INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS
(IRISH GROUP)

Presents

Groundwater: A Resource at Risk?

PROCEEDINGS

OF THE

IAH (IRISH GROUP)

28th ANNUAL GROUNDWATER CONFERENCE

Tullamore Court Hotel, Co. Offaly

Tuesday 22nd and Wednesday 23rd April 2008

2008 IAH (Irish Group) Tullamore Conference Organising Committee:

President: Paul Johnston, Trinity College Dublin
Secretary: Morgan Burke, Greenstar, Dublin
Treasurer: Orla Mc Alister, Tobin Consulting Engineers, Dublin
Burdon Secretary: Morgan Burke, Greenstar, Dublin.
Publicity & Education Secretary: Ulrich Ofterdinger, Queen's University, Belfast
Fieldtrip Secretary: Peter McConvey, EHS/GSNI, Dublin
Northern Region Secretary: David McLorinan, McLorinan Consulting, Belfast

***Conference Secretary*:** Pamela Bartley, Hydro-G, Galway.
***Conference Sub-Committee*:** Yvonne Cannon, RPS Group, Dublin; Michael Gill, Hydro Environmental Services, Waterford; Natalya Hunter Williams, Geological Survey of Ireland, Dublin; Niall Mitchell, RPS Group, Dublin; Suzanne Tynan, Jennings O'Donovan, Sligo; Ania Kuczyńska, Hydro-G, Galway.

The IAH would sincerely like to thank Linda McHugh of the Department of Civil, Structural and Environmental Engineering, Trinity College Dublin, for her help and efficiency in administering the Conference registration.

Proceedings cover designed by Taly Hunter Williams

Published by: International Association of Hydrogeologists (Irish Group)

ISSN – 1393-1806.

Annual seminar - International Association of Hydrogeologists. Irish Group

Additional copies of these Proceedings can be purchased for €45 (€25 if requested by a Conference delegate) (plus postage €3) by application to the Education and Publicity Secretary. For contact details, see the IAH (Irish Group) website: http://www.iah-ireland.org/

Printed by Snap Printing, Rampart Lane, Donnybrook, Dublin 4

www.iah-ireland.org

www.iah-ireland.org

PROGRAMME DAY I - TUESDAY 22nd APRIL

09.30 **Registration & Tea, Coffee, & Exhibits**

10:30 **Welcome and Introduction**
Paul Johnston, President,
Irish Group of the IAH

10.45 **Ecohydrology & Groundwater Dependent Terrestrial Ecosystems**
Prof Okke Batelaan,
University of Brussels

SESSION I – Legal Framework

11:20 **Legal Issues on Groundwater and Sustainability in Ireland**
Dr Yvonne Scannell, Department of Law, TCD & Arthur Cox Solicitors

11.50 **Legal Aspects peculiar to Groundwater Development**
Dr Peter Howsam, Reader in Water & Environmental Law, Cranfield University

12:20 **Groundwater Supply: Local Authority Experience**
Kieran Madden, Senior Engineer, Water Services, Roscommon County Council

12.50 Panel Discussion

13.00 Buffet Lunch & Exhibits

SESSION II – Groundwater Health

14:00 **Tracking the Source of Faecal Contamination of Water**
Prof. Martin Cormican, Professor of Clinical Mircobiology, and Siobhan Dorai-Raj, Environmental Change Institute, NUI Galway

14:30 **Microbial Communities in Groundwater and their Potential use for Biomonitoring**
Dr Nico Goldscheider, University of Neuchâtel Switzerland

15.00 **An Overview To Human Health Risk Assessment Tools And Application To Assessing Chronic Risks To Human Health From Contaminated Shallow Water**
Yolande Macklin, Atkins (UK)

15:30 Panel Discussion

15:45 Tea, Coffee & Exhibits

SESSION III – Contaminated Land

16:15 **Remediating Environmental Damage under the Environmental Liability Directive: a competent authority strategy**
Johnathan Derham, Environmental Protection Agency

16:45 **The use of 'Magic Numbers' in the assessment of contaminated land in Ireland**
Kevin Cleary, White Young Green

17.15 **Enhanced bioremediation and chemical oxidation of chlorinated organic compounds in groundwater**
Sean Moran, O'Callaghan Moran & Associates, and Dr. Tevfik Arguden, Eric Nichols

17:45 Panel Discussion

18:00 Wine Reception Sponsored by Alcontrol

8 pm Evening Social Event

PROGRAMME DAY 2 - WEDNESDAY, 23rd APRIL

SESSION IV – Groundwater & Infrastructure

9:15 **Construction Dewatering for Basements in Gravels**
Dr Toby Roberts, WJ Groundwater ltd.

9:45 **The Symbiotic Relationship between Groundwater and Geotechnical Engineering**
Gordon Clarke, QUB

10:15 **Groundwater Impacts from Engineering Projects**
Martin Preene, Golder Associates (UK)

10:45 Coffee, Tea and Exhibits

SESSION V – GW & Infrastructure continued

11.15 **The Importance of Understanding Recharge When Considering Groundwater Risk Assessment**
Shane Herlihy, RPS

11.45 **Planning, Development & Hydrogeology**
Mary Cunneen, Senior Inspector,
An Bórd Pleanála,
David Ball, Hydrogeologist

12:15 Panel Discussion

12:40 Closing Address - Paul Johnston

12.45 Buffet Lunch

www.iah-ireland.org

www.iah-ireland.org

www.iah-ireland.org www.iah-ireland.org

INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS

IRISH GROUP

28th Annual Conference

Groundwater: A Resource at Risk?

TULLAMORE COURT HOTEL
TULLAMORE
Co. OFFALY

Tuesday 22nd & Wednesday 23rd April, 2008

WHO SHOULD ATTEND?

It is expected that these topical issues will be of great interest to local authority engineers and scientists, planning officials, environmental scientists, public health officials, consultants, architects and hydrogeologists.

The International Association of Hydrogeologists (IAH) was founded in 1956 to promote co-operation amongst hydrogeologists, to advance the science of hydrogeology world wide, and to facilitate the international exchange of information on groundwater. The IAH is a worldwide scientific and educational organisation with more than 3,500 members in 135 countries.

The Irish Group of the IAH was started in 1976 and has over 130 members. It hosts a well-attended, annual groundwater conference in the Irish Midlands, and holds technical discussion meetings on the first Tuesday of every month between October and June, in the Geological Survey of Ireland in Dublin. The following members are serving on the 2008 IAH (Irish Group) Committee:

President: Paul Johnston, Trinity College Dublin	☎(01) 896 1372
Secretary: Morgan Burke, Greenstar	☎(086) 388 7909
Treasurer: Orla Mc Alister Tobin Consulting Engineers	☎(01) 803 0401
Fieldtrip Secretary: Peter McConvey DoENI	☎+4428 9038 8445
Education Publicity: Ulrich Ofterdinger, Queen's University Belfast	☎+442890 974 517
Burdon Secretary: Morgan Burke, Greenstar	☎(086) 388 7909
Northern Secretary: David McLorinan, McLorinan Consulting, Belfast	☎+44 2890 454 738
Conference Secretary: Pamela Bartley, Hydro-G	☎(091) 704848

Conference Subcommittee:

Taly Hunter Williams, GSI	☎(01) 678 2780
Yvonne Cannon, RPS Group Ltd.	☎(01) 288 4499
Niall Mitchell, RPS Group Ltd.	☎(01) 288 4499
Michael Gill, Hydro Environmental Services	☎(058) 44122
Suzanne Tynan, Jennings O Donovan	☎(071) 9129292
Ania Kuczynska, Hydro-G	☎(091) 704841

Please contact Conference Secretary Pamela Bartley for all Conference Enquiries

CONFERENCE OBJECTIVE

The theme of the 28th Annual Conference of the IAH (Irish Group) is the assessment of risk as applied to groundwater. Public awareness of damage to groundwater resources has become more acute in recent times arising from cases of cryptosporidium in water supplies, reports of increasing biological contamination of private well waters and reported environmental impact of road construction on wetlands and all in the face of generally increasing water demand. Management of these issues depends on the evaluation of risk which the various hazards represent. Risk can be said to describe the threat that a hazard presents to a receptor or target. The concept of risk also underlies much environmental legislation and regulation, including that relating to groundwater. While the ultimate receptor or target at risk is often assumed to be human, the target may also be seen in ecological terms or as the groundwater body itself. The threat may also be to the yield of an aquifer or to the quality of the groundwater, which in turn may affect downstream receptors. Such risks to groundwater have long been recognised but many are becoming more acute in the light of changing conditions such as growth in infrastructure construction and in population patterns or intensification of agriculture, apart from the effects of any future changes in climate. Moreover, the evaluation of risk may be from an economic perspective, from a legal standpoint or from a human or ecological health point of view.

The conference is therefore divided into topical sessions:

- Legal Aspects of Groundwater Management
- Groundwater and ecological risk
- Health Risks and Groundwater Quality
- Contaminated Land: Risk Assessment by Regulators & Consultants
- Groundwater Management & Engineering Projects.

Hydroecology is a growing field of endeavour and will be introduced from an international perspective. Groundwater differs from surface water in the way in which it is viewed legally – which is the subject of one session. The groundwater pathways in relation to quality and human health are not so well understood and the session on this risk assessment is both from the human health standpoint as well as from the perspective of groundwater protection. The evaluation of risk to groundwater and the criteria to be used in the context of remediating contaminated land is assigned to another session. Finally, engineering infrastructure, such as roads and quarries, presents its own risk assessment issues, which are still evolving in terms of understanding threats to groundwater.

www.iah-ireland.org www.iah-ireland.org

Table of Contents

.

Introduction

ECOHYDROLOGY AND GROUNDWATER DEPENDENT TERRESTRIAL ECOSYSTEMS

Prof. Dr. Okke Batelaan (1) and J.P.M. Witte (2)

(1) Department of Hydrology and Hydraulic Engineering, Vrije Universiteit Brussel, Pleinlaan 2, 1050 Brussels, Belgium; e-mail: batelaan@vub.ac.be; and Department of Earth and Environmental Sciences, K.U.Leuven, Celestijnenlaan 200e - bus 2410, 3001 Heverlee, Belgium; e-mail: Okke.Batelaan@ees.kuleuven.be

(2) Kiwa Water Research, PO Box 1072, 3430 BB Nieuwegein, The Netherland; e-mail: Flip.Witte@kiwa.nl

ABSTRACT

Throughout the world ecohydrology has lately been discovered, and tightly embraced, as a new scientific discipline. Several authors have stressed its importance to the progress of hydrology and ecology but there appears to be a wide range of ideas on the topics ecohydrology is supposed to include. Elements of the history of ecohydrology are described here and different ecohydrologic schools are distinguished. One of the roots of ecohydrology is based on the dependence of phreatophytic plant species on groundwater. In the first half of the 20th century plants were regularly used as indicators in groundwater investigations by hydrologists. More recent the interest in phreatophytes in general revived again, following the interest in groundwater dependent ecosystems. A case study is used to show the benefit of use of phreatophytes in hydrological studies. It is argued that a well balanced use of 'soft' phreatophytic information can be complementary to 'hard' groundwater data and analysis techniques and help to understand more profoundly groundwater dependent ecosystems.

INTRODUCTION

Throughout the world ecohydrology has lately been discovered, and tightly embraced, as a new scientific discipline. Several authors have stressed its importance to the progress of hydrology and ecology but there appears to be a wide range of ideas on the topics ecohydrology is supposed to include. Here elements of the history of ecohydrology are described and different ecohydrological schools are distinguished.

In arid regions there is a relatively simple relationship between plants and the occurrence of groundwater. In more humid areas the situation is much more complex: The vegetation, has especially in discharge areas lavish available water for its growth and transpiration, it specialises in a wide range of species adapted to different environmental site or local conditions. This results in highly valued groundwater dependent wetlands with a high biodiversity, which is the main reason for their protection. However, mankind is changing the hydrological system and consequently the site conditions and hence plant occurrences. Hence, the scientific challenge is what role can plants play in the study of groundwater?

ECOHYDROLOGY DEFINED

In the last 10 years several definitions have been published of what ecohydrology is supposed to mean.

- Wassen and Grootjans (1996): 'An application driven discipline aiming at a better understanding of hydrological factors determining the natural development of wet ecosystems, especially in regard of their functional value for natural protection and restoration'.
- Baird and Wilby (1999): 'Eco-hydrology is the study of plant-water interactions and the hydrological processes related to plant growth'.
- Rodriguez-Iturbe (2000): 'Eco-hydrology seeks to describe the hydrological mechanisms that underlie ecological pattern and processes'.
- Nuttle (2002): 'Eco-hydrology is … concerned with the effects of hydrological processes on the distribution, structure and function of ecosystems, and on the effect of biological processes on the elements of the water cycle'.

Since 2000 ecohydrology in hydrological literature tends to be dominated by dryland hydrology, that means soil moisture limited evapotranspiration processes. Eagleson and Rodriguez-Iturbe are the main authorities in this version of ecohydrology (Rodriguez-Iturbe and Porporato, 2004; Eagleson, 2002). Did ecohydrology as it appears from these recent references pop out of the sky? To investigate the roots of ecohydrology or its 'founding father' it is very useful to look back in time, following the geological principle of the past is the key to the future.

Pre-historic man must have had some ecohydrological consciousness, since he was able to recognise plants to warn him against dangerous places where he could drown, or to find food. However, he did not publish his observations and hence he was not a scientist and therefore he cannot be regarded as the founding father of ecohydrology. Ross (2007) interprets and translates the Hebrew bible text of Isaiah 44 in modern language as: 'I will pour out My spirit as suddenly and overwhelmingly as a rainstorm in the desert. After such a storm, the willow does not fade like grass, but is kept green for many years by groundwater that recharges in the storm'. Obviously, the prophet made accurate observations relating rainfall-recharge-groundwater and plant species occurrence. Vitrivius, roman architect and engineer in the 1st century published the following remark concerning exploration of drinking water: 'One of the indications where groundwater can be found is the occurrence of small rushes, willows, alder, vitex, reeds and ivy'. It is significant to notice that he remarks: 'one must not rely on these plants if they occur in marshes, which receive and collect rain water'. Hence, he was well aware of the relativity of the plants as indicators for good quality groundwater.

In the famous work of Darcy (1856) is, besides the well known column tests in the appendix, also a description of the search for drinking water by the spring seeking 'Father Paramel'. It is written that he infers from the nature and strength of the plants, the probable presence of water, and even the approximate depth of the water below the ground surface. Schimper (1898) made a difference between wet, hygrophyte and dry, xerophyte plant species. The important difference lies in the physiology: if a soil contains too much salt the plants cannot absorb the water and hence it is physiologically dry. All soils which are physically dry are also physiologically dry; and hence only the physiological dryness or wetness of soils need be considered in ecology. Schimper used the term xerophytes to include plants, which live in soils which are physiologically dry, and the term hygrophytes those which live in soils which are physiologically wet or damp.

Oscar Edward Meinzer (1923), the father of modern groundwater hydrology, was the first to define the term phreatophyte as a plant that habitually obtains its water supply from the zone of saturation. In 1927 he wrote a whole book about these phreatophytes. He describes the principle phreatophytic species, like common salt grass (Distilchlis spicata) and their occurrence in the arid and semi-arid regions of the US (Meinzer, 1927). In these days plants were for groundwater hydrologists clearly indicators for locations of groundwater resources. After the first half of the 20th century it seems that hydrogeologists lost their interest in the use of phreatophytes in groundwater studies, however ecologists continued the study of their habitat requirements (Londo, 1988; Ellenberg et al., 1992).

Phytosociologists started in the 1950's the research on the relationship between vegetation types and groundwater dynamics. Ellenberg (1948, 1950, 1952, 1953, 1974) and Tüxen (1954) were the first to systematically study the relationship between groundwater level and the occurrence of vegetation types.
The first publication in which the word 'ecohydrology' is mentioned is from the Dutch author van Wirdum in 1982 (van Wirdum, 1982). It is an annual report of the activities in 1981 of the Ecohydrology section of the Dutch national institute for nature research. Frequently used in ecohydrology is his simple and elegant diagram of electrical conductivity versus ionic ratio in which a groundwater sample can be plotted to infer directly its position in the hydrological cycle between rainwater (atmotrophic water), groundwater (lithotrophic water) and seawater (thalossotrophic water). More recently, the interest in phreatophytes in general has revived, following the interest in groundwater dependent ecosystems (Batelaan et al., 2003a; Witte and von Asmuth, 2003, Loheide II et al., 2005).

This very short overview did not give tribute to many important contributions like the pioneering work of Russians and other scientists on bogs and fens. However, what this tells us is that ecohydrology and especially the groundwater versus plant species relationship is not new. Its scientific content has grown over the ages, while the recognition of its scientific importance in the wider hydrological community is only now realized. It also tells us that potentially a lot of interesting and useful information for groundwater studies is contained in the ecological knowledge. Ecologists build more and more complex vegetation prediction models based on groundwater level and chemistry dynamics: for understanding the differences in groundwater chemistry and levels more hydrogeological support is urgently needed.

APPLICATION: LINKING VEGETATION, GROUNDWATER FLOW AND GEOCHEMISTRY

The relationships between soil, water characteristics and nature quality (i.e. diversity of vegetation) of three Flemish groundwater dependent wetlands were examined (Huybrechts et al., 2000). These wetlands are the Doode Bemde in the valley of the Dijle River, Vorsdonkbos in the valley of the Demer River, and Zwarte Beek Valley along the Zwarte Beek River, a tributary of the Demer River (Fig. 1). Large parts of these wetlands are groundwater saturated for most of the year, therefore they are mainly occupied by phreatophytic vegetation types such as reed lands, brook forests, sedges, etc. It is observed that there is a large diversity in vegetation types between the areas (Fig. 2). While the Doode Bemde is mainly dominated by reed and grasslands, it appears that Vorsdonkbos has a lot of brook forests and large sedges and Zwarte Beek is dominated by smaller sedges. Since regional land use, soil and climate is not significantly different, it is hypothesized that these vegetation differences are due to differences in groundwater fluxes and qualities. A groundwater modelling exercise was performed to investigate the differences between the areas with respect to the connected groundwater system.

The groundwater seepage in all three wetlands is sourced from recharge in the surrounding hills. Subsequently, it moves through sandy aquifers towards the wetlands. In the Doode Bemde these aquifers belong to the Brussels Formation (Eocene). In the Valley of the Zwarte Beek they belong to the Diest Formation (Miocene) and in the Vorsdonkbos to both. Batelaan et al. (2003b) describe the groundwater model for the area in detail. The recharge for the model was simulated on basis of distributed land use, soil, topography and hydrometeorology with the spatially distributed WetSpass modelling procedure (Batelaan and De Smedt, 2007). The discretisation used for the groundwater modelling was 20 m. The level at which the groundwater will seep at the land surface, in drainage ditches or wetlands is defined as the maximum seepage level. This level has been determined by way of an Arc/Info Topogridtool interpolation of contour lines of 1:10,000 scale topographic maps. Locally, in the study area, measured topographic levels were also included in this interpolation, as well as a high resolution topographic database of the Demer valley obtained from aerial laser altimetry. The USGS modular three-dimensional finite difference groundwater model, MODFLOW (Harbaugh and McDonald, 1996) has been used to simulate the groundwater flow extended with a

SEEPAGE package (Batelaan and De Smedt, 2004) to accurately delineate the groundwater discharge areas. A MODPATH (Pollock, 1994) simulation was performed to determine by particle tracking the recharge area and flow times.

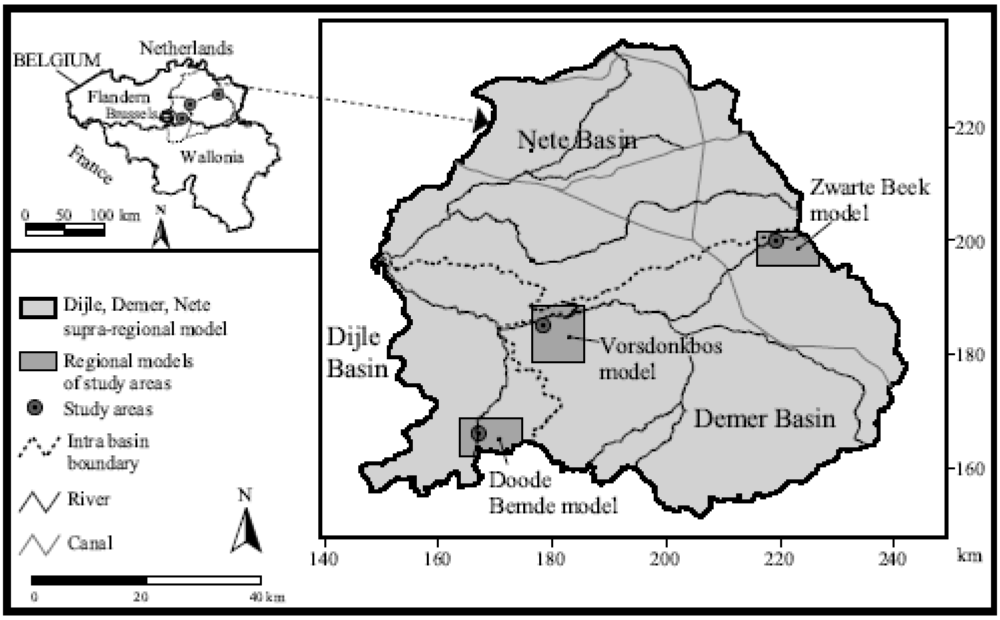


Fig. 1: Location of the three study areas and their regional groundwater models within the supra-regional model for the Dijle, Demer and Nete Basin.

