comply with EU Draft Waste Legislation including the Draft EU Landfill Directive.*

- *Scenario 2: Directive to achieve the maximum realistic level of recycling thus exceeding the mandatory recycling targets.*

- *Scenario 3: To meet mandatory recycling targets, comply with EU Draft Waste Legislation and to achieve efficient bulk waste reduction through thermal treatment*

- *Scenario 4: To reach maximum realistic level of recycling to comply with EU Draft Waste Legislation and to achieve bulk waste reduction through thermal treatment. Minimise reliance on landfill*

These four scenarios were compared on the basis of meeting statutory requirements in terms of technical capacity, environmental acceptance and cost incurrence. This assessment included a cost benefit analysis having regard to environmental concerns such as global warming, acidification, nitrification, photo-chemical ozone formation, heavy metals and dioxin generation.

9. RECOMMENDED WASTE STRATEGY

The recommended strategy having regard to the objectives of the study is Scenario 4 above as it is reckoned to the Best Practicable Environmental Option and most likely to provide a robust sustainable waste management system for the region in accordance with legal and practical requirements. It will also make provision for the required increased regulation of waste movement by the local authorities.

The central recommendations of the waste management strategy are as follows:

- ***Waste Minimisation:***

  - *That Dublin adopt a 'Green Region' approach in promoting waste reduction/minimisation initiative with the support of resident associations, community and business groups.*

  - *This process should be assisted by new public education units within re-organised Waste Services Departments of the four local authorities staffed with Community Environmental Officers.*

  - *There is a need to create a new focus on waste management obligations of householders, institutions, commercial and industrial enterprises.*

- ***Waste Collection:***

    - *Extension of home address source separation and collection of recyclables to the Dublin region to cover 80% of the population.*

    - *Upgrading and extending of bring bank centres in centre city areas*

    - *Provision of 10 new Waste Recycling Centres distributed throughout the Region to where the public can bring various types of recyclable wastes and bulky items which cannot be handled by conventional waste collection methods.*

    - *Source separation and separate collection of household organic waste based on alternative weekly collection using a dual bin system is recommended but based on initial piloting of the collection system.*

    - *Source separation of commercial and industrial waste*

    - *Recommendations are made with regard to the source separation of harmful household wastes (e.g. batteries, oils, paints, etc.)*

    - *The source separation of construction/demolition waste from building sites and major infrastructural projects should commence leading to increased recycling and minimised landfilling of this high volume waste stream*

- ***Waste Recycling/Recovery and Disposal:*** *the following recommendations are made with regard to waste recycling, recovery and disposal:*

    - *Provision of additional sorting facilities for recyclables*

    - *Provision of green garden waste depots and composting facilities to collection and treat green waste from parks and garden waste delivered by residents to Waste Recycling Centres*

    - *Facilities for the biological treatment of household organic waste using composting or biological digestion methods*

    - *The provision of a central thermal treatment facility with energy recovery because of the critical shortage of available landfill capacity and in order to satisfy the requirements of EU Legislation*

    - *Provision of facilities for the reception, sorting and recycling of construction/demolition waste together with market creation for the recycled products to incorporate into future road, footway, cycle-way and parks projects*

    - *The requirement to create an additional unbaled landfill facility for the disposal of wastes which cannot be recycled or recovered together with catering for residual wastes resulting from recycling. This additional* *requirement of some 10 – 11 million tonnes of void space needs to be selected having regard to the EPA Guidelines on Landfill Site Selection.*

These recommendations represent the best environmental approach to future waste management in the Dublin area. In addition, it also results in the highest recycling levels and the lowest quantities going to landfill. It therefore meets strategy requirements best as the Best Practicable Environmental Option.

The overall effect of the strategy will be to dramatically cut dependence on landfill from approximately 80% for all wastes in 1997 to some 16% by the year 2004 provided that new recycling projects are put in place together with construction of a new thermal treatment plant. It should be noted that the provision of thermal treatment will not adversely effect progress on recycling.

Table 2 below shows the breakdown of how the new recycling targets can be met.

	Form of Treatment/Disposal		
Source	**Recycling**	**Thermal**	**Landfill**
Households	60%	39%	1%
Commerce and Industry	41%	37%	22%
Construction/Demolition	82%	0%	18%
Total	**59%**	**25%**	**16%**

Table 2: Dublin Waste Strategy Targets for Year 2010

10. PAYING FOR WASTE MANAGEMENT

The study examined the annual costs of the proposed strategy including annual operating costs plus annual investment costs. The total investment costs of new waste facilities up to year 2011 is IR£253 million - IR£29 million (12%) on collection, IR£165 million (65%) on recycling and recovery and IR£59 million (23%) on landfill. This shows that current expenditures which are in the order of £24 million per annum will increase by approximately £20 million over the first five years and by over £30 million to almost £60 million per annum in conjunction with proposed major investments. In addition, £5 million is required in the initial 2 to 3 years to fund the setting up of new planning, regulation and public education units together with significant community based initiatives in waste reduction and minimisation, school programmes, media promotion, etc.

Faced with an additional £20-30 million annual outlay even in the initial years, the local authorities in the region will require to put in place appropriate charging mechanisms to provide funding. The bulk of this funding should be provided through charges levied on waste producers which include both householders and industrial and commercial users of the system. Such charges should be related to the waste volumes produced and should provide incentives for waste minimisation and recycling.

It may also be necessary to use *"gate fee"* levels as an economic mechanism to divert waste from less desirable options (landfill disposal) to more favourable options (recycling). This may involve higher landfill disposal charges to subsidise more environmentally desirable elements of the strategy.

11. PUBLIC AND COMMUNITY ASPECTS

A primary objective for the strategy is the minimisation of waste produced through support for community and waste producer initiatives by the local authorities in the region. A *"Green Region"* or similar identification is necessary in ensuring that the four Dublin local authorities lead this debate as the primary waste planners under new legislation. **Central to the strategy is the essential requirement that the four authorities adopt a single Regional Waste Management Plan**. Allied to this, it is of fundamental importance that there be a common approach towards planning, regulation and user charges in the four local authority areas.

A substantial package of public education is required over the period 1998 to 2000 to prepare Dublin for waste management in the new millennium. This will include appointment of waste education officers, public education programmes to encourage waste minimisation, home composting, etc. and support for community initiatives operating to Local Agenda 21 objectives.

The public information role is seen as particularly important in ensuring a full understanding among waste producers generally of the nature and volume of waste being produced, the options for its reduction and recycling and the need for waste treatment and disposal facilities. A clear link between the relative levels of service delivery/environmental protection and the resulting costs must be generated in the public mind.

12. REGULATORY ASPECTS

Under the Waste Management Act 1996, local authorities are required to regulate waste practice and movements throughout their administrative areas. This includes, where deemed necessary:

- *Making local bylaws requiring all holders of household waste to present such waste for collection in a fit manner, e.g. type of bins, bags, etc.*

- *Requiring certain wastes to be brought to certain recycling or disposal sites where considered to be required for effective and orderly management of such wastes*

In addition, all waste collectors including recycling organisations must be granted waste movement permits by the local authorities who may attach specific conditions to such permits.

In future, all major waste recovery and disposal sites must be licensed by the Environmental Protection Agency.

13. CORE RECOMMENDATIONS FOR STRATEGY IMPLEMENTATION

In the best interests of the Dublin Region, it was recommended that:

1. A single Regional Waste Management Plan be adopted by the four Dublin Local Authorities - this should be agreed in principle by each authority early in 1998 and preparations set in train for the formulation and presentation of a Draft Regional Waste Management Plan including public consultation

2. That the recommendations of this strategy report form the basis for the preparation of the Regional Waste Management Plan

3. That the Polluter Pays Principle applies. There is a clear necessity for waste producer charges. Volume or use related charges are recommended on the basis of fair and equitable apportionment. These charges need to apply to all producers of waste at household, commercial and industrial level

4. In the context of the Regional Plan, a consistent policy on waste planning, regulation and user charges needs to be implemented across the Dublin Region

14. STEPS TOWARDS STRATEGY IMPLEMENTATION

The following phased approach was required to implement the strategy;

1. Waste Strategy Report to be published early in 1998. At the same time, Elected Members of each local authority to be briefed on the contents of the Draft Strategy early in 1998

2. Arrangements to be made for public consultation by facilitating public displays and access to the Draft Waste Strategy Report dated December 1997

3. Each local authority to note the Draft Strategy proposals and each to agree in principle to make a single Regional Waste Management Plan in accordance with the Waste Management Act 1996

4. Each Local Authority to fund and set up new Waste Services Departments to include new waste planning, regulation and public education units early in 1998 to commence recommended initiatives on waste minimisation/recycling particularly community initiatives

5. Draft Regional Waste Management Plan to be formulated and placed on public display in accordance with the 1996 Act, also seeking public submissions on same - target date 1st July, 1998 for public display

6. Discussion and adoption of Regional Waste Management Plan at Elected Member level by 31st December, 1998 (at which date the current Dublin Corporation Waste Plan expires)

15. CONCLUSION

The challenge to plan for improved waste management in Ireland is one of the greatest facing environmental engineers and scientists as we approach the new millennium. The regional approach is recommended for reasons of partnership, economy of scale and finding environmentally sustainable alternatives to landfill. However, landfill must remain for the bulk of Irish wastes in the short-term and for residual wastes in the longer term. Planning for its provision will never be easy but must be pursued in a logical consultative way for the common good having regard to best hydrogeological siting and design practice.

ACKNOWLEDGEMENTS

The permission of the four Dublin Local Authorities through the Dublin Waste Steering Group (Chairman, Mr. Derek Brady, Deputy City Manager and Mr. John Singleton, Project Engineer) to present this paper is gratefully acknowledged.

The MCCK Consultancy Group is an Irish-Danish partnership led by M. C. O'Sullivan & Co. Ltd., Consulting Engineers and COWI Consulting Engineers and Planners Copenhagen, assisted by City of Copenhagen's Environment Protection Agency and KPMG Management Consulting.

The full reports on the study (4 volumes) are available for public consultation at each of the Dublin Public Libraries.

Paper No. 10.

Water and Environmental Health - The Wider Dimensions.

Anne Deacon, South Eastern Health Board.

WATER AND ENVIRONMENTAL HEALTH - THE WIDER DIMENSIONS

Anne Deacon, Senior Environmental Health Officer, South Eastern Health Board

Abstract

Water is essential for life, it is one of mans most basic needs. Environmental Health deals with all aspects of the environment that can impact on health. In the Health Strategy the Department of Health acknowledges that there are wider dimensions to Health than the mere curing of diseases - one of these would be the effect water quality has on health. An Environmental Health Officer working in a Health Board assesses water as a food ingredient and as a vector for infectious diseases. They have no function with regard to public supplies unless the supply has been proved to be "contaminated". They are barred as a profession from working directly for Local Authorities by the Department of the Environment and only a minority of Local Authorities contract out the monitoring of public and private water supplies to them. Some Health Boards provide a water analysis service for private individuals.

Groundwater quality is disimproving as the statistics are improving. Groundwater behaves as a complex biological system. It can be preserved, conserved and managed as a resource but it cannot be controlled by the use of waste and/or water treatment plants.

Groundwater management requires input at many levels including planning control, waste management, health surveillance, hydrogeology, stock conservation and quality control.

Local knowledge and comprehensive national surveillance are both essential. A multi disciplinary approach is required but there has to be a lead authority and a definition of roles.

INTRODUCTION

Once upon a time not so long ago a clerical officer named Cathy in the Health Board decided she would buy a house out of town within a five mile radius. Since Cathy was a major asset to the Environmental Health Department they were determined that she would have the benefit of their best processional advise. Cathy was delighted with this - at first.

The first advise was not to buy any house until they had sampled the water and checked our the septic tank, etc. for her. So far, so good.

After the first weekend's searching Cathy came back with a house overlooking the town with a lovely view. Unfortunately the reason it had a lovely view was because it was on the side of a mountain and over the years people has moved out of town and built "single houses" in groups of thirty along both sides of every road that went up, down or across the mountain. So by the end of her tea-break Cathy was informed that her initial choice was a disaster due to the fact that she would end up drinking sewage from <u>all</u> the neighbours houses above her and that the only houses she should consider were those on the upper side of the top road. Also, she should ensure that she installed a sewage treatment plant and that there was no existing house directly below her when she moved in. If not someone below her could claim she was polluting their drinking water at a later date. No to the NORTH.

Next Cathy tried the SOUTHERN edge of town and selected an old Garda station house in a village. However, when the E.H.O.'s heard about this they pointed out that the septic tank was too near her well and given the age of the building there was likely to be at least two soakpits on site which could almost definitely pollute her well (major lecture on water table levels and rainfall and flash floods underground followed).

So to the EAST. Cathy was relieved to find that none of the E.H.O.'s had ever tested the well at this house for any reason. .However, they had tested the wells in the commercial and public buildings in the village and in recent years these had all given unsatisfactory results so she got instructed on how if she turned on her taps and the school was not using water she could take in their polluted water or if the shop ___ FORGET IT!

So it was a case of to hell or the WEST. Before she even started looking she asked about the water. No one has ever detected a problem with sewage but iron and manganese could be a problem. Not a health problem you understand, just if you wanted to use a dishwasher, wash any clothes, not have staining on sanitary ware or blocked plumbing. Oh, and check the plumbing was all plastic because the water was acid and would eat our all her heating and water pipes otherwise. But if she saw a house she liked, be sure and let them know so they could test the water and see how much of a water treatment plant she would need. As well as a sewage treatment plant? Well, that would depend on the size of the site and whether adjacent sites were developed and ....

WATER & ENVIRONMENTAL HEALTH - THE WIDER DIMENSIONS

I'm here as an Environmental Health Officer who has absolutely no responsibility for the quality of drinking water. I have observer status but as a professional observer I am worried about the deterioration in quality of our drinking water in recent years.

We all need to look at the overall picture - by we I mean every Well Driller, Engineer, Environmental Health Officer, Hydrogeologist, Planner, Environmental Technician or Scientist and policy makers. Not just the day to day problems or even special problems but the global picture.

We need to look at it from the laymans viewpoint. We need to remember how important water is and by implication how important our role is.

I am going to confine my remarks to groundwater rather than surface water and within groundwater to drinking water.

I am going to use a number of case studies to illustrate what I am talking about. These are based on real life situations and are representative of the types of problems being experienced around the country. A large number of scientific papers have been published already and what I wish to do here today is to represent the situation being faced by the ordinary man in the street. This is not a scientific paper.

At the end of the talk I want you to have proof that our drinking water quality is deteriorating, to know what some of the causes are and why we all need to get together now and play our part in ensuring that in Ireland it will not become a case of "water, water everywhere and not a drop to drink".

WATER - HOW IMPORTANT IS IT?

It is the second most important need for sustaining life. Only by depriving ourselves of air can we die more quickly. It is more important than any other food. We are water! - 70% of your body weight is water.

Drinking water should therefore be so much in our consciousness that money and personnel should be liberally assigned to its management and if money is stretched the water budget should be the very last budget to be cut. Water production and protection and monitoring should always be handled by the most senior and most experienced of staff. It should never be left to chance.

It is more important than housing even, though this too is a primary need. Planning and development and the creation of jobs are secondary needs. Aesthetically pleasing buildings, art, music and politics are tertiary needs.

Everyone, no matter who they are, rich or poor, wants safe drinking water.

ENVIRONMENTAL HEALTH

Environmental Health concerns itself with the effects the physical environment has on all aspects of the health of the public. Even if we take the narrowest definition possible of health, i.e. the absence of disease, we cannot have good health without safe drinking water. Environmental Health Officers are professionally trained to be general practitioners with regard to drinking water. Environmental Health has a role to play with regard to groundwater and its protection, conservation and management both at a practical level and at a policy level. From an officers point of view planning and development control on public health grounds, infectious disease monitoring, wellhead protection, catchment surveillance, quality monitoring of public and private supplies and waste management are all within his competence and his area of professional interest. He does not expect to be the only professional or policy maker with a valid input but he can be certain also that his input is from a unique perspective. Over the years the profession has gained the man in the street's confidence as an authority on the safety of drinking water.

It is not in the public's interest therefore that Environmental Health Officers are barred as a profession from working directly for Local Authorities by the Department of the Environment. Only a minority of Local Authorities contract out the monitoring of public and private water supplies to the Health Board and the same applies to planning, development, waste control or water protection issues.

THE WIDER DIMENSIONS

Because Environmental Health Officers are barred from working with Local Authorities the majority work for Health Boards. Despite the man in the streets perception that safe water is a health issue Environmental Health Officers have very little legal responsibility when it comes to drinking water.

1. Drinking water used is a food premises for which the E.H.O.'s are responsible must be of drinking water quality and those on private wells are therefore tested approximately annually.

2. If a public supply becomes contaminated and the Health Board is aware of it they will work with the Local Authority to help solve the problem.

That's it except for "the wider dimensions". The Health Strategy for the Health Boards talks about the wider dimensions. All Health Boards will test water when investigating infectious disease cases. Some will test schools on private wells and hospital supplies periodically. Some will test water for private individuals . Our office stopped testing drinking water for private individuals last year. This has resulted in us referring them to private laboratories if they can afford it and then we usually end up explaining the results to them on the phone or generally discussing treatments on the phone without the benefit of a site visit or details of sampling procedures. Obviously this is not satisfactory from the public's point of view or ours and hopefully we can persuade our employers to reinstate the service at a charge with concessions available for poorer people. The wider dimensions theory is all about intersectoral co-operation with the purpose of influencing environmental policy and practice to protect and promote the public health. Obviously drinking water is central to public health. But since it is not a core function the Health Board cannot be the lead authority. However, the Health Board's participation can be legitimately asked for if there is a definite threat to any water supply either public or private in their functional area both for practical help and policy formation. As well as Environmental Health Officers some departments of Public Health Medicine may also offer assistance. For anything other than fire brigade action however they need a system to link into.

GROUNDWATER QUALITY

Groundwater is disimproving but the Health Board statistics are improving. How? Once an Environmental Health Officer establishes that a private commercial supply is polluted he/she serves a notice under food law requiring a treatment plant. The plant goes in, the results improve but we know the groundwater continues to disimprove.

Case Study A - Village Gone Bad

An E.H.O. tests private commercial wells and comes up with unsatisfactory results. Makes everyone cover their wells properly and disinfect and follows up. Some give 3 clear bacteriological samples and are left as OK. Shop requires treatment which is installed only for shop, not for 3 houses also on the well. Some chemical tests are also taken. All OK.

6 years later, E.H.O. does more testing. 1 school fails. Treatment plant installed. 2nd school fails. Abandons well and shares with 1st school. Pub, another shop and take away fail. Private house exceeds guideline for nitrates but not EC maximum. Nitrates in wells have gone up from 7mg/l to 27mg/l. Health centre and 2 private houses fail and are connected up to council well. Only 1 restaurant, and 2 lightly used wells at top of village remained untreated.

Statistically there are 2 bad wells in the village at present, but the groundwater had disimproved dramatically. Also iron would seem to be causing a problem periodically which would lead one to suspect occasional acidic pulses of pollution but unfortunately UV tubes are the treatment plant installed leading to worries about coating of the tubes longterm. The ordinary man in the street cannot be expected to know the niceties of choosing one type of treatment above another. The chances of a water treatment plant being operated effectively are 50/50 from my experience. The reasons for lack of efficiency are cost of chemicals, not a mechanical type of person, fed up doing, laziness, forgetfulness, not convinced of necessity, bad maintenance by company.