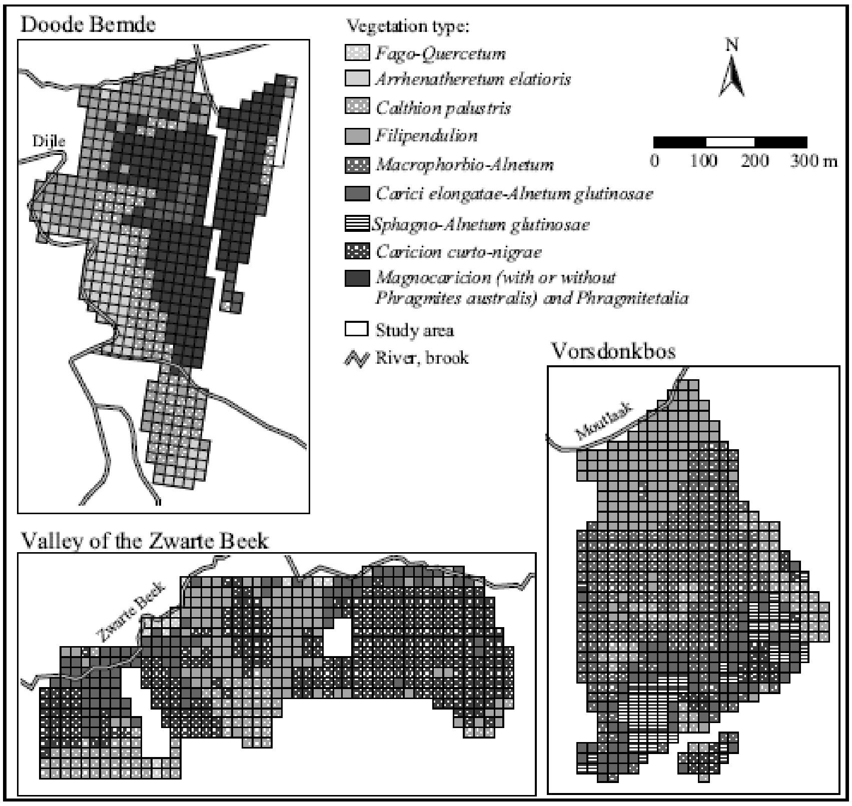


Fig. 2: Vegetation types, derived from cluster analysis of species mapping, for the study areas Doode Bemde, Vorsdonkbos and Valley of the Zwarte Beek.

RESULTS AND DISCUSSION

Figure 3 shows for the three study areas, the calculated groundwater discharge areas, while Fig. 4 shows the simulated recharge areas and flow times of the discharge areas. The sizes of the study areas and the discharge zones in each area are very similar. The average discharge flux however varies much more due to the strongly varying size of the recharge areas and the average flow times from recharge to discharge area. If the discharge map (Fig. 3) is compared to the vegetation map (Fig. 2) it is clearly observed that the patterns of the discharge correlate well with the patterns of phreatophyte occurrence. However, it does not explain the diversity of the vegetation.

The shallow groundwater quality (Fig. 5) on the other hand clearly shows that the three groundwater dependent wetlands receive groundwater with quite different qualities. The acidic groundwater type 1a occurs only along the hill side of the wetlands, the comparable (but less acidic) type 1b also more inside the valleys. Both types are dominant in Vorsdonkbos, and calcium is the major cation. It is counteracted equally by chloride, bicarbonate and sulphate. In groundwater types 2, 3 and 4 calcium and bicarbonate dominate, but these types differ in total ionic concentration, acidity (pH), and the significant sulphate concentration in groundwater type 4. Groundwater type 2 has the lowest ionic concentration of all, type 4 the highest. The acidic groundwater type 2 dominates in the Zwarte Beek Valley, the more neutral, calcareous groundwater type 3 in the Doode Bemde. Groundwater type 4 is found in the Doode Bemde, but also in the Vorsdonkbos.

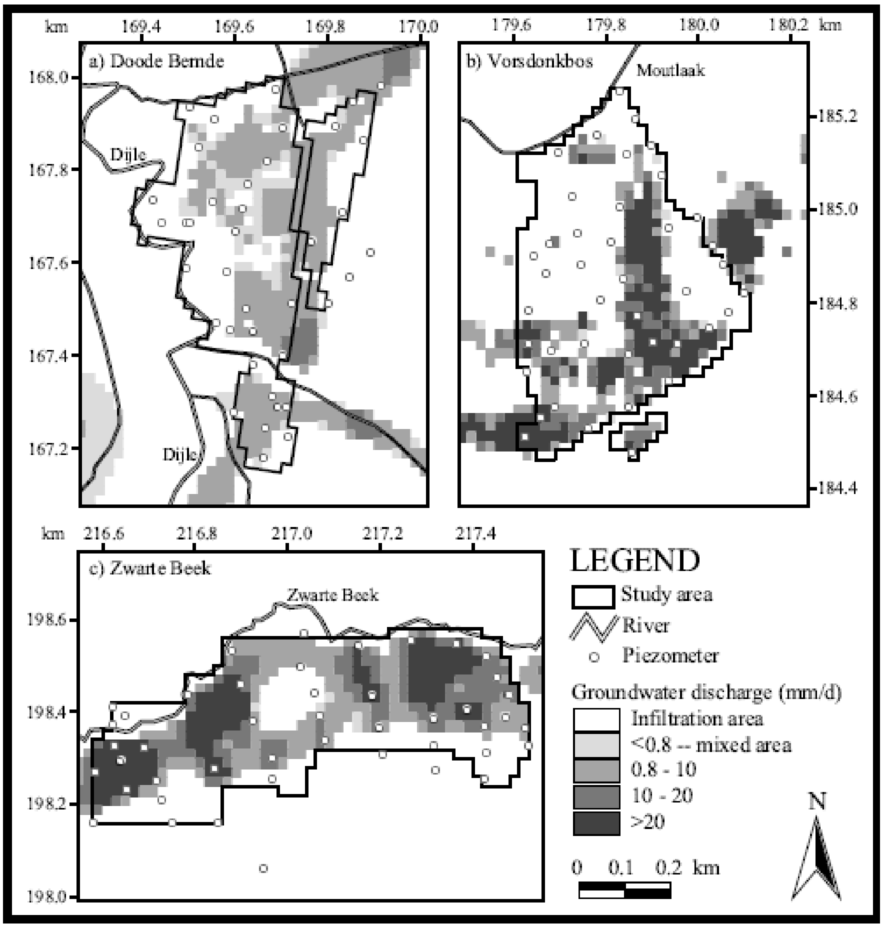


Fig. 3: Simulated discharge areas and fluxes in a) Doode Bemde, b) Vorsdonkbos and c) Zwarte Beek.

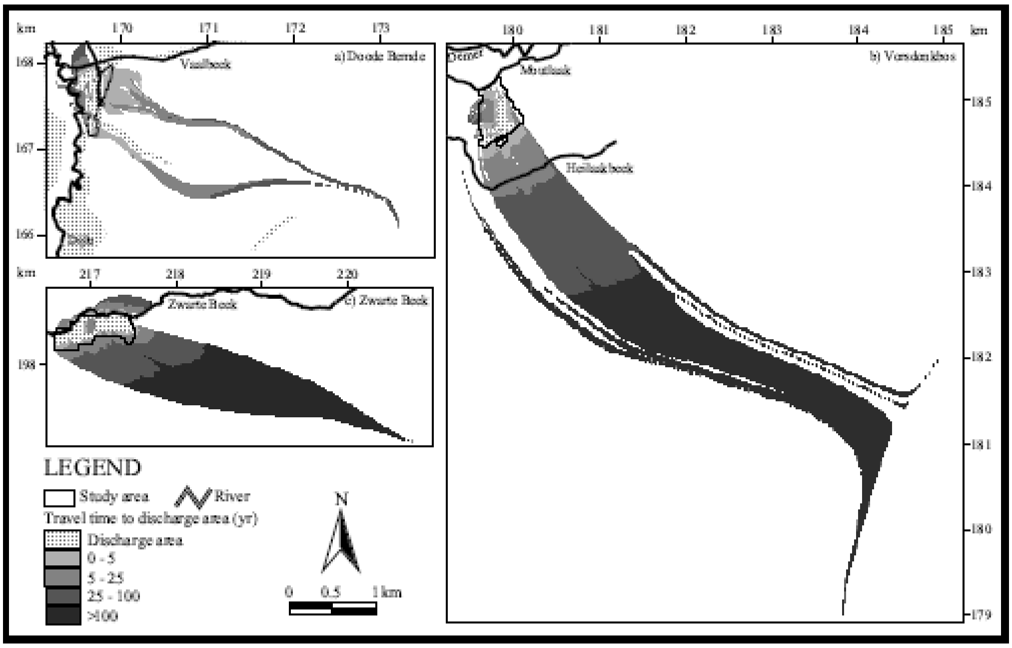


Fig. 4: Simulated groundwater flow systems and travel times from recharge to discharge location for: a) Doode Bemde, b) Vorsdonkbos, and c) Zwarte Beek study area.

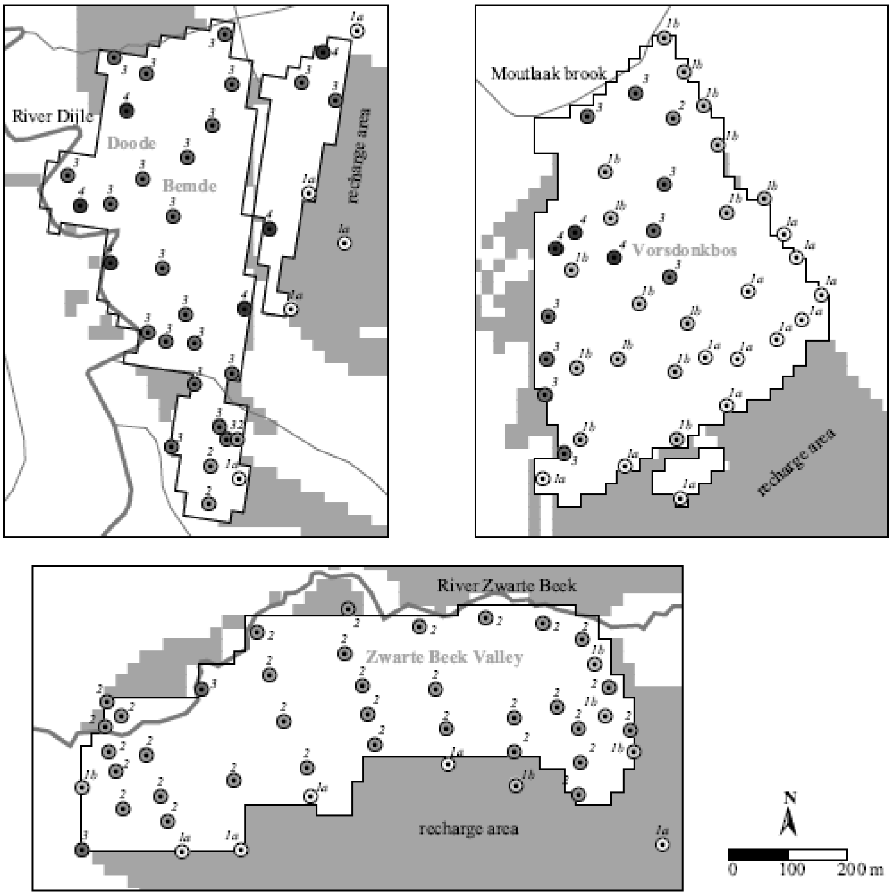


Fig. 5: Distribution of the four groundwater types (indicated by 1a, 1b, 2, 3 and 4) in piezometers of the Doode Bemde, the Vorsdonkbos, and the Zwarte Beek Valley.

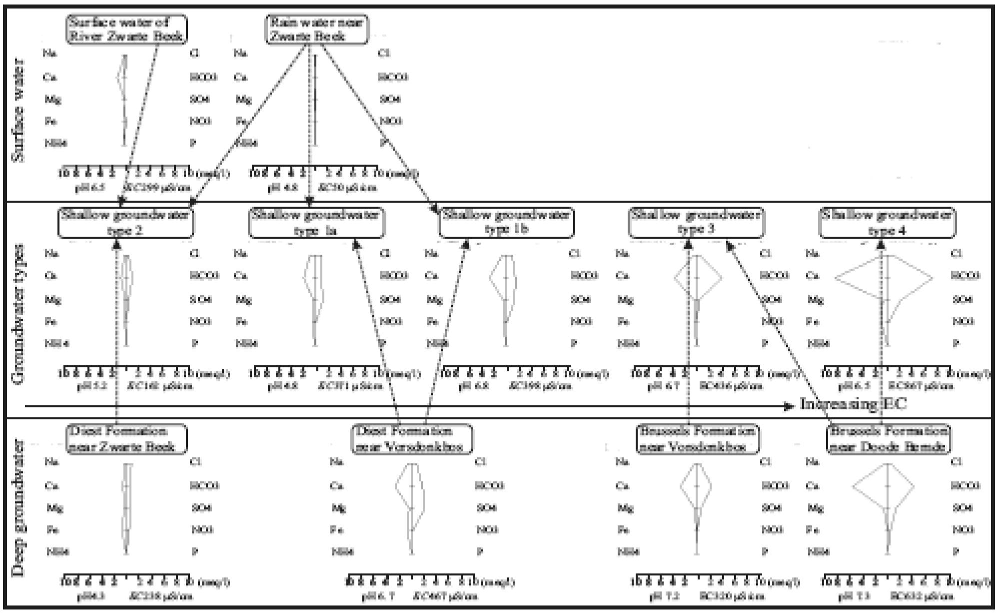


Fig. 6: Stiff diagrams for the groundwater types in the wetlands, the deeper groundwater in the aquifers, the surface water in the River Zwarte Beek, and the local rain water near the Zwarte Beek Valley (Huybrechts et al., 2000).

Figure 6 shows that the cause of the varying shallow groundwater quality lies in the geochemical composition of the feeding aquifers. Interaction between the flowing water and the porous media of the Diest or Brussels Formations appear to have a major impact on the resulting shallow water quality. Van Rossum et al. (2000) shows that the mineral reactivity determines the possibility for dissolution of minerals in the groundwater and that flow time and distance is of secondary importance. The Brussels Formation contains more soluble minerals than the Diest Formation and is the main aquifer for the Doode Bemde area, while for Vorsdonkbos it is one of the two feeding aquifers. The Diest Formation also feeds Vorsdonkbos, and it is, as well, the main contributor to the valley of the Zwarte Beek. Together with groundwater, which is very little mineralized, having atmotrophic qualities from the very short flow paths and times, the vegetation in these different wetlands is, nevertheless, highly determined by the groundwater discharge from these qualitatively different sources.

CONCLUSIONS

The investigated vegetation diversities are mainly determined by regional factors such as topography, hydrology (recharge areas and groundwater-flow times) and hydrogeochemistry (mineral reactivity in the aquifers). Soil moisture dynamics for the groundwater dependent wetlands is of much less importance.

Important is that it is shown that by synthesizing data and methods from different fields of sciences (i.e. ecology and hydrology) new insights in the functioning of ecosystems can be obtained. It is, therefore, in line with Harte (2002), advocated that more integration of ecological and hydrological sciences will benefit problems in earth system sciences.

REFERENCES

Batelaan, O., De Smedt, F. and L. Triest, 2003a. Regional groundwater discharge: phreatophyte mapping, groundwater modelling and impact analysis of land-use change. Journal of Hydrology, 275(1-2): 86-108.

Batelaan, O., Asefa, T., van Rossum, P. and F. De Smedt, 2003b. Groundwater flow modeling of three wetland ecosystems in river valleys in Flanders, Belgium. In: Verhoest, N.E.C., Hudson, J. and De Troch, F.P. (eds), Monitoring and Modeling Catchment Water Quantity and Quality, Proceedings of 8th Conference of the

European Network of Experimental and Representative Basins (ERB), Ghent (Belgium), 27-29 September, 2000, UNESCO IHP-VI Technical Documents in Hydrology 66, 1-7.

Batelaan, O. And F. De Smedt, 2004. SEEPAGE, a new MODFLOW DRAIN Package. Ground Water 42 (4), 576-588.

Batelaan, O. and F. De Smedt, F., 2007. GIS-based recharge estimation by coupling surface-subsurface water balances. Journal of Hydrology, 337(3-4), 337-355.

Baird, A.J. and R.L. Wilby, 1999. Eco-hydrology: plants and water in terrestrial and aquatic environments. Routledge, London.

Darcy, H.P.G., 1856. The public fountains of the city of Dijon. English translation by P. Bobeck, 2004. Kendall/Hunt Publishing Company, Dubuque, Iowa, 506 pp.

Eagleson, P.S., 2002. Ecohydrology: Darwinian expression of vegetation form and function. Cambridge University Press, Cambridge.

Ellenberg, H., 1948. Unkrautgesellschaften als Mass für den Säuregrad, die Verdichtung und andere Eigenschaften des Ackerbodens. Ber. Landtechnik 4: 2–18.

Ellenberg, H., 1950. Landwirtschaftliche Pflanzensoziologie. I: Unkrautgemeinschaften als Zeiger für Klima und Boden. Stuttgart.

Ellenberg, H., 1952. Landwirtschaftliche Pflanzensoziologie. II: Wiesen und Weiden und ihre standörtliche Bewertung. Stuttgart.

Ellenberg, H., 1953. Physiologisches und ökologisches Verhalten derselben Pflanzenarten. Ber. Deutsch. Bot. Ges. 65: 351–361.

Ellenberg, H., 1974. Zeigerwerte der Gefässpflanzen Mitteleuropas. Scripta Geobotanica 9: 1–97.

Ellenberg, H., Weber, H. E., Düll, R., Wirth, V., Werner, W. and D. Pauliszen, 1992. Zeigerwerte der Gefässpflanzen (ohne Rubus). Scripta Geobotanica, Verlag Erich Goltze, Göttingen, 9-166.

Harbaugh, A.W., McDonald, M.G., 1996. User's documentation for MODFLOW-96, an update to the U.S. Geological Survey modular finite difference groundwater flow model, Open-File Report 96-485, 56.

Harte, J., 2002. Toward a synthesis of the Newtonian and Darwinian worldviews. Physics Today, 55(10), 29-34.

Huybrechts, W., Batelaan, O., De Becker, P., Joris, I. and P. van Rossum, 2000. Ecohydrological research of wetland ecosystems in river valleys in Flanders (in Dutch). Final report VLINA project C96/03, Institute of Nature Conservation, Brussels, Belgium.

Loheide II, S.P., Butler, J.J.J. and Gorelick, S.M., 2005. Estimation of groundwater consumption by phreatophytes using diurnal water table fluctuations: A saturated-unsaturated flow assessment. Water Resources Research, 41(7): W07030.

Londo, G., 1988. Nederlandse freatofyten. Pudoc, Wageningen, 107 pp.

Meinzer, O.E., 1923. Outline of ground-water hydrology with definitions. U.S. Geol. Surv. Water Supply Paper, 494: 71.

Meinzer, O.E., 1927. Plants as indicators of ground water. U.S. Geological Survey Water-Supply Paper 577.

Nuttle, W.K., 2002. Is ecohydrology one idea or many? Discussion. Hydrological Sciences Journal, 47(5): 805-807.

Pollock, D. W., 1994. User's guide for MODPATH/MODPATH-PLOT, Version 3: A particle tracking postprocessing package for MODFLOW, the U.S. Geological Survey finite difference groundwater flow model. Reston, Virginia, U.S. Geological Survey.

Rodriguez-Iturbe, I., 2000. Ecohydrology: A hydrologic perspective of climate-soil-vegetation dynamics. Water Resources Research, 36(1): 3-9.

Rodriguez-Iturbe, I., and A. Porporato, 2004. Ecohydrology of Water Controlled Ecosystems: Soil Moisture and Plant Dynamics. Cambridge University Press, Cambridge.

Ross, B., 2007. Phreatophytes in the Bible. Ground Water 45(5): 652-654.

Schimper, A.F.W., 1898. Pflanzen-Geographie auf physiologischer Grundlage. Fischer, Jena.

Tüxen, R., 1954. Pflanzengesellschaften und Grundwasser-Ganglinien. Angewandte Pflanzensoziologie, 8, 64-97.

van Rossum, P., Asefa, T., Batelaan, O. Huybrechts, W., De Becker, P. And F. De Smedt, 2000. Hydrogeochemistry of three wetland ecosystems in river valleys in Flanders, Belgium. In: Verhoest, N.E.C., Van Herpe, Y.J.P. and De Troch, F.P. (eds), Book of abstracts of European Network of Experimental and Representative Basins (ERB) Conference: Monitoring and Modeling Catchment Water Quantity and Quality, Ghent, Belgium, 27-29 September, 2000, 65-69.

Van Wirdum, G., 1982. The ecohydrological approach to nature protection, Annual report 1981, Research institute for nature management, Arnhem, Leersum and Texel.

Wassen, M.J. and A.P. Grootjans, 1996. Ecohydrology: an interdisciplinary approach for wetland management and restoration. Vegetatio, 126: 1-4.

Witte, J.P.M. and von Asmuth, J.R., 2003. Do we really need phytosociological classes to calibrate Ellenberg indicator values? Journal of Vegetation Science, 14(4): 615-618.

Session I

LEGAL ISSUES ON GROUNDWATER AND SUSTAINABILITY IN IRELAND

Dr Yvonne Scannell, Arthur Cox, Solicitors[1], Earlsfort Plaza, Earlsfort Terrace, Dublin 2.

Yvonne.scannell@arthurcox.com

ABSTRACT

This article briefly describes aspects of the law peculiarly relevant to groundwater, refers to some concepts and arrangements which should be re-examined in the light of modern scientific knowledge and obligations under EU legislation and describes some of the legal lacunae and deficiencies which militate against the establishment of a robust management system in Ireland to protect groundwater resources and to ensure their sustainable management and wise use.

INTRODUCTION

Groundwater defined in article 2 of Directive 2000/60/EC (the Water Framework Directive) as all water beneath the surface of the ground in the saturation zone and in direct contact with the ground and subsoil. An aquifer is defined in the same article as an underground layer of rock or geological strata of sufficient porosity and permeability to allow a significant flow of groundwater or the abstraction of significant quantities of groundwater. The definition of an aquifer in Irish water pollution law is a little different. Section 1 of the Local Government (Water Pollution) Act 1977 defines an aquifer as "any stratum or combination of strata which stores or transmits groundwater". This definition is a bit wider. About 30% of people in Ireland are dependent on groundwater for drinking water supplies. This percentage is likely to increase as sources of surface water dry up or are exploited.

The purpose of my paper is to outline some of the legal issues involved when trying to ensure the sustainable use and development of groundwater. This is a vast topic so only an outline can be given of elements of the relevant legal framework. At the outset it must be stressed that the common law on groundwater is very uncertain. Most of it developed in the 19th century when conditions were every different to today and most of it is English law. It is not at all certain that our courts would follow some of the English precedents.

WHO OWNS GROUNDWATER?

WATER IN DEFINED CHANNELS

Running water is ownerless. There are two reasons for this:

- water us an essential life resource and many consider that should not be owned,
- practicality, how can one assign ownership of individual molecules in a flowing stream?

However, although water is not owned, some people have property rights in water and rights to use water. The law also makes a clear distinction between water flowing in defined and known channels

[1] I have gratefully relied on the work of Brendan Slattery, Solicitor, Arthur Cox, and Rory Mulcahy B.L. in preparing this paper. The structure of the paper, cases cited and many of the ideas on the common law in it are substantially derived from Bryan Clark's excellent article, "Water Use Reform in Scotland: a critical analysis" *19 Journal of Environmental Law* 376 – 406 which is essential reading for all interested in this area.

and water flowing in undefined and/or unknown channels. The former type of water, whether flowing on the surface or beneath it is subject to riparian rights. Riparian rights are described as natural rights, and a riparian owner is entitled to the use of water flowing on his land. However, it is clear that no riparian rights exist in relation to water flowing in undefined and unknown channels.

At common law, landowners have unfettered rights to exploit water resources flowing in defined channels under the ground for domestic purposes. If the defined channel is underground, the person must prove this because there is a legal presumption that all underground water is percolating.[2] This right can be limited by servitude, liability considerations and by statute. There is no absolute right to exploit water for industrial purposes but a reasonable amount can be abstracted. There must be enough left for the uses of inferior proprietors[3] unless industrial abstraction rights have been acquired by prescription i.e. long use.

PERCOLATING WATER

There are no riparian rights to percolating water in undefined channels but a landowner may abstract as much of this water as he likes even if it deprives others of water.[4] So, in *Chasemore v Richards* the question was whether the Plaintiff had any cause of action against the Defendant, the local Board of Health, in circumstances where the Defendant in abstracting water from a well to supply water to the local town had caused the Plaintiff, a mill-owner, to lose the use of a stream which was fed by percolating underground water, and which had been used in the course of the mill-owner's business for over sixty years. The House of Lords held unanimously that he did not have any cause of action. Indeed a landowner may even abstract water maliciously. This happened in *Bradford v Pickles*[5] where a landowner drained percolating groundwater so that it did not reach another landowners well in an attempt to force the other to sell his land. The court found that the defendant's motive was irrelevant. Indeed although it found that Pickle's motives were "churlish", "selfish and grasping" and "shocking to the moral philosopher", it refused to prevent him exercising what it considered to be his property rights. *Chasemore* has been followed in this jurisdiction in a number of cases.[6] More recently, *Chasemore* has been applied by the High Court in England in the case of *Rugby Joint Water Board v Walters*[7], but, in so far as I am aware, there have been no similar cases in this jurisdiction.

There was some reference to the effect that development might have on water rights in *Scott v An Bord Pleanála* [1995] 1 ILRM 423. In *Scott*, the Court held that development within the meaning of the Local Government (Planning and Development) Act 1963 meant the carrying out of works on the land and not merely the consequences of those works, and therefore any adverse effects that the proposed development might have on the applicant's water rights did not constitute a basis for challenging the decision to grant planning permission for a lead and zinc mine.

WHY THESE RULES?

The law relating to the ownership of groundwater owes much to the Victorian notion that property rights are sacrosanct. It was also developed for practical reasons. Judges found it impracticable to have the same rule for percolating water and water in defined streams because of the difficulties in proving the nature and extent of percolating water.[8] Groundwater was too "unknowable" and "occult" in its

---

[2] *Black v Ballymena Township Commissioners* (1886) 17 LR Ir 459 at 474-475 per Chatterton VC.

[3] *Marquess of Breadalbane v West Highland Railway* [1885] 22 R 307 per Lord Ordinary (Wellwood) at 310.

[4] The authority for this proposition is the case of *Chasemore v Richards* [1859] 7 HLC 349, a case which Lord Wensleydale described in his judgment as "of the greatest importance… No question that has occurred in my time has been so worthy of the most careful consideration. "

[5] [1895] A.C. 587 HL.

[6] *Ewart v Belfast Poor Law Guardians* (1882) 9 LR (Ir) 172 and *Black v Ballymena Town Commissioners* (1886) 22 LR (Ir) 459 are often cited as Irish authority for the proposition put forward in *Chasemore*.

[7] [1967] Ch 397.

[8] Lord Cranworth picturesquely described the rationale for distinguishing between the situation of water percolating underground and that of water flowing in defined channels. "The right to running water has always

hydrological aspects. Determining the directional flow and volumes of groundwater was at that time impossible. So the easiest thing was to make a black and white rule that all would know and which would avoid litigation involving the unknowable. There were clear rules and a minimisation of disputes. The judges were reluctant to have "men of science" debating issues in their courts. Another reason was the courts' desire to facilitate rapid industrialisation in the 19th century - it would have placed enormous obstacles in the way of the developers of roads, railways, mines, and reservoirs if they could be made responsible for drawing away water from other landowners.

THE PROBLEM WITH COMMON LAW RULES

The common law rules are based on scientifically ignorant assumptions. It is wrong to distinguish percolating from surface water in hydrological terms because they are inextricably linked. The common law rules can be unfair to other water users who may have begun abstracting before a new abstractor who interferes with their abstraction. The common law rules may also operate inequitably because they have no regard to the needs or difficulties of other abstractors. Even the riparian approach (which has some regard to downstream users by not giving *carte blanche* to industrial abstractions who have not got prescriptive rights), can result in depriving a downstream user when too much water is abstracted for domestic uses. Moreover, the common law rules have no regard to the requirement of sustainability: a landowner can abstract without regard to the fact that he may be exhausting a groundwater resource.

If groundwater resources are to be protected from depletion, the law must, as Clark has argued, move from "property absolutism to property obligationalism." In other words, the nature of property rights over groundwater and the extent to which these can be regulated will have to be re-examined.

THE POSITION ELSEWHERE

The majority of countries today designate their water resources as being in public ownership, with government having the overall responsibility for resource management. The right to abstract (or divert) and use water (including groundwater) is granted to individuals, public entities or private corporations, under certain terms or conditions, and such rights are generally issued by the water resources authority or by the law courts directly. A 'water right' usually constitutes the right to use (but not ownership of) the water itself. Lawyers call this a 'usufructuary right'. Grants to abstract and use groundwater are instrumented through permits, licenses, concessions or authorisations, generally called here 'water rights'. A system of groundwater rights (permits to abstract and to use groundwater) is often first introduced as a means to reduce interference, avoid counterproductive conflicts and resolve emerging disputes between neighbouring abstractors. However, the development of a stable system of water rights has far wider benefits because it provides a sound foundation for the development and protection of water and its sustainable use and development.

IRELAND POST THE WATER FRAMEWORK DIRECTIVE

Obligations under the Water Framework Directive may soon require Ireland to legislate to prevent depletion of groundwater resources. Article 1 of the Directive seeks to prevent the further deteoriation of groundwater (something which will happen if the resource is depleted beyond its regenerating capacity) and it promotes the sustainable use of groundwater based on long-term protection of available water sources. Article 4 requires the achievement of good groundwater status by 2015. For the first

---

been properly described as a natural right, just like the right to the air we breathe; they are the gifts of nature, and no one has the right to appropriate them. There is no difficulty in enforcing that right, because running water is something visible, and no one can interrupt it whether he does or does not do injury to those who are above or below him. But if the doctrine could be applied to water merely percolating, as it is said, through the soil, and eventually reaching some stream, it would be always a matter that would require the evidence of scientific men, to state whether or not there had been an interruption, and whether or not there had been injury. It is a process of nature not apparent, and therefore such percolating water has not received the protection which water running in a natural channel on the surface has always received. If the argument of the Plaintiff were adopted, the consequences would be that every well that ever was sunk would have given rise, or might give rise, to a cause of action." (at p. 381)

[1] Hector Garduño1 Stephen Foster, Charles Dumars, Karin Kemper Albert Tuinhof: *Groundwater Abstraction Rights: from theory to practice.*

time, quality and quantity of surface and groundwater must be considered together. Article 11 requires Member States to establish controls over the abstraction of groundwater including a register of groundwater abstractions and a requirement of prior authorisation for abstractions and impoundments. Member States can exempt abstractions and impoundments which have no significant impact on water status. Current Irish law does not meet all of these requirements.

CONSTITUTIONAL ISSUES - PROPERTY RIGHTS AND GROUNDWATER PROTECTION

Ideally, for proper control, all groundwater should be in State ownership. This would be impossible, however, because Article 10 of the Constitution vests in the State "all natural resources, including the air and all forms of potential energy…..subject to all estates and interests therein for the time being lawfully vested in any person or body." Interests in groundwater are lawfully vested in landowners. Articles 40.3 and 43 acknowledge an individual's right to hold property and the State is mandated to protect that right. But nothing in the Constitution confers property rights where none existed before. Nonetheless, the case of *Webb v Ireland* and various statutory provisions (e.g. Minerals Development Acts 1940 – 1979, Planning and Development (Strategic Infrastructure) Act 2006, s.48) appear to imply that restrictions can be placed on certain property rights for environmental and other reasons which, broadly speaking, can be termed the common good. Action to promote sustainable development is undoubtedly an objective (indeed one of the most compelling objectives) for the common good. One approach adopted in the Planning and Development (Strategic Infrastructure) Act 2006 was for the State to legislate a presumption that that the ground under 10 metres has a *nil* value unless the landowner could establish otherwise. Another adopted in the Minerals Development Act 1979 was to vest the right *to work* minerals in the Minister for Energy and to except existing mining operators currently or just about to exercise the right[9]. I am not sure that these particular provisions would survive a challenge under the European Convention on Human Rights but to date nobody has challenged their constitutionality.

There is a reference in section 213 of the Planning and Development Act 2000 to "water-rights" and provision for compulsory purchase of such rights. However, the reference cannot be read as creating a right to water which did not exist prior to the coming of the Act. In this regard, the Act merely empowers a local authority to acquire an existing water right, which must relate to an existing property right.

Section 61 of the Public Health (Ireland) Act 1878 gives an urban authority the power to provide water within its district and the power to "Construct and maintain waterworks, dig wells, and do any other necessary acts,…."

The power to compulsorily acquire land includes acquisition for the purpose of supplying water. The Water Services Act 2007 also provides explicit powers for the acquisition of land by a water services authority for the purpose of performing any of its functions under that Act. These functions include providing water supplies. Where the proposed abstraction rate is above a particular threshold, an EIS must be prepared. If the EIS does not establish that an abstraction complies with the principles of sustainable development, the planning authority will usually insist on mitigation measures or refuse permission.

TENTATIVE CONTROLS OVER WATER CONSUMPTION

Although there are implied powers under Planning and Water Pollution legislation to control water consumption in various ways, the Water Services Act 2007 addresses the matter more comprehensively. Each water services authority has specific duties to ensure the sustainable management and use of water resources, including powers to ensure groundwater quality. They also have powers to prohibit or restrict the use of a water supply, where necessary, to protect public health or the environment. It is envisaged that those powers could be applied to follow up a water quality incident or at times of drought or other events to protect the integrity of the water supply and related ecosystems. The Act places a new duty of care on owners and occupiers regarding the sustainable use of water services on their premises. Section 70 specifically obliges occupiers and owners to maintain water treatment systems in such condition as to avoid nuisance or risk to human health or the environment.

[9] Minerals Development Act 1979, ss12, 13, 14.

Section 56 provides extensive new powers for the purpose of conserving water supplies. An authorised person is enabled to direct the owner or occupier of premises to take corrective action to prevent water from being wasted or consumed in excessive amounts. Such officers will also have powers of direction regarding the restriction of water use. Exercise of those powers will be subject to appeal to the District Court, except in times of emergency, and an authorised person will have power to cut off or restrict supply pending compliance.

At present sanitary authorities can charge non- domestic users for water supplied to them. The current exemption of domestic users is widely regarded as ill advised. However, there is an emerging human right to a sufficient quantity of water for domestic uses and it may be that there is some justification for limited exemptions from the obligation to pay for a minimum quantity of water for personal uses.

THE STATUTORY POSITION ON ABSTRACTIONS

INFORMATION ON ABSTRACTIONS

In Ireland we have no legislation to ensure the sustainable use and environmental protection of groundwater holistically and realistically. Statute law does not *explicitly* regulate the right to abstract groundwater. There are a number of provisions in various pieces of legislation that may be operated to limit abstractions indirectly. There are also some explicit but somewhat limited controls relating to abstractions. So a local authority has power under section 23 of the Local Government (Water Pollution) Act 1977, as substituted in 1990, to serve a notice on any person abstracting waters in its area requiring specified information in relation to water abstraction activities or practices within a specified period. Failure to comply with this notice is a criminal offence punishable by a maximum fine of £1,000 and/or six months imprisonment. If an IPPC licence is involved, the EPA must exercise the powers of the local authority under this section.[10]

REGISTERS OF ABSTRACTIONS

Section 9 of the Local Government (Water Pollution) Act 1977 and Part V of the Local Government (Water Pollution) Regulations 1978-92, require local authorities and sanitary authorities to keep registers of water abstractions other than abstractions which do not exceed 25 cubic metres in any 24 hour period. The register must be in a prescribed form and contain specified prescribed particulars and it must be made available for public inspection at all reasonable times. Fees are payable for a copy of any entry in the register.

GROUNDWATER AND THE PLANNING AND DEVELOPMENT ACT 2000

DEVELOPMENT PLANS AND GROUNDWATER PROTECTION

Under this Act all local authorities are required to plan for the proper planning and *sustainable* development of their areas. The Act requires local authorities who are also planning authorities to make provision for waste water services and other matters relevant to water management in development plans[11] and enables them to control - through their development plans and otherwise- the location of developments likely to cause water pollution or inimical to water management objectives, and to refuse permission for, or to permit subject to appropriate controlling conditions, developments which may cause water pollution or impair water management objectives.[12]Local authorities are obliged to prepare development plans for their areas and to take such steps within their powers as may be necessary to secure the objectives of development plans and are prohibited from carrying out any development

[10] Environmental Protection Agency (Extension of Powers) Order 1994, article 4.

[11] See in particular section 10(2) (b), First Schedule, Part 1, paras 7, 10, 11; Part 11, paras 6; Part 111, para 2; Part 1V, paras 1, 3.

[12] See in particular Fourth Schedule, paras 1(a)-(d), 3, 6, 9, 10(g), 19, Fifth Schedule, paras 1, 2, 7, 9, 10, 12, 15, 16, 18, 22.

which materially contravenes their plans.[13] Objectives in plans must include objectives for the sustainable development of groundwater. Those that appear particularly relevant for this purpose are objectives for:

- *regulating, restricting or controlling development in order to reduce the risk of serious danger to human health or the environment,*
- *regulating, promoting or controlling the exploitation of natural resources,*
- *regulating and controlling in accordance with the principles of proper planning and sustainable development the provision of water facilities.*
- *protecting and preserving the quality of the environment, including the prevention, limitation, elimination, abatement or reduction of environmental pollution and the protection of waters, groundwater…*
- *prohibiting, regulating or controlling the deposit or disposal of water materials …….. the disposal of sewage and the pollution of waters.* [14]

Many of these obligations overlap with obligations under Water Law, particularly those under the Water Framework Directive, and the integration of groundwater management measures in the two regimes is essential.

GROUNDWATER PROTECTION SCHEMES

Local authorities have adopted groundwater protection schemes in development plans and have regard to the need for groundwater protection in their decision-making under the Act.[15] Groundwater Protection Schemes subdivide the land surface on the basis of aquifer type (groundwater resource value), vulnerability and source protection area. They subdivide regions into three zones corresponding to regionally important aquifers (Zone 2), locally important aquifers (Zone 3) and poor aquifers (Zone 4). Groundwater Protection Responses list the generally acceptable and unacceptable activities in each zone. Some designations of environmentally sensitive areas required by EU or Nature Conservation legislation integrated into planning legislation or development plans may also mandate the protection of groundwater, for example, designations under the Habitats Directive or the Urban Wastewater Treatment Directives. Maps accompanying planning applications must show septic tanks and percolation areas, bored wells and other features on, adjoining or in the vicinity of the structure or land to which the application relates.[16] Specific notice of any development which (a) might cause the significant abstraction or addition of water either to or from surface or ground waters, whether naturally occurring or artificial or (b) of any development which might give rise to significant discharges of polluting matters or other materials to such waters or to be likely to cause serous water pollution or the danger of such pollution or (c) of any development in, over or along or adjacent to the banks of such waters or of any structure in , over or along the banks of such waters which might materially affect such waters, must be given to the appropriate Regional Fisheries Authority and in certain cases to Waterways Ireland.[17]Applicants for planning permission for developments who will not be connected to a public sewer are almost invariably conditioned to prevent the contamination of groundwater by providing suitable wastewater treatment or other pollution prevention or abatement systems.

DUTY TO NOTIFY STAKEHOLDERS OF ABSTRACTIONS

Planning authorities are obliged to notify the appropriate Regional Fisheries Board or Waterways Ireland, as appropriate, when they receive applications for development which:

(i) might cause significant abstraction or addition of water either to or from surface or ground waters, whether naturally occurring or artificial,

[13] Sections 15 and 178.
[14] See First Schedule, Part 1,clauses 10, 11: Part 11, clause 6(b); Part IV, clauses 1 and 3.
[15] GSI *Groundwater Newsletter*, January 1994, 3-5.
[16] Planning and Development Regulations 2001, article 23(1) as substituted by the Planning and Development Regulations 2007
[17] Ibid., article 28 as substituted by the Planning and Development Regulations 2007.

(ii) where the development might give rise to significant discharges of polluting matters or other materials to such waters[18] or be likely to cause serious water pollution or the danger of such pollution, or
(iii) where the development would involve carrying out works in, over, along or adjacent to the banks of such waters[19], or to any structure in, over or along the banks of such waters, which might materially affect such waters.[20]

PLANNING DECISIONS

Planning permission may be refused for any development if it would endanger public health or cause serious water pollution. Compensation is not payable when planning permission is refused because the proposed development would cause serious water pollution.[21] But there does not seem to be any explicit mandate to planning authorities to ensure that groundwater resources are not depleted although the obligation to ensure the proper planning and sustainable development of their areas surely implies that this is the case. Despite this, in practice, planning authorities sometimes have regard to the fact that a proposed development will dewater other lands[22] and sometimes developers volunteer, or are required to ensure, that this does not happen or that alternative water supplies are provided if it does happen. In my experience this has happened with mining and quarrying permissions.[23] In this way, groundwater quantity control is to some extent integrated into the physical planning process.

The various provisions relating to groundwater in the Planning Act imply that the protection of groundwater resources is a relevant planning concern. Unfortunately this concern is not expressed clearly enough and the Act should be strengthened to ensure that groundwater is sustainably managed, used and developed.

GROUNDWATER QUALITY

The discussion above discussed *quantitative* sustainability issues. Sustainable development principles require qualitative protection of groundwater so that it must also be protected from pollutants. This is achieved, to some extent, by a whole range of laws designed to protect beneficial uses of water or, since the Water Framework Directive came into force, to ensure that water attains or maintains good status. It would be too tedious to describe these laws here. Many statutes including the Planning and Developments Act 2000 - 2006, the Local Government (Water Pollution) Acts 1977 - 1990, Waste Management Acts 1996 - 2006 and the Environmental Protection Agency Acts 1992- 2006 seek to control discharges to groundwater in various authorisations granted or by the enactment of bye-laws or the enforcement of protective measures. Regulators have more or less *carte blanche* when imposing groundwater protection controls in authorisations granted and the law in this area is adequate (if properly applied) to protect groundwater quality. Courts rigorously ensure that regulatory decisions are made fairly but they rarely examine whether the decisions themselves are fair. The doctrine in *O'Keeffe v An Bord Pleanala* has more or less ensured that regulatory decisions on groundwater protection are immune from successful challenges unless made in the wrong way. Thus, we have many environmental decisions set aside for inadequate publication of public notices, failures to supply information, invalid fees paid, appeals given to doormen not employees of the regulator, addresses not supplied, but we only have about three decisions ever overturned regulatory environmental decisions on their merits. There is effectively no judicial remedy where regulators have used shifting and inconsistent rationales, imposed burdens that appear to have no clear environmental advantage, employed confused and capricious

[18] This includes groundwater.
[19] Ibid.
[20] Planning and Development Regulations 2001, article 28(1) (g).
[21] Planning and Development Act 2000, s. 190 and Fourth Schedule, clause 9, Fifth Schedule.
[22] It appears from the case of *State(Boyd) v An Bord Pleanala*, High Court, unreported, 18 February 1983, that this is a legitimate concern of a planning authority in making a planning decision although it is also arguable that disputes about water abstraction rights are private law matters which should not be considered by planning authorities unless issues of sustainability are at stake.
[23] The Minister for Energy may also have regard to this consideration when considering granting authorisations under the Minerals Development Acts.

analysis or failed to provide a coherent discussion of the issues. The courts do not want to become involved when the decisions of regulatory bodies are challenged on substantive grounds. They prefer to leave these matters to the "experts" notwithstanding that An Bord Pleanala or the EPA may be somewhat deficient in experts!

As well as this, there are many voids and defects in water law which until recently has largely ignored nonpoint sources of pollution (such as runoff from agricultural lands or construction sites) and groundwater quality. As a result there has been an ominous spread of groundwater pollutants and deteriorating groundwater quality. The scientific complexities of underground water transport make it difficult for regulators and courts to deal with groundwater problems. It may be hard to identify the real culprits or to confront them when they are identified. There may be multiple groundwater polluters but objectors will campaign only against new potential polluters, usually landfills or quarries. No matter how good the plans of the project planner, no matter that the prospect of pollution is only a cloud on the horizon dominated by the developers' good intentions, objectors will pursue him to the last. If action is taken about a potential nuisance the courts will wistfully suggest that if things go wrong, the objectors can sue later.[24] And in fact they can. Regulatory regimes have remedied common law deficits by providing effective and efficient remedies for past, present or anticipated polluting events and empowering local authorities to take anticipatory preventative actions.[25] There are numerous remedies under the Local Government (Water Pollution) Acts 1977 to 1990 and elsewhere that can be used not only to punish groundwater contamination, but to compel the polluter to remediate the pollution and provide alternative sources of water and pay damages to any person who has suffered injury to his person or property. And most of these remedies (except civil remedies) are available to any person regardless of his interest in the matter. So, for example, in *Thornton v Meath County Council*[26], Thornton was compelled to provide an alternative water supply to residents whose groundwater had allegedly been polluted by leachate from a landfill. Citizen initiatives to enforce water laws are encouraged by extensive rights to participate in decision-making and to enforce almost all the important statutory controls over groundwater quality.

PROBLEMS WITH CAUSATION

There are lots of remedies available if groundwater is polluted, but the problem is in proving causation. Occasionally causation can be established by a simple sniff- and- taste test (oil contamination) but more likely the issue will boil down to a battle of the experts. The court will be treated to heated arguments over the direction and flow of groundwater, the slopes of the land, alleged connections between the source of pollution and the polluted groundwater, the existence of impermeable barriers of various kinds, alternatives sources of contamination etc. Sometimes, the court will find a link between the pollution and the alleged polluter relying on circumstantial evidence but then there will be a further problem proving a link between the pollution and the damage suffered. So, in one case where a plaintiff successfully linked the contamination of his well to a rock salt storage facility only to fail to prove that his children's' illnesses were caused by the contaminated water.[27]

CONCLUSION

I hope I have illustrated some of the lacunae and deficiencies in Irish law which militate against the establishment of a robust management system to protect groundwater resources, especially against unsustainable depletion. It is obvious that the courts and the legislators should have a great deal more to do with "men and women of science" if the objective of achieving the sustainable development of groundwater resources is to be achieved!