Also, may I appeal to well drillers please install 2 sampling taps, 1 before and 1 after a treatment plant. It enables us to keep an eye on groundwater quality if we get the mandate to do so and enables us to get the bad results required to keep the village water on the agenda.

Case Study B - Village Flooded with Invisible Water!

There was a village where the water gave sporadic coliform readings but no E.Coli. Then last August Bank Holiday weekend it rained and rained and rained. Water samples taken as late as October showed the water to be grossly polluted and the wells haven't recovered. Treatment plant are being installed on the 2 bigger wells. No flood health warning was issued to people using wells, yet practically every well in the county must have been affected

SOME RIDDLES ASKED BY A LAYMAN

- What is a single swelling? A house on a suitable site which is surrounded by 4 undeveloped sites of equal suitability and size.

- What size should a single site be? 10 acres for sustainability? 2.5 acres for waste assimilation capacity? 1 acre for Devs sake? No less for God's sake!

- If 3 caravans constitute a caravan park by law because as temporary dwellings they would cause sanitary problems without provision of communal services why doesn't 3 houses constitute a housing estate?

- If 5 houses built at the one time require communal treatment for public health reasons why is it OK for 5 houses built incrementally on the same spot to have 5 separate systems?

- One house per hill? Just ask Cathy!

- If the Local Authority will only build a single house in a rural area for farmers, fishermen or foresters and require all other housing applicants to live in housing schemes and villages adjacent to facilities, how come better off people are not confined to developing the urban fringe?

- How come in some European counties the developers must install water supply, sewerage and roads first before he builds a house but in Ireland they can do in the other way around and have been known to abandon without ever completing the scheme? This is happening even today when whole estates have been sold off the plans at inflated prices. They have all the money before they even lay a brick.

WATER IS A COMPLEX BIOLOGICAL SYSTEM

Treatment plants both water and waste have their place in the protection of groundwater but it is always wise to remember that man has never tamed nature nor is he ever likely to be able to do so. Man can intervene in to natural water systems but must be very careful.

- The system is a circle

  Put literally "Water is the one substance from which the earth can conceal nothing. It sucks out it's innermost secrets and brings them to our very lips" and that ladies and gentlemen incudes sewage and chemicals and whatever else goes into the ground from septic tanks, sewage treatment plants, landfills and illegal dumps and graveyards.

- "Man may come and man may go, but I go on forever" Tennyson

  Water there will always be but will it be safe to drink? Every decision made that has even a knock on effect on the water system will have long term additive effects. These effects may take time to show up and it is guaranteed that when they do they will take a long time to cure. Prevention is the only answer.

- There is no machine that can be built that will control groundwater and make it drinkable. We are working with a complex biological system which you can't see and which is not fully understood even by hydrogeologists. If you can't fix it don't break it. It requires a multi-disciplinary approach and sharing of everyone's information and skills.

WASTE TREATMENT PLANTS

- Compliance with an Agreement Certificate may not protect groundwater or public health.
- All treatment plants must have outfalls. Whatever comes out of the outfall goes into the groundwater. It is never of drinking water quality standards.
- Intermittently used biological treatment plants do not work properly and may even die.
- The cowboys have arrived. We now have "cheap" plants. In the absence of hard scientific data that a particular plant is capable of producing best effluent practicable it should bot be allowed. Any treatment system will not do. Particular sites will have particular requirements.
- In the absence of effective planning requirements the dictum that all plants are OK because the engineer or architect submitting the plan has done the calculations and they are qualified people is asking for trouble.
- On small sites where public health nuisances exist if it is not an option to knock down the development a treatment plant producing best effluent practicable is a better option than a septic tank.
- To specify as a planning condition that a plant would treat effluent to a 20:30 standard when discharging to groundwater does not protect it adequately. This is a standard for discharge to surface water where the dilution factor is greater than 8 times the volume of effluent discharged from a point source outfall. What is the dilution provided by groundwater under my site? Which way is the groundwater flowing? Treatment plants would never be allowed to discharge into a water course upstream of a surface drinking water intake point. A line of outfalls discharging to surface water would require that the overall assimilative capacity of the water was not breached. Yet a line of sewage outfalls to groundwater does not warrant the same attention. Why not?

Case Study C - New Technology, New Problems

A large commercial premises had a waste treatment plant 1000' away and downhill from the newly bored well which is 200 feet deep. A water treatment plant to treat acid pH, iron, manganese and bacteria is installed. The bacteria are treated using a UV tube. All the health parameters are OK but the Chloride has gone from 12mg/l to 50mg/l to 78mg/l in three months. Are we recycling treated urine, recycling backwash water from the water treatment filter or ...? We're miles from the sea on top of a hill so saltwater intrusion is out or is it?

FROM TUNNELS TO HONEYCOMB?

We are all working in our own tunnels. Many organisations are involved at some level but there is no co-ordination. As well as Health Boards, Local Authorities, Teagasc, EPA, Department of Marine, Department of Agriculture, Department of Energy and Department of the Environment all have information and monitoring systems. The Department of the Environment have the most responsibility but because they are responsible for producing quality water and have several other conflicts of interest they cannot be the lead authority for comprehensive national surveillance and policy making. The specialists are the hydrogeologists and there is only one group of them within the public service and that is in the groundwater section of the Department of Energy. They will have to be the lead authority. To do the job they will require to tap all sources of local knowledge and this will require both computer power and manpower. They will need to get everybody working together locally and nationally and provide overall direction. Everyone will need to know where they fit into the honeycomb. I use the ideology of a honeycomb because I believe everybody should have their own particular blind tunnels reshaped to fit in with everybody else and there has to be a be a queen bee.

There are 2 grades of information available. Grade 1 information is from those laboratories which are certified by Europe for water testing and Grade 2 information is from uncertified laboratories. If Grade 2 information indicates a bad trend then it can be investigated using Grade 1 laboratories. Fewer better laboratories may be the answer. When all public laboratories are in the system, certified private laboratories should also be encouraged to join the network.

Having interested individuals in all the relevant authorities is not enough. The bodies that they work for must be officially involved and accept their role and allocate resources to that role.

This process is likely to take a couple of years form inception and given the current state of our groundwater needs to be started now. We cannot wait until the well runs dry.

Paper No. 11.

Constructed Wetlands for Wastewater Management.

Ciaran J. Costello, Maxpro Engineering.

Constructed Wetlands
For
WastewaterManagement

Ciaran J. Costello Maxpro Engineering

Abstract

Mathematical models of Constructed Wetlands, gravel substrata Reed Beds and other natural waste water treatment systems are required to give an insight into the effect the various design parameters have on the sizing and performance of such systems. Models should also be used to assist with the operation and performance evaluation of the systems. A review of the literature on the subject shows that there are many problems, both in the models proposed and in their application. The performance evaluation of systems reported in the literature have for the most part not been designed to test the models or evaluate the design parameters. In many cases, when designing Constructed Wetlands, models have either not been used or else incorrectly applied. This paper reviews the literature and proposes some variations to the model for Irish Conditions

CONTENTS

Introduction

The value of the environment in which we live is not measured in the nations Gross National Product. A country can cut down it's forests, erode it's soils, pollute its aquifers, rivers, lakes and estuaries and hunt it's wildlife and fisheries to extinction, but it's measured income is not affected as these assets disappear. Impoverishment is taken for progress. In Ireland this is nowhere more evident than in the deterioration of our water quality in respect to its economic, recreational, amenity, and aesthetic value. The incidence of pollution is a disincentive to tourism, to the location of certain sectors of industry, to the development of aquaculture and to the recreational use of waters. Moreover, the country's clean environmental image is a valuable factor in marketing agricultural and other products and in expanding tourism. There are also problems in respect to the disposal of agricultural and industrial wastes. Agricultural waste, which aside from the occasional discharge due to mismanagement, is a non point source of pollution. It is a major factor in the eutrophication of some Irish lakes and in the nitrate contamination of groundwater. At present there is no general strategy for dealing with non point source pollution.

The magnitude of the task and the limited financial resources available in Ireland indicate that alternative effective low-cost treatment systems, which could contribute to a solution of these problems, should be studied. This paper deals with the possible contribution of one such system Treatment Constructed Wetlands to the solution of these problems

CONSTRUCTED WETLANDS

Constructed Wetlands are man-made systems that are specifically designed to emulate the ability of natural wetlands to purify water. They are designed as natural biological reactors using selected substrata, plants, and engineering configurations. Constructed Wetlands are former terrestrial environments that have been modified to create a hydraulically controlled system containing selected substrata wetlands flora and fauna for the primary purpose of contaminant or pollutant removal from wastewater. Constructed wetlands are designed to transform many pollutants into gaseous forms for release to the long-term biogeochemical reservoir in the atmosphere or to trap others in the substrata and are designed and operated as wastewater treatment systems, though many systems do support other functional values. In these systems which include constructed wetlands, anaerobic, high-rate and facultative ponds, mass culture of higher plants and animals and land application, wastewater is principally treated by bacterial metabolism, physical sedimentation and chemical reactions. The different unit process operations may be arranged in sequence to create integrated treatment systems. Wetlands are those areas where at least periodically, the land supports predominantly hydrophates and the substrata is predominantly undrained hydric soil or the substrata is non-soil and is saturated with water or covered by shallow water at some time during the growing season of each year. Natural wetlands have and continue to support wetlands flora and fauna. Natural wetlands can be and are designed to operate as wastewater treatment systems and continue to function as habitats for wild life and support other functional values.

Constructed Wetlands are conceived as thin film natural biological reactors using selected substrata, plants, and engineering configurations and must be designed using sound engineering practice to ensure the effectiveness of the wastewater treatment. The productivity of wetlands can be very high. They receive, hold, and recycle nutrients continually washed from upland regions. As a result of their high productivity and diversity of plant life, wetlands are also rich habitats for insects, micro-organisms, zooplankton, crustaceans, reptiles, amphibians, fish, birds and animals. It is the diversity of wetlands, and the ability of the aquatic plants both to survive in a saturated environment and at the same time create an aerobic zone at the rhizosphere by transporting oxygen from the leaves to the roots that give wetland ecosystems their unique ability to remove aquatic pollutants. The oxygen transported to below ground tissues of wetland plants can leak out of roots and oxidise the surrounding sub-strata. (See Figure 1.) Oxygen is also introduced at the water surface by wind action. This oxidation supports aerobic microbial populations in the rhizosphere and the water column that modify nutrients, metallic ions and trace organics. The microbial decomposition occurring in wetlands is shown in Figure 2.

WASTE WATER TREATMENT.

To understand how Constructed Wetlands work and the role they play in the treatment of municipal wastewater we must understand how sewage treatment systems work. Conventional treatment systems vary from simple direct discharge into the receiving waters, which allow, by adequate dilution, for the treatment of the contaminants, through to the fully integrated systems. A full conventional wastewater treatment plant is a complex bioengineering process involving the use of bacteria that alter contaminated substances to obtain nutrients or energy to live.

Construction and costs are high as the number of unit operations such as screening, pumping, primary treatment, secondary treatment, advanced treatment and sludge processing involve a considerable amount of Process Engineering equipment and design. Operation and maintenance costs are also high as they require considerable energy, chemicals, and management inputs. For optimum performance a steady state flow of nutrients is required[1].

The sludge arising from wastewater treatment processes must also be disposed of. Disposal is controlled by EC directive No. 86/278/EEC, which requires that the sludge, unless it is injected or ploughed immediately into the soil, must undergo biological, chemical or heat treatment, long term storage or other appropriate processes. The object of this treatment is to kill off pathogens. While it is also possible to treat these sludges in an anaerobic digester or to incinerate the sludge, both options have high capital and management cost factors. Effluents from conventional treatment installations contain both unstable organic matter and nutrients.

Biological Treatment.

Secondary or biological treatment systems are designed to bring wastewater into contact with Hetrotrophic bacteria which use organic matter as both an energy and carbon source for synthesis. These bacteria are subdivided into three groups:

1 Aerobic bacteria requiring free dissolved oxygen to decompose organic matter to gain energy for growth and reproduction.
2 Anaerobic bacteria oxidise organics in the complete absence of dissolved oxygen by using oxygen bound in other compounds, such as nitrate and sulphate.
3 Facultative bacteria use free dissolved oxygen when available but can also live in its absence by gaining energy from anaerobic reaction.

The development and successful design and operation of biological processes, including Constructed Wetlands, depends on an in-depth understanding of the process involved, at all levels, from biochemistry to ecology. The system must be based upon the fundamentals of microbiology and on the transformations in biological waste treatment. Without this understanding the designs can only be based on input/output analyses of established systems. This may not translate to different wastes and environments.

In biological systems, microorganisms utilise waste to synthesise new cellular material and to supply the energy for synthesis and respiration. Microorganisms can, especially in the absence of external food supplies or exogenous food sources, use internal or endogenous food sources for their respiration.

These principles are the same for all biological systems and can be stated in the form of two equations as follows:

Energy in metabolisable waste + microorganisms --> Energy- New microorganisms + products of metabolisation (1)

Microorganisms --> products of metabolisation + fewer microorganisms (Endogenous respiration) (2)

The rate of these reactions is affected by the ability of microorganisms to assimilate the food, the presence of toxic materials, the temperature and pH of the system and the availability of nutrients, carbon, nitrogen, phosphorus, and trace minerals.

Biological Wastewater treatment processes are defined by the presence (aerobic) or absence of dissolved oxygen (anaerobic), by their photosynthetic ability and by the mobility of the organisms that are either suspended or attached growth. Aerobic microorganisms are found in activated sludge, trickling filters and oxidation ditches and anaerobes predominate in sludge digestion and anaerobic filters. Facultative bacteria are active in both aerobic and anaerobic treatment units. Activated sludge and oxidation ditches are examples of suspended growth systems and trickling filters; rotating biological contactors and Constructed Wetlands are examples of attached growth systems.

The amount of energy biologically available from a given quantity of matter depends on the oxygen source used in metabolism. The greatest amount is available when dissolved oxygen is used and the least amount is derived from anaerobic metabolism. Microorganisms growing in wastewater seek the greatest energy yield to maximise synthesis.

Fresh aerated wastewater placed in a closed container will be decomposed by aerobic and facultative bacteria thus depleting the dissolved oxygen. Aerobic bacteria will then cease to function and facultative bacteria operating anaerobically will first use the oxygen bound in nitrate releasing nitrogen gas. The next most accessible oxygen is available in sulphate by conversion to hydrogen sulphide.

[1] Wastewater Engineering Metcalf & Eddy McGraw Hill International ISBN -007-041690-7

Simultaneously other facultative and anaerobic bacteria decompose organic material to organic acids, alcohol, and other compounds. Methane forming bacteria will then convert the organic acids to methane and carbon dioxide. Protozoa graze on bacteria for energy to reproduce. Some of the new microbial growth dies, releasing cell contents for resynthesis. The oxidation of organic carbon-containing compounds is the mechanism by which heterotopic organisms obtain energy for synthesis. Under anaerobic conditions, organic carbon is converted to microbial solids, carbon dioxide, methane and other reduced compounds.

Ammonia nitrogen is the main soluble nitrogen end product in anaerobic units. The release of ammonia nitrogen to receiving waters creates an oxygen demand. The oxidation of 1 g of ammonia nitrogen to nitrate nitrogen requires 4.57 g of oxygen. The pollution of the environment by Nitrogen is different from other biogenic pollutants such as carbon and phosphorous, because not only can it trigger eutrophication of water bodies and effect the soil and atmosphere, but high nitrate levels in drinking water are toxic and some forms (ammonia, NO_2) can be toxic to various life forms.

Compared to other biogenic inorganic ions the nitrogenous end products of the degradation of inorganic nitrogen except for dinitrogen are very soluble and can therefore reach high concentrations. Ammonia and nitrite are the most toxic to aquatic life concentrations of 0.1 -1.0 mgNH_3 are considered lethal. Nitrite reacts with chlorine and can cause problems in disinfecting facilities.
The transformations can be summarised as follows:

Aerobic:

Organic carbon + O_2 --> $C_5H_7O_2N + CO_2$ + Energy (3)

Organic N --> Ammonium --> Nitrite N --> Nitrate N (4)

NH_3 + Oxygen ----Nitromsonas---> NO_2- + Energy

NO_2- + Oxygen ----Nitrobacter----> NO_3- + Energy (5)

Anaerobic:

Organic carbon --> Microbial cells + Organic acids, aldehydes, + alcohols, etc (6)

Organic acids + oxidized organic carbon---> Microbial cells + CH_4 + CO_2 + energy. (7)

Organics + NO_2 ---> CO_2 + N_2 + energy (8)

Organics + SO_4 ---> CO_2 + H_2S + energy (9)

Autotrophic bacteria oxidise inorganic compounds for energy and use carbon dioxide as a carbon source. Nitrifying bacteria oxidize ammonium nitrogen to nitrate in a two-step reaction. The first step in the nitrification is the oxidation of ammonia to nitrite carried out for instance by various species of the genus Nitromsonas. Two enzymes catalyse two key reactions:

NH_3 + 2H+ + 2e- + O_2 -------> $NH_2OH + H_2O$ (10)
(ammonia monooxygenate)

$NH_2OH + H_2O$ -------> HNO_2+4H^++4e- (11)
(hydroxylamine oxdoreductase)

Only the second reaction provides the energy requirements of the ammonia oxidising bacteria. The second step carried out by members of the Nitrobacter and Nitrospira genre is the oxidation of nitrite to nitrate:

$HNO_2 + H_2O$ --------> HNO_2 + 2H+ + 2e- (12)
(nitrite dehydrogenase)

Oxidation of ammonia to nitrite requires one atom of oxygen derived from O_2 and one atom derived from water. Water is also the source of the oxygen incorporated into nitrate during the oxidation of nitrite.

Other autotrophic bacteria such as sulphur and iron bacteria are also important but are not limiting factors in design or operation.

CONSTRUCTED WETLANDS REMOVAL MECHANISMS.

Constructed Wetlands can be designed to have a variety of processes, habitat conditions and species diversity for the effective biological stabilisation of these effluents and the removal of nutrients. Each one of the processes and removal mechanisms (Table 1) must be separately designed and analysed

BOD Removal

In the BOD reduction phase Constructed Wetlands are secondary or biological treatment systems and the reduction of BOD in is brought about by bring wastewater into contact with Hetrotrophic bacteria which use organic mater as both an energy and carbon source for synthesis. All three groups of these bacteria, Aerobic, Anaerobic, and Facultative exist in Constructed Wetlands.

In Constructed Wetlands wastewater is brought into contact with microbial growths which adhere to the surface of a supporting medium. The supporting medium can be the substrata and submerged plant roots and/or the dense plant growth or litterfall in the water column.

The wetland plants in Constructed Wetlands grow and die down each year and the litterfall or detritus from the plants forms a layer on the soil surface and in the water column. This layer has a large surface area per cubic meter and create a system to which microorganisms adhere. Wastewater flows through this layer and air can enter the water from the exposed surface. The slow flow of the wastewater through the plant growth litterfall to which the microbial growth is attached forms part of the thin layer bio-reactor on which the organic loading rate is based.

Some or all of the water flow is through the substrata and the wetland plant roots. The substrata and the plant root hairs also provide a surface to which microorganisms become attached and forms the other part of the thin layer bio-reactor.