---

[24] *McGrane v Louth County* Council High Court, 9 December 1983.
[25] See, e.g. Local Government (Water Pollution) Act 1977, sections 11, 13.

[27] *Meehan v State of New York*, 95 Misc.2d 678, 684.

LEGAL ASPECTS PECULIAR TO GROUNDWATER DEVELOPMENT

Dr Peter Howsam,

Reader in Water & Environmental Law, Natural Resources Management Centre, Bldg 42a,

Cranfield University, Bedfordshire, MK43 0AL, UK.

p.howsam@cranfield.ac.uk

Discussion Points

Firstly what is peculiar about groundwater? Well despite the fact that it is highly valuable (one estimate = €8b to the UK economy), it is, compared to surface water,

(a) Scientifically less well understood,

(b) Liable to public misconception,

(c) Complicated by much longer timescales,

(d) Legally less well protected, and

(e) More difficult to manage: largely and simply because it is out-of-sight and therefore out-of-mind.

This presentation will look at three aspects of groundwater development from the legal context, touching on international, EU, national and local jurisdictions. These aspects are:

(1) Resource management (quantity) – abstraction controls/regulations;

(2) Quality protection – pollution/contamination prevention and remediation controls/regulations;

(3) Exploration and development (drilling, testing and monitoring) controls/regulations.

NOTES:

GROUNDWATER SUPPLY – THE LOCAL AUTHORITY EXPERIENCE

Kieran Madden, Senior Engineer,

Water Services, Roscommon County Council, The Courthouse, Roscommon, County Roscommon

kmadden@roscommoncoco.ie

ABSTRACT

This paper is intended to briefly outline some practical aspects on how groundwater protection is handled in County Roscommon through the use of the County Development Plan, the Water Framework Directive and other legislation currently at our disposal.

INTRODUCTION - WHY PROTECT GROUNDWATER?

Groundwater is an important natural resource, which supplies some 20 to 25% of drinking water in Ireland and is also important in maintaining wetlands and river flows through dry periods. Groundwater is the main source of drinking water in County Roscommon. Twelve of the eighteen public water schemes (i.e. over 80% of the water abstracted) and forty seven of the fifty six private group schemes are supplied solely by groundwater. In addition all areas not supplied by either public water or group schemes rely on individual wells as their source of water. The use and the protection of groundwater in County Roscommon is thus of relatively greater significance than in other parts of the country. Groundwater and groundwater catchments have an inherent ecological and economic value and are a major resource that needs to be protected. Groundwater contributes to rivers and lakes and therefore influences their amenity and recreational value. Roscommon County Council is responsible for the protection of all waters, including rivers, lakes and groundwater. This responsibility includes implementation of pollution control measures, licensing of effluent discharges, implementing and monitoring compliance with environmental regulations and the drawing up of pollution contingency measures.

THE GROUNDWATER SITUATION IN COUNTY ROSCOMMON

In June 2003 the Geological Survey of Ireland, in collaboration with Roscommon County Council published ‘The County Roscommon Groundwater Protection Scheme’. Six of the major regional water supply schemes, all of which are solely supplied by groundwater, had individual groundwater source protection zone reports, with associated mapping, prepared for them. These are:

- Roscommon Central Regional Water Supply Scheme,
- Ballinlough Water Supply Scheme,
- Boyle/Ardcarne Water Supply Scheme,
- Killeglan Water Supply Scheme,
- Castlerea Water Supply Scheme (2 No. Springs)
- Mount Talbot Water Supply Scheme.

The groundwater protection scheme report stated that a large portion of the county is classed as having either extreme or high vulnerability, with risk of contamination. In particular, areas in the north of the county, where rock is generally at or close to the surface, are extremely vulnerable. The current County Development Plan (CDP) in County Roscommon is the 2002 plan. As the groundwater protection scheme was only in draft form when this CDP was adopted, there is only one mention in the 2002 plan of the protection scheme.

A Draft County Development Plan has been prepared under Section 11 (5) of the Planning and Development Acts 2000-2006. This plan is currently being reviewed and it is expected to be adopted by June 2008. This draft plan includes vulnerability mapping, makes extensive reference to the County Groundwater Protection Scheme, and includes paragraphs such as:

"*Groundwater and major surface water sources are important to the development of the county. The protection of these resources is of major concern to the council. The proposed County Development Plan 2008-2014 will take cognisance of the groundwater protection plans and groundwater vulnerability in the county and shall adopt a Water Quality Management plan for the County.*"

HOW IS THIS IMPLEMENTED?

Although diffuse sources, for example the spreading of organic and in-organic fertilisers and pesticides, are considered a threat to groundwater, the main threat in the Roscommon is considered to be posed by point contamination sources such as farmyard waste (mainly silage effluent and soiled water), septic tank effluent, leakages, spillages and leachate from waste disposal sites.

PLANNING

All planning applications to Roscommon County Council are referred to the Area Engineers for comment. In addition, all farmyard related applications are referred to the Environment Section for their assessment and report. Generally, where landspreading of effluent is the proposed solution for disposal, maps and a nutrient management plan for all of the lands intended for use in landspreading are requested. This information is closely scrutinised and compared to the maps in the County Groundwater Protection Scheme or the individual source protection schemes. In many instances specific site investigation work is requested including the use of percolation tests and trial holes. The response matrix for landspreading is applied. In more recent times where planning permission for the application is being granted but where restrictions are to be placed on the lands used for land spreading, a Section 47 Agreement under the Planning & Development Act 2000 is generally included as a planning condition. In relation to applications for single houses, each is specifically examined with particular emphasis on those houses proposed within the source protection zones. Again the Groundwater Protection response matrix for On-Site Wastewater Systems for Single Houses is used for all assessments and where planning permission is being granted the use of a proprietary sewerage treatment plant is generally a condition of the permission.

LANDFILLS

One of the main point loading threats to groundwater was escape of leachate from landfills. Roscommon County Council used to operate 5 landfills in the County. All of these landfills, with the exception of Ballaghaderreen, have now been closed for more than six years. The landfill in Ballaghaderreen is now licenced by the Environmental Protection Agency (EPA). All the new cells are fully lined and collect all leachate. The older part of the landfill has been completely lined and capped. All leachate is directed towards the leachate lagoon and is pumped to the sewerage treatment works in Ballaghaderreen for full treatment.

Roscommon town landfill, the only other landfill which had been licenced by the EPA, was closed some 7 years ago. This landfill has also been lined and capped, with a leachate collection system constructed and here again all leachate is collected and taken by tanker to the Roscommon Wastewater Treatment Plant for treatment.

THE USE OF SEWAGE SLUDGE IN AGRICULTURE

Roscommon County Council prepared a Sludge Management Plan in 2004. The mainstay of this plan was the provision of a sludge treatment centre adjacent to the existing sewerage treatment works at Roscommon town to be provided under a design, build and operate contract. Roscommon County Council has employed a Client's Representative who is currently in the process of preparing an Environmental Impact Statement for this sludge treatment plant. In the meantime however all of the sewage sludge in Roscommon is being removed by contract and being disposed of through the use of agricultural land spreading in accordance with the Good Agricultural Practise Guidelines. In late 2002 a controversy arose in County Roscommon in relation to the use of "Sludge Lagoons". These lagoons were being used for the purpose of storing sewage sludge from Roscommon and from other counties before final spreading. These lagoons were in breach of the Planning Act and at the time, notices, both under the Planning & Development Act 2000 and the Waste Management Acts 1996-2001 (Section 55), were served on the landowners to have them removed. The episode did however serve to heighten our own awareness and made us look much more stringently at the groundwater protection responses. It also led to significant improvements in the methods adopted in Roscommon for assessing lands to be used for the landspreading of sewage sludge. At the time specific site investigations were carried out. Nutrient management plans were immediately put in place for all lands in the county being used for the use of landspreading of sewage sludge. Sampling points were also put in place and a much tighter regime in relation to our records and the sewage sludge register was introduced. In the recent past Roscommon has introduced an alternative contractor for the sludge being disposed of from the Monksland Wastewater plant in Athlone. This sewage sludge is being taken on a pilot basis for the use of alternative bio-energy crops, for example, production of back oats and willow coppicing. This pilot has been quite successful to date but any extension of the scheme depends greatly on the farming community buying into the planting of these energy crops.

GROUNDWATER UNDER OTHER LEGISLATION

Both the Nitrates Directive (91/676/EC) and the Water Framework Directive (2000/60/EC) have major implications for the protection of groundwater. In Roscommon a considerable number of farm surveys have been carried out to date. Much of this work has been focused on the pollution of surface waters such as lakes, rivers and streams. However, all new planning applications for farmyard development are also being assessed under the Nitrates Directive. Closer attention is paid to storage requirements and the protection distances required for landspreading beside all lakes, rivers, streams and water sources. Currently there is a requirement to report all farms that are not complying with the Nitrates Directive to the Department of Agriculture and Fisheries.

In the Group Scheme sector where new water sources have been identified the imposition of source protection under the Nitrates Directive has led to some problems in the acquisition of land. Because of the sterilisation of lands required around any new source, the cost of land acquisition has increased exponentially. It should be pointed out that it is not just farm developments that are being scrutinised under the Nitrates Directive. Commercial industries such as abattoirs and meat plants that have a storage requirement and who land spread wash water are also being examined. In one recent instance, Roscommon County Council assisted a meat plant that did not have the required storage by allowing the wash water effluent to be taken by tanker to one of the sewerage treatment plants in Roscommon for treatment.

WHERE DOES ROSCOMMON GO FROM HERE?

In the planning context I am aware that some of our neighbouring counties have introduced a "Panel of Experts" for the specific site assessment for single houses necessary in order to comply with the

EPA guidelines. Roscommon County Council are currently examining this option and also looking at the professional indemnity cover required for persons carrying out these assessments.

Many of the existing water schemes in Roscommon are included in the Water Services Investment Programme for Renewal/Upgrading. These schemes are at various stages of planning or construction. In each case, because of the abundance of groundwater available in County Roscommon, trial wells were drilled and were either proven or rejected. Roscommon County Council have currently applied for five Water Abstraction Orders on various schemes, four of which relate to groundwater sources. In the past groundwater was considered the cheaper option for some counties because of the high quality of water being extracted. In many schemes no further treatment other than chlorination was required. This is reflected in the cost of 82 c/m^3 charged by Roscommon County Council to all non domestic customers for 2008, one of the lowest in the country. However because of improvements in water quality required under the Drinking Water Regulations (most recently S.I. No. 278 of 2007), and highlighted by the recent outbreak of cryptosporidiosis in Galway, full treatment is required on most sources to ensure compliance with all the parameters listed. For this reason risk assessments (including Cryptosporidium Risk), are carried out on all existing and potential sources and in general full treatment is being recommended. As most new schemes are undergoing the design, build, operate (DBO) route, whereby a service provider will be contracted for 20 years to ensure good quality water, there are serious financial implications for counties like Roscommon into the future. I foresee a much greater emphasis in many counties, particularly Roscommon, on source protection and water conservation.

CONCLUSION

I have taken you through a whirlwind tour of our experience in Roscommon in relation to groundwater supplies and issues. From my point of view I see no diminution in the next 20 years in Roscommon’s dependence on groundwater and I thus see a much-enhanced regulatory regime into the future to protect that resource.

.

Session II

TRACKING THE SOURCE OF FAECAL CONTAMINATION OF WATER

Siobhan Dorai-Raj & Professor Martin Cormican

Departments of Microbiology and Bacteriology, National University of Ireland, Galway, & Galway University Hospital

siobhan.dorairaj@nuigalway.ie, martin.cormican@hse.ie

INTRODUCTION

Faecal contamination of water represents a significant risk to human health. In Ireland, of the four types of water supplies, it is private Group Water Schemes (PrGWS) that are most frequently contaminated by faeces. PrGWS are formed when a group of households, mainly in rural areas, come together to source and distribute their own water supply. PrGWS serve approximately 6.4% of the population. The Environmental Protection Agency's annual reports on the quality of drinking water in Ireland consistently show that the quality of the water supplied to the public by PrGWS is unacceptably low. Overall, in 2006 nearly 36% of the PrGWS that were tested were contaminated with *E. coli* at least once. Detection of *E. coli* indicates that the water is unfit for human consumption. In assessing the potential risks of faecal contamination and protection of source water, identification of the source of the pollution is an important element. The most widely used faecal indicator microorganisms (coliforms, faecal coliforms, *Escherichia coli* and enterococci) are found in the faeces of both human and animals and thus their detection does not allow differentiation between possible sources of faecal contamination. The need to identify the origin of faecal contamination in water has led, in recent years, to the development of many different types of faecal source identification methodologies, a field known as microbial source tracking (MST).

PROJECT BACKGROUND

Recent EPA funded work at NUIG Galway has focused on bacteria of the order *Bacteroidales* and on the genus *Bifidobacterium.* The two largest genera in the order *Bacteroidales* and the most important for MST purposes are the genus *Bacteroides* and the genus *Prevotella*. Members of the genus *Bifidobacterium* and the *Bacteroides-Prevot*ella group are strict anaerobes, are restricted to warm blooded animals, and make up a significant portion of faecal bacteria. Most importantly, some species of the microorganisms are of human origin, whereas others are exclusively found in animals. The use of these organisms as faecal contamination indicators, however, has been limited because strict anaerobes are often difficult to grow. Using molecular methods such as PCR to detect the organisms can circumvent these difficulties.

METHODOLOGY

One element of this project focussed on microbiological contamination of raw and piped water from three rural PrGWS. This involved bimonthly testing of samples of raw and piped water from three PrGWS (C, K and M) for total coliforms, *E. coli* and enterococci over a 12 month period. The level of *E. coli* contamination in the raw water samples was then correlated with the level of precipitation for the three PrGWS.

The second part of the project concentrated on identifying host-specific 16S rDNA sequences from members of the order *Bacteroidales*, and using these sequences to design host-specific PCR primers. The first step in this endeavour involved amplifying an approximately 1060 bp section of the 16S rRNA gene from members of the *Bacteroidales* order from human sewage and cow and sheep faecal samples. These PCR products were cloned into plasmid vectors and used to generate three clone libraries (human sewage, cow faeces and sheep faeces). The cloned 16S rDNA was analysed by amplified ribosomal DNA restriction analysis (ARDRA) and unique operational taxonomic units (OTUs) were sequenced. This sequence data was analysed by ClustalW alignments, comparison to sequences in online databases and phylogenetic tree construction.

Next, a sequence of approximately 1060 bp of the 16S rRNA gene from members of the *Bacteroidales* order, was PCR amplified from human sewage and cow and sheep faecal samples using the same PCR primers used for generation of PCR products for the clone library construction, except this time, the PCR primers were fluorescently labelled. This allowed for the examination of these PCR products by terminal restriction fragment length polymorphism (TRFLP) analysis. The TRFLP analysis revealed ruminant-specific terminal restriction fragments (TRFs) which were correlated to the sequence data generated as described above. Cloned sequences from cow and sheep faeces which corresponded to ruminant-specific TRFs were aligned with cloned sequences from the human sewage library and areas of DNA, where the 16S rDNA gene sequences from ruminant faeces differed from all the 16S rDNA gene sequences from human sewage, were used to locate putative ruminant-specific PCR primers.

In total, 20 putative ruminant-specific PCR primers were designed, which were paired with each other and with previously published non-host specific *Bacteroidales* primers to give novel ruminant-specific PCR assays. After initial evaluation of 29 assays, the six assays with the most promising performance characteristics (utility in differentiation between ruminant and human faecal samples and consistency of positive results with target samples) were chosen for more comprehensive optimisation and for evaluation. Six previously published PCR assays (five human-specific, one ruminant-specific), designed to detect putatively host-specific members of either the order *Bacteroidales* or the genus *Bifidobacteria*, were also evaluated. In addition, the limit of detection of the six ruminant-specific assays and the six previously published assays was evaluated. All the putatively host-specific assays were also tested on naturally contaminated water samples from the three PrGWS supplies described above.

RESULTS

In the three PrGWS source waters tested, in general, increased precipitation resulted in increased levels of *E. coli*. In the groundwater source, precipitation data over a longer period of time was more significantly correlated with increased levels of *E. coli* whereas in surface water sources precipitation data closer to the time of sampling was more closely correlated with levels of *E. coli*. The microbiological results confirmed that the frequent finding of faecal contamination on previous intermittent random sampling of PrGWS is probably representative of the general condition of these supplies. The results emphasise the need to upgrade the source, treatment infrastructure and monitoring of rural water supplies.

Overall, the analysis of the *Bacteroidales* 16S rDNA cloned sequences from human, cow and sheep faeces revealed a highly diverse group of organisms, many of which were not represented by previously characterised 16S rDNA sequences. The use of TRFLP analysis, combined with sequence analysis of clone libraries, proved to be a good method of identifying host-specific species of bacteria in faeces. The TRFLP analysis showed that certain species of *Bacteroidales* display a level of host-specificity that can be exploited for use in faecal source tracking.

The six novel ruminant-specific PCR assays developed in this study amplified DNA from almost all of 74 ruminant faecal samples tested, none of the 59 human sewage/faecal samples tested and very few of the non-target animal faecal samples (1-5 samples out of 44). Using these PCR assays it was possible to detect from 7.3×10^{-3} to 7.3×10^{-6} g (dry weight) of faeces per litre and from 1×10^{2} to 1×10^{4} copies of the 16S rRNA gene per PCR assay volume. Of the five putatively human-specific published PCR assays evaluated in this study, the PCR assay which targeted the *Bifidobacterium catenulatum* group was the most promising. Land use patterns indicated ruminant faeces as the likely main source of faecal pollution in the raw water. Using the most sensitive of the novel ruminant-specific assays it was possible to detect ruminant pollution when approximately 50 *E. coli* per 100 ml were present in the water. All of the assays developed in this study compared favourably to the previously published ruminant-specific assay tested.

CONCLUSION

In conclusion, the ruminant-specific PCR assays developed in this study show good specificity, sensitivity, have a low limit of detection and have been used to amplify putatively ruminant-specific *Bacteroidales* species from naturally contaminated water samples. These assays show promise for use in faecal source tracking studies and merit further field testing and specificity and sensitivity evaluation.

MICROBIAL COMMUNITIES IN GROUNDWATER AND THEIR POTENTIAL USE FOR BIOMONITORING

Dr Nico Goldscheider(1), Michiel Pronk (1) & Jakob Zopfi (2)

(1) Centre of Hydrogeology, University of Neuchâtel, Switzerland, nico.goldscheider@unine.ch

(2) Laboratory of Microbiology, University of Neuchâtel, Switzerland

ABSTRACT

This communication discusses the current knowledge on biocenoses in aquifers, their potential use for groundwater biomonitoring and the limitations of this approach. On this basis, a case study from a karst system is presented, focusing on microbial communities in spring water that were characterised using methods from molecular microbiology, particularly DGGE fingerprinting. It was possible to show that the microbial communities are clearly related to the contribution of frequently contaminated water from a swallow hole. This finding shows that microbial communities could potentially be used for groundwater biomonitoring, although both the analytical and interpretive methods need to be further developed before this could be used as a standard technique.

INTRODUCTION

Aquifers are not only drinking water resources but are increasingly recognised as ecosystems. Ecosystems consist of habitats (i.e. the physical and chemical environment) and biocenoses (i.e. the organisms that live there), which interact in manifold ways. This new ecological approach to groundwater protection is also reflected in the Swiss Water Protection Ordinance, which demands that the biocenoses in aquifers should be in a natural state adapted to the habitat and characteristic of non-polluted groundwater. This has two possible implications: on one hand, the biocenoses could be considered as something valuable that should be protected; on the other hand, the biocenoses could be used as indicators for groundwater quality.

As there was relatively little knowledge about microbial biocenoses in groundwater and their potential use for biomonitoring, we carried out a detailed literature review (Goldscheider et al. 2006), followed by a research project in which we investigated the dynamics of microbial communities in a karst aquifer system and how these communities are related to hydrological and physicochemical parameters and to contamination (Pronk et al. 2009).

AQUIFERS AS ECOSYSTEMS

Aquifers are heterogeneous on all scales and can be structured, from an ecological point of view, into a variety of macro-, meso- and microhabitats (Fig. 1). The contact or transition zones between habitats can be described as ‘ecotones’ and are often characterised by increased biological activity and diversity. Ecotones can also be observed on all scales, e.g. contact zones between different aquifers and aquicludes, or between sand layers and clay lenses. Steep chemical energy gradients occur in such zones, for example at the groundwater table and in the capillary fringe, where reduced groundwater may come in contact with oxygen-rich percolation water. On a microscopic scale, interfaces between organic or inorganic particles and pore water can be considered as micro-ecotones.

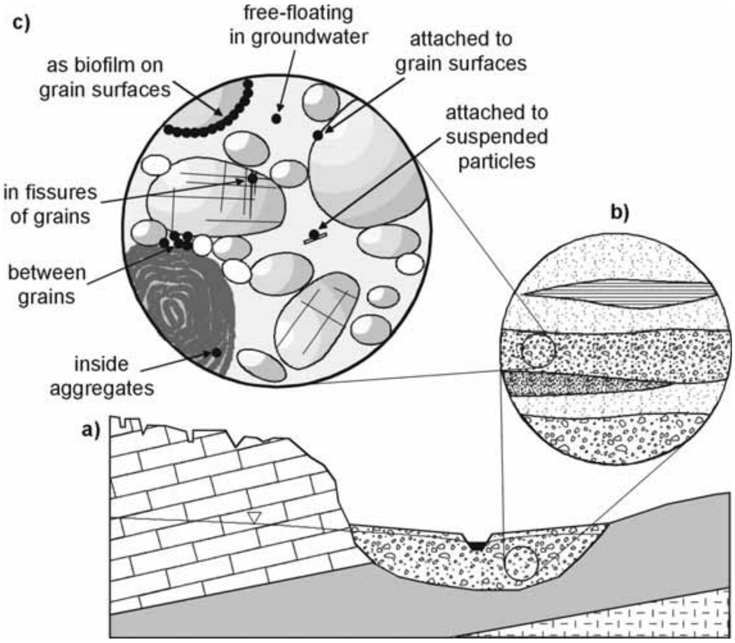


Fig. 1: Aquifer heterogeneity on all scales results in a great variety of macro- (a), meso- (b) and microhabitats (c). The groundwater table and hyporheic zone are important ecotones on a macro- to meso-scale (modified after Goldscheider et al. 2006).