If organic matter i.e. carbonaceous waste, BOD, or COD is added at a constant rate to a continuous flow biological treatment unit, the unit will eventually reach equilibrium conditions. Until equilibrium conditions occur, the microorganisms will respond to the waste addition and synthesise new organisms until the microbial mass is in equilibrium with the available food supply. At equilibrium the net microbial concentration in contact with the waste is related to the available substrate and to the decay rate or the endogenous respiration rate of the organisms. This basic relationship can be expressed as follows:

Net microbial growth per unit time = (waste utilised per unit time) x (microbial cell yield coefficient) x (organism decay coefficient) x (microbial mass in the system) (13)

The theoretical expression for reaction rates can be written as the decrease in concentration of the reactant (A_c), C_A/dt and is found to be dependent on the product of concentration terms.

The order of the reaction is defined as the sum of the exponents of the concentration terms in this rate law. The order of a reaction need not be a whole number and is determined solely by the best fit of a rate equation with the empirical data. Experience with organic wastes has indicated that the change in the BOD of the waste is characterised by a first order equation:

$$dC/dt = -\kappa C. \tag{14}$$

For BOD removal in a Constructed Wetlands this equation is written as:

$$Q_t(C_t-C_b) = Q_o(C_e.C_b)e^{-\kappa_t R_t} \tag{15}$$

Where:

Q_O = Flow m^3/day at time $R_t = 0$ (Inlet)

Q_t =Flow m^3/day at time $R_t = t$ (days)

C_O =Concentration at time t = 0 (Influent BOD mg/l)

C_t =Concentration remaining at time t.

C_e =Concentration at outlet. (Effluent BOD mg/l)

C_b =Background Concentration. (BOD mg/l)

κ_t =Rate constant at temperature T. (1/day)

$\kappa_t = \kappa_{20}(\theta)^{(T-20)}$ (16)

θ = Temperature coefficient. Range 1.05 to 1.08

f_t =Temperature factor = $(\theta)^{(T-20)}$

R_t = Hydraulic Residence Time days.

Q_{pe} =Flow m^3/day per person equivalent = 0.2 m^3

Q_a = Average Flow m^3/day= $(Q_O-Q_t)/2$

Equation (16) is a standard biological equation for correlating reaction rates calculated at 20°.C to those experienced at other temperatures. It is generally estimated that biological reaction rates double for every 10°. rise in temperature.
The water balance of a constructed wetlands is:

$$Qt_2 = Qt_1 + Qp - Qev - Qg - Qm - Qd \tag{17}$$

Qt_1 = Flow m^3/day at time t_1

Qt_2 = Flow m^3/day at time t_2.

Qp = Precipitation Flow m^3/day.

Qg = Leakage to ground water m^3/day.

Qev = Evaportranspiration m^3/day.

Qm = Increase in Soil and biomass storage m^3/day.

Qd = Increase in level and surface storage m^3/day.

To establish a constructed wetlands material balance, all of the above flows as well as the BOD and residence time in the wetlands must be measured. Neither the evapotranspiration, nor ground water leakage, nor the variation in water storage can be directly measured in an operating system. The most common instruments employed for flow measurement in Constructed Wetland systems is a Vee notch weir, the accuracy of which in a field installation is plus or minus 15%. Furthermore the integrated or average hourly flow rate obtained from such flowmeters is too inaccurate to be employed for mass balance calculations due to the inaccuracy at low flow rates.

A review of the literature shows that data collected from Constructed Wetlands is not sufficiently accurate or consistent to determine the reaction rate. As Constructed Wetlands are designed with residence times in excess of 4 days for BOD reduction a system mass balance should be conducted over a period of not less than 8 days. Furthermore such variables as the surface area for microbial attachment, the water column depth, and climatic conditions are rarely if ever measured. Another problem that arises in obtaining an accurate mass balance is the repeatability of BOD measurements at low BOD levels and the natural background BOD level of the wetland.

The reaction rate is generally approximated by assuming $Q_O = Q_t$ and rewriting the decay equation as :

$$C_t - C_b = (C_o - C_b)e^{-\kappa_t R} t \quad \text{or} \quad [\ln(C_o - C_b) - \ln(C_t - C_b)] = \kappa_t R_t \tag{18}$$

The reaction rates quoted in the literature have a larger spread. This is due to:

a) Variations in the inherent reaction rate of the different systems due to variations in contact surface area for attached microbial growth.
b) The actual residence time as opposed to the calculated residence time.
c) The fact that the temperature dependence of the reaction rate is generally not taken into account in evaluating operating systems.

The effectiveness and capacity of a wetland system to remove or retain contaminants is a function of the reaction rate, the residence time and the temperature of the system. The reaction rate (κ) is a function of biological activity and this is dependent both on the specific surface area available for attached microbial growth and on the state of subdivision of the reactants and the degree of contact between the microorganisms and their source of energy in the wastewater. Tests have shown for example that the reaction rate increases if the wastewater is passed through fine gravel. This is because the greater surface area available to anchor the microorganisms and the greater subdivision of the wastewater allows more rapid reaction than in coarse gravel. The above model assumes that BOD reduction is a first order biological process only and no allowance is made for sedimentation, filtration, absorption or other physical or chemical processes which may have a different order of reaction.

For 95% BOD removal the residence time in constructed wetland wastewater treatment systems is reported at between 4 and 6 days depending on the system, while at the same time the reaction rate constant is reported as been between 0.6 and 1.1. In view of this confusion it is prudent to design the system based on the lowest quoted reaction rate of i.e. 0.5. This corresponds with a half-life for BOD removal of 1.4 days. Furthermore in designing wastewater treatment systems it is important to allow for increases in loading rates and so a half-life at 10°C of 2 days for a Constructed Wetland treatment system is used. The reaction rate constant κ_t at an ambient temperature of 10°C corresponding with this half life may be calculated as follows: $\kappa_t R_{t1/2} = \ln 2 = 0.693$, $\kappa_t = 0.693/2 = 0.356$. This figure is based on the assumption that the system is operated in such a way as to achieve an adequate specific surface area for attached microbial growth at all times.

Residence Time and Reaction Rate.

To evaluate or design a Constructed Wetlands treatment system we must know both the residence time and the reaction rate constant. They are interdependent and both depend on the design and operation of the system as will be shown below. The reaction rate is a function of the specific surface area for attached microbial growth, which in turn is a function of the depth of flow and the flow pattern through the system as well as the properties of the medium. For a given Constructed Wetlands treatment area and wastewater flow the residence time also depends on the depth of flow and the flow pattern through the system.

Designing on the basis of residence time or hydraulic loading rate alone is unsafe; the variations of the reaction rate with depth must also be taken into account. The available surface area for attached microbial growth increases with increasing depth of water column until the dense layer of litterfall and surface vegetation is covered and decreases thereafter. The maximum surface area for attached microbial growth in a mature marsh type wetland occurs at a depths between 200 and 300mm depending on season.

Residence time, an important factor in the performance of a Constructed Wetlands, is a function of the system hydraulics. In designing Constructed Wetlands the calculation of the residence time is generally based on the assumption:

a) That plug flow (isotropic and homogeneous flow) exists
b) That the various hydraulic flows other than the wastewater flow can be neglected
c) That the design depth of surface flow wetlands is 250mm or less.

The residence time in constructed wetlands may be approximated by assuming that Darcy's law applies and where:-

W = Width of Constructed Wetland (m)

d = Water plug flow depth (m)

L = Length of Constructed Wetland. (m)

H = Hydraulic head (m)

k = Hydraulic Conductivity of the medium (m/day)

s = Hydraulic Slope H/L

s_b = Gradient of the Constructed Wetlands bed base.

η = Porosity of Constructed Wetland bed.

η = Hydraulic Volume/Total Volume of bed..

The plug flow through a homogeneous and isotropic Constructed Wetland bed is given by:

$$Q = (W)(d)(k)dH/dL \qquad (19)$$

$$Qa = (W)(d)(k)s \qquad (20)$$

The residence time in a Constructed Wetlands bed is: -

$$R_t = \text{Hydraulic Plug Flow Volume/Flow} \qquad (21)$$

$$R_t = (L)(W)(d)n/Qa \qquad (22)$$

$$R_t = (L)(W)(d)(n)/(W)(d)(k)(s) = L(n)/(k)s$$

$$L = R_t x k x s/n \qquad (23)$$

Hydraulic conductivity (k) is a function of both the medium and fluid properties. Porosity (η) is function of the grain size in the case of Gravel Beds and of the vegetation density in the case of surface flow Constructed Wetlands. The following table[2] gives typical values of these properties.

	k m/day	Porosity η
Clay	0.03	0.45

[2] Hydrology Rafael L. Bras Addison-Wesley ISBN 0-201-05922-3

Silty Loam	0.3	0.36
Sandy Loam	3.0	0.25
Fine Gravel	15	0.22
Medium Gravel	50	0.20
Gravel	100	0.18
Surface Vegetation	5000	0.85
Dense Vegetation	10000	0.85
Floating Vegetation	20000	0.90

Taking into account the increase in the reaction rate with the available area for attached microbial growth the following graph has been prepared showing the approximate width per pe and the area per pe for the various substrata. As this graph and the following table show it is only possible to achieve a balance between bed slope, hydraulic conductivity and porosity for gravel bed systems for gravel with a Hydraulic Conductivity of 80 to 120 m/day. Furthermore, the removal rate constant is a function of the surface area for attached microbial growth, which for gravel based systems, is a function of the porosity and size of the gravel. For increasing gravel size the hydraulic conductivity and porosity increase and the surface area for attached microbial growth decreases, hence the removal rate constant (κ_t) decreases.

For example a gravel bed system with a hydraulic conductivity (k) = 80m/day, a Porosity (η) = 0.2 and designed to treat 100 pe at 200l/pe with a residence time R_t = 3 days.

$s = s_b = 0.01$ and from (31) $d = D = 0.6$

$W = Qa/(0.6)(80)(0.01) = Qax2 = 20x2 = 40m$ (24)

$L = 3(k)(0.01)/(n) = 3x80x0.01/0.2 = 12m$ (25)

In surface flow systems the hydraulic conductivity of the bed generally less than 3.0 m/day and the wastewater flow is mostly through the dense vegetation and plant detritus on the surface of the bed. The bed slope sb is designed to be zero and the hydraulic conductivity of the medium k is greater than 10,000 m/day. The depth is adjusted to maintain dense vegetation and plant detritus in the water column. Increasing the depth will increase the residence time but reduce the density of the material in the water column and hence the specific surface area for attached microbial growth and therefore the reaction rate. The outlet depth required to achieve these conditions is between 200 mm. and 300 mm and depends both on the season and on number of years of plant growth.

$L = R_t(k)s/\eta = Rt(5000)H/L(0.85)$ for $H = 25mm$

$L^2 = R_t(125)/(0.87)$, $L = 12\sqrt{(Rt)}$

The plug flow depth of a surface flow system is equal to the adjusted outlet depth and therefore ;-

$d = 200\ mm = 0.2$

$W = Qa/(0.2)(5000)(s) = Qa(L)/(0.2)(5000)(H)$ and for H= 25 (26)

$W = Qa(L)/25$ (27)

$LxW = Qa\ R_t(125)/(22) = Qa\ R_t x5.7$ (28)

For Qa=Qpe=0.2m3 and (κ_{20}) = 0.7 and C_b =0 we can calculate the Constructed Wetland residence time :-

$R_t = [\ln(C_o-C_b)-\ln(C_t-C_b)]\ /\kappa_t = (\ln Co-\ln Ce)/0.7xf_t$ At an operating temperature of 10^oC $f_t = 0.55$

An inlet BOD Co = 120mg/l and an outlet BOD Ce = 20mg/l equation and C_b = 2mg/l results in a residence time :-

$Rt = =[\ln(C_o-C_b)-\ln(C_t-C_b)])/0.7x0.55 = 1.88/0.45 = 4$ days

Constructed Wetlands Area per pe = $LxW = Qpe(4)(5.8) = 0.23x232 = 5.2\ m^2/pe$ (29)

Sub-surface Flow Systems [3]

Sub-surface flow systems are designed to have wastewater flow under the surface and through the bed at all times. In order to achieve sub-surface flow the hydraulic conductivity of the bed, the length and gradient of the Constructed Wetlands bed are all selected and designed for the designed flow. To

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[3] European Design and Operations Guidelines for Reed Bed Treatment Systems P. F. Cooper (water research Centre, Swindon U.K

optimise the performance and utilisation of the bed in a Sub-surface Flow Systems or gravel bed system the hydraulic slope should equal the bed slope. For gravel bed system the bed slope is generally designed to be less than or equal to 1% and the bed depth to be 600mm

In sub-surface flow systems where the gradient of the Constructed Wetlands bed s_b is equal to or less than the hydraulic slope (s) the hydraulic depth is:-

$$d = D + L(s_b) - L(s) = D - L(s-s_b) \quad (30)$$

$$d = D - R_t(k)(s)(s-s_b)/\eta \quad (31$$

$$W = Qa/(d)(k)(s) \quad (32)$$

$$LxW = QaxR_t/(d)(\eta) \quad (33)$$

For an increased flow or a decreased conductivity the required hydraulic head is greater than the bed slope so that the increased waste water will flow over the surface of the bed. The extent of this surface flow can be adjusted by decreasing the level at the outlet so that the flow for the first section of the bed will flow over the surface until a point at which the flow, head and hydraulic resistance are in equilibrium (see Fig 9b). Using the previous example but with the flow increased by 10% and the discharge level adjusted to obtain some sub-surface flow the flow depth (d) will be less than the bed depth (D).

If $s_b = 0.01$ and $s = 0.0125$ for example, then for the above example $d = Qa/(W)(k)(s)$
$= 100x0.2/40x80x0.0125 = 0.5$ and length of sub surface bed = $(0.6-0.5)/0.0125 = 8$m.

If the required hydraulic head is less than the bed slope then the waste water will flow down through the gravel bed until the flow and flow depth are in equilibrium with the remaining hydraulic resistance. The total equivalent reaction bed depth will be determined by the outlet level and the degree of mixing with that section of the bed under the flow level. Any variation from design flow or in the Hydraulic Conductivity of the gravel due to sedimentation with use or stratification in the selection or laying of the gravel will cause a variation in the residence time and hence the performance of the system.

Suspended Solids Removal.

Wetland vegetation influences water movement through the substrata, and along with the substrata causes significant reduction in suspended solids. The removal of Suspended Solids occurs simultaneously with the BOD removal and as it is not normally the limiting factor is not taken into account in the design or modelling of Constructed Wetlands. The removal of suspended solids is by gravity sedimentation and filtration in the dense plant growth and litterfall in the water column and or in the substrata through which the water flows.

Wastewater velocities through constructed wetlands are typically between 2m/day and 2m/hr. This low velocity causes settling and significant reduction in suspended solids. The removal of suspended solids occurs simultaneously with the BOD removal and as it is not normally the limiting factor is not taken into account in the sizing or modelling Constructed Wetlands. The removal of suspended solids is by gravity sedimentation and some adsorption on the dense plant growth and litterfall in the water column and or in the substrata through which the water flows. The introduction of a pond for nitrogen removal will result in the production of algae and plankton that will increase the TSS in the water. A final marsh is required to settle the suspended solids formed in the pond

Nitrification/Denitrification.

Nitrogen is an essential constituent of all plant and animals principally in proteins and nucleic acids. Ammonia is produced when any nitrogen containing organic material decomposes in the absence of air. Organic nitrogen is transformed through ammonification, nitrification and dentrification to N_2. Nitrification is the breakdown of ammonia to form NO_3 and NO_2 by nitrifying bacteria. The resulting NO_3/NO_2 may be further broken down (denitrified) to release N as a gas. A combination of aerobic and anaerobic conditions is required to achieve an advanced level of biological organic waste treatment. [4]Case studies have shown that nitrification and denitrification can be achieved by combining shallow water emergent vegetation with deep-water submergent vegetation areas within Constructed Wetlands.

Nitrification can occur under the conditions of low organic BOD loading and aerobic conditions. The oxygen demand for nitrification is 4.3g oxygen per 1g of ammonia and reaction rates are low at temperature less than $1O^0$ C and at pH less than 6.0 pH. After the BOD in wastewater is reduced suitable conditions for nitrification could be created in a pond planted with oxygenating plants.

[4]

Varying depths in the pond can create anaerobic site and together with a carbon source in the plant litter and benthic layer provide sites for denitrification, which occurs when oxygen supply is low. Fig. 3 shows the nitrogen cycle in wetlands.

Ammonification transformations can be faster than nitrification and ammonia will thus initially increase along the wetland. Ammonification rates depend on temperature and pH Nitrification can occur under the conditions of low organic BOD loading and aerobic conditions. The oxygen demand for nitrification is about 4g of oxygen per gram of ammonia. Nitrification rates are low at temperatures less than 5 ° C and at a pH less than 6.0 pH. After the BOD in wastewater is reduced suitable conditions for nitrification could be created in a pond planted with oxygenating plants. Varying depths in the pond can create anaerobic sites and together with a carbon source in the plant litter and benthic layer provide sites for denitrification, which occurs when oxygen supply is low. Comparative studies on the performance of three higher aquatic plant types *Scirpus Validus*, *Phragmites Australis* and *Typha Latifolia* in the removal of nitrogen via sequential Nitrification-Denitrification, BOD and TSS from primary municipal wastewater have shown a 94% to 80% reduction in mean ammonia concentration with mean BOD effluent levels of 5.3 and 22.2mg/l for *Scirpus Validus* and *Phragmites Australis* respectively. CW – 001 shows the nitrogen cycle in wetlands. To achieve good nitrogen removal or evaluate the performance of a Constructed Wetlands for nitrogen removal the Wetlands must be specifically designed for this purpose.

The plug flow mass balance TN equation for a surface flow marsh wetland is given by:[5]

$$Axk_{Tn} = Q\ln[(C_{TNi} - C_{TNb})/(C_{Tno} - C_{TNb})] \quad (34)$$

Where:

A	= Surface flow marsh area m^2
Q	= Flow m^3/year
C_{TNi}	= Initial TN concentration mg/l
C_{TNb}	= Background TN concentration mg/l
C_{TNo}	= Final TN concentration mg/l
k_T	=Area-based, first order total nitrogen rate constant m/yr

To achieve good nitrogen removal or evaluate the performance of a Constructed Wetlands for nitrogen removal the Wetlands must be specifically designed for this purpose.[6]

Phosphorus Removal.

Phosphorus is present in all sanitary and farm wastes. With short rivers fast rivers, Phosphorus is not usually regarded as a dangerous water pollutant. If however, the watershed drains to inland lakes or reservoirs, phosphate levels steadily rise as small amounts of leached phosphate accumulate and can be of great concern over eutrophication of the water. Phosphorus is removed in natural systems by vegetation uptake, adsorption, complexation, and precipitation.

The annual die back of the vegetation returns the phosphorus to the water column and to the bed of humus being formed by the decaying vegetation where some phosphorus is bound into the humus and some is recycled. To permanently bind phosphorus it must be either precipitated or immobilised into the substrata or humus. This can be achieved by passing the wastewater through a bed of gravel rich in Aluminium, Iron or Calcium where the PO4 anions and combines with the metal cations to form an insoluble precipitate.