Microorganisms predominate in groundwater environments, mainly bacteria (true Bacteria and Archea) but also protozoans and others. Small invertebrates are also often present, while larger animals are restricted to the wider voids in karst aquifers and the hyporheic zone. Due to the absence of light, groundwater biocenoses entirely depend on chemical energy, which is, however, scarce in non-polluted oligotrophic aquifers. All animals, protozoans and most bacteria are heterotrophic, i.e. they need organic carbon. Organic carbon generally originates from the land surface and soil zone, and its concentration is rapidly decreasing within the unsaturated zone and in the aquifer. Autotrophic bacteria are independent of organic carbon as they can synthesise biomass from CO2 and inorganic energy sources (Fig. 2). The heterogeneous distribution of energy and nutrient sources further increases the heterogeneity of the habitats and biocenoses. Examples of small-scale heterogeneity include:

- Locally reducing conditions around organic particles in a generally oxidising aquifer;
- Locally acid conditions around pyrite grains;
- Locally nutrient-rich conditions around weathering apatite grains, etc.

Groundwater macro- and microorganisms have developed specific adaptation strategies to survive nutrient-poor conditions: reduced organism size and activity, low population densities and reproduction rates, long lifetimes, and the effective use of energy resources.

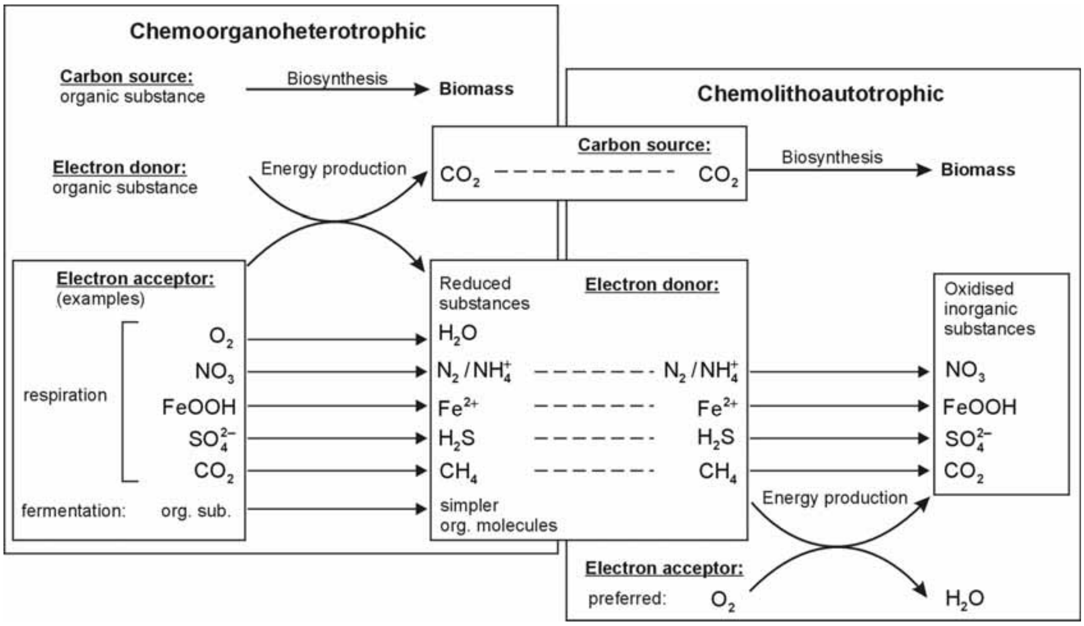


Fig. 2: Illustration of metabolic pathways in aquifer biocenoses. Reduced substances produced by heterotrophic bacteria can serve as electron donors for autotrophic bacteria (modified after Goldscheider et al. 2006).

The groundwater fauna includes many rare and endemic species. There are also endemic genera, families and orders that only occur in groundwater, often in very restricted geographical areas. Many species are yet to be discovered. Groundwater environments contribute significantly to global biodiversity, although most species are very small and largely unknown invertebrates ("hidden biodiversity").

BIOMONITORING

The basic idea of biomonitoring is to use specific organisms or groups of organisms as bioindicators for environmental quality. Biomonitoring is frequently used in surface water, while there is no commonly accepted method for groundwater. Passive biomonitoring is based on naturally occurring organisms, while active biomonitoring uses standardised organisms that are introduced into the ecosystem. Compared to chemical analyses of water samples, biomonitoring delivers less precise information on water quality, as it does not allow specific contaminants to be identified and quantified. The major advantage of biomonitoring is that it indicates the actual impact of contaminants on living organisms, including:

- Information about unknown contaminants and substances that are not measured;
- Integral information about contaminant mixtures;
- Temporally integrated information.

There are four approaches to groundwater biomonitoring, some of which are relatively well established, while others are in an early stage of development:

1) Active drinking water monitoring using trout or daphnia: For example, the water supply works of the city of Zurich, Switzerland, pump groundwater from an alluvial aquifer and use a 'daphnia toximeter' to control the water quality. Changes in the behaviour of the daphnia make it possible to conclude on changes in groundwater quality.

2) Passive groundwater monitoring using the natural invertebrate fauna, as it is routinely done in surface waters: This type of monitoring is less feasible in groundwater than in surface water, because most invertebrate species in groundwater are very small, difficult to identify, rare and often endemic to small areas; the population densities are often low, particularly in deep and fine-grained aquifers. Therefore, it is difficult or impossible to establish universally applicable indicator organisms.

3) Faecal indicator bacteria (FIB): The presence of E. Coli or other bacteria of faecal origin indicates contamination and thus the possible presence of pathogens from untreated wastewaters or agriculture. Other contaminants from the same sources might also be present, such as hormones, antibiotics, etc. Although this is not biomonitoring in the proper sense of the word, it is a powerful monitoring technique using biological indicators.

4) Passive groundwater monitoring using microbial communities: This is a promising new approach, because microorganisms (bacteria) exist virtually everywhere, even in deep, fine-grained and thermal aquifers. Most bacteria in groundwater are viable but non-cultivable, and thus often overlooked, but methods of molecular microbiology make it possible to characterise the entire microbial communities. Contamination, particularly organic substances, is likely to influence the microbial biocenoses. However, the natural variability and heterogeneity is probably often higher than the influence of contamination.

The case study presented in the next sections focused on this latter monitoring approach.

MICROBIAL COMMUNITIES IN A KARST AQUIFER SYSTEM

The test site is a karst aquifer system near the city of Yverdon in Switzerland. In a simplified way, the system consists of a swallow hole (input) connected to two karst springs (output), one of which is tapped as a drinking water source for the city. The stream sinking into the swallow hole drains an agricultural area and is often contaminated with high levels of faecal bacteria, total organic carbon (TOC), turbidity and nitrate, and probably several other contaminants for which the study did not check (Pronk et al. 2006). Four tracer tests between the swallow hole and the springs, along with water quality monitoring, made it possible to show that the sinking stream is by far the most important source of spring water contamination. After intense rainfall, two types of turbidity can be observed at the springs:

- A primary or autochthonous turbidity peak results from the remobilisation of sediments from karst conduits near the springs due to increasing flow rates. This type of turbidity is “clean”, i.e. bacteria, TOC and nitrate levels remain stable and low.
- A secondary or allochthonous turbidity signal can clearly be related to the arrival of surface water from the sinking stream. This type of turbidity coincides with bacterial contamination and increased levels of TOC and nitrate.

During low-flow conditions (i.e. intense rainfall following a relatively long dry period), the two signals can easily be differentiated; during high-flow periods, they tend to overlap. It was possible to demonstrate that particle-size distribution (PSD) allows the two types of turbidity to be separated: autochthonous turbidity is a mixture of all particle-size classes, while allochthonous turbidity is characterised by a relative increase of fine particles (around 1 μm). Therefore, PSD can be used as a reliable early-warning parameter for the arrival of contaminated water from the sinking stream (Pronk et al. 2007).

The diversity and temporal variability of microbial communities in the spring water was assessed by means of DGGE (Denaturing Gradient Gel Electrophoresis) profiles (or fingerprints) of 16S-rDNA PCR (Polymerase Chain Reaction) products (Fig. 3). DGGE profiles illustrate the diversity of a microbial community in a water sample and make it possible to compare the communities found in different samples, but do not allow the species to be identified. The spectrum of bacterial species was determined in a small number of water samples, using the relatively laborious cloning/sequencing technique (Pronk et al. 2009).

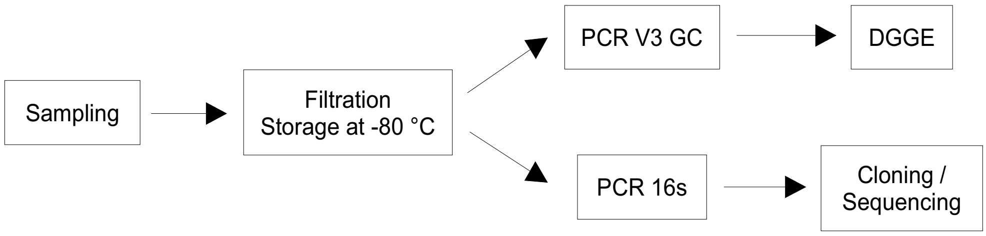


Fig. 3: Simplified illustration of the two employed methods from molecular microbiology.

Only the DGGE profiles are discussed here. The degree of similarity between the microbial communities found in different samples was analysed statistically using the software Gel Compare, which also groups the profiles into clusters. Fig. 4 shows selected results. The different clusters clearly correspond to distinct hydrological conditions. In particular the microbial communities found in water samples, during periods with a very low flow contribution from the sinking stream, form one cluster, while all samples corresponding to a high flow contribution from this frequently contaminated sinking stream are grouped into another cluster, which is very clearly separated from all other clusters. Thus, the DGGE fingerprints make it possible to assess the inflow of contaminated surface water, suggesting that this technique has the potential to be further developed and used for biomonitoring.

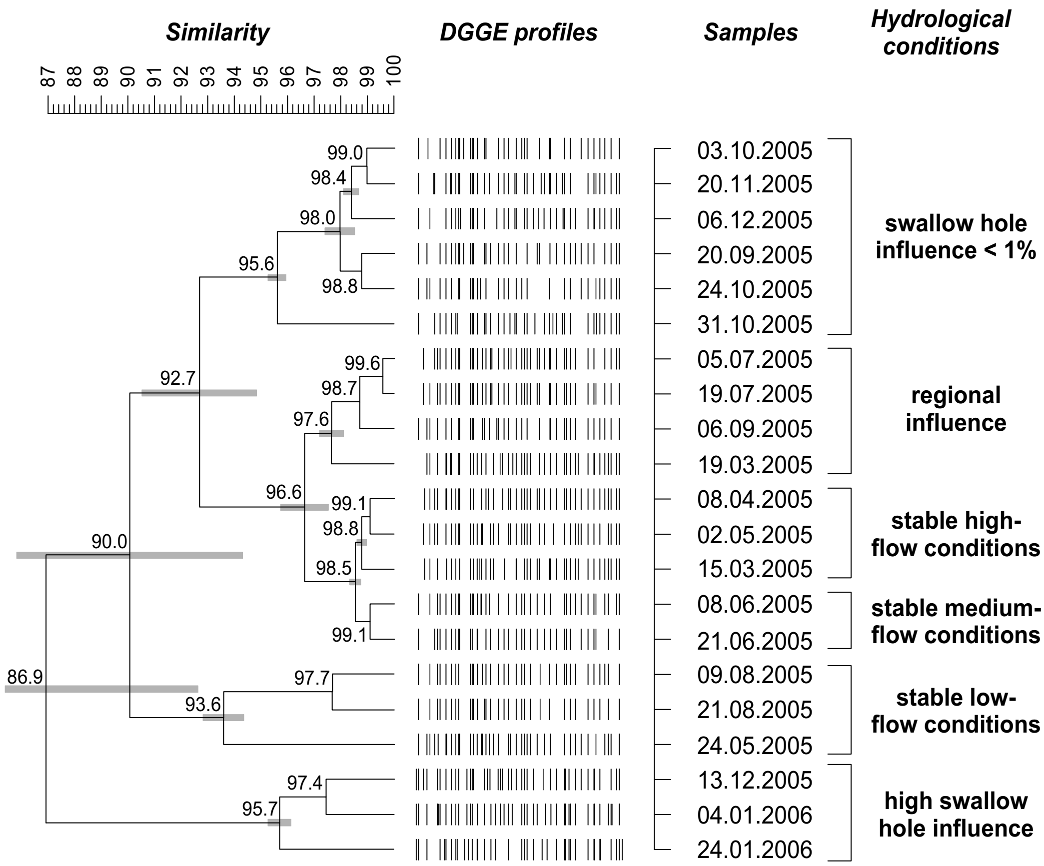


Fig. 4: Cluster analysis of the DGGE profiles from the Moulinet spring.

CONCLUSIONS

Aquifers are ecosystems harbouring biocenoses mainly consisting of microorganisms and small invertebrates. The invertebrate fauna include many rare, endemic and undiscovered species, which contribute to global biodiversity. The biocenoses in groundwater could also be used for biomonitoring, although there have been no generally applicable methods until the present day. Invertebrates are successfully used for surface water quality monitoring, but the small sizes and low population densities of invertebrates in groundwater, and the high proportion of endemic species, hamper their use for groundwater biomonitoring.

In a case study, we assessed the diversity and dynamics of microbial communities in a karst aquifer system using methods from molecular microbiology. The DGGE profiles (fingerprints) can be grouped into several clusters, which clearly correspond to distinct hydrological situations. The contribution of frequently contaminated surface water from a sinking stream to the karst spring water is clearly reflected in the clustering, thus illustrating that this approach could potentially be used for biomonitoring. However, both the analytical and interpretation methods need to be further developed before this approach can be used for standard applications. Groundwater biomonitoring will never replace chemical and bacteriological analyses and measurements, but it could deliver additional, integrated information on the quality status of the groundwater.

REFERENCES

More information can be found in the following four papers, on which this communication is largely based. The review article (Goldscheider et al. 2006) also includes an exhaustive reference list with more than 200 papers and books:

Goldscheider N, Hunkeler D, Rossi P (2006) Review: microbial biocenoses in pristine aquifers and an assessment of investigative methods. Hydrogeology Journal, 14(6): 926-941.

Pronk M, Goldscheider N, Zopfi J (2006) Dynamics and interaction of organic carbon, turbidity and bacteria in a karst aquifer system. Hydrogeology Journal 14: 473-484.

Pronk M, Goldscheider N, Zopfi J (2007) Particle-size distribution as indicator for fecal bacteria contamination of drinking water from karst springs. Environmental Science & Technology, 41(24), 8400-8405.

Pronk M, Goldscheider N, Zopfi J (2009) Microbial communities in karst groundwater and their potential use for biomonitoring. Hydrogeology Journal, invited contribution, accepted, in press.

AN OVERVIEW TO HUMAN HEALTH RISK ASSESSMENT TOOLS AND APPLICATION TO ASSESSING CHRONIC RISKS TO HUMAN HEALTH FROM CONTAMINATED SHALLOW WATER

Yolande Macklin, Senior Environmental Scientist, Atkins Ltd, Woodcote Grove, Epsom, Surrey KT18

yolande.macklin@atkinsglobal.com

ABSTRACT

The increased use of brownfield sites for housing has lead to the need for rigorous human health risk assessment from soil and groundwater contaminants and one of the main questions is 'might this land be a risk to human health?. Within the uk the approach developed by the department for environment, food and rural affairs (defra) and the environment agency was to provide a suite of seven soil guideline values (sgvs) in 2002 together with a human health risk assessment model. However there are currently no sgvs or uk risk assessment tool which assesses the potential risk to human health from inhalation of groundwater vapour from vapour intrusions. Vapour intrusion is the migration of volatile chemicals from the subsurface into overlying buildings. Volatile chemicals may include volatile organic compounds (e.g. Benzene), select semi-volatile organic compounds (e.g. Naphthalene), and some inorganic analytes, such as elemental mercury and hydrogen sulphide. The main concern is usually whether or not these volatile contaminants are causing long-term chronic health effects. In order to assess this pathwa,y other commercial risk assessment tools have to be considered, such as riscworkbench and rbca toolkit. The algorithms and pathways within these models vary and careful consideration of their fate and transport algorithms is required in order to determine whether they are appropaite for use at a site. Modelling of the human health risks from groundwater vapour has limitations and consideration of these should be taken into account when assessing a site.

INTRODUCTION

The increased use of Brownfield sites for housing has lead to the need for rigorous human health risk assessment from soil and groundwater contaminants. Throughout the UK and other countries there are numerous sites where land has become contaminated by human activities, including industry, chemical, oil spills and where waste has been buried. Examples of contaminative land uses include gas works, tanneries, petrol stations; landfill sites and scrap yards. Most commonly the concentrations of contaminants are not sufficient to pose an acute risk to human health and the main concern is the long term (chronic) effects from sustained exposure. The main question when dealing with a brownfield site is, *'might this land be a risk to human health?'*. To answer this question, data on contamination in soil or groundwater must be compared to appropriate screening criteria.

Within the UK the approach developed by the Department for Environment, Food and Rural Affairs (Defra) and the Environment Agency was to provide a suite of seven Soil Guideline Values (SGVs) in 2002 together with a human health risk assessment model called Contaminated Land Exposure Asessment (CLEA) 2002 model. The SGVs were derived using the according to three typical land-uses, applicable to long-term human exposure to soil contaminants. The three types of land-use are:

- Residential with plant uptake (for example, home grown vegetables) or residential without plant uptake;
- Allotments; and
- Commercial/industrial.

In November 2005 an update to the CLEA 2002 software was released, called CLEA UK. The Environment Agency withdrew the CLEA 2002 in October 2006.

The CLEA UK model (Figure 1) was intended to be used to derive generic assessment criteria where no SGVs are available. However the CLEA UK model is only applicable for select pathways and all these pathways relate to soil. One of the major pathways which is not included within this model is inhalation of shallow water vapour via vapour intrusion. This pathway is commonly present at sites and can be present in the form of perched water (which may be associated with the geology or due to former structures on the site) or shallow groundwater. Therefore, in order to assess this pathway, other commercially available risk assessment tools need to be considered. This paper concentrates on the commercially available risk assessment tools that are applicable to the inhalation of water vapours pathway and presents a case study where one of these tools has been applied.

Figure 1 CLEA UK Main Screen

VAPOUR INTRUSION

Vapour intrusion is the migration of volatile chemicals from the subsurface into overlying buildings (Ref. 1). Volatile chemicals may include volatile organic compounds (e.g. benzene), select semi-volatile organic compounds (e.g. naphthalene), and some inorganic analytes, such as elemental mercury and hydrogen sulphide. In extreme cases, these vapours may accumulate at concentrations that pose near-term safety hazards (e.g. explosions or acute health effects) or aesthetic problems (i.e. odours); however, it is more likely that the chemical concentrations will be low, if detectable at all. In the case of lower concentrations the main concern is usually whether or not there is an unacceptable chance of longer-term chronic health effects (Ref. 2). The inhalation of vapours indoors is typically considered more critical than outdoors as mixing with outdoor air results in large dilutions of gaseous contaminants so that their concentrations in the outdoor breathing zone are usually negligibly small (Ref. 3).

MECHANISM OF GROUNDWATER VAPOUR ENTRY INTO BUILDINGS

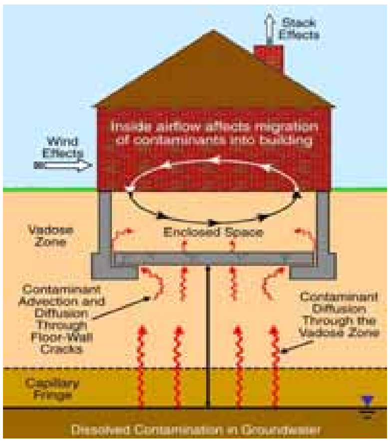


Figure 2 Illustration of the mechanisms of vapour entry into buildings (source: Interstate Technology Regulatory Council, 2007)

At a site where groundwater is impacted with volatile contaminants, the volatiles will volatilise beneath a building and diffuse toward regions of lower chemical concentration (e.g. the atmosphere, conduits, basements). Vapour from groundwater can flow into a building due to a number of factors, including barometric pressure changes, wind load, thermal currents, or depressurisation from building exhaust fans (Ref. 4). The rate of movement of the vapours into a building is a difficult value to quantify and depends on soil type, chemical properties, building design and condition, and the pressure differential. Upon entry into a structure, volatile contaminants mix with the existing air through the natural or mechanical ventilation of the building. There are two mechanisms by which volatile contaminants can enter a building: diffusion and advection. Diffusion is the mechanism by which soil gas moves from high concentration to low concentration due to a concentration gradient. Advection is the transport mechanism by which soil gas moves due to differences in pressure. These pressure differences can be generated by atmospheric pressure changes, temperature changes creating natural convection in the soil, or forced pressure changes due to building ventilation systems.

Advective transport is likely to be the most significant in the region very close to a basement or a foundation, and soil gas velocities decrease rapidly with increasing distance from the structure (Ref. 1). Once soil gases enter the "building zone of influence", they are generally swept into the building through foundation cracks by advection due to the indoor-outdoor building pressure differential. The reach of the "building zone of influence" on soil gas flow is usually less than a few feet, vertically and horizontally (Ref. 5). The mechanisms of transport are demonstrated in Figure 2.

Human health risk assessment models use a number of different algorithms to model this process of vapour intrusion. These algorithms use a combination of factors such as soil properties, building properties, fate and transport (e.g. Partitioning of the contaminant) to estimate the concentration in indoor air from volatilisation of volatiles in groundwater.

KEY COMMERCIAL HUMAN HEALTH RISK ASSESSMENT TOOLS

BP RISC (RISC WORKBENCH)

The RISC Workbench (version 4.03) (Figure 3) programme was developed by BP Oil International Ltd to promote a consistent and transparent approach when dealing with its portfolio of sites (Ref. 6). It is based on the American Society for Testing and Materials (ASTM) Risk Based Corrective Action (RBCA) methodology (Ref. 7), with additional pathways, fate and transport and contaminant information. The default exposure and contaminant related parameters within the model are based on US policy and therefore the model requires considerable adaption when used within the UK or other European countries. The model contains a number of pathways including:

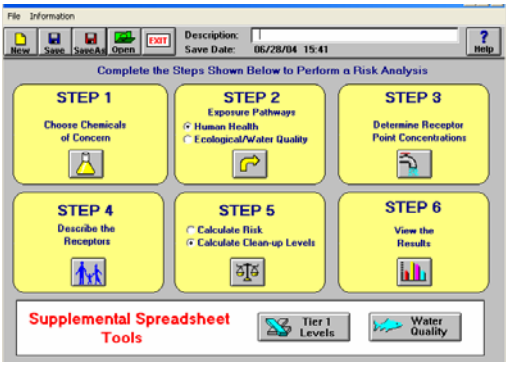


Figure 3 BP RISC Front Screen

- Direct ingestion of contaminated soil;
- Dermal contact with contaminated soil;
- Ingestion of vegetables grown in contaminated soils;
- Ingestion of contaminated groundwater;
- Inhalation of contaminated groundwater during showering;
- Inhalation of soil vapours in outdoor air;
- Inhalation of soil vapours in indoor air;
- Ingestion of surface water (during swimming);
- Dermal contact with surface water (swimming);
- Ingestion of groundwater used for irrigation by children playing under a sprinkler;
- Inhalation of volatile components of groundwater used for irrigation;
- Dermal contact with sprinkler;
- Ingestion of vegetables irrigated with contaminated groundwater.

This model enables the user to calculate assessment criteria for soil or groundwater (which are termed "Site Specific Target Levels" (SSTLs)) which can be compared to the soil or groundwater concentrations present at a site. The prediction of the concentration of vapour in indoor air is modelled based on a modified version of the Johnson and Ettinger algorithm (Ref. 8). The Johnson and Ettinger algorithm uses building properties and soil properties to express the proportionality between the concentration in soil vapour and the consequent concentration in indoor air. It derives two related factors, which are, Q: building, the volume of air exchange moving through the building (expressed as litres per minute) and Q: soil, the volume of vapour moving through the soil into the house (also expressed as litres per minute). The ratio of these two factors leads to the 'soil vapour to indoor air attenuation factor', or alpha (α). This algorithm assumes that there is a ground bearing slab and the groundwater source is directly beneath the building. The algorithm only accounts for volatiles dissolved in groundwater, not present as free product. The user is able to adjust parameters relating to building dimension, soil type and depth of contaminant to ensure the fate and transport algorithm reflects the site being assessed. However the user is unable to enter a pressure differential for the building and consequently advective flow is not included as a mechanism of entry of vapours into the building. As discussed above advective flow is a major route of entry of vapours into buildings, therefore groundwater assessment criteria derived from the RISC model to assess the risk to human health from groundwater vapours are likely to underestimate the potential risk.

RBCA (RISK BASED CORRECTIVE ACTION)

The RBCA tool kit model (version 1.3b) (Figure 4) is an Excel based model, which can be used to calculate assessment criteria (termed "Site Specific Target Levels" (SSTLs)) for soil or groundwater. The model is based on the Tier 1 and 2 processes defined within the ASTM (American Society for Testing and Material) (Ref. 7). The RBCA tool kit was developed for use in the USA and as such incorporates U.S. regulatory policy. However the default parameters within the model can be adapted. The model contains a number of pathways including:

- Ingestion of groundwater;
- Inhalation of groundwater vapour;
- Discharge of contaminated groundwater to surface water;
- Ingestion/dermal contact via swimming;
- Ingestion via fish consumption;
- Inhalation of soil vapour and particulates;
- Direct dermal contact with soil;
- Ingestion of soil and dust (incidental);

The prediction of the concentration of vapour in indoor air from volatilisation of groundwater vapours is, like RISC Workbench, modelled based on a modified version of the Johnson and Ettinger algorithm (Ref. 8). Again the user is able to adjust parameters relating to building dimension, soil type and depth of contamination to ensure the fate and transport algorithm reflects the site being assessed. The main difference from RISC Workbench is that the user is able to enter a pressure differential for the building and consequently advective flow is activated as a mechanism of entry of vapours into the building. Consequently the difference in numbers generated between the RBCA model and the RISC model are significant for this pathway. Figure 5 demonstrates the difference in groundwater assessment criteria derived from the two models adapted for a UK standard residential land use (as defined within CLR9 and CLR10 (Ref. 9 and 10).

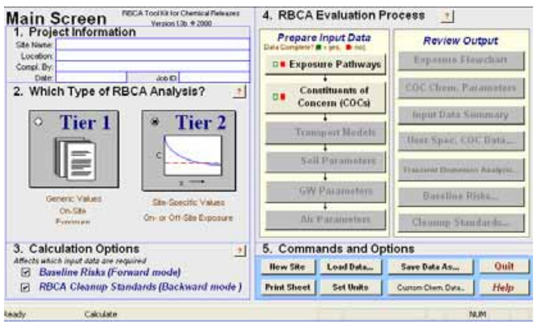


Figure 4 RBCA Tool Kit main screen

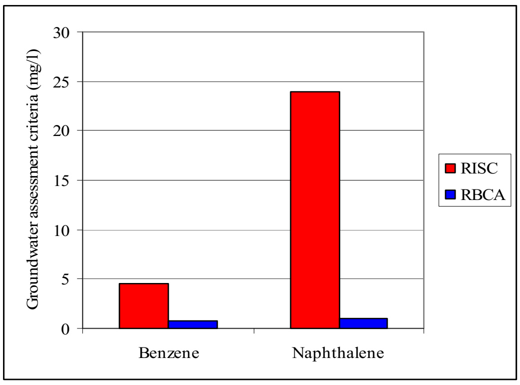


Figure 5 Comparison of groundwater assessment criteria derived from RISC workbench and RBCA Toolkit. Values have been derived using a standard residential land use within CLR10 (including the updates within CLEA Briefing Notes 2 and 3 (Ref. 11 and 12)) and includes modification of the toxicological data to include UK policy within CLR9. The source depth is assumed to be 1m below ground level.

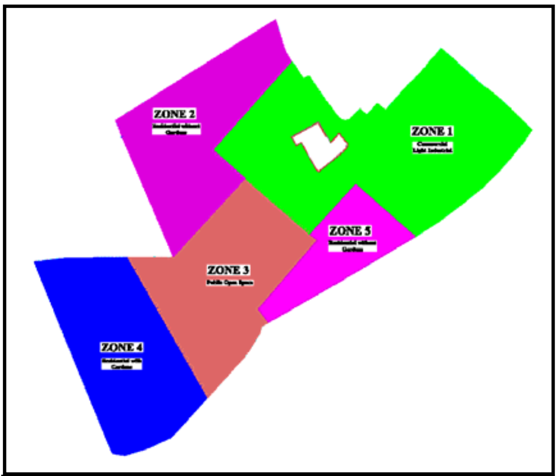
CASE STUDY

The site in question is a former gasworks in Southern England. A number of site investigations were undertaken at the site from 1988-2006 and volatile contaminants such as benzene, naphthalene and light Total Petroleum Hydrocarbons (TPH) fractions were identified as the key contaminants of concern in both the groundwater and the soil. The soil and groundwater were found to be heavily impacted in several areas of the site (Figure 6). Shallow groundwater was found to be present within gravel deposits at an average water depth of 1.5m bgl. The general direction of shallow groundwater flow within the gravel deposits was found to be south to north, becoming more north-westerly in direction towards the western edge of the site. The site was proposed to be redeveloped for mixed land use which was to include residential housing (Figure 7). In order to assess the potential risk to future human receptors from the inhalation of groundwater vapours indoors, risk assessment modelling was undertaken.

Figure 6 A Trial Pit with heavily impacted soils and groundwater

METHODOLOGY

The RBCA model was selected as a suitable risk assessment tool for deriving assessment criteria for the groundwater in order to determine the remedial requirements at the site. This is due to the inclusion of the advective flow pathway within the model algorithms. The RBCA model was fully adapted to be consistent with the UK toxicological approach outlined in CLR9 (Ref. 9) and UK default input parameters (where appropriate) outlined in CLR10 (Ref. 10). Assessment criteria were derived for both a commercial/industrial and residential land use. The results of the risk assessment modelling were plotted graphically to assess the potential risk within each zone from the groundwater concentrations but also to determine whether there was a potential risk from the migration of higher groundwater concentrations downgradient to a more sensitive zone (i.e. migration from less sensitive land use commercial/industrial to residential). This is demonstrated within Figure 8, which illustrates the concentrations of TPH Aliphatic >C10-C12 compared to the appropriate derived groundwater assessment criteria for each zone.



Zone 1- light commercial/industrial
Zone 2- residential without plant uptake
Zone 3- public park
Zone 4- residential with plant uptake
Zone 5 - residential without plant uptake

Figure 7 Proposed redevelopment areas for a former gasworks site

RESULTS

The results of the risk assessment indicated that there was a potential risk to human health from the inhalation of groundwater vapours pathway from several volatile contaminants, which are, TPH Aliphatic >C8-C10, TPH Aliphatic >C10-C12 and benzene. Therefore the results of the risk assessment modelling were used to inform a suitable remediation strategy for the site.

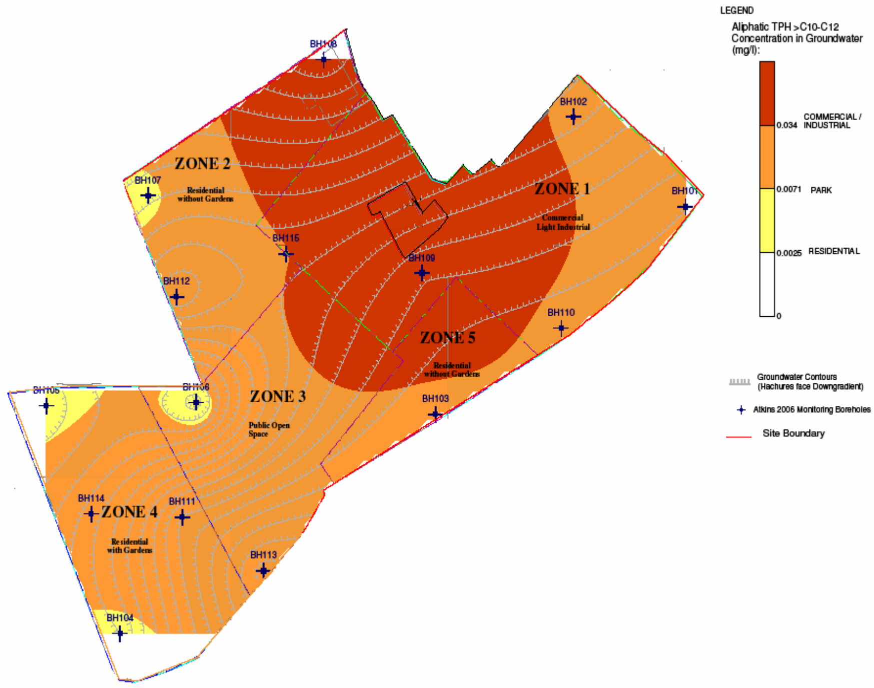


Figure 8 Groundwater plot illustrating the groundwater concentrations of TPH Aliphatic >C10-12 compared to the derived assessment criteria from RBCA

LIMITATIONS OF RISK ASSESSMENT MODELLING

As discussed in detail above the Johnson and Ettinger algorithm is a one-dimensional model which predicts the concentrations of volatiles in indoor air from a volatile source. The 'soil vapour to indoor air attenuation factor', or alpha (α) factor calculated from Q:soil and Q:building is generally considered to be conservative in most circumstances as it makes assumptions regarding the partitioning process. There is evidence to suggest that the Johnson and Ettinger model may over predict indoor air concentrations (Ref. 13). In reality there is a complex relationship between soil properties, building factors and atmospheric factors and typically the concentrations are much lower within a building than what may be detected within the soil or groundwater. Figure 9 demonstrates a study carried out within the U.S. for Trichloroethylene (TCE), which illustrates this point (Ref. 14). The study indicated that there was a large difference between what was measured in indoor air and what was present beneath the floor slab.

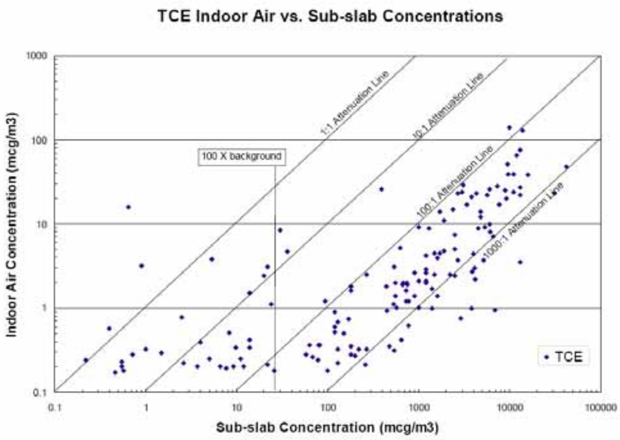


Figure 9 TCE concentrations in the Indoor Air vs Sub Slab Concentration (source: McDonald G.J. and Wertz W.E., 2007)

In addition, the Johnson and Ettinger model does not consider advective water movement within the soil column, nor does it consider transformation processes (e.g., biodegradation, hydrolysis, etc.). The Johnson and Ettinger model was based on a number of simplifying assumptions (e.g., homogeneity, diffusion-only through subsurface, uncontaminated capillary fringe, etc.). Conditions under which the Johnson and Ettinger model should not be used include (Ref. 15):

- Presence or suspected presence of Non Aqueous Phase Liquids (NAPLs);
- Heterogeneous geology, fractured media, karst or macropores;
- Sites where significant lateral flow of vapours may occur (e.g., utility conduits);
- Very shallow groundwater that wets building foundation;
- Very shallow groundwater source;
- Very small building air exchange rates (e.g., <0.25/hr);
- Buildings with crawlspaces, earthen floors, stone floors, etc.;
- Contaminated groundwater sites with large fluctuations in water table elevations; and
- Sites with time-varying flow rates and/or concentrations for which a steady state assumption is not conservative.

SUMMARY AND CONCLUSIONS

The increased use of Brownfield sites for housing has lead to the need for rigorous human health risk assessment. One of the major pathways which pose a risk to human health is the inhalation of shallow water vapour via vapour intrusion. This pathway is commonly present at sites and can be present in the form of perched water (which may be associated with the geology or due to former structures on the site) or shallow groundwater. Volatile contaminants within the groundwater can migrate upwards into buildings through a variety of mechanisms. The principal pathway is advective flow where vapours are drawn into the building through cracks and openings due to a pressure gradient. There are several commercially available risk assessment tools which can model the potential risk to human health from inhalation of vapours from groundwater, including RISC Workbench and RBCA. The RISC Workbench model does not include advective flow into buildings from groundwater vapours and therefore underestimates the potential risk when compared to models such as RBCA toolkit which does include advective flow. Whilst risk assessment modelling of this pathway is a useful tool for predicting human health risks and informing remedial strategies at a particular site, it is important to note that modelling has its limitations. There is evidence to suggest that the Johnson and Ettinger algorithm used within the RBCA and RISC model can over predict in certain circumstances. In addition there are several situations where the application of the algorithm may not be appropriate to the site in question.

REFERENCES

1. USEPA. 2004b. *User's Guide for Evaluating Subsurface Vapor Intrusion into Buildings*.Revised. Office of Emergency and Remedial Response. (www.epa.gov/oswer/riskassessment/airmodel/pdf/2004_0222_3phase_users_guide.pdf)
2. Johnson P (2002) Identification of Critical Parameters for the Johnson and Ettinger (1991) Vapor Intrusion Model. A summary of research results from API's soil and groundwater technical task force.
3. Ferguson C, Krylov V and McGrath P (1995) Contamination of Indoor Air by Toxic Soil Vapours: a Screening Risk Assessment Model. Building and Environment. Vol 30, No 3, pp 375-383.
4. Interstate Technology Regulatory Council (2007) Technical and Regulatory Guidance. Vapour Intrusion Pathway: A Practical Guideline.
5. Environment Agency (2002) Vapour Transfer of Soil Contaminants. R&D Technical Report. P5-019/TR.
6. BP Oil International Ltd. RISC 4.0 Manual, available from BP Oil International Ltd.
7. American Society for Testing Materials (ASTM) (1995) Risk Based Corrective Action (RBCA) E1739-95. Standard Guide for Risk Based Corrective Action Applied at Petroleum Release Sites.
8. Johnson and Ettinger (1991) Heuristic model for predicting the intrusion rate of contaminant vapours into buildings. Environ. Sci. Technology. 25:1445-1452.
9. Defra and Environment Agency (2002). "Contaminants in the soil: Collation of Toxicological Data and Intake Values for Humans. Report CLR9.

10. Defra and Environment Agency (2002). The Contaminated Land Exposure Assessment model (CLEA): Technical Basis and Algorithms. Report CLR10.
11. Environment Agency (2004). CLEA Briefing Note 2: Update on Estimating Vapour Intrusion into Buildings. Version 1.1, July 2004.
12. Environment Agency (2004). CLEA Briefing Note 3: Update of Supporting Values and Assumptions Describing UK Building Stock. Version 1.1, July 2004.
13. Thomas E. McHugh, John A. Connor, and Farrukh Ahmad (2004) An Empirical Analysis of the Groundwater-to-Indoor-Air Exposure Pathway: The Role of Background Concentrations in Indoor Air. Environmental Forensics 5:33-44, 2004.
14. Gerald J. McDonald, William E. Wertz (2007) PCE, TCE, and TCA Vapors in Subslab Soil Gas and Indoor Air: A Case Study in Upstate New York. Ground Water Monitoring & Remediation 27 (4), 86–92.
15. US Environment Protection Agency (EPA) Evaluating Vapor Intrusion into Buildings from Contaminated Groundwater and Soils. http://www.epa.gov/athens/learn2model/part-two/onsite/jne_background_reverse.htm (viewed April 2008).

Session III

REMEDIATING ENVIRONMENTAL DAMAGE UNDER THE ENVIRONMENTAL LIABILITY DIRECTIVE: A COMPETENT AUTHORITY STRATEGY

Dr Jonathan Derham, Pgeo, Eurgeol,

Senior Inspector, Environmental Protection Agency[1]*,*

Licensing Unit, Office of Climate, Licensing & Resource Use

PO Box 3000, Johnstown Castle Estate, Wexford, IRELAND

j.derham@epa.ie

ABSTRACT

Emergency management is a key challenge and priority issue for Ireland (Irish Government, 2006). In recent years, EU law such as the Environmental Liability Directive (2004/35/EC), the IPPC Directive (96/61/EC), the SEVESO Directive (96/82/EC), the Habitats Directive (92/43/EC), the International Biodiversity Convention, as well as the national major emergency management protocols, all require incident or accident related environmental pollution risks to be evaluated, mitigated and remedied. The EU legislation is particularly concerned with protection of non-market environmental public goods such as biodiversity, ecosystems, protected species and natural resources (e.g. water). The competent authorities for these various requirements include the Heritage Service, the Local Authorities, the Health & Safety Authority and the Environmental Protection Agency. In the event of an emergency incident involving environmental damage it is essential that a scientifically and economically logical, robust and quantifiable process is followed, so as to assess and value such damage and to ensure an appropriate remedial strategy and outcome. The preferred approach, where considered cost beneficial, is to require that damaged environmental ecosystems or resources are returned to baseline status. In the event that this is not possible this paper sets down a paradigm for competent authorities to follow, which is protective of public welfare and ensures that habitats or environmental resources of equivalent status or value are provided. This thesis adds to the knowledge base necessary for competent authorities to adequately provide for public management of the planning for, and recovery stages of an emergency management plan where pollution damage is involved. Such incidents have happened in the past, and will happen again.

INTRODUCTION

Over the last 10 years the State has introduced a range of environmental regulatory instruments[2] that, *inter alia*, require the operators of potentially polluting activities (industry, waste operations, farming, genetically modified organism use, dangerous goods transport, etc.,), to put in place measures to prevent accidents, as well as measures to deal with the consequences of accidents when these occur. Some of the major landfills and hazardous waste treatment facilities, as well as pharmaceutical and

[1] The views expressed herein may not necessarily reflect those of my employer.

[2] e.g., Waste Management Acts 1996 – 2007; Environmental Protection Agency Acts 1992 & 2003; Health, Safety and Welfare at Work Act 2005; Planning & Development Act 2000; and associated regulations.

mining activities are typical – and would make up the bulk - of the operations subject to this new raft of environmental legislation.

Such operations typically require an environmental permit from the Irish Environmental Protection Agency – so called Waste, and Integrated Pollution Prevention Control (IPPC), licences[3].

In the IPPC licensing system, Section 83(5)(a)(ix) of the Environmental Protection Agency Act 1992-2003 (EPA Act) specifically prohibits the Environmental Protection Agency (the EPA) from granting an operating licence to a facility unless:

> *[the]* necessary measures will be taken to <u>prevent</u> accidents in the carrying on of the activity and, where an accident occurs, to <u>limit</u> its consequences for the environment and, in so far as it does have such consequences, to <u>remedy</u> those consequences.

Key elements of this binding obligation are the requirements to 'limit consequences' of accidents, and to ensure that any such consequences 'are remedied'.

An equivalent provision – if perhaps somewhat less explicit - is contained in Section 40(4)(h) of the Waste Management Acts 1996 – 2007 and speaks to waste facility licences issued by the EPA[4].

It is also necessary to mention the major industrial accident control legislation operated by the Health & Safety Authority (HSA) arising out of the implementation in Ireland of the EU 'SEVESO' Directive[5] - so named after a major industrial accident in Italy. The implementing regulations are known as the *European Communities (Control of Major Accidents & Hazards Involving Dangerous Substances) Regulations (SI 74 of 2006)*. This legislation seeks to ensure the protection of man and the environment in the event of a major industrial accident. These regulations, which are enforced by the Health and Safety Authority, set out certain obligations on operators of specified industrial/chemical storage operations in relation to the 'limitation of the consequences' of accidents. Many of these industrial operations are also subject to the aforementioned IPPC licensing system[6] operated by the EPA within which – as noted above - reside similar accident consequence limitation provisions. The net effect of this regulatory regime is that there is a dual competency between the EPA and the HSA in relation to accident consequence mitigation/limitation at most industrial sites. The SEVESO Regulations also place burdens on Local Authorities to prepare external emergency plans to cater for accidents on industrial sites.

All Local Authorities in the State are required under national Emergency Management initiatives to prepare Major Emergency Plans for their areas which, *inter alia*, take account of environmental risk scenarios including those associated with industrial accidents or incidents or other environmental risk scenarios such as chemical/fuel transportation. The national guidance on emergency planning (Irish Government, 2006a) opens by stating 'major emergency management is a key challenge and a priority issue for Government'.

It is necessary for the EPA, as a competent authority, to articulate for itself and indeed any environmentally regulated activity in the State, what is *understood* by the clause 'limit its consequences' and, in particular, what is *expected* by the clause 'remedied' or 'remediated'.