Pathogen Removal

Pathogenic organisms are present in both wastewater and sludges. The control of pathogens is a perquisite of wastewater treatment. The removal of pathogens in natural systems is due to die off, sedimentation and adsorption. Parasitic cysts and eggs will settle to the bottom in the quiescent zone of ponds. Numerous studies have shown that the removal of faecal coliform and enrtric virus is dependant on residence time and temperature and that the various natural treatment systems are very effective in the removal of pathogens. With hydraulic residence times of 5 to 6 days Constructed Wetlands are capable of removing bacterial and viral indicators of pollution at efficiencies of 99%. Chlorination or other disinfection means will further reduce virus levels in wetland effluents to below 0.1 (pfu) plaque forming units per litre.

SUMMARY

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[5] Treatment Wetlands Robert H. Kadlec, Robert L. Knight (Chapter 13) ISBN 0-87371-930-1

[6] Designing Constructed Wetlands for Nitrogen Removal Donald A. Hammer, Robert L. Knight

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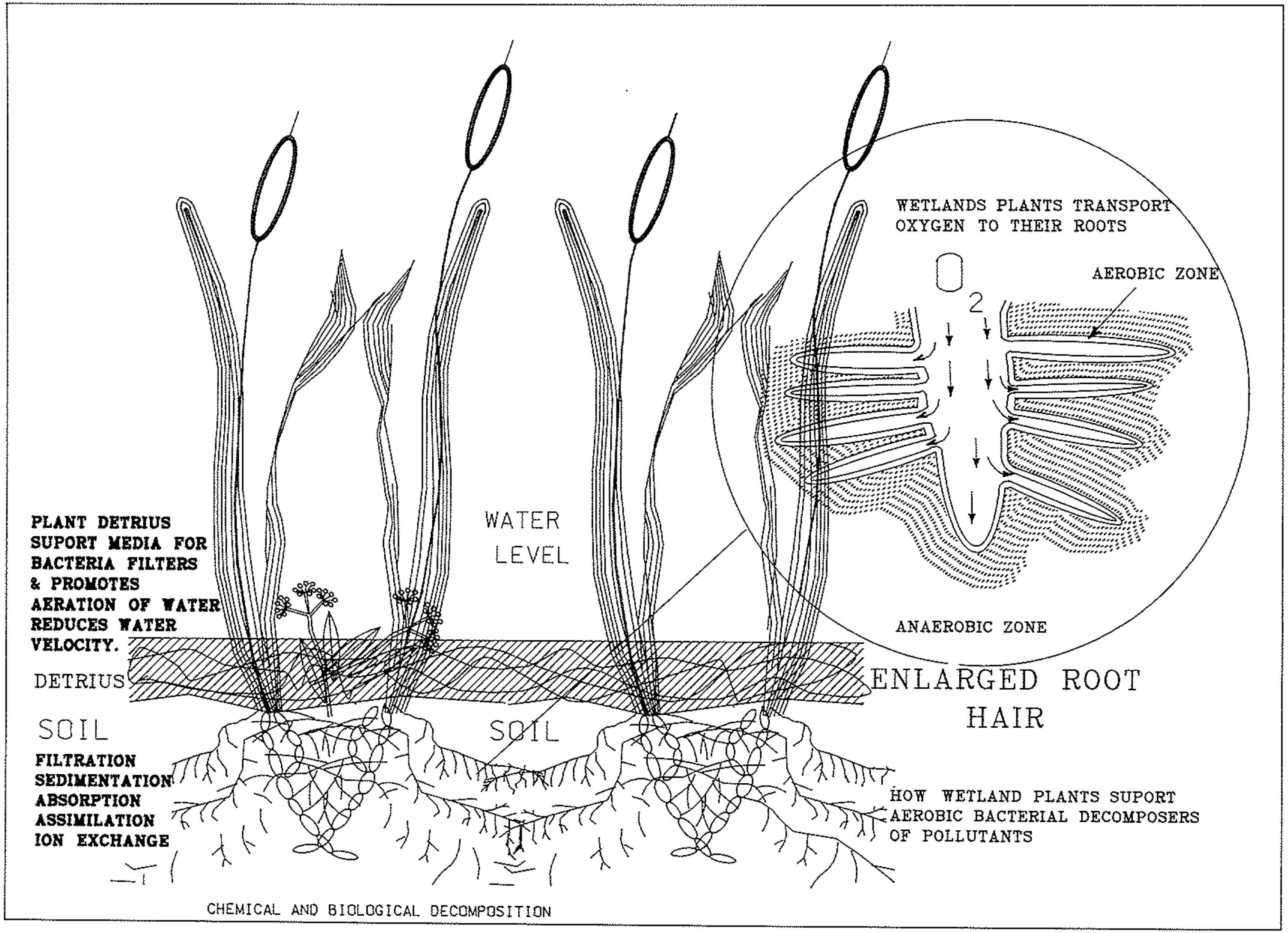
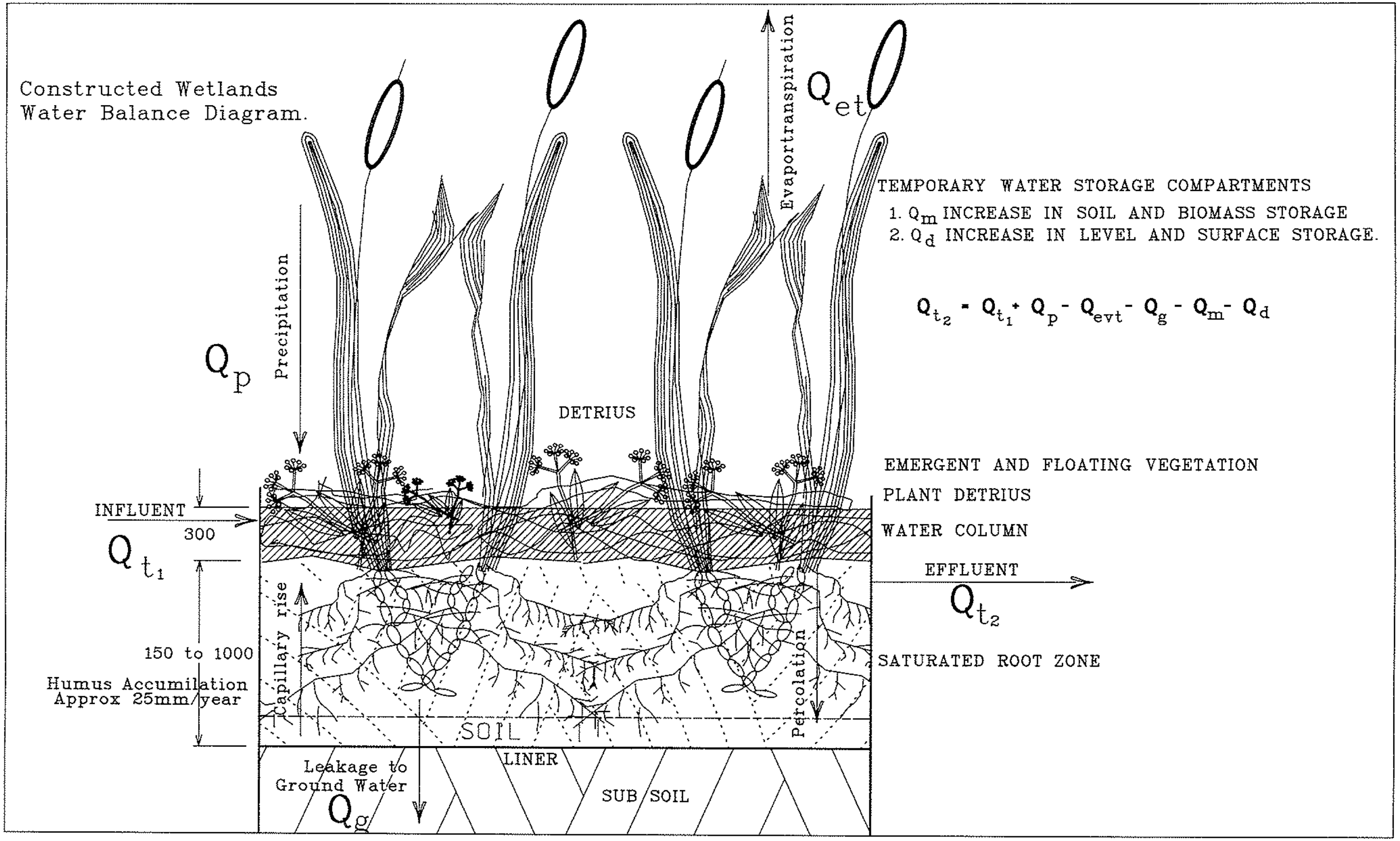
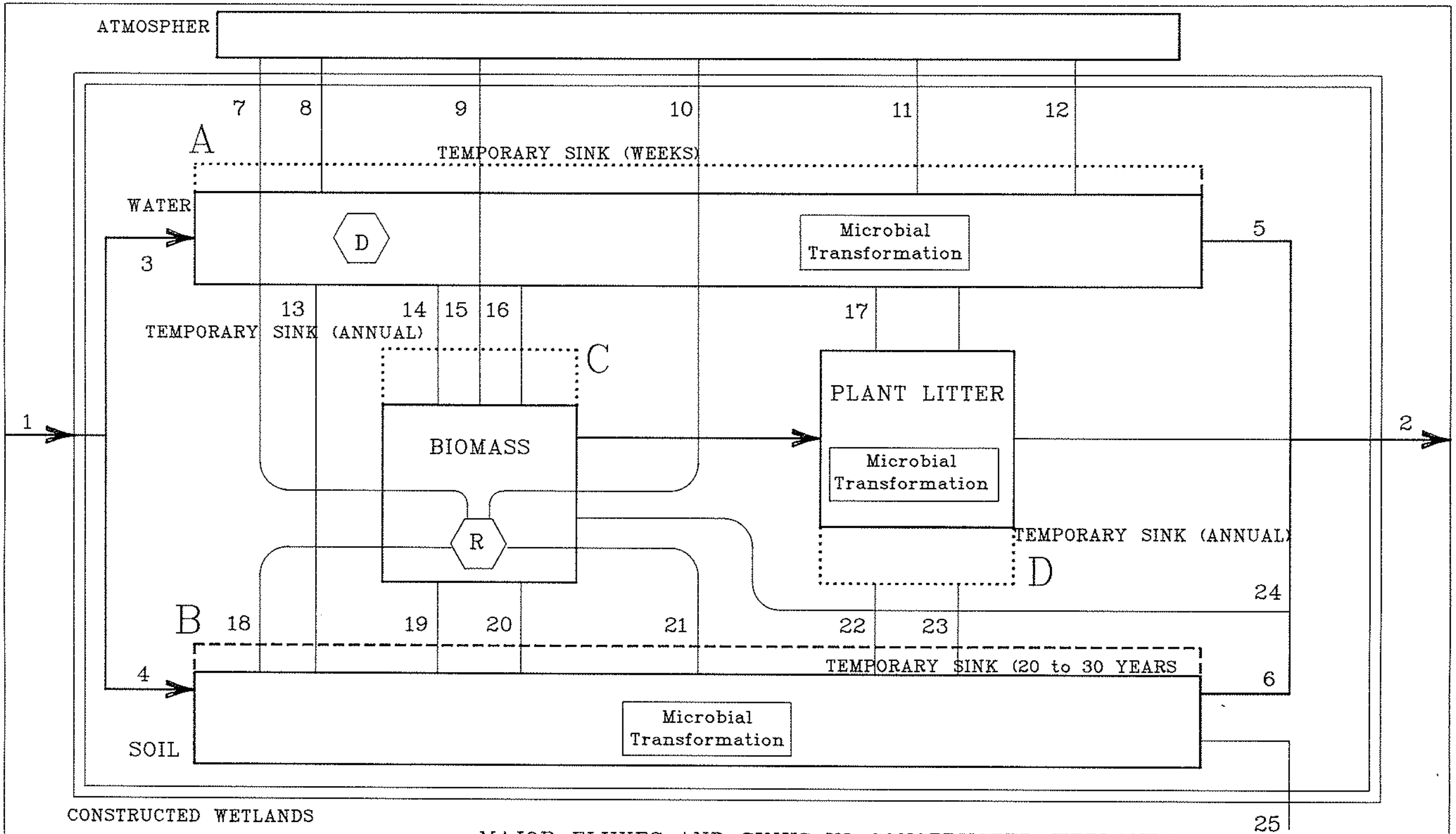
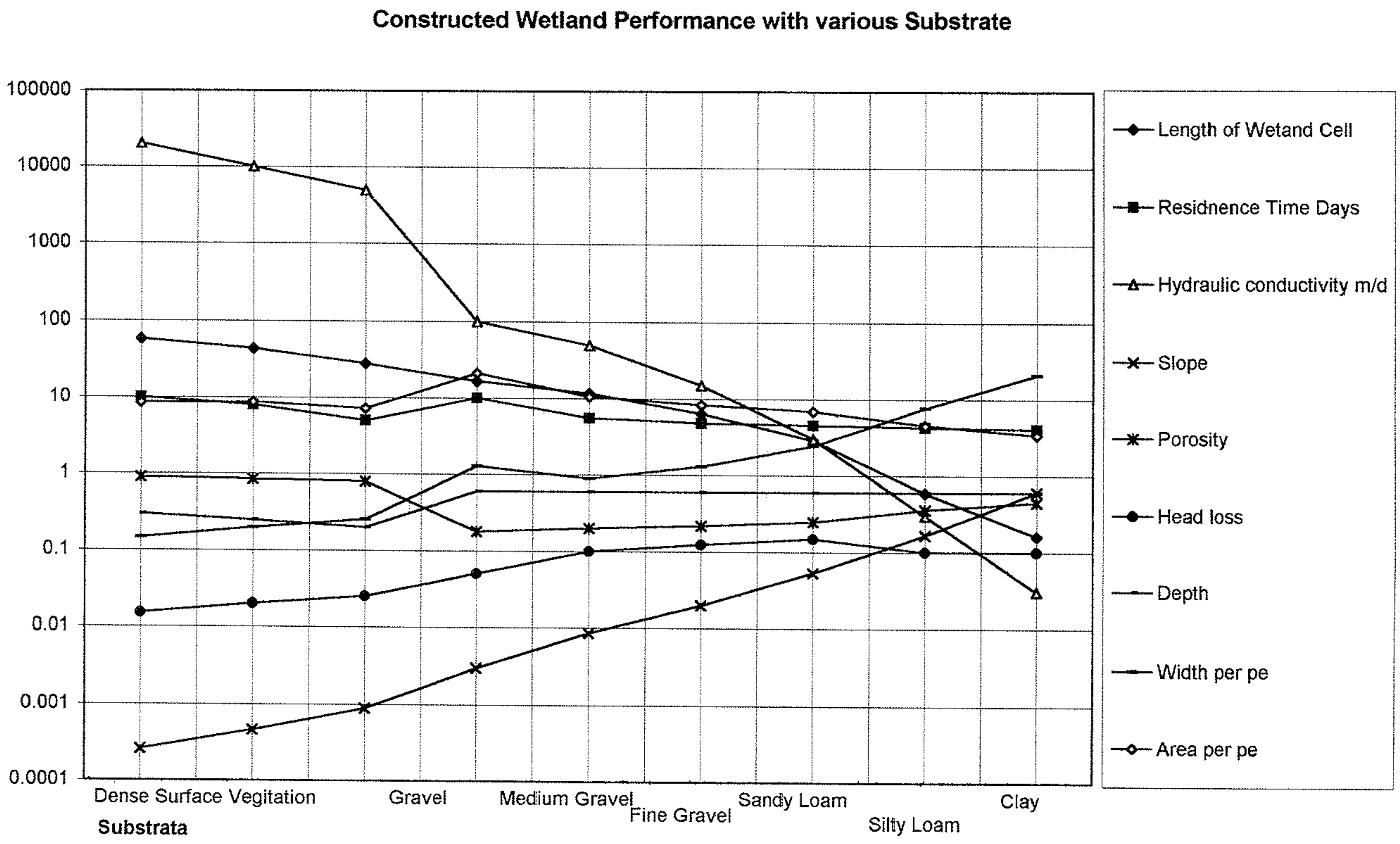



Constructed Wetlands
Water Balance Diagram.
Evaportranspiration
Q_{et}
TEMPORARY WATER STORAGE COMPARTMENTS
1. Q_m INCREASE IN SOIL AND BIOMASS STORAGE
2. Q_d INCREASE IN LEVEL AND SURFACE STORAGE.
$Q_{t_2} = Q_{t_1} + Q_p - Q_{evt} - Q_g - Q_m - Q_d$
Precipitation
Q_p
DETRIUS
EMERGENT AND FLOATING VEGETATION
PLANT DETRIUS
WATER COLUMN
INFLUENT
300
Q_{t_1}
EFFLUENT
Q_{t_2}
Capillary rise
150 to 1000
Humus Accumilation
Approx 25mm/year
Percolation
SATURATED ROOT ZONE
SOIL
LINER
Leakage to
Ground Water
Q_g
SUB SOIL



MAJOR FLUXES AND SINKS IN CONSTRUCTED WETLANDS


Constructed Wetland Performance with various Substrate
100000
10000
1000
100
10
1
0.1
0.01
0.001
0.0001
Dense Surface Vegitation
Gravel
Medium Gravel
Fine Gravel
Sandy Loam
Silty Loam
Clay
Substrata
Length of Wetand Cell
Residnence Time Days
Hydraulic conductivity m/d
Slope
Porosity
Head loss
Depth
Width per pe
Area per pe

Paper No. 12.

Septic Tank Systems: Recent Advances.

Hubert Henry, Bord na Mona.

SEPTIC TANK TREATMENT SYSTEMS - RECENT ADVANCES

DR. HUBERT HENRY, BORD NA MÓNA, ENVIRONMENTAL DIVISION

ABSTRACT

Septic tank systems have been widely used in both developed and developing countries for the treatment of domestic wastewater in rural areas for over 100 years. In recent decades they have become increasingly popular in suburban areas not serviced by public sewer systems. The widespread use of the septic tank system has continued in the face of a consistent history of failure, with severe localised groundwater pollution, and almost unanimous disapproval by researchers in the field. The feasibility of using septic tank systems as a method of treating domestic wastewater was being questioned as early as 1956 when Kiker suggested that 'at best, a septic tank is a poor substitute for centralised sewage collection and should be avoided whenever possible'.

As a primary treatment system septic tanks do not significantly reduce the polluting potential of the wastewater. The bulk of the treatment takes place in the soil through various physical, chemical and biological interactions between the effluent and soil colloids. In the United States approximately three billion m^3 of septic tank effluent is discharged into the soils for treatment annually (Bitton & Gerba, 1984). However, less than 50% of these soils are through to be capable of achieving an adequate reduction in the pollution potential of waste. In Ireland there are an estimated 350,000 septic tank systems serving a population in the region of 1.2 million people (1991 census). Again only half of these soils are considered capable of providing sufficient treatment to prevent groundwater pollution.

In the past number of years there have been a variety of significant developments in the area of onsite wastewater units which has resulted in a welcome advancement in our knowledge of the workings and failings of such systems in addition to providing better site evaluation techniques and proprietary treatment technologies. This paper summarises the, by now, well accepted problems associated with conventional systems and outlines a number of recent advances which will greatly to improve the current situation.

SEPTIC TANKS - EFFICIENCY OF TREATMENT

A septic tank functions primarily as a settlement chamber and as such only affords limited digestion of the wastewater. The efficiency of treatment within the tank depends on many factors, primarily the design, construction and maintenance of the system. The volume and nature of the waste is also important.

In general, approximately 50% of the solids will be removed but this can increase to 70%, in a well constructed two chamber tank. BOD removal within the tank is considerably less, ranging from 15 to 30%, although this can also be extremely variable.

The effluent from a septic tank is of poor quality and highly polluting if it reaches surface or groundwaters. The effluent contains high numbers of faecal bacteria and viruses and large amounts of phosphorous and nitrogen (mainly as ammonia), as well as having a high BOD and SS content. It is a common misconception that the tank will effectively remove the bacteria and other micro-organisms contained in the waste. Studies have shown that the removal of these organisms within the tank is negligible. Even the most efficient tank can only offer partial treatment, hence the physical, chemical and biological quality of the effluent is such that it cannot be discharged directly to surface or groundwaters without further treatment. This treatment takes place in the solid treatment system into which the effluent is channelled on leaving the tank.

THE SOIL TREATMENT SYSTEM

The soil is an integral part of the process by which the effluent strength is reduced before reaching the saturated zone. Once the effluent leaves the septic tank it enters the soil treatment system which it interacts with the soil colloids. There are two types of solid treatment systems commonly in use:

i. Soakage pits
ii. Distribution fields (also called percolation, tile or absorption fields)

The first system simply allows the effluent to flow into an excavated hole filled with stone or rubble. The main disadvantage of this is that the effluent is concentrated into a small area which may become clogged and quickly fail. The use of soakage pits as a disposal option is not recommended.

Absorption fields are designed to event distribute the effluent through a large volume of soil via perforations in a pipe distribution network. The soil's ability to effectively treat the waste depends on the design, configuration and loading of the pipe distribution network, maintenance of the tank, and the constituents of the waste (in addition to the soil characteristics).

The extent to which attenuation of the effluent takes place in the regolith (soil and overburden) depends on the ion exchange capacity, the porosity, permeability and texture of the regolith, its thickness beneath the site, the dept of the water table and the slope of the ground surface.

FAILURE OF THE SOIL TREATMENT SYSTEM

Not all soils are capable of effectively treating septic tank effluent. More than half the soils in the United States are unsuitable for septic tank systems with respect to percolation rate. Failure of the systems has been reported to be between 25 and 50%. It is estimated that half of these failures can be attributed to the location of absorption fields in soils of low permeability, a characteristic in soils of over 50% of Irish soils. Another major reason for failure is location in an area with a high water table. This can cause ponding of the effluent on the surface with resulting health hazards. In addition, failure can occur if the density of septic tank systems in the area is too high, causing the soil to be overloaded.

Failure can also occur in a septic tank system situated in a soil with high permeability. Although, it is unlikely to become clogged, severe groundwater pollution can occur by the rapid passage of wastewater through the unsaturated zone without sufficient contact time with the soil for treatment.

Septic tank systems are the most frequently reported source of groundwater contamination. Many public health workers feel that the most critical effect of septic tank systems is the contamination of private water wells. The human health implications of such contamination are considerable. Outbreaks of typhoid fever, infectious hepatitis, gastrointestinal infections and infantile methaemoglobinemial have all been linked to malfunctioned septic tanks. Almost half the reported water disease outbreaks in the US every year are due to the consumption of contaminated groundwater. Overflow from septic tanks was responsible for 42% of the reported outbreaks of disease.

Pollution of groundwater by septic tank effluent can be chemical or biological in nature, or both. The poor microbiological quality of domestic well water supplies has been well documented. A study of rural groundwater sources in the US showed 92% to be contaminated with coliform bacteria (Bitton & Gerba, 1984), while a similar study in western Ireland found that 68% of all rural groundwater supplies contained faecal coliforms, faecal Streptococci or both. Septic tank effluent was believed to be the main source of contamination in both cases.

Inundation of soil disposal systems with primary wastewaters results in the formation of a clogged zone or biomat at the wastewater/soil interface. The mat is formed by three distinct processes:

(i)	Physical	-	Where solids in the effluent clog soil pores
(ii)	Chemical	-	Where soil colloids swell as a result of chemical processes
(iii)	Biological	-	Where bacterial biomass or microbial breakdown products reduce pore size.

The biomat has many implications for the degree and effectiveness of effluent attenuation. It plays an important role in reducing the faecal bacterial numbers in the percolating effluent and has also been reported to remove other contaminants in the percolating effluent by various sorption reactions. However, if the biomat reaches a stage where the infiltration rate of the soil is seriously reduced and the effluent can no longer percolate into ground, the system will quickly fail. The immediate results of this failure is often the saturation of surface soil with raw sewage which may move by overland flow to contaminate surface waters. There is some evidence that the "resting" of a disposal field which has become clogged can result in the breakdown of the biomat with a subsequent return to the soils original permeability. It is for this reason that most guidelines recommend the installation of a reserve percolation area.

Because the vast majority of the physical, chemical and biological renovation or attenuation of the raw wastewater occurs within proprietary treatment units themselves there is no formation of biomat (secondary biomat) and as such the potential for failure in the soil treatment unit has been greatly reduced.

RECENT ADVANCES

The area of onsite sewage treatment has in recent times been the focus of significant scientific and engineering research. These efforts has resulted in a number of significant developments/advances. These are outlined below under the following headings:

(i) Onsite treatment systems
(ii) Site evaluations
(iii) Contaminant tracing

ONSITE TREATMENT SYSTEMS

Much of the scientific research into the performance and operation of onsite treatment systems has been carried out in the US. This is not surprising as the US has the largest 'unsewered' population in the developed world. It is estimated that less than 75% of the total population in the US are connected to mains sewers compared to >95% in Germany, Britain, and much of the EU. The relative importance of subsurface disposal technology in the rural/suburban US economy has resulted in much published material by eminent scientists and engineers in addition to numerous detailed and informative text books on the subject.

Similarly, in Europe there has been a renewed focus of attention on the application of established effluent treatment technologies to smaller populations. The technologies currently available can be broadly categorised into the following groups:

- Suspended growth aeration systems
- Fixed film aeration systems
- Rotating biological contactors
- Biological filters
- Constructed wetlands

Research has conclusively shown that when any of the above units are designed, operated and maintained correctly they can greatly reduce the pollutant load of primary wastewaters with an associated improvement in the problems encountered in failing septic tank systems. In addition, it is now possible to permit development in areas which hitherto were excluded due to their inherent unsuitability for onsite sewage disposal.

It is important to note that the utilisation of available treatment technologies does not present the solution to all onsite problems. This is particularly true in cases where treated effluent can not be discharged to surface receiving waters and must be disposed to the subsurface. In such cases, it must be accepted that some sites are simply unsuitable for onsite sewage disposal due to any of a range of factors including soil restrictions, ecological sensitivity, groundwater vulnerability, etc. In certain instances treatment technologies can be modified to incorporate tertiary or advanced features such as bacterial reduction, phosphate removal, denitrification, etc.

In order to safeguard the future development of the onsite treatment technologies business sector and at the same time maximise the environmental benefits of their use, it is imperative that an integrated site evaluation/assessment program is developed in conjunction with the proprietary technologies. This program must recognise the multi-disciplinary nature of the onsite sewage treatment/disposal issue and, if possible, include a performance based certification scheme for appropriate treatment technologies. Efforts to harmonise and certify proprietary technologies is currently underway at European (CEN) and National (Agrèment) levels.

SITE EVALUATIONS

Recent developments in Ireland will without doubt result in a significant improvement in the evaluation of site suitable for the treatment and disposal of onsite wastewaters. Following on from a comprehensive study on small scale wastewater treatment systems, co-ordinated by staff from the University of Galway, the Environmental Protection Agency are proposing to issue a set of guidelines on site selection and the use of appropriate treatment technologies. In addition, a site evaluation certification program funded by FÁS is at an advanced stage of preparation. In both cases the complex multi-disciplinary approach to the onsite sewage treatment/disposal practices will be fully addressed. Furthermore, it is accepted that in order to effectively assess the suitability of a site for onsite effluent disposal and subsequently recommend a suitable technology for same, it is necessary to adopt the principles of a number of professions as presented in the table below.

Profession	Task
Engineering	Design of treatment system
Environmental Science	Treatment system efficiency, contamination attenuation
Geology/Hydrogeology	Contaminant movement/groundwater resource and vulnerability
Soil Science	Soil mechanics, contaminant migration and attenuation

A comprehensive site evaluation will, as a minimum, investigate the suitability of a location under the following headings:

Treatment System (Hazard)	• Size/capacity • Performance • Seasonality • Maintenance
Disposal System (Vector)	• Soil type/depth • Water table/unsaturated zone • Bedrock • Vertical separation • Site slop/hydraulic gradient • Setback distances
Receiving Waters (Impact)	• Pollutant assimilation/attenuation • Groundwater resource/vulnerability

In the US, for example, the site evaluation approach is being developed along the risk assessment model:

$$\text{Hazard} \rightarrow \text{Vector} \rightarrow \text{Impact}$$

CONTAMINANT TRACING

The use of a selection of chemical and biological tracers in the detection of contamination from failing onsite systems in addition to predicting the migration/movement of same in 'green field' (pre development) situations, has attracted a great deal of attention from academics/researchers in the onsite sewage treatment field over the past number of years. Unfortunately, the techniques have not, as yet, gained widespread usage in regulatory or site evaluation circles. There are a wide range of tracer materials which are commercially available and can be used to mimic the migration and attenuation of a septic system contaminating plume in subsurface soil and groundwater systems. The range of tracer materials available are summarised in the following table:

Tracer Type	Material
Chemical	
Ionic	NaCl, KCl, Li, Br, $NaNO_3$, Conductivity
Dye	Optical Brightener, Sodium Fluorescene, Rhodanine
Biological	Bacterial, Endospores, Viruses

Research has shown that a single tracer type cannot be used to accurately monitor the movement of a complex multi-component contaminant plume. It is, therefore, contended that a combination of chemical and biological tracer materials must be used to gain a more realistic representation of the movement of specific pollutant types in effluents. The combined use of *Bacallus globigii* endospores and bromide and/or nitrate ions is recommended as being likely to yield an accurate indication of the migration patterns of septic tank effluent in a range of soil overburden types and hydrogeological conditions. It is concluded that the use of tracer techniques will provide a useful tool to regulation and site evaluation in the assessment of the impacts of onsite developments.

CONCLUDING REMARKS

Onsite sewage disposal will remain the most commonly used means of wastewater treatment in rural and suburban areas for the foreseeable future.

The use of appropriate onsite treatment technologies in place of dated septic tank units alone, has proven to be a useful tool in alleviating the problems encountered with failing onsite systems. It has also permitted the development of marginal sites which hitherto were considered unsuitable due to sewage disposal limitations

The use of treatment technologies is not the solution to all onsite problems. Some sites are inherently unsuitable for sewage disposal of any kind due to severe subsoil limitations or extreme sensitivity/vulnerability.

Onsite sewage treatment is a complex multi-disciplinary task requiring inputs from engineering, science, geology/hydrogeology and soil science professionals. In order to ensure that the developed technologies are used to their maximum potential. It is imperative that comprehensive site evaluation/assessments precede their installation.

The siting, selection and management of onsite systems should be approached along a risk based assessment model such as the Hazard (Sewage) - Vector (Subsurface) - Impact Groundwater

Paper No. 13.

Mining Waste and the Groundwater Environment.
James Dodds, Steffen, Robertson and Kirsten (UK) Limited.

MINING WASTE AND THE GROUND WATER ENVIRONMENT

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ABSTRACT

This paper discusses the sources of water borne contaminants in a mining environment and the pathways via which these contaminants may be transported. The topics discussed are presented in the context of Irish geological settings and mining industry. Mining produces many types of waste, from waste rock extracted with ore, through to tailings, the reject portion from the milling and processing systems. In each form of waste, minerals are exposed to water and air, as well as chemical reagents. Oxidation and other processes act to release and adsorb metals and other compounds. The very act of mining introduces pathways for the transport of contaminants and therefore creates a risk of impact on the environment. The nature of this risk varies during the mining project and this variation must be understood when planning and managing the mine waste disposal.

INTRODUCTION

Ireland has experienced mining activity for many centuries. Over the last 30 years mining has had a higher profile with the opening of major lead zinc mines at Navan, Galmoy and Lisheen and an increase in gold exploration. Historic copper mining in the Wicklow mountains continues to give rise to metal rich discharges and the legacy of mining at Tynagh and Silvermines is still evident after many years of closure.

All the problems experienced and many of the concerns raised during recent mining related planning application determinations can be attributed to the impact of waste disposal on ground water supplies and surface waters in general. The issue of mining and waste disposal in the context of the impact on the water environment therefore has a high profile in Ireland. In this context this paper broadly discusses the issues of the potential impact of mine waste disposal on the water environment. The key issue of mine waste geochemistry is presented in summary form in order that that the source and control of contaminants can be understood. Pollutant pathways during and after mining are discussed, the context of changes over time. The concept of changes in geochemical, hydraulic and geological controls during and after the life of a mining project are fundamental to the successful planning and control of mitigation measures.

As the twenty first century approaches the pressure to ensure the protection of the natural environment is becoming more intense. All industries which produce hazardous wastes are effected, but the mining industry are received particular attention (Kraicheva, 1996; Ricks and Connelly, 1996; Sides, 1996). Not only is it a large industry with a very visible profile, but it suffers from a legacy of poor historical husbandry of the environment around the world. Therefore it is appropriate to review aspects of waste disposal in the mining industry, the processes which result in environmental impact and present them in the context of modern waste disposal risk minimisation practices.

TYPES OF MINING

OPEN PIT

Open pit mining comprises surface excavation which can range from a small shallow pit of tens of metres across and 5 m deep to large pits which may be of the order of 2000 m across and 400m deep. A special case would be strip mining of coal where a relatively narrow but long 'strip' is open at any one time but the face is continuously moving forward and waste material piled on the other side of the excavation. The waste pile also moves forward and isprogressively rehabilitated as mining continues. In the conventional open pit, the ore will generally form only a part of the excavated material and large volumes of waste are generated which have to be placed in waste piles outside the pit. In the case of industrial minerals and aggregates, little waste is generated, therefore a large hole remains at the end of working. Backfilling of large open pits is not often undertaken due to the high cost of this procedure. In Ireland present open pit mining is limited to the quarrying of industrial minerals and aggregates. Historical open pit mining sites exist however e.g. Silvermines.

UNDERGROUND

This covers mining of an underground ore body accessed either by drifts or tunnels into hillsides or inclined or vertical shafts to deeper ore bodies. Depth of mining can be from very close to surface such as coal mining at depths of 20m, to the deepest mines in the world which are the gold mines in South Africa at depths of around 4000m. Some methods of underground mining, such as block caving induce collapse of the overlying material into the mine. This collapse is progressive as the mine develops and probably represents the worst case of impact. Other methods involve leaving pillars of intact rock to support the roof either permanently or temporarily. The amount of movement of surrounding rock is a function of the geology, mine geometry and mining method but there will be some movement which may or may not be reflected in ground surface subsidence. Long wall mining of coal is designed to totally extract the coal from a panel. This induces total collapse of the mining horizon which may reflect as surface subsidence. The amount of surface subsidence is a function of lithology of the roof rocks, bulking and separation and the depth below surface. Present mining in Ireland is not designed to induce collapse, but rather retain the structural stability of the overlying rock mass.

TYPES OF MINING WASTE AND DISPOSAL OPTIONS

WASTE ROCK

Waste rock is material excavated as overburden or rock outside the ore body that must be moved to gain access to the ore but would normally not contain any ore material of any significance. The rock would be placed in specific sites on surface around the mining operation, out side the area that any future development may take place in. Unless a specific mineral which is perceived to be a potential environmental problem, is present in the rock, no detailed studies of a geochemical nature are usually carried out. Some waste rock is taken undergroundto provide backfill and roof support, although this is generally a small proportion of that extracted. Waste rock is also used as sources of construction material if in a suitable locality. The waste rock generally poses a low level of risk due to it's low mineral content, but it is placed in the open and will remain on surface in perpetuity.

LOW GRADE ORE

Low grade ore will usually be placed in piles, close to the process plant. These will be similar to the waste rock piles but kept separate for possibleore processing at a later stage, usually towards the end of the mining operation or to maintain bulk feed through a processing plant. They therefore represent a source of contaminants due to exposure of the ore and gangue minerals to chemical weathering.However, as the plan will be to process the ore it would not be expected that the stock piles would remain after mine closure. Should for unexpected reasons this not be the case, then these stockpiles could present long term sources of contaminants.

TAILINGS

Tailings and slimes represent the waste product after crushing, grinding and processing of the ore. Disposal can be in the form of dry dumps or hydraulically placed in "tailings dams". Tailings dams are traditionally the major source of potential problems from escape of contaminants into the environment as they contain the greatest concentration of gangue minerals as well as reagents introduced during processing. They do not always contain problem minerals, but generally a sulfide ore will generate a tailings with a content of reactive sulfide gangue. Tailings dams constructed more than 20 years ago generally did not have any particular protection to control the escape of leachate. Nowadays, consideration is given to preventing the loss of any leachateto the environment as well as methods of dam construction to minimise development of contaminant leachate during the mine operation as well as after closure. In modern mine planning the siting, design, management and closure of tailings facilities, probably takes up the greatest amount of environmental effort.

Depending on the waste product and the mining operation, a proportion of the tailings may be placed underground in worked out areas to assist in mine roof support. The tailings often have Portland orpolymer based cements added to provide strength and lock in moisture. Depending how this is done and the nature of additives to achieve strength, the stabilisation can be a source of long term leaching to ground water, or create a largely inert barrier to ground water flow.

SOURCES AND SINKS OF CONTAMINANTS

The transport, mobilisation and precipitation of an element is a balance between processes which release the element from its precursor, through dissolution and processes which scavenge or fix the element through mineral precipitation or adsorption. The actual mechanisms involved are complex and detailed reviews have been published elsewhere (Lowson, 1982; Nordstrom, 1982; Blowes and Jambor, 1994; Bowell and Fuge, 1996; Bowell et al., 1996).

The most common source of contaminants from metal or coal mines is the oxidation of FeS_2 which is the dominant control on drainage pH. Pyrite and/or marcasite generate the acidity of the mine waters and simultaneously supply large quantities of Fe and sulfate and consequently produce large volumes of ochres. The production of ochres, which primarily comprise ferric hydroxides together with the dispersal of clays, is an important process in that it creates suitable mineral surfaces for adsorption of metals to take place and therefore creates a sink for contaminants.

The principal pyrite oxidation reactions are:

a) $FeS_2 + 3 1/2\ H_2O = Fe^{2+} + 2SO_4^{2-} + 2H^+$

b) $2FeS_2 + 7O_2 + 2H_2O = Fe(SO_4) + 2H_2(SO_4)$

followed by ferrolysis of ferrous to ferric iron:

a) $Fe^{2+} + 2\frac{1}{2}H_2O + \frac{1}{4}O_2 = Fe\,(OH)_3 + 2H^+$

b) $2Fe(SO_4) + H_2(SO_4) + \frac{1}{2}O_2 = Fe_2(SO_4)_3 + H_2O$

Both pyrite oxidation and ochre precipitation are pH controlled. Where alkalinity-acidity is not balanced an acidic discharge will be emitted. Where neutralisation occurs metals and sulfates are precipitated forming a range of mineral precipitates providing sinks for the potential contaminants. In the case of ochres, many oxide surfaces change from being positive at low pH (thus attracting anions) to negative at high pH (attracting cations), thus changing their status from a sink to a source of contaminants. For example at low pH only a small proportion of the base metals present will be retained, sorbed onto the Fe hydroxide, but elements such as AS, Sb, W, U, V and Mo will be retained, adsorbed as oxyanions. Athigher pH's precipitated Fe oxyhydroxides can adsorb substantial concentrations of liberated metals from the mine waters.