The requirement to understand these concepts is also relevant to requirements under the EU Habitats Directive (92/43/EC) as well as adoption of the Convention on Biological Diversity[7] ratified by

---

[3] See www.epa.ie/licensing

[4] Waste Management Acts 1996-2007 Section 40(4)(h):- *necessary measures will be taken to prevent accidents in the carrying on of the activity concerned and, where an accident occurs, to limit its consequences for the environment,*

[5] EU Council Directive 96/82/EC (as amended) on the Control of Major Accident Hazards involving Dangerous Substances. OJ L10, 14 January 1997.

[6] Not all SEVESO sites are IPPC activities, and vice versa. For example large petroleum storage depots are SEVESO sites, but are not regulated by IPPC as there is no associated industrial manufacturing process.

[7] Un Convention #30619. Done at Rio de Janerio on 5 June 1992, by Albert Reynolds, etc.

Ireland in 1996, and requiring, *inter alia*, national measures to protect and restore damaged ecosystems. This is even more pressing given the forthcoming transposition into Irish law of a very significant – and controversial – piece of European Union environmental law. I speak of Directive 2004/35/CE on *Environmental Liability with Regard to the Prevention and Remedying of Environmental Damage*[8]: known as the Environmental Liability Directive (ELD).

THE ENVIRONMENTAL LIABILITY DIRECTIVE

The fundamental aim of this Directive is to hold operators (public & private) whose operations have caused environmental damage financially liable for remedying that damage. The ELD also has elements dealing with preventative actions in the event of an imminent threat of environmental damage.

As is the case with some of the existing national provisions discussed above, the scope of the ELD is wide and covers the major potential sources of harm to the environment and human health, including the majority of industrial activities in the State, the Local Authority and private sector waste activities, activities involving genetically modified organisms, marine discharges, activities associated with the transport of dangerous goods, municipal waste water treatment plants, bulk chemical and fuel storage, etc.

The ELD does not seek to levy criminal penalties, nor does it deal with compensation for loss of private goods, income, etc., due to environmental accidents. Such matters are dealt with by other legislation including civil and criminal law provisions. The main aspect of the environment that the ELD seeks to protect is that which is considered *public good* natural resources. The ELD is intended to cover damage to protected species, natural habitats, and natural resources such as air, surface and groundwater and soil. The ELD also introduces something new to the concept of liability for the first time across the Union, and that is a scheme to remedy damage to biodiversity. Diffuse sources of pollution (e.g. from cars exhausts, or nutrients from agricultural communities – i.e. non point sources) are not addressed in the ELD.

It is expected that the requirements of the EU environmental liability directive will sit along-side of such existing national provisions - where indeed they exist – in relation to preventing and limiting the consequences of accidents, and the remedying of any consequences.

The influence of the ELD will span not only all the EPA regulated activities (potentially 2000+ private and public sector operations including IPPC, Waste, GMO & Urban Waste Water Treatment Plants) but also a great many activities regulated by the Local Authorities, the Department of Marine, Communications & Natural Resources, and the Health and Safety Authority. All HSA regulated SEVESO sites are subject to the requirements of the ELD. The ELD, therefore, is the common thread that binds various regulatory authorities into a shared purpose, with the consequent need to articulate collective understandings of environmental imperatives for pollution incidents. These accident related environmental damage imperatives can be articulated thus:-

… to devise a common understanding of the scope and meaning of;

- Environmental damage risk assessment,
- Environmental damage management/mitigation,
- Damage consequence assessment, and
- Damage recovery management.

The latter two are the main elements discussed in this paper.

[8] Directive 2004/35/CE, OJ L143/56, of 30 April 2004.

A MODEL FOR NATURAL RESOURCE DAMAGE VALUATION AND REMEDIATION

In order to value damage it is firstly important (but not essential) to have evaluated risk scenarios and vulnerable receptors. The role of risk assessment in 'context' and 'baseline' establishment for vulnerable environmental receptors will enhance the ability of the competent authorities to accurately define a remedial strategy. This risk assessment step, which naturally precedes any consequence evaluation and remedial action, is not covered in this paper

This paper sets out a protocol or model to frame the requisite activities and actions of a competent authority within the Recovery Stage of a major pollution incident.

The practice in the US and also that mandated by the EU Environmental Liability Directive (ELD)[9] for Member States, is that the remedial strategy for damaged environmental resources or habitats – following an anthropogenic pollution incident – can only result in three possible alternatives:-

- **Primary Remediation** – restoring the damaged natural resource back to original state (baseline);
- **Complementary Remediation** – where it not practical or possible to restore the damaged environment to its baseline state, the solution can involve the provision of a similar level (quality or type) of natural resource at an alternative location;
- **Compensatory Remediation** – where the remediation to baseline conditions may not be technically or economically possible, or take an extended period (i.e. decades) the solution can include provision of, or improvements to, other sites, or the introduction of additional features to the damaged site as a mechanism to compensate for interim losses.

These 'solutions' can be represented graphically. Figure 1 is adapted from models presented by the US National Oceanic and Atmospheric Administration web pages[10] and from similar figures presented in the EU funded REMEDE web site[11], and a consultants report for the Commission (EU Commission, 2001).

The choice as to which option is acceptable has to be made by the competent authority(ies). The Government has yet to decide who the competent authority for the ELD will be, but the EPA is likely to be one of the principal authorities for its implementation. Other State bodies such as the Office of Public Works, the National Parks & Wildlife Service, Fisheries, Teagasc, Local Authorities, the Health and Safety Authority, etc., will all likely have a role in advising and informing the competent authority in relation to selection of the optimum solution.

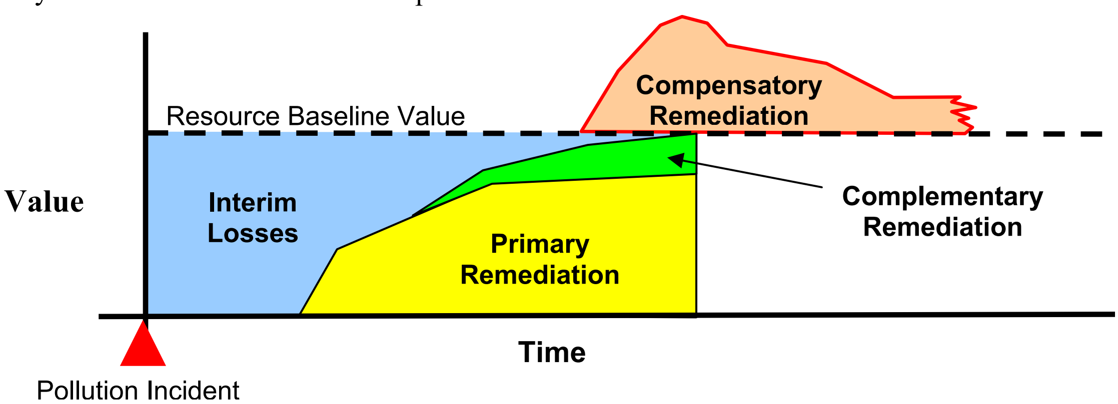


Figure 1: Diagrammatic relationship between Primary, Complementary & Compensatory Remediation, value and time.

---

[9] Directive 2004/35/CE
[10] www.csc.noaa.gov/coatal/economics/habitatequ.htm
[11] www.remede.eu REMEDE stands for Resource Equivalency Methods for Assessing Environmental Damage in the EU. An EU Commission funded initiative.

There are three main measures for environmental ecosystems/resources: Type, Quality and Value. Assuming that the default position for environmental trustees is that the ultimate goal is to generate a remedial strategy that delivers a solution which is of equivalent Value, then the deciding factors for differentiating between Primary, Complementary and Compensatory remediation can be thought of as Quality and Type. Figure 2 explains this relationship.

		TYPE (of remediated ecosystem/resource)	
		Same	Different
QUALITY (of remediated ecosystem/ resource)	Same	Primary Remediation	Complementary Remediation
	Different	Complementary Remediation	Compensatory Remediation

Figure 2: Regulatory preferences for remediation, relationship between restoration objective, type and quality. Assumes equivalent value.

Competent authorities – or environmental trustees - being the guardians of environmental public goods, have, following a pollution incident, responsibilities to see that the damage is remediated (directly or by replacement), and to seek on behalf of society adequate compensation for lost ecological or resource service/value – so termed interim losses. Interim losses embrace losses for the period following damage until baseline or equivalency is achieved, as well as any estimate of infinite losses due to irreparable damage. This is a complex area of governance requiring a range of specialist expertise. The mechanisms for determination of habitat or resource equivalency, or for assessment of the service value or utility of such biodiversity or environmental resources is scientifically, legally, economically and ethically complex (e.g., EU Commission (2001); UK Government (2006); US NOAA, 1997; etc.,). Non-monetary resource-to-resource, service-to-service, and value-to-value assessments are usually based on expert judgement, whereas monetary value is based on individual preferences (e.g. deliberative and participatory valuation methods). To quantify environmental damage and determine the most appropriate remedial strategy, access to technical skills in, *inter alia*, law, science, economics, engineering, and ecology will be necessary.

In order to assist discussions, and to support the setting out of a necessary protocol for the competent authorities for post-incident Recovery Stage management, a process flow diagram is presented in Figure 3 that plots the various incident Recovery Stage management sub-steps or Phases. In addition, this Figure sets out the typical actions that would accompany each phase.

As stated above, the default objective of the environmental trustees must be to achieve Primary Remediation (the baseline status having being established (usually) as part of a pre-incident risk assessment process) but not at any cost. There are two main reasons for not pursuing Primary Remediation:- (1) it may not be technically possible, and (2) the cost of the primary remedial strategy is excessive to the extent that the value for money is questioned. In relation to *Phase III – Remediation Planning* in Figure 3, it is possible to further 'explode' this phase to provide enhanced visualisation of the individual elements of this Phase, see Figure 4. Note in particular the 'nodes' at which technical feasibility and cost/value are evaluated. The EU Environmental Liability Directive recognises the role of Cost Benefit Analysis (i.e. *is a given objective worth achieving?*) and Cost Effectiveness Analysis (i.e. *what is the most cost efficient way of achieving the objective?*), in assessing restoration proposals.

PHASES	ACTIVITIES
Phase I - Damage Assessment Initiation	- Quantification of needs / scoping - Team establishment - Legal requirements - Scheduling
Phase IIA - Damage Assessment	- Deployment of Technical Experts - Consultation with relevant competent agencies (standards, designations, etc) - Data collection - Gather data on 'expectations' of environmental trustees, as well as those of public and political representatives -
Phase IIB - Damage Reporting	- Report findings - Assess damage - Select 'metric' - Scale damage - Assess/scale against baseline
Phase III - Remediation Planning	- Define remediation objectives / requirements / Expectations - Develop remediation alternatives - Undertake HEA / REA on alternatives - Apply CBA / CEA / VEA on alternatives as appropriate - Select preferred remediation solution - Select compensatory solution (for interim losses) - Ensure acceptance by environmental trustees & interested parties
Phase IV - Remediation Implementation	- Assign resources - Scheduling - Implement agreed remediation plan - Contracting & supervision - Finance
Phase V - Remediation Monitoring & Validation	- Establish monitoring and reporting programme for remediation plan - Report to technical expert steering group (incl. competent authority) - Re-evaluate remediation objectives and refine remediation plan as necessary

Figure 3: Phased approach to post incident Remediation planning

Where Primary Remediation is considered cost excessive or not technically possible, then in such circumstances Complementary Remediation is contemplated, it too being subjected to a Cost Benefit Analysis (refer also to Figure 4). It is significant to note that Complementary Remediation may not take place at the site of damage: most likely for the same reasons that primary restoration would not be possible there – e.g. groundwater too damaged by DNAPLs to be usable. Moreover it may not be possible to provide by Complementary means, a habitat or resource equivalent (in type of quality) to that which was damaged, i.e. it may not be technically possible or cost excessive. For Compensatory Remediation, equivalency analysis is used to provide habitat, resource or value estimates of equivalency (for damage endured). Compensatory remedial mechanisms can then be by the provision of 'resource' or 'service' or 'value' (ecological or monetary) equivalency to that lost or endured.

EQUIVALENCY ANALYSIS

This reference to 'equivalency' and equivalency analysis seen in Figure 4 above, is a fundamentally important process in the planning of a remedial strategy under the ELD. Habitat Equivalency Analysis (HEA) and Resource Equivalency Analysis (REA) are - as noted by the EU Commission REMEDE[12] working group (EU REMEDE 2007a) – favoured in various EU ecological and environmental Directives[13]. The alternative approach, i.e. the calculation of a monetary value for ecological damage, is not without its problems; principal amongst which is due to market failure for such public goods (biodiversity, habitats, etc.,). HEA and REA in economical terms represent a non-monetary service-to-service, resource-to-resource evaluation approach. It can be considered 'replacement in kind' – i.e. of a type or quality that was lost. In Figure 2 we saw that resource or habitat 'Quality' and 'Type' were central to the determination of equivalency in relation to regulatory preferences for remedial strategy selection. The key to the equivalency methods is the identification of a 'metric' (or surrogate), to compare lost resource and required replacement, e.g. total number of species, number of a particular species, m^2 of wetland, m^3 of groundwater, etc.

HEA is used where the task is to estimate the losses following a pollution event and gains due to remedial activities, and where the metric is in units of habitat. Whereas in REA the losses and gains are measured in units of resource (e.g. a specific species, or m^3 of groundwater). In the case of 'Compensatory' remedial activities, Value Equivalency Analysis (VEA) is used, where methodologies such as Contingent Valuation are applied to scale the compensatory activities against losses endured or observed. VEA is used where HEA/REA is not possible, i.e. where the resource or service is not technically (e.g. irreplaceable elements, unique species lost) or economically possible to remediate or where remediation of a like or equivalent kind is not possible (i.e. remediation to same type or same quality not possible – c.f. Figure 4). Contingent Valuation methodologies, despite their well published contribution to ecological damage valuation exercises (e.g. Arrow et al., 1993) will unlikely ever capture the total value or service that biodiversity or ecosystems provide to humanity – because this is probably not fully understood. The human comprehension of biodiversity value, as expressed perhaps in willingness to pay and willingness to accept evaluations, may well be rational but is likely to be a very shallow understanding of - it is acknowledged – an unquantifiable reality. It is important that the competent authorities, or environmental trustees, who pursue the Compensatory measures on behalf of society understand these limitations.

EU REMEDE (2007a) sets out three main steps for HEA / REA:-

- Quantification of the effects of environmental damage in terms of lost resources or services.
- Identification and evaluation of remedial options in terms of quantity and quality of service or resource replacement required.
- Scaling of the remediation to assist determination of the necessary compensation for interim losses.

[12] Resource **E**quivalency **M**ethods for Assessing **E**nvironmental **D**amage in the **E**U
[13] Environmental Liability Directive, Habitats Directive, Wild Birds Directive, etc.

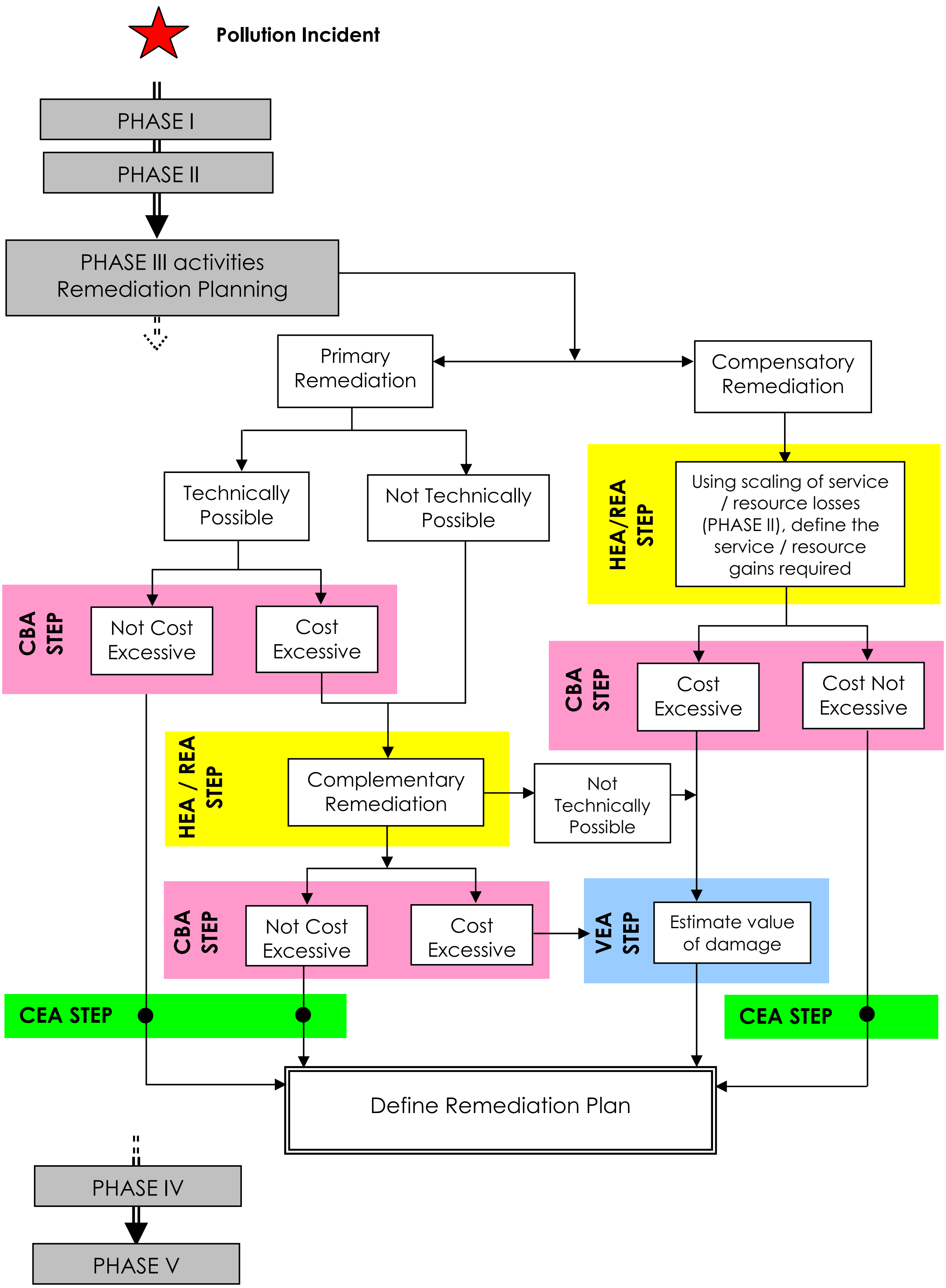


Figure 4: Decision tree for Phase III Activities (Selection of Remediation Plan)

The selection of an appropriate 'metric' or 'scalar' to be used to scale damage and remedial efforts will have to be a matter for expert decision by the technical professionals available to the competent authority. Different metrics can give an indication of the primary productivity or health of an ecosystem or resource. Naturally the distillation of the value or contribution of an ecosystem to only one 'metric' is a gross oversimplification of the complexity of interaction and interdependency in biological and natural resource systems. Nevertheless it is incumbent on the technical specialists assisting the competent authority for the recovery stage of an emergency incident, to identify 'a' or 'the' keystone metric that is acceptable to all as the means to scale damage and remedial effort (i.e. the surrogate).

In addition to the requirement for a 'metric', and as noted in US NOAA (2006) and EU REMEDE (2007a), HEA and REA are only considered to be best applied for the estimation of remediation where:- (a) there is an approximate service equivalency (or equivalency potential) between the anticipated remediated site and that possessed by the pre-damage site, and (b) where the technical experts can practicably obtain (or calculate) sufficient data on pre- and post-damage site characteristics to input into the equivalency model (such as baseline level of service, nature and extent of injury, percentage reduction in service, projected recovery period, recovery objectives, etc.,). These model conditions are necessary for HEA/REA to yield an appropriate remedial strategy. And as noted previously, if the remedial strategy indicates that a resource or service of some equivalence to that lost is not technically reasonable or achievable, or/and financially practicable (CBA), then the use of VEA in compensation assessment is to be recommended.

There is substantial North American and EU experience on how to undertake HEA / REA (e.g. US EPA (undated); US NOAA, 1997, 1999, 2004, 2006; EU REMEDE 2007a, 2007b, 2007c). In the case of valuation techniques (e.g. VEA), published work by Arrow et al., (1993), Carson et al., (1992), Carson et al., (1995), Breffle & Rowe (2002), Breffle et al., (2005), and Lazo et al., (2005) and others on Contingent Valuation and other deliberative and participatory valuation techniques, gives a good overview of the methodologies available. Accordingly it is not proposed herein to repeat the detail of how such methodologies are applied, but rather - and as suggested in the process flow diagrams, c.f. Figures 3 & 4 - this paper has set out a guide as to the role and place (sequencing) for these processes in support of decision making for a post-pollution incident recovery programme.

A final point in relation to the management process flow charts presented in Figures 3 and 4, is to note that a Cost Recovery phase is not shown. However, it is important to realise that at some stage in the process (either before implementation of remedial plan, or after successful implementation, or staggered throughout) that someone has to pay. The competent authorities will have to ensure that the responsible party(ies) – should there be any – underwrite the Recovery Stage; and it is likely that the final amount will end up being decided in court. There could well be legal challenge to the metric (or scalar) used, the application of CBA, and most probably to the VEA methodologies, i.e. the compensatable value. Suffice at this stage to note that the costs of all aspects of the incident management and recovery should be covered, but only in-so-far as that management deals with the public goods. Private goods are a matter for private compensation. Such *Public* costs might include:-

- Costs of immediate emergency response and containment of incident
- Costs of immediate gross pollution clean-up
- Costs of damage assessment
- Costs of remediation planning
- Costs of remedial plan implementation
- Costs of monitoring and validation

Mindful of this burden, the competent authority charged with management of the Recovery Phase will have to ensure that amongst the technical experts retained to advise and manage the remedial efforts, one should also include expertise in accountancy and book-keeping. Such skills along with those of the economists will also be relevant to the certain requirement for Cost Discounting in relation to the evaluation of remedial strategies. Future benefits and costs must be discounted so-as to be expressed in present value terms. Remedial plans can take many years to be completed, with probable significant 'future' spend events.

Pollution remediation projects carry a certain amount of Risk and Uncertainty. These too will have to be accounted for by the experts in project planning and budgeting. Examples of risk/uncertainty might be:-

- Baseline quality not known.
- Precise timeframe to achieve objectives not known.
- Total cost to end of remediation unpredictable.
- Likelihood of certain strategies being a complete success not known.
-

Project timeframe is one of the principal risks with pollutant remediation projects: they can take a very long time, are complex, and often project goals may have to be redefined at various stages. This protracted effort (20 to 30yrs plus) will require stamina and sustained commitment from the competent authority and its technical advisors. Project management, knowledge management and management continuity planning are therefore additional competencies that need to be factored into the remedial plan management architecture.