Acid-neutralisation reactions result from mineral buffering of pH in drainage. The major mineral phase which consumes acidity is calcite by the reaction:

$$CaCO_3 + H_2SO_4 + H_2O = CaSO_4.\,2H_2O + CO_2$$

As the chemical system is stressed, it tries to maintain a chemical equilibrium. The neutralising reactions are primary control on the equilibrium of the system and essentially operate by mineral dissolution (source) and mineral precipitation (sink) reactions. Carbonate minerals have a varying degree of acid neutralisation capacity. The order of ranking is calcite>dolomite>ankerite>siderite. In the case of calcite, dissolution is rapid and generally sufficient to maintain water pH in the range 6.5-7.5. If all available calcite is removed then pH will decrease to a dolomite buffer range of pH 6-7. When dolomite is depleted pH will fall to the siderite buffer regime of pH 4.8-6.3. In the carbonatebuffer zones the precipitation of metal hydroxides are promoted with dissolved Fe derived from sulfides, Mn and Al from wall rock oxides and silicates. As acid generation continues and carbonate minerals are depleted pH will fall until the hydroxide buffer zones are reached, for $Al(OH)_3$ this is the pH range 4-4.3 and for $Fe(OH)_3$ the pH range 2-4. Under very low pH conditions, the dissolution of Aluminosilicates can be an important acid neutralisation mechanism. Thus the neutralising potential of a waste is important in controlling the release of contaminants.

The final important sink and source of contaminants are the Acid Volatile Sulfates. On weathering in addition to the products describe above, sulfides produce a range of sulfates, hydroxides and oxides which are stable in oxidizing, acidic pH conditions. A good example is the formation of römerite from the oxidation of pyrite:

$$3FeS_2 + 11O_2 + 16H_2O = Fe^{2+}Fe^{3+}{}_2(SO_4)_4.14H_2O + 2SO_4^{2-} + 4H^+$$

In this reaction a proportion of the sulfate is "stored" as a unhydrolized, partly oxidized iron mineral. This sulfate mineral is included in the term Acid Volatile Sulfates and includes minerals such as jarosite and copper and nickel sulfates. These minerals are highly soluble, so can represent an instantaneous sourceof acidic, sulfate rich water upon dissolution and hydrolysis. Hence Acid Volatile Sulfates are important as both sinks on precipitation and as sources of acidity, sulfate and possibly metal ions, and rapid release on exposure to moisture (Cravotta, 1991, 1994; Olyphant et al., 1991). This is a particularly important source of contaminants when underground workings are flooded on closure and from the walls of open pit mines.

This brief discussion on the geochemistry of mining waste demonstrates several important points. Firstly, that although the chemistry is complex, compared to domestic waste there are fewer chemical processes active and therefore the overall system is more predictable. Secondly, the nature of the chemical pathways is such that potentialcontaminants can be stored or released from the system depending on the physical and chemical conditions acting at any particular time. Therefore, in a dynamic mining environment, the chemical processes will be continually changing as chemical equilibriaare maintained. It must also be considered that the controlling reactions are dependent on sources of minerals for oxidation and neutralisation. These sources are finite and with time will become depleted. Therefore, again the nature of the chemistry willchange with time.

PATHWAYS

Following the widely accepted environmental risk analysis technique of "Source – Pathway – Target", the hydrogeological environment provides pathways for the contaminants described above to move to the target or receptor. By linking the hazard created by the source, to the target via a pathway, a risk is created. The pathways are created by a combination of geological and hydraulic controls. Mining in Ireland is carried out in a hard rock environment and therefore the geologicalcontrols on ground water flow are fractures rather than intergranular pore spaces. Fractures capable of transmitting water occur in many different forms, from micro fractures in deep igneous or high grade metamorphic rocks, to large solution weathered caverns in limestone. Generally in the Irish setting, the principal area of mining interest is the Carboniferous Limestone terrain and the Dalradian meta sediments. As such, consideration of Irish mining waste disposal must take account of the special circumstances relating to palaeo and active karstification of the limestone and dolomite sequence. In addition to the natural pathways, the mining activity will create new pathways. These may comprise the mine void itself, as well as new fractures created above the workings and the opening of small or micro fractures present prior to mining.

The nature of mining in terms of operations and timing places stresses on the aquifer system which are unlike the hydrogeological settings of most waste disposal sites. In most cases waste disposal sites are sited in areas of hydraulic steady state, outside the capture zone of principal pumping boreholes. This is not the case in the mining scenario. Although general waste disposal often takes place in voids created by shallow'mining', such as quarrying or borrow pit development, the voids are usually free of ongoing development. In the case of mining, the extraction phase and waste disposal operations go hand in hand. The effect of these differences is that the nature of the hydrogeological pathways changes during a mining operation.

Figure 1 shows a schematic mine layout and the typical sources of contaminants and pathways. Figure 1 clearly shows that the two principal pathways during active mining are runoff and seepage. Inthe context of this discussion, we shall concentrate on the seepage as it is this source which enters the ground water system.

A hydrogeological pathway can be considered as a vector, with both a value (flow rate) and a direction. The movement of a volume of water along a particular pathway, in a particular direction can be called a flux. During the life of a mining project and on cessation of mining and closure, pathways will be created and closed and will change in terms of their ability to transport contaminants and their direction. Thus, the fluxes will also change. All mining operations, below the 'water table', reduce the level of the phreatic surface, whether inflows into the mine are significant or not. As such, when mining commences and by definition waste is generated, there is a hydraulic gradient toward the centre of mining. The area affected will be dependent on the depth of mining and the hydraulic properties of the aquifers. However, it can be expected that the zone of influence of the dewatering effect will extend several hundred metres from the edge of mining. Within this zone the ground water flux will be toward the mine and therefore any contaminants introduced into this zone will move toward the mine. The contaminants are removed by the mine drainage system and if anticipated can be controlled and/or treated by an appropriate process.. In actively dewatered situations, the zone of influence could extend for several kilometres and capture seepage from all the mining related sources of contamination. During active mining therefore, the risks of contaminants arriving at receptors is reduced. The process of mining opens pathways which may not have been open previous to mining activity. The new pathways may comprise the physical mine openings and fractures which have opened due to changes in the stress regime created by the mine. In a similar way, pathways which may have existed prior to mining may also close. Such a situation exists in areas of historic mining, where mine drainage was often facilitated by gravity. Specific drainage adits were driven from the mine to low points in valleys, which remained open on closure. Such a situation can create an ongoing contaminant problem, by preventing the workings from fully flooding, thereby keeping mineral surfaces available to oxygen, while at the same time maintaining an active pathway for the contaminants to be dispersed. Figure 2 shows typical pathways relating to the post closure situation. Waste tips and tailings can continue to contribute contaminants to the system if maintained in an oxygenating environment. Unsealed shafts can link a vertical aquifer sequence and shallow unsaturated flow zones to deeper saturated flow paths, providing flow paths for contaminated water from the shallow system to the surface water system. However, on full recovery of the ground water system after cessation of pumping, the deep mining system may become part of a long flow path along which the oxidation potential decreases as dissolved oxygen in the recharge water isdepleted by ongoing reactions, thereby increasing the likelihood of mineral precipitation within the old workings. A modern form of mine waste disposal is the placement of tailings underground in order to provide roof support and to reduce the cost and environmental impact of surface disposal. Portland cement or polymer based cementing products are often mixed with the tailings in order to provide strength and bind in the water content. This also reduces the permeability of the material. In such circumstances the pathways that may have been present by virtue of the mine openings will be essentially sealed, reducing the overall flux to very low levels. There may however, be the potential for low levels of contaminant transport by diffusion from the backfill.Although generally considered good practice, the sealing of underground workings may divert other seepages to a more shallow flow path than would otherwise have been the case.

The pathways that are active during mining and post closure will be very different. Therefore, by definition, the risks to the environment will be different. It is important that these differences are understood during the design of waste management facilities and the design of the mine closure plan.

MITIGATION

The previous sections have discussed the contaminants and pathways that occur within a mining environment. Together these combine to form a risk which must be mitigated to an acceptable level. As the sources and pathways change during mining so the mitigation measures must take account of the changes.

SEEPAGE

The risk to the ground water system is mitigated in the first place in the design of liners. Like all waste disposal facilities the liner is designed to minimise the seepage from the facility. The nature of the wastes, the geochemical process and the time related nature of the risk must be considered when designing liners. For instance, it may not be necessary to construct a complicated liner for a short lived low grade ore dump, if the flux transporting seepage will betoward the mine water management system and that the source of contaminants will be removed prior to mine closure. Tailings in particular behave differently from other types of waste. They are more homogenous and when deposited subaqueously have a low hydraulic conductivity due to the high proportion of silt and clay sized particles. With time the hydraulic conductivity falls further due to consolidation by overlying tailings. The containment system can be designed to take account of these properties of thewaste materials. In the planning applications for the Tara tailings dam extension and the Lisheen Tailings management facility, the properties of the tailings and underlying materials were exploited in the designs, to increase the level of environmental protection.

As discussed above, oxidation is a primary mechanism for the mobilisation of contaminants. It is therefore important to exclude oxygen and infiltration from waste dumps and tailings as much as possible, where it is appropriate. Operationally this can be achieved by early restoration of waste dumps and the placement of 'capping' material which minimises oxygen transfer into the waste pile. Tailings are often deposited and kept under water in order to minimise oxidation, and on closure the cappingand rehabilitation of the impoundment will incorporate layers which will maintain a low Eh environment at the top of the tailings.

RUNOFF

During operations modern mine sites incorporate managed drainage systems and potentially contaminated drainage fromthe mill site, low grade ore dumps, waste rock dumps or any other area is collected and either stored prior to treatment and discharge or used in the mill circuit and other non-potable water circuits. By utilising such water, the mine minimises the amount of make up water that is required from 'fresh' sources.

CONCLUSIONS

Mining produces several different types of waste. All the types react with air and water in a complex manner. These reactions can result in both the adsorption and the potential for release of contaminants. In order for these contaminants to pose a risk to the environment they must be transported, via pathways to the receptors. A principal pathway is via the aquatic environment and mining has a major influence on ground water systems inthe vicinity of the mine. The nature of mining is such that hydrogeological pathways and fluxes change during the mining development. The changing chemical, geological and hydraulic controls on the movement of contaminants away from the mine and it's environs must be understood in order to be able to predict the risk posed to the wider environment. By understanding the processes and the temporal nature of the risks to the environment, appropriate mitigation measures can be designed and installed to reduce the risks to acceptable levels.

ACKNOWLEDGEMENTS

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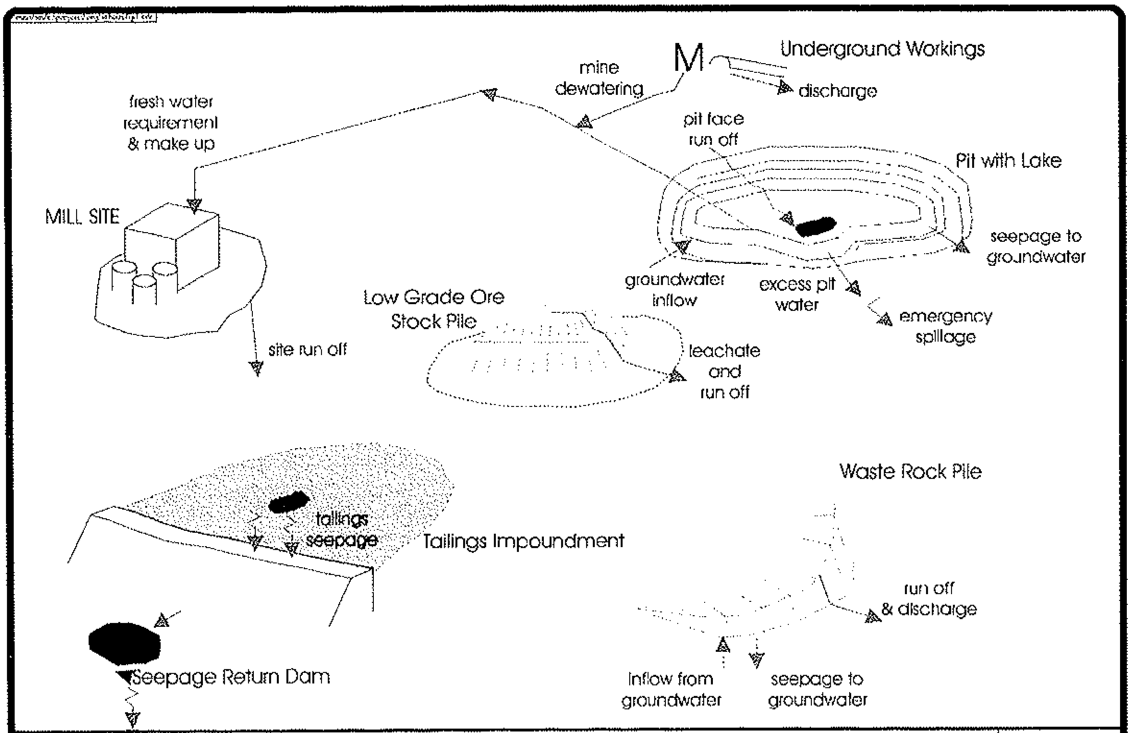
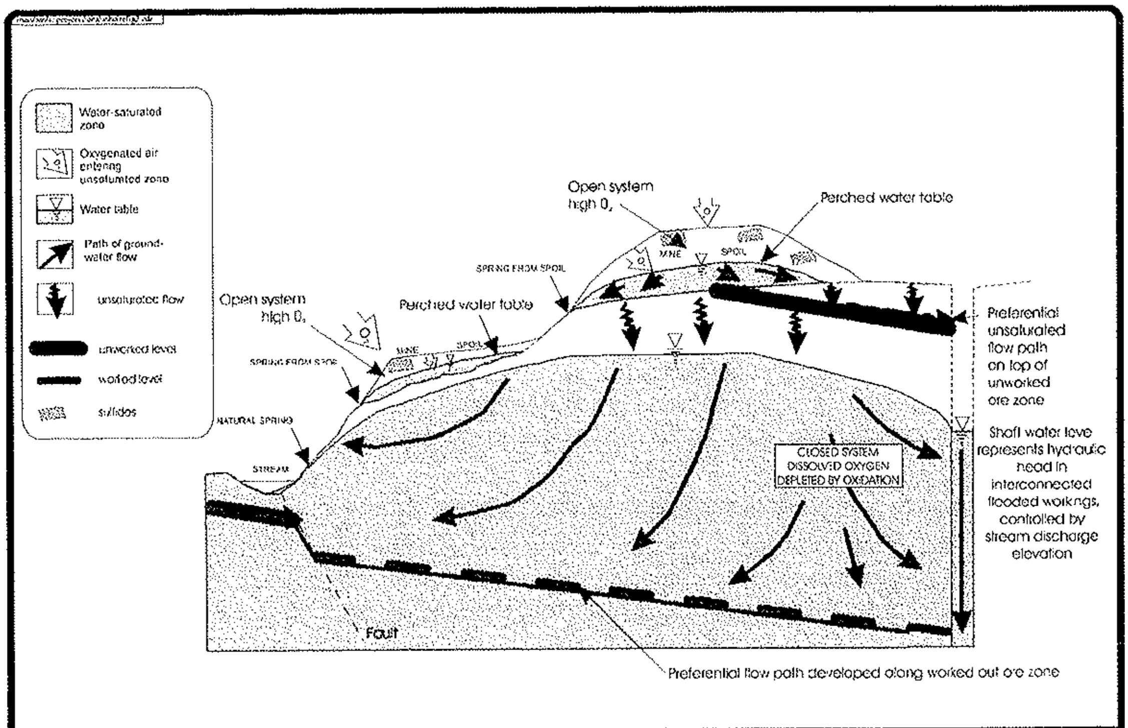


SOURCES AND PATHWAYS FOR MINE WATER CONTAMINATION

FIGURE 1



TYPICAL MINE WATER PATHWAYS AFTER CLOSURE

FIGURE 2

<u>Paper No. 14.*</u> ***Regional Lecture:***

Flow Modelling in the Karst Terrain of South Galway.

Paul Johnston, Department of Civil, Structural &
Environmental. Engineering, Trinity College.

*This paper was not available at the time of print and is therefore not bound in these Proceedings.
The paper was distributed separately during the seminar.

GROUNDWATER MODELLING IN THE KARST LIMESTONES OF THE GORT LOWLANDS

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Abstract

Flooding in the Gort-Ardrahan area of South Galway has caused significant disruption to the local communities in recent years. On the other hand the karstic limestone lowlands around Gort give rise to a characteristic landscape dotted with seasonal turloughs which have ecologies worthy of conservation. The hydrology of the area comprises runoff from upland areas of sandstone discharging into a fissure network in the lowland limestones, ultimately dischaging to the sea through subterranean conduits. A recent major study of the hydrology of the area evolved a conceptual model of the hydrogeology which suggested that flood prediction might best be achieved through a pipe network model rather than by a conventional porous media approach. The role of turloughs as flood attenuation devices in the network and the connection between these hydrological dynamics and their characteristic ecology is fundamental. The establishment of a conduit model, its calibration and validation has demonstrated its success and utility for application in other similar karstic areas.

Introduction

In recent years, the Gort-Ardrahan area of South Galway has experienced several severe flooding episodes. Until the winter of 1989-90, major flooding in the area had only occurred twice before in living memory, in 1924 and 1959. More recently, however, the frequency of flooding appears to have increased significantly. Major flooding during the winter of 1989-90 was repeated in early 1991 and 1994 and, worst of all, in early 1995. Each event has caused severe local disruption to the community and a characteristic of the flooding has been that large areas of agricultural land have been inundated for several weeks. Alongside this flooding problem is the unique landscape in which the hydrological regime operates - the ecology of the limestone plateau, dotted with seasonal lakes (turloughs), is widely recognized as deserving conservation under the EU Habitats Directive. It is the apparently inextricable link between the hydrology and the ecology that was one of the reasons behind the commissioning of a hydrological study of the region. A principal objective was to try and understand the causes of the flooding and the hydrological dynamics controlling the ecology, particularly in the turloughs.

Geology and hydrogeology

The topographic catchment within which the flooding occurs is over 500 km^2 in area. The region is bounded on the east by the Slieve Aughty mountains and by the edge of the Burren in the west and drainage is northwestward across the lowlands around Gort towards the sea in Galway Bay (Figure 1). Elevation ranges from a high of 368m OD(Poolbeg) in the east to sea level at Kinvara with the lowlands rarely exceeding an elevation of 30mOD. This characteristic topography is reflected in the geology, the mountains being of relatively low permeability sandstones and mudstones of Devonian age (Old Red Sandstone) and the lowlands of higher permeability Carboniferous limestones, in many places intensively karstified. This change in geology occurs quite abruptly at the stratigraphic top of the sandstone along a line running approximately northeast-southwest across the area, at the foot of the Slieve Aughty mountains, and has a major effect on the hydrology. Nearly half the catchment area is on the sandstones and is hydrogeologically distinct from the remaining lowland limestones.

The mountains are drained by three streams, the Owenshree, the Boleyneendorish and the Owendalluleegh, flowing westward with a dendritic pattern onto the limestones where the effects of karstification dominate. The streams then frequently 'disappear' into underground fissures and conduits and reappear in surface reaches or in the glacially formed depressions known as turloughs. Eventually this drainage converges with a northward flowing stream system (Cloonteen) on the limestones to form Lough Coole, the hub of the drainage network, northwest of Gort. Although appearing once more in Caherglassaun Lough, the combined drainage becomes totally subterranean, reaching the sea in the vicinity of Kinvara via coastal and subsea springs. The flooding mainly occurred in the lowlands in certain areas of the karstified limestones which could not take the increased flows being delivered from the catchment upgradient. Significantly little flooding occurred between Kinvara and Caherglassaun, suggesting the underground flow capacity is very large.

The objective of an appropriate numerical simulation of the hydrological behaviour of such a system depends strongly on the soundness of the underlying conceptual model. In this context, an understanding of the nature of the karst geology and its genesis is essential to that model. The karstification in the area was not spatially homogeneous in its development but is complex, and seems to have evolved mainly under lithostratigraphic control. Certain cherty beds within the limestone stratigraphy appear to have exerted a strong developmental control. Folding and faulting associated with the regional north-northwest trending Fergus shear zone also affected the development of the karst. Moreover, there appears to be at least two levels or ages of karstification - a shallow 'epikarst' associated with the last glaciation and at least one level of palaeokarst, perhaps associated with a lower base level as it occurs up to 50m below current sea level.

The dip of the limestone sequence is commonly shallow, 1 or 2 degrees north/northwestward, but can steepen significantly as a result of folding, particularly close to the sandstone boundary. In accordance with the lithostratigraphic control, karst fissure development has largely been along strike. Where the drainage flows change direction, across strike, as in the vicinity of Coole, it appears that structural 'lineaments' have had a predominant influence on fissure development.

Most of the karst development in the lowlands is epikarst, often less than 15m in depth. The fissure openings themselves are very variable, ranging from millimetres in width to many metres when they become recognized as 'conduits' or even caves which occur along the main drainage pathways. Where collapses occur or fissures intersect the surface, any associated topographic depressions act both as a focus or collector for incoming rainfall as well as an emergent route for subsurface flows already in the fissure network. The larger depressions, or turloughs, flood in the winter when water levels in the ground rise and drain or dry out in the summer. Turloughs can thus temporarily store significant volumes of water and are key components of the hydrogeological regime. Some of the turloughs behave as integral parts of the principal drainage pathways and behave hydraulically as surge tanks in the system, attenuating flows in the fissure network. On the other hand, any constrictions in the downgradient geometry of the fissure network can have a reverse control on the flood levels in the turlough. Water levels in these turloughs can change very quickly, in response to a hydrological event. Other turloughs, particularly in the northern lowlands, act more or less in isolation. They have typically developed on muddy Calp limestone and are filled seasonally from drainage from adjacent glacial surficial deposits or from very poorly developed pathways in the limestone subcrop. Consequently, the water levels change much more slowly over a winter season. Hydrologically, the different types of turlough, a few permanent lakes (particularly Lough Cutra) and several bog areas (particularly at the foot of the Slieve Aughty mountains) all act as buffer storages in the overall drainage and are important components in any simulation of the system.

The surficial deposits in the area played a relatively minor role in the overall hydrological response of the catchment but could be locally important in providing groundwater storage and slowing the runoff rate. Most of the outcrop and limestone pavement is exposed west of a northeast-southwest line passing through Coole. East of this line are a number of drumlins and modest thicknesses of sandy till along the limestone/sandstone contact. In the west, along the margin of the Burren, drilling showed evidence of a former glacial outwash channel with an excess of 40m thickness of sands and gravels. The relevance of the distribution of surficial deposits lies in their control of direct recharge to the underlying hydrogeological regime and which will require representation in a hydrological model.

This geological framework emerged from the extensive investigation undertaken as part of the hydrological study of the area undertaken by Southern Water Global, UK and Jennings and O'Donovan, Sligo (1995) for the Office of Public Works building on previous work by the Geological Survey of Ireland (Daly,1992). Drilling, surface geophysics, geological and topographic mapping all contributed to this understanding and it provided the basis for a hydrogeological conceptual model.

A hydrogeological conceptual model

The fissured nature of the limestone bedrock in much of the area and the thin cover of glacially derived subsoils meant that the hydrological regime could not be neatly divided between surface water and groundwater components. There is a merging of traditional boundaries and modelling needed to take account of this integration. The geological investigation also showed that the drainage of the area was likely to be via discrete pathways both surface and underground. Direct recharge was likely to be routed quickly to these main pathways either through overland flow on outcrop, or through thin soil or peat cover. It was the surcharging of these discrete drainage pathways which was likely to have been causing the flooding. Thus, groundwater flow could not readily be represented by a conventional 'equivalent porous media' (EPM) model. In this sense, a map of contoured groundwater levels in the limestones, gained from point observations could be misinterpreted as it tends to mask the effects of the constrained pathways of much of the subsurface flow. If flood response was to be simulated at points within the system, the flow routes needed to be identified and modelled discretely. A network model was chosen as the most suitable approach for simulating flow in the karstified limestone area where conduit flow mechanisms dominate and distributed groundwater flow comprises only a minor component.

Hydrometry

The application of a network model to the karst hydrogeology of the area required identification of the topology and geometry of the appropriate network nodes and their interconnections. Calibration and validation of the model would then be required to characterize the hydraulic behaviour of the links in the network. Topology was primarily identified through the use of tracers. Strategic hydrological monitoring of discharges and water levels at accessible points in the area was then undertaken in order to evaluate the hydraulics of the conduits and associated turlough storages. Hydrometeorological variables (rainfall and evapotranspiration) were assessed from sparse historical records and from setting up five new recording raingauges from which direct recharge could be estimated.

Guided by a tentative understanding of the geology of the karst, tracing studies were undertaken mainly to establish point-to-point flow routes (Figure 3). Dyes (leucophore, rhodamine WT and fluorescein), salt and bacteriophages were used and were effective in revealing some unexpected flow routes. The tracing showed that groundwater flow velocities can be very high (hundreds of metres per hour were estimated in the vicinity of Lough Coole) and can increase with increasing stage. It was also possible to identify the key conduits or fissure systems which were taking the majority of flows. Moreover, the tracing helped to establish the stages at which certain flow routes became operative or ceased to operate.

Hydrometric measurements involved evaluation of water level dynamics in turloughs, conventional stream gauging of key surface water flows, borehole water level monitoring and the measurement of water levels in caves. The measurement of stage in turloughs presented particular challenges as water levels could range from zero in summer to 15m or more (Hawkhill) in flood conditions and water level changes could be as fast as 9m in 48 hours (Blackrock) over areas of tens of hectares. The objective for these key turloughs was a relationship between stage and volume through which discharge dynamics could be evaluated from stage recession curves and, ultimately, could be incorporated in a hydraulic model.

Important stream gauging stations were established to measure flows from the three rivers draining the Slieve Aughty mountains at the points where they crossed the sandstone-limestone boundary. Rating curves were established so that input flows to the karstic flow model could be evaluated from logged stages. Stations were also established on the Cloonteen river system which is located entirely within the karst limestone area and at the exit to Lough Cutra in order to assess the effect of the storage provided by the 4km^2 lake.

Water levels in some 50 borehole were monitored in order to establish regional flow gradients and directions, notwithstanding the conduit nature of the pathways. While water levels at a point could vary by several metres seasonally, the hydraulic pattern confirmed overall drainage focussed on an outlet at the sea around Kinvara. Hydrochemistry from many of the monitoring points also confirmed the very rapid travel of peaty water from the uplands to the northwestern extremities of the karstic conduit system. Water level monitoring in the caves south of Kinvara, which were thought to be part of an older karstification than most of the active conduits in the area, showed clear effects of the cyclical tidal head to the north. Turloughs such as Caherglassaun and Hawkhill near Gort also exhibited tidal influence on their water levels.

The most important missing link as far as the establishment of a model was concerned was the absence of any reliable discharge measurement at the catchment oulet, that is at the springs at Kinvara. The distributed nature of the springs and the location of many of them below low tide level made realistic discharge measurement impractical. Thus the downstream boundary condition in the model would have to be represented by the known cyclical tidal head only. This condition made the hydrometric monitoring at intermediate points in the karstic network critical for the calibration of the model.

Supporting all the hydrometric work and essential to the analysis of the data were accurate elevations for each monitoring point. Available topographic information had inadequate resolution, so extensive levelling was carried out using a two-station Global Positioning System from which a vertical resolution of two centimetres was possible. These data were also used to fix a digital terrain model of the catchment which, in turn, was used as a basis for a GIS approach to calculating recharge to the network model and to ascertaining the extent of any flooding predicted by the model.

Accurate spatial rainfall estimation was clearly essential to the calibration and operation of the model. Available historical rainfall data within the catchment area was limited. Although there were 23 raingauges within the area with variable lengths of record between 1941 and 1997, less than 9 gauges were operational at any one time. Correlation studies were carried out to link the five new rainfall stations with the existing network and to infill missing data. An analysis of the historical data showed that flooding was always associated with high winter rainfall amounts, particularly when amounts exceeded 550mm during the December through February period (Figure 2). A spatially variable stochastic model of daily rainfall was established through which synthetic sequences of rainfall input to the model could be generated. Different flooding scenarios involving various sequences of rainfall or events of estimated frequency of recurrence could therefore be simulated.
Daily evapotranspiration was estimated based on disaggregated monthly data from Shannon airport and was assumed to apply to the whole catchment area.

The hydraulic conceptual model
Implementation of a network model for groundwater flow in the karst limestone of the Gort lowlands was thus predicated on a sound geological and hydrometric foundation. While a network approach is more appropriate than a conventional porous media model in this conduit-dominated karst, other modelling approximations might have been used. Dreiss (1982) employed a linear systems approach, deriving a kernel function to relate rainfall input to a point output (discharge) at a karstic spring, but the concept is a lumped model and interpretation of intermediate flows and levels is difficult. Some attempts at pipe network representation of karst flow have been undertaken (Thrailkill,1974, Thrailkill et al, 1991, Smart, 1988) but require detailed knowledge of the geometry of the 'pipes' and are not readily utilized in a large complex regional system such as around Gort.

A network representation is an intermediate approach in which individual links can be characterized by a hydraulic resistance derived from an understanding of the nature of the link and fitting the model to hydrological response data collected in the field. Detailed geometry of the link is not required and, indeed, a given link may be a single conduit or an active fracture system operating under a pressure head. Moreover, nodes in the network need to be able to accommodate open water storages such as the turloughs or lakes. A suitable analagous system hydraulically is a sewerage network capable of incorporating storage elements in its structure. Simulation modelling packages for sewer networks are readily available commercially but the one selected as suitable in this study was HYDROWORKS produced by Wallingford Software and widely used in the UK for urban storm sewer design.
The essential conceptual model of groundwater flow in the limestone then becomes a network of conduit links connecting nodal storages (Figure 4). The outlet is controlled by a cyclical tidal head and inflow comes from recharge from net rainfall and from surface water inflows from the three streams draining the Slieve Aughty catchments. A separate model is required for each of these inflow components.

Surface water flows:
Although the permeability of the sandstone in the Slieve Aughty catchments is low, these upland areas have a cover of peat and glacially derived soils, partially planted with forestry, and they respond conventionally to a rainfall input, producing hydrographs with relatively rapid runoff and a baseflow. A pipe network model cannot easily incorporate baseflow, so a rainfall-runoff model was chosen to yield continuous simulation of runoff which would provide a suitable input into the karst groundwater system at the three gauging stations. The rainfall-runoff model adopted was IHACRES of the Institute of Hydrology, UK (Jakeman,1993) which is an integrated catchment response model made up of two components. A nonlinear loss model has rainfall as input and effective rainfall as output. The second component routes the effective rainfall into two linear conceptual storages, one representing a 'quickflow' and the other a baseflow. The combined output provides the required integrated runoff. The nonlinear loss model has a single parameter which characterises the rate at which the catchment wets up or dries out after rainfall events. The operating equation is a simple parameterisation of the well known Antecedent Precipitation Index, API, (Shaw,1994) which attempts to quantify the current soil moisture status of the catchment in order to determine how much of any present rainfall will be available for runoff. The parameter is determined by simple iterative optimisation. Each of the parallel linear storages is characterized by two parameters which are estimated using time series analysis and least squares regression. Rainfall data from the five installed raingauges was reduced to areal rainfall using Thiessen polygons and combined with discharge data from the gauging stations, on an hourly increment, to fit these models to each of the three catchments and to establish surface water inputs to the network model. In the case of the Owendalluleegh River, the runoff had to be routed through Lough Cutra using a level-pool reservoir routing model (utilizing a non-linear storage-discharge equation). Surface water inputs from the 180km^2 of catchments on the sandstone could then be generated from historical rainfall records or from synthesized rainfall records.

Direct recharge model:
Direct recharge to the network model was calculated on the basis of 250m^2 grid squares under a Geographical Information System (GIS). Soil moisture deficits were calculated on the basis of rainfall and Penman-Grindley estimates of evapotranspiration, thereby determining the water available for direct recharge. For purposes of the recharge model, rainfall was regionalized on the basis of six subregions, whose delineation was determined by similsr or likely influences on rainfall depths and frequency such as elevation or topography. Mapped estimates of the nominal thickness of surficial deposits combined with four land use types yielded six landuse categories to which were assigned rooting constants and wilting points. These in turn allowed calculations of soil moisture balance on the grid. Finally, the gridded estimates were consolidated into 23 direct recharge areas. Daily values of direct recharge were input to the network model at nodal points. Significantly, calibration of the surface runoff models for the Slieve Aughty catchments and of the direct recharge model indicated that approximately half the flows in the network/conduit system were coming from the uplands and half from direct rainfall recharge.

The groundwater network model
HYDROWORKS is essentially a hydraulic pipe network model employing the St Venant equations to simulate the flow in the pipes, open channels and storage elements (reservoirs). Free surface conditions (unconfined groundwater flow) and surcharged conditions (confined groundwater) are both simulated in the model with a smooth transition between the two. All flow inputs (upland catchment runoff and direct recharge) are modelled separately, so the structure of the network model is relatively simple. Data is required to describe the properties of all links and nodes and to provide water level information at the outlet of the network (sea level). Applying a model of this type to a regional system of karstic fissures represents a significant change in scale from the conventional applications. Thus, a suitable time step for the hydraulic calculations was chosen so that the model could run for a minimum 6-month winter period without becoming unstable or consuming too much time. A simulation time step of 15 minutes was selected and the inflow and level data were input at time increments of one hour. A six-month simulation run of the model took approximately 2-3 hours on a 90Hz Pentium desktop computer. Analagously to a sewer system design, results could be displayed in several ways, typically as time histories of water levels or flows for any element in the system.

The links in the conduit model may be pipes or open channels, connecting two nodes. The nodes may represent large defined storages such as turloughs or they may have limited storage and purely represent the junction of two or more links. 'Standard' links, pipes or open channels, are defined by their shape, size, gradient, roughness and the two nodes which they connect. The gradient is defined by the invert levels at each end of the pipe or open channel. Nodes were assigned to all defined storages in the network including recharge input locations, locations where three or more conduits meet, locations where the characteristics of connecting links change significantly (eg where a surface channel becomes a conduit), the gauging stations and the downstream outlets (Kinvara springs). Since the maximum allowable length of link is 5km, nodes have occasionally been inserted to subdivide a longer length of conduit. Storages at nodes can be characterized by stage-area relationships as determined from field measurement. Conceptual orifices or weirs may be inserted to represent chokes (flow restrictions) or to represent threshold overflows into another link or node.

The karstic features and tracing studies largely determined the geometry of the network defined to represent the hydraulic behaviour. However, the properties of the karst conduit connections or links were essentially determined by calibration. The most sensitive parameters were the nominal pipe diameters and gradients which were adjusted to achieve observed hydraulic responses. Conduit proerties such as head loss coefficients and shape were found to be less sensitive and were estimated and held constant for all links unless specific information suggested otherwise. A hydraulic roughness equivalent to a Manning's n of 0.05 was adopted for all links and open channels. Open channel links were assumed to have a trapezoidal cross section with a side slope of 4:1. The downstream boundary condition was simulated using an hourly timestep derived from a tidal prediction program known as TIDECALC, produced by the Hydrographic Office of the UK Ministry of Defence. Inflow data consisted of continuous hydrographs at 29 nodes which included the flows from the upland catchments and the direct recharge nodes within the network.

Results

The philosophy adopted in calibrating the model was to start with the simplest form of representation of the conceptual model that could be envisaged. Simple initial conditions at the start of modelling gave way to progressively more complicated features as success was achieved in calibration. The final version of the model was calibrated using data from the winter of 1996/7, from 21st September to 16th April. The results of this calibration exercise indicated that the model appeared to be functioning well, accurately producing correct hydrograph shapes and generally the correct hydrograph levels. There was some discrepancy in simulation at the end of the winter with underprediction and a recession which begins too early. These effects are probably attributable to the underestimation of rainfall and inappropriate modelling of storage in the epikarst and in bogs and possibly in unknown distributed groundwater.

Validation of the model was carried out for the flows in the winter of 1994/5 which contained the largest flood on record. Predicted water levels in the turloughs agree remarkably well with observations at the time (MacDermot, Geological Survey of Ireland). Peak levels in Blackrock agreed within one metre and water levels in the other key turloughs (Coy, Coole, Hawkhill) exhibited excellent agreement (Figure 5). Some problems occurred in matching discharges particularly from Lough Cutra but much of this discrepancy is attributed to uncertainty in the extrapolated rating of the gauging station.

Conclusions

These results confirm the utility and power of a network approach in modelling the integrated surface and groundwater flows in karst regions. It appears to be the first time a network model has been applied to such a complex regional system for flood prediction purposes. Certainly refinement (particularly of storage definition) would improve the predictive capacity of the model especially at extreme rainfalls and discharges. Nevertheless, agreement between observed and calculated levels is generally excellent and demonstrates the value of the model in engineering flood analysis. As with any pipe network model, much of its value in prediction depends on the ability to model the inflows accurately. While the conduit flow model has been demonstrated to be able to route flows extremely well, the importance of the associated inflow models representing surface water inputs and direct recharge is basic to that success. As an excellent model of karstic groundwater flow, it cannot be easily separated from its surface water inputs, as the two components are inextricably linked.

Acknowledgements
This paper represents an extraction and presentation of a methodology developed during the hydrological study of the Gort-Ardrahan area conducted by Southern Water Global and Jennings O'Donovan and Partners. Many people were involved in that study in the field and in analysis but particular thanks are due to Sally Watson, Nicholas Price and David Ball for their assistance in the context of a new modelling approach.

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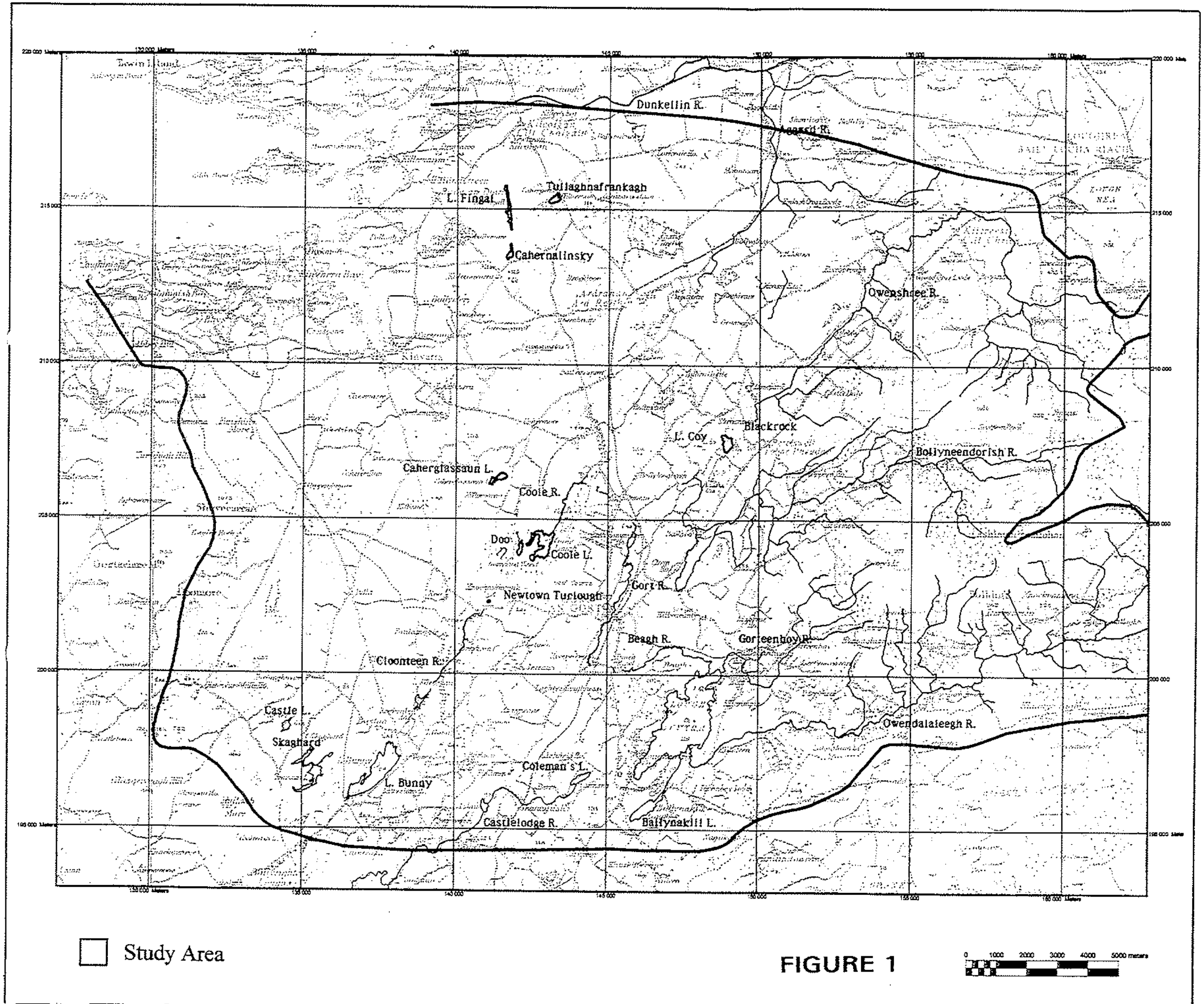
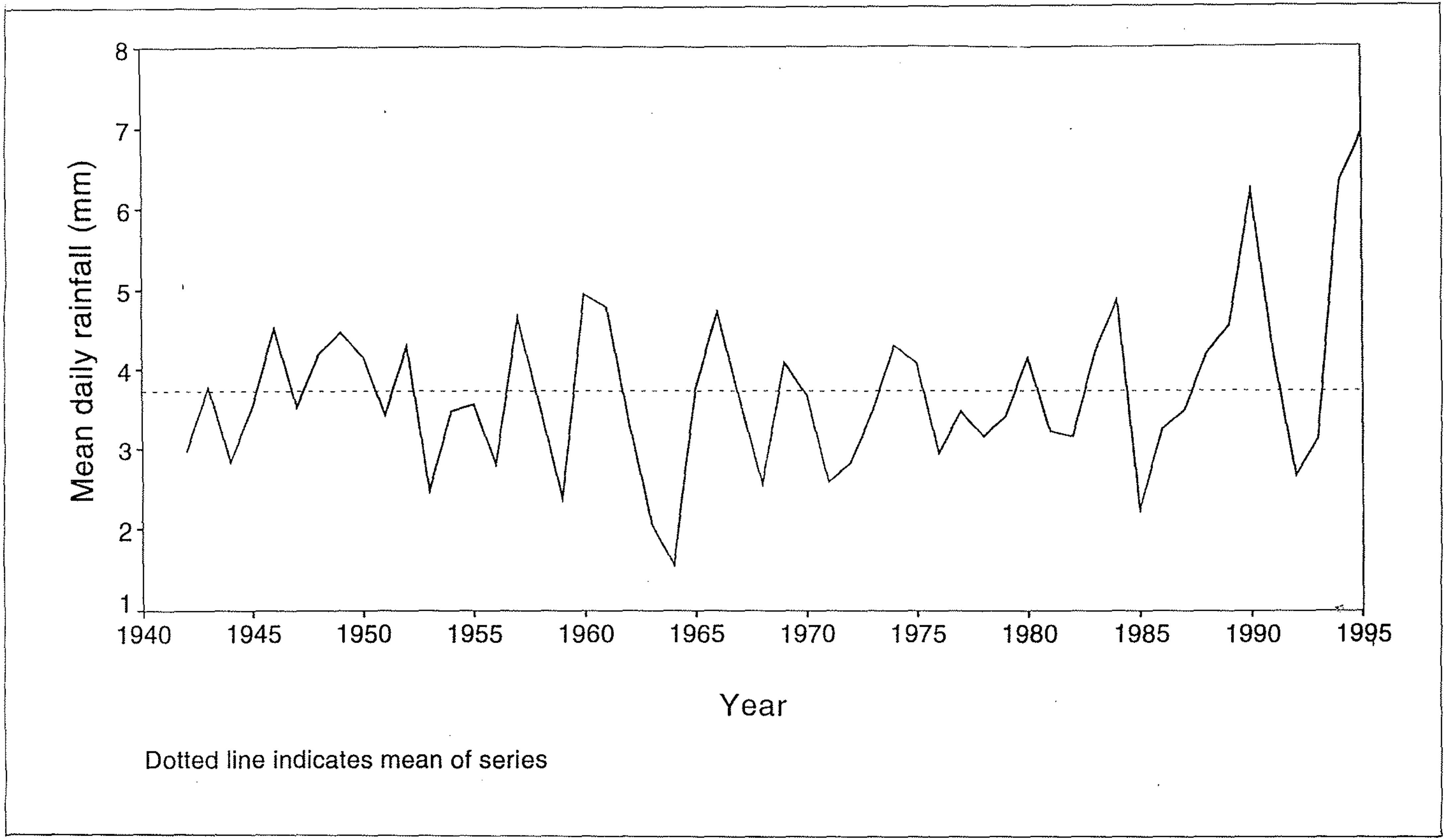


Study Area

FIGURE 1



MEAN DAILY RAINFALL, DURING THE DECEMBER-FEBRUARY PERIOD, OVER THE ENTIRE STUDY AREA FOR THE YEARS 1941-1995

FIGURE 2

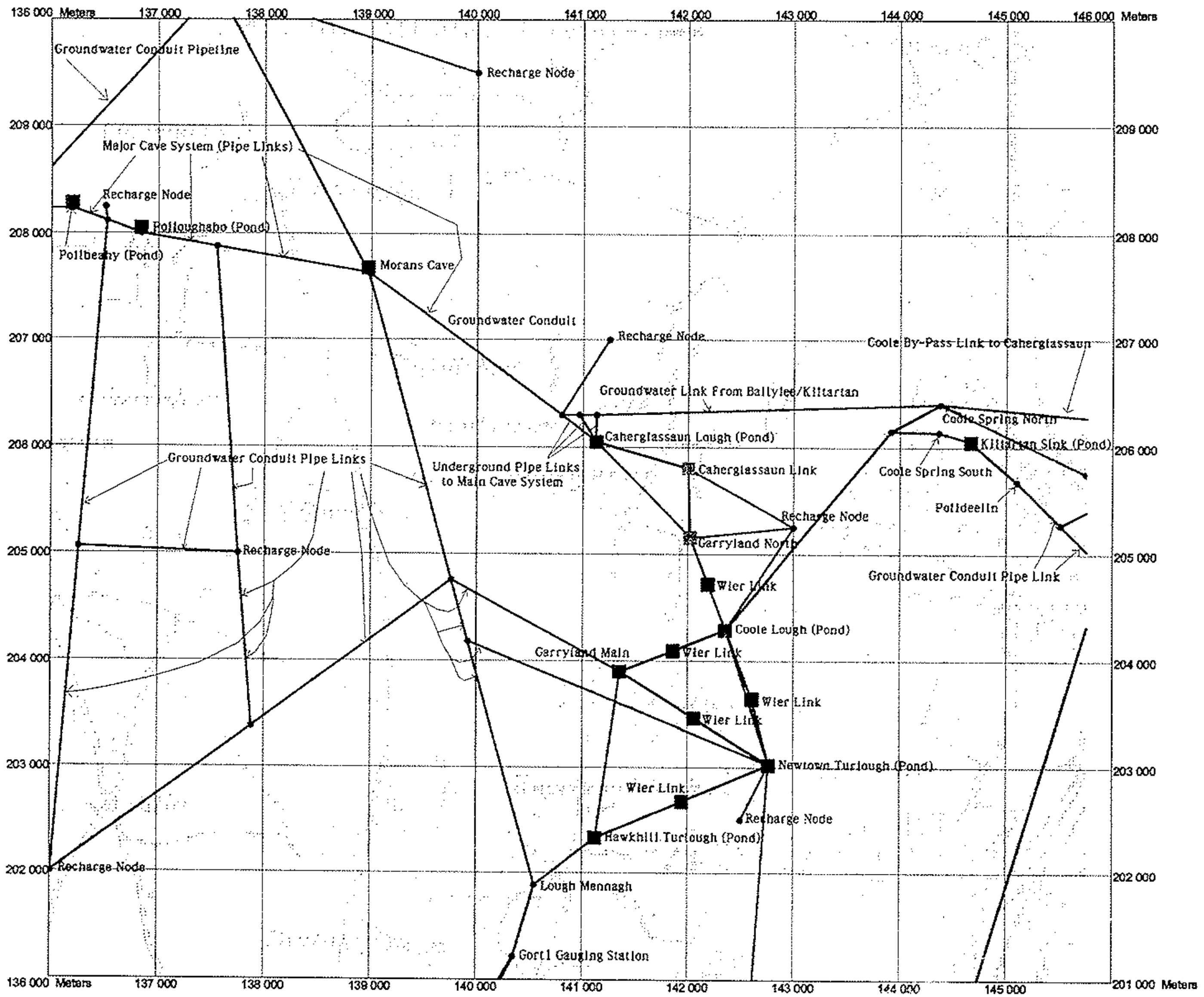
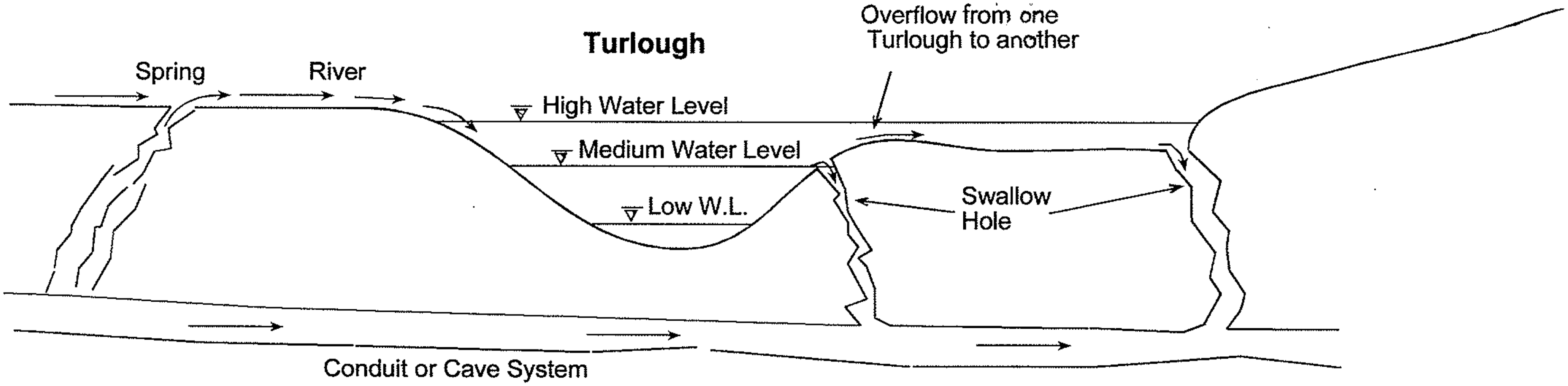
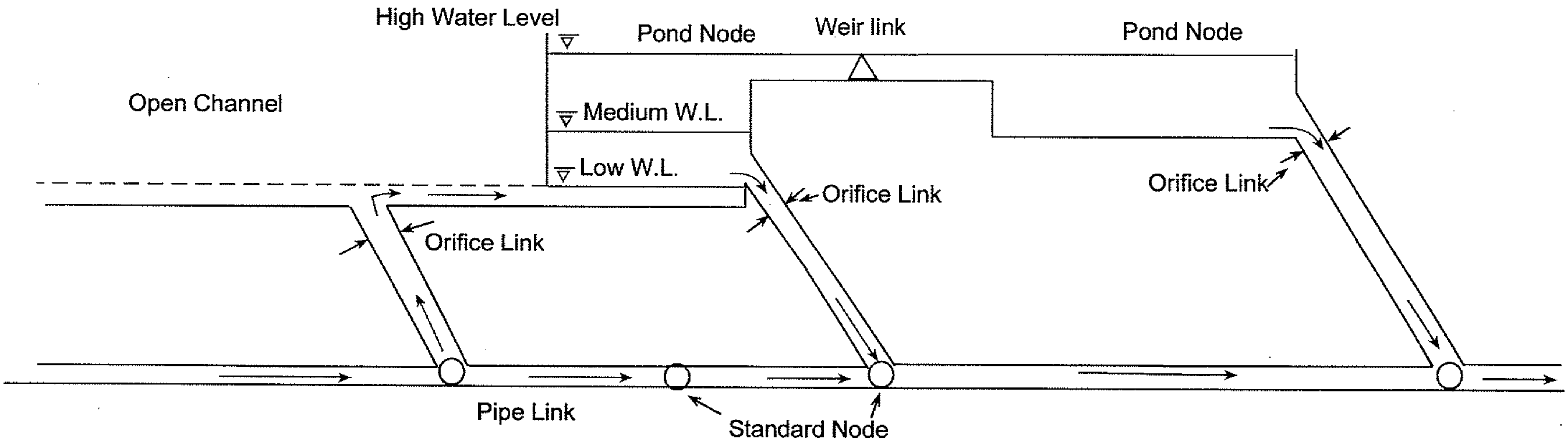


FIGURE 3

Network/conduit representation of the central area

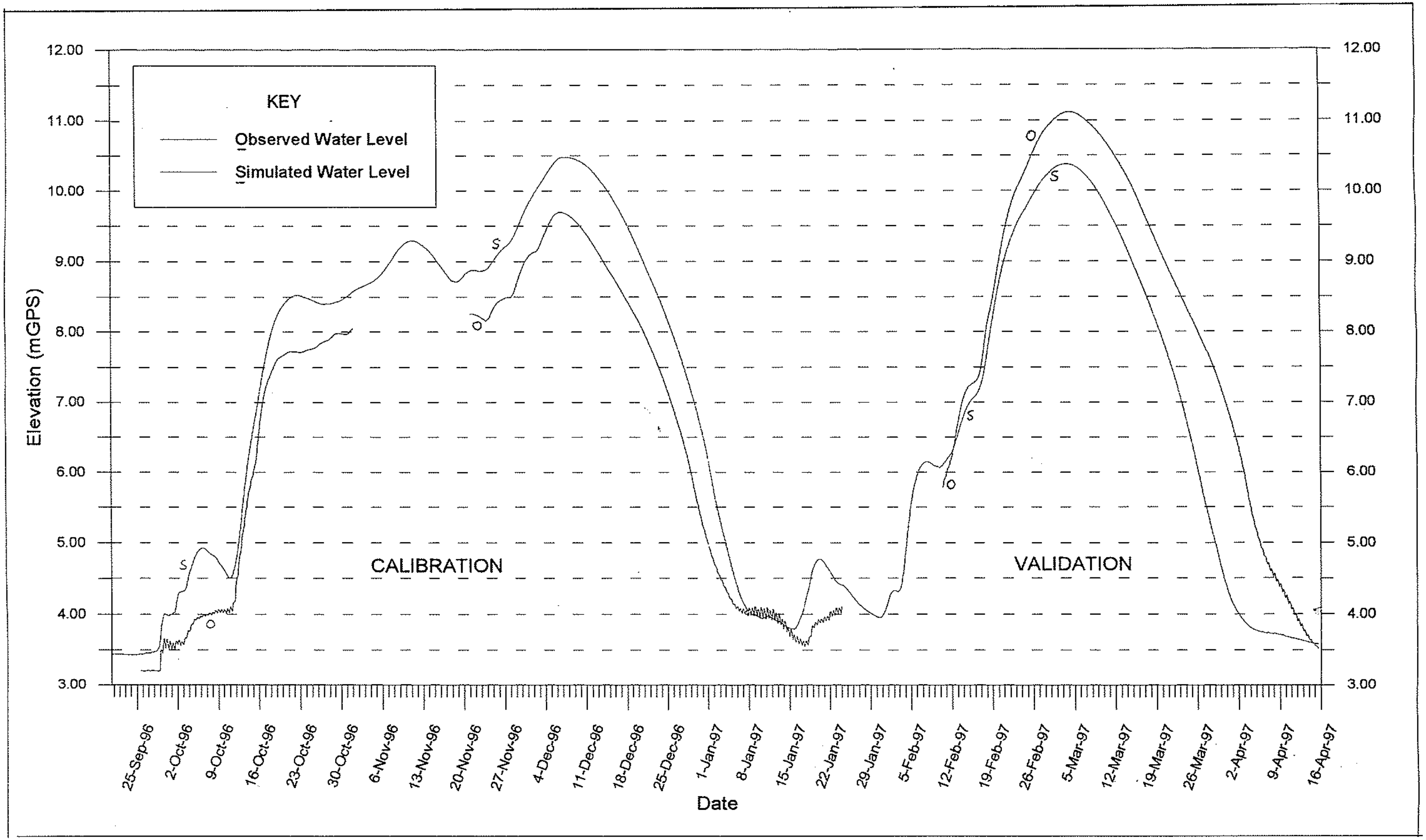


a, Schematic of Part of Typical Karst System



b, Representation in HYDROWORKS

FIGURE 4



Model Calibration and Validation Water Levels for Garryland North 21/09/96 - 16/04/97

FIGURE 5