CONCLUDING REMARKS

In Ireland, Government initiatives have evolved a range of mechanisms which provide for the prevention and mitigation of pollution events, and the remediation of any damage caused - primarily via the making and enforcement of laws. National and EU law requires competent authorities to ensure adequate measures are put in place to limit the consequences of pollution incidents and to provide for remediation of same. This paper articulates an understanding as to what would be expected by the clause 'remediated'. The environmental damage evaluation and valuation method of Habitat or Resource Equivalency Analysis was introduced and their important role in achieving a satisfactory outcome in the public interest demonstrated. It is clear from the damage assessment (including valuation) and damage recovery paradigms presented herein that the processes are highly technical and require a wide range of skills (scientific, legal, engineering, managerial, financial, etc.,). Environmental damage valuation is a complex area of governance, and is an area that the competent authorities as well as advisors to regulated industries will have to familiarise themselves with, if for not other reason but to ensure that risk activities are properly indemnified. Pollution incidents will occur again in the future and the EU Environmental Liability Directive, amongst other legal obligations, sets a high bar in relation to remediation expectations.

REFERENCES

Arrow, K., Solow, R., Portney, P., Leamer, E., Radner, R., and Schumann, H. (1993). *Report of the NOAA panel on contingent valuation*. Federal Register **58**: 4601-4614, January 15th 1993. [*the Blue Ribband Panel*]

Breffle, W., & Rowe, R. (2002). 'Comparing choice question formats for evaluating natural resource tradeoffs'. *Land Economics* **78**(2), pp298-314.

Breffle, W., Morey, E., Rowe, R., and Waldman, D. (2005). 'Combining stated-choice and stated-frequency with observed behaviour to value NRDA compensable damages: Green Bay, PBCs and fish consumption advisories'. In *The Handbook of Contingent Valuation*, D. Bjornstad, J. Kahn, and A. Alberini (eds). Edward Elgar Publ.

Carson, R., Mitchell, R., Hanemann, W.M., Kopp, R., Presser, S., and Ruud, P. (1992). *A Contingent Valuation Study of Lost Passive Use Values Resulting from the Exxon Valdez Oil Spill*. Report to the Attorney General of the State of Alaska.

Carson, R.T., Wright, J.L., Csrson, N.J., Alberini, A., and Flores, N.E. (1995). *A Bibliography of Contingent Valuation Studies and Papers*. La Jolla, California: Natural Resource Damage Assessment Inc.

EU Commission (2001). *Study on the Valuation and Restoration of Damage to Natural Resources for the Purpose of Environmental Liability*. Report prepared for EU Commission by MacAlister Elliot & Partners and EFTEC, reference B4-3040/2000/265781/MAR/B3. Brussels. Refer also web site at http://europa.eu/environment/liability/index.htm

EU REMEDE (2007a). *Deliverable #6A: Review Report on Resource Equivalence Methods and Applications*. Report prepared under the EU Commission 6th Framework Programme by *Status Consulting Inc.,* in support of the implementation of the EU Environmental Liability Directive. Published on the EU funded website www.remede.eu

EU REMEDE (2007b). *Deliverable #8: Draft Toolkit Document (outline).* Document prepared under the EU Commission 6th Framework Programme by consultants *Eftec* in support of the implementation of the EU Environmental Liability Directive. Published on the EU funded website www.remede.eu

EU REMEDE (2007c). *Deliverable #6B: Use of Natural Resource Equivalency Methods in Environmental Damage Assessment in the EU with respect to the Habitats, Wild Birds and EIA Directives*. Report prepared under the EU Commission 6th Framework Programme by consultants *Eftec* in support of the implementation of the EU Environmental Liability Directive. Published on the EU funded website www.remede.eu

Irish Government (2006). *A Framework for Major Emergency Management*. Inter Departmental Committee on Major Emergencies[14] National Working Group, chaired by the Dept. of the Environment, Heritage & Local Government, Dublin. www.emergencyplanning.ie

Lazo, J., Allen, P., Bishop, R., Beltman, D., and Rowe, R. (2005). 'Determining economic trade-offs among ecological services: planning for ecological restoration in the Lower Fox River and Green Bay'. In, *Economics and Ecological Risk Assessment, Applications to Watershed Management*. R. Bruins and M. Heberling (eds). CRC Press, FL, USA.

UK Government (2006). *Valuing our Natural Environment*. Report prepared for UK Department of the Environment, Food & Rural Affairs, by Eftec in association with Environmental Futures Ltd. Report Reference NR0103, dated 20th March 2006. Defra, London.

US EPA (Environmental Protection Agency) (undated). EPA web resource pages on Natural Resource Damage Assessments and the Superfund sites programme. See www.epa.gov/superfund/programs/nrd/primer.htm and related linked pages.

US NOAA (1995, revised 2000 and **2006**). *Habitat Equivalency Analysis: An Overview*. Unites States National Oceanic and Atmosphere Administration Damage Assessment & Restoration Programme; Policy and Technical Papers Series No. 95-1. Washington.

[14] The interdepartmental committee (DoEHLG, DoJELR, DoHC) has subsequently been replaced by the National Steering Group for emergency preparedness.

US NOAA (1997). *Natural Resource Damage Assessment Guidance Document: Scaling Compensatory Restoration Actions (Oil Pollution Act of 1990)*. Unites States National Oceanic and Atmosphere Administration Damage Assessment & Restoration Programme, Washington, D.C.

US NOAA (1999). *Discounting and the treatment of uncertainty in Natural Resource Damage Assessment*. Unites States National Oceanic and Atmosphere Administration Damage Assessment & Restoration Programme, Washington, D.C.

US NOAA (2004). *Habitat Equivalency Analysis*. Unites States National Oceanic and Atmosphere Administration Damage Assessment & Restoration Programme web resource at www.csc.noaa.gov/costal/economics/habitatequ.htm

THE USE OF 'MAGIC NUMBERS' IN THE ASSESSMENT OF CONTAMINATED LAND IN IRELAND

Kevin Cleary(1) & Grace O' Shea(2) , White Young Green

(1) Environmental Director, Eastgate House, Lock Quay, Limerick, kevin.cleary@wyg.com

(2) Senior Environmental Scientist, Unit 2, University Technology Centre, Curraheen Rd, Cork, grace.oshea@wyg.com

ABSTRACT

The assessment of contaminated land internationally has evolved significantly in the last 10 years with increased awareness and scientific knowledge on human health and environmental risks associated with the presence of contamination. Without pressure from the European Commission, Ireland has been slow to embrace changes and some risk assessors and regulators continue to use and misuse outdated methodologies to assess the risks associated with contaminated land. In Ireland a variety of soil screening values (SVs) from other European countries are being used, which in some cases are incorrectly seen as magic numbers to be achieved on remediation projects. It is generally accepted that the purpose of SVs is to provide nationally consistent guidance on the likely need for soil remediation without the very substantial costs associated with developing site specific assessment criteria. This paper provides a brief history and discussion of the methodologies and soil screening values used in Ireland. In the absence of national standards it is recommended that Irish regulators and risk assessors adopt the UK approach to deriving SVs.

1.0 INTRODUCTION

The Republic of Ireland does not have specific contaminated land legislation, however, it is acknowledged that existing legislation provides a considerable range of powers for dealing with contaminated land and has implications for any remedial actions that may be required[1]. In the absence of specific environmental legislation, the Irish Environmental Protection Agency (EPA) have stated that their approach to contaminated land is in line with other European countries, in that it encompasses pollution prevention, the polluter pays principle, the precautionary principle and the use of risk assessment to identify and prioritise sites requiring remedial action.

The EPA indicated to the Commission of the European Communities that they were "considering setting non-statutory guideline values for contaminants, both in soil and groundwater; which were to be derived from risk-based generic guideline values adopted in other European countries" (Ferguson, 1999). These values were to be tailored to meet Irish conditions and policies through a process of consultation with relevant bodies. The EPA envisaged that the guideline or screening values (SVs), when exceeded, would act as triggers to indicate whether further site specific investigation and evaluation is required. On sites where the identified contaminants were found to

---

[1] Existing Irish Legislation that provides powers to deal with contaminated land includes - The Waste Management Act 1996; The Environmental Protection Agency Act 1992; The Local Government (Water Pollution) Acts 1977-1990; Building Control Act 1990; and The Air Pollution Act 1987.

exceed the SVs then a site-specific risk assessment (SSRA) was deemed to be required to determine the actual risks to human health and the environment posed by the identified conditions. Considering the land use, appropriate remedial actions are then based on the findings of the SSRA. This approach is generally recognised as being best practise in Europe for the assessment of contaminated land. That said Irish SVs for contaminated land have yet to be developed and in their absence the common practise is to use the Dutch values, which are perceived by some Irish regulators as being a panacea when considering targets for remedial strategies at contaminated sites.

More than ever before, the approach to managing contamination must be thorough, appropriate and technically justified. Until the EPA develops a set of SVs for Ireland there is a need for an agreed set of SVs based on those available in other European countries. There is also a need for these SVs to be used correctly. The objective of this paper is to highlight the misuse of the commonly used SVs and to propose the use of the UK approach and values in the interim. It should be noted that best practise in the field of contaminated land is not static and there is a need for regulators and practitioners to remain up to speed with changes and improvements.

2.0 HISTORICAL REVIEW OF SOIL SCREENING VALUES IN IRELAND

The high population density and environmental legacy resulting from major industrial development in the UK and the Netherlands has meant that they have had a high incidence of contaminated sites. These sites are being remediated for environmental reasons and to facilitate development mainly in urban areas and as a result the UK and the Netherlands have led the way in developing SVs. Ireland on the other hand, has not had the same urgency in developing brownfield sites and consequently environmental policy for contaminated land management has not been a priority. In the absence of national SVs the Irish contaminated land industry therefore adopted the UK and Dutch guidance.

When contaminated land started to become an issue in Ireland in the mid 1980s the approach was to use the UK Interdepartmental Committee on the Redevelopment of Contaminated Land (ICRCL) guidance values for soils. These values presented generic standards termed ‘threshold levels’ and ‘action levels’. The ICRCL approach defined that, concentrations above ‘threshold levels’ could ‘usually be disregarded because there is no significant risk that the hazards(s) will occur’. When contaminant concentrations exceeded ‘action levels’, the ICRCL defined that ‘the risk of the hazard(s) occurring are sufficiently high that the presence of the contaminant has to be regarded as undesirable or unacceptable’ and that some kind of response was required. The range of contaminants for which action levels were required was limited and many regulators incorrectly interpreted exceedances in ‘threshold levels’ as an indication that remediation was required. The ICRCL guidance was formally withdrawn by the UK Department for Environment, Food and Rural Affairs (DEFRA) in 2002.

In the mid 1990s the general approach in both Ireland and the UK was to move away from the ICRCL values and to adopt the use of Dutch target and intervention values, which are discussed in more detail later in the paper. In 2000 the UK adopted contaminated land legislation and in 2002 it introduced a new methodology for establishing SVs, known as soil guidance values (SGVs). This move away from the Dutch guidance has resulted in the current situation in Ireland where a variety of SVs are being used and misused.

3.0 EU APPROACHES TO SOIL SCREENING VALUES

In general, a risk-based tiered process is used across Europe to assist in decision making on contaminated land sites. The source-pathway-receptor model is widely recognised for the assessment of risk to an identified receptor in the development of a site specific conceptual model. These three elements can exist independently, however only represent a risk when they are linked together, that is, to use the UK Environment Agency (EA) terminology, where a ‘pollutant linkage’ has been established (EA, 2004). In other words, a contaminant source must affect a receptor through a particular pathway, for example a pollutant (source) has to be proven to be affecting a water course (receptor) to present a hazard. This is also the general approach advocated by the EPA in Ireland.

In 2007 the European Commission published a research document (Carlon, 2007) which reviewed the use of SVs across Europe. The research identified the lack of a coherent framework in Europe for the derivation and use of SVs. But it was noted that SVs are derived for different purposes, with reference to various levels of risk. In general three levels were distinguished: negligible, intermediate (or warning) and potentially unacceptable risk. The level of risk is usually related to the intended application of the SVs.

In most EU countries, federal/national laws determine the legal framework of soil guideline values for contaminated land. As previously stated SVs are established to screen soil concentrations with reference to specific intended land use. Exceedances of SVs are analysed further and if necessary a site specific risk assessment through the use of computer models is used for the derivation of site specific target levels (SSTLs). Where an SSTL is exceeded, remediation is usually required.

Although the general approach to the assessment of contaminated land is consistent there are a large variety of SVs adopted by European countries. This has raised questions among risk assessors and regulators and led to a drive within the EU to standardise approaches within member states.

The number of substances for which SVs are available varies widely within the EU, ranging from less than 20 in some countries to a maximum of 234 (EU, 2007). In countries where SVs have been developed or adopted, SVs are generally available for 60 of the most common contaminant parameters (EU, 2007). These include heavy metals and metalloids, aromatic hydrocarbons, polycyclic aromatic hydrocarbons, chlorinated aliphatic and aromatic hydrocarbons, pesticides, dioxins and polychlorinated biphenyls.

National legislation across the EU varies as a result of differences in factors such as:

- Geology
- Planning policy
- Political decision making
- Industrial history
- Socio-cultural variability
- Approaches taken by the scientific community on aspects such as:
- Definitions of contaminant exposure
    - ➢ Receptor sensitivity
    - ➢ Definitions of critical receptor
    - ➢ Computer models for guideline value derivation

Most EU member states have adopted land use specific SVs with the exception of the Netherlands and Slovakia, where SVs are independent of land use. The most common classifications are outlined in Table 3.1 (Carlon, 2007).

<table>
<tr><th>COUNTRY</th><th colspan="7">LAND USE APPLICATION FOR SVs</th></tr>
<tr><th></th><th colspan="2">agricultural</th><th>natural</th><th>recreational</th><th colspan="2">residential</th><th>industrial</th></tr>
<tr><td>Austria</td><td colspan="3">agricultural or gardening purposes, as well as non-agrarian ecosystems</td><td colspan="3">residential areas, sport fields, playgrounds</td><td></td></tr>
<tr><td>Belgium – Flanders</td><td colspan="2">agricultural</td><td>nature</td><td>recreational</td><td colspan="2">residential</td><td>industrial</td></tr>
<tr><td>Belgium – Walloon</td><td colspan="2">agricultural</td><td>natural</td><td>recreational</td><td colspan="2">residential</td><td>industrial</td></tr>
<tr><td>Czech Republic</td><td colspan="2">agricultural</td><td>natural</td><td>recreational</td><td colspan="2">residential</td><td>industrial</td></tr>
<tr><td>Denmark</td><td colspan="2"></td><td colspan="5">Generic</td></tr>
<tr><td>Finland</td><td colspan="2"></td><td colspan="5">residential</td></tr>
<tr><td>Germany</td><td colspan="2">agricultural</td><td>green land</td><td>parks/recreation</td><td>Play ground</td><td>residential</td><td>industrial</td></tr>
<tr><td>Italy</td><td colspan="2"></td><td colspan="4">Residential/green areas</td><td>Commercial/industrial</td></tr>
<tr><td>Lithuania</td><td colspan="6">Agricultural, recreational and residential</td><td></td></tr>
<tr><td>Poland</td><td colspan="2">Agricultural and urbanized land</td><td>Nature and groundwater protection</td><td colspan="3">Agricultural and urbanized land</td><td>Industrial, mining and transportation</td></tr>
<tr><td>Slovak Republic</td><td colspan="2">Agricultural</td><td colspan="5">Generic</td></tr>
<tr><td>Spain</td><td colspan="2"></td><td>natural</td><td colspan="3">urban/residential</td><td>industrial</td></tr>
<tr><td>Sweden</td><td colspan="2"></td><td colspan="4">Sensitive land uses</td><td>Less sensitive with or without GW protection</td></tr>
<tr><td>Netherlands</td><td colspan="2"></td><td colspan="5">Generic</td></tr>
<tr><td>United Kingdom</td><td></td><td>Allotments</td><td>Natural (ERA SSVs)</td><td colspan="2">Residential with plant uptake</td><td>Residential without plant uptake</td><td>Commercial/industrial</td></tr>
</table>

Table 3.1: Land use applications of screening values in the a number of EU countries (Carlon, 2007)

Table 3.2 provides an illustration of the variable approaches of member states with reference to soil type (Carlon, 2007). In the Flemish Region of Belgium, the Netherlands and the UK, the SVs are corrected for parameters such as percentage clay, organic matter content and pH. In the case of Poland different values are provided according to depth of contamination and the hydraulic conductivity.

Country	Soil type function	
Austria	No	
Belgium - Flanders	Yes	clay and organic matter
Belgium - Walloon	No	
Czech Rep.	No	
Denmark	No	
Finland	No	
Germany	No	
Italy	No	
Lithuania	No	
Netherlands	Yes	clay and organic matter
Poland	Yes	saturated hydraulic conductivity
Spain	No	
Sweden	No	
United Kingdom	Yes	organic matter and pH

Table 3.2: Countries where different screening values are provided according to soil type and soil properties considered (Carlon, 2007)

The derivation process for SVs is far from consolidated within the EU. The European Commission is therefore currently aiming to establish a coherent framework for this process in the form of the Soil Framework Directive, discussed in Section 4.0.

4.0 EU SOIL FRAMEWORK DIRECTIVE

In response to concerns about the degradation of soils in the EU, the European Commission (EC) adopted a communication, 'Towards a Thematic Strategy for Soil Protection' in April 2002, which was supported by member states. Five technical groups with representatives from member states were assembled to help develop the Thematic Strategy. Following a public consultation process and consideration of consultation results, the Commission proposed a Soil Framework Directive and a non-legally binding thematic strategy in September 2006. At the recent EU Environment Council (December 2007), however, Environment Ministers were unable to reach political agreement on the EC proposals for an EU Soil Framework Directive. According to the UK's DEFRA, the UK would not be in a position to support the current EC proposals 'without further changes to bring it into line with the principles of better regulation and subsidiary in order to avoid unnecessary additional administrative burden and disproportionate costs' (DEFRA, 2008). The Netherlands, France, Germany and Austria also did not support the current proposals. The Soil Framework Directive proposals are therefore currently on hold pending an indication from the EC on when future work on outstanding issues will be taken forward.

5.0 DUTCH GUIDELINE VALUES – USE AND MISUSE

In the Netherlands generic risk-based soil screening values have been derived by the National Institute for Public Health and the Environment (RIVM) and the Ministry of Housing, Spatial Planning and the Environment (VROM). These are represented by two screening values; a target value and intervention value, both of which are defined based on *potential risks*. The Dutch VROM and RIVM bodies published revised Dutch guideline values in draft in December 2007 and these new guideline values maintain the standard soil but have changed from the multi-functionality approach to end-use specific values for industry, residential areas and nature/agriculture. For nature and agriculture background values are being used. At the time of writing, the 2007 document is expected to be translated into English sometime in 2008.

5.1 USE OF DUTCH GUIDELINE VALUES

Under the VROM (2000) guidelines that are being used in Ireland, in addition to the target and intervention values, intermediate values have been derived. These represent the numerical average between the target and intervention values. The use of the values is independent of land use. The Dutch values are employed as follows as outlined by Swartjes and Walthaus (2007) of RIVM and VROM respectively:

- Concentration below target value (clean soil) means no restrictions.
- Concentration above target value and below intermediate value (slightly contaminated soil); (Minor) restrictions can be imposed on site management.
- Concentration above intermediate value and below intervention value (*slightly contaminated soil*) implies the triggering of a further investigation.
- An average soil volume concentration of at least 25m^3 (for soil quality assessment) or an average concentration in the pore water of a water saturated soil volume of at least 100m^3 (for groundwater quality assessment) above intervention value (*seriously contaminated soil*) means that in principle remediation will be necessary; the urgency of remediation has to be determined.

Swartjes and Walthaus (2007) describe that the purpose of determining the urgency of remediation is to distinguish between two urgency classes: *urgent* and *non-urgent* cases of serious contamination. A non-urgent classification means the timeline for remediation to commence is not specified, while in the case of *urgent* classification, remediation must be commenced within four years. The urgency of remediation is determined on a basis of *actual* risk, that is, risk which must be assessed on a site specific basis and is higher for the most sensitive receptors.

5.2 MISUSE OF DUTCH GUIDELINE VALUES

The often quoted VROM (2000) has historically been adopted by risk assessors in many member states including Ireland. Oversimplification of the Dutch guidance is widespread however, with 'magic numbers', being used in many cases, without a full (or even basic) understanding of the meaning of guideline values or how they should be applied. For instance, the authors have experience of some local authorities insisting on target values being used as clean-up values for remediation. There is often a lack of understanding in the industry that exceedances in generic standards, does not always mean that remediation is required. There are cases where homes have been demolished based on slight exceedances of the target values. Nathanail and Earl (2001) outline other common examples of misuse of the Dutch values include:

- Use of Dutch values without correction factor for soil composition.
- Use of Dutch intervention value where receptor driving the value is not human health.
- Use of Total PAH or TPH measurements without reference to the nature of the hydrocarbon being considered.
- Ignoring the influence of soil conditions such as pH, soil organic matter or particle size distribution.

In many cases, stakeholders undertaking work on guideline values are reluctant to issue generic guidance values, as a direct result of the numbers being used incorrectly. Consequently, the process of updating guidance values in accordance with best practice in the industry is increasingly protracted.

6.0 UK APPROACH

In recent years the UK has moved away from using Dutch values for the assessment of the risks associated with contaminated land. In terms of the legal framework within the UK, Part 2A of the Environmental Protection Act 1990, introduced in 2000, has led to a new statutory regime for the identification, assessment and remediation of contaminated land in England and Wales. In response to the new regime, DEFRA and the EA have adopted a three tier risk-based procedure for the management of contaminated land:

- Tier 1 - Preliminary Risk Assessment - soil concentrations from a potentially contaminated site are screened against SVs to determine if further assessment is required.
- Tier 2 - Generic Quantitative Risk Assessment - involves a more detailed assessment of the site to assess potential pollutant linkages between identified sources, pathways and receptors. Following completion of a Tier 2 assessment, where remediation may be deemed necessary, a cost-benefit analysis is generally undertaken to assess whether a Tier 3 assessment is required.
- Tier 3 - Detailed Quantitative Risk Assessment- involves a detailed site specific assessment and generally the derivation of SSTLs for the site. If SSTLs are exceeded, then remediation is necessary.

The EA and DEFRA document, *Model Procedures for the Management of Land Contamination (CLR 11),* provides an overall framework for the contaminated land management which is summarised in Figure 6.1.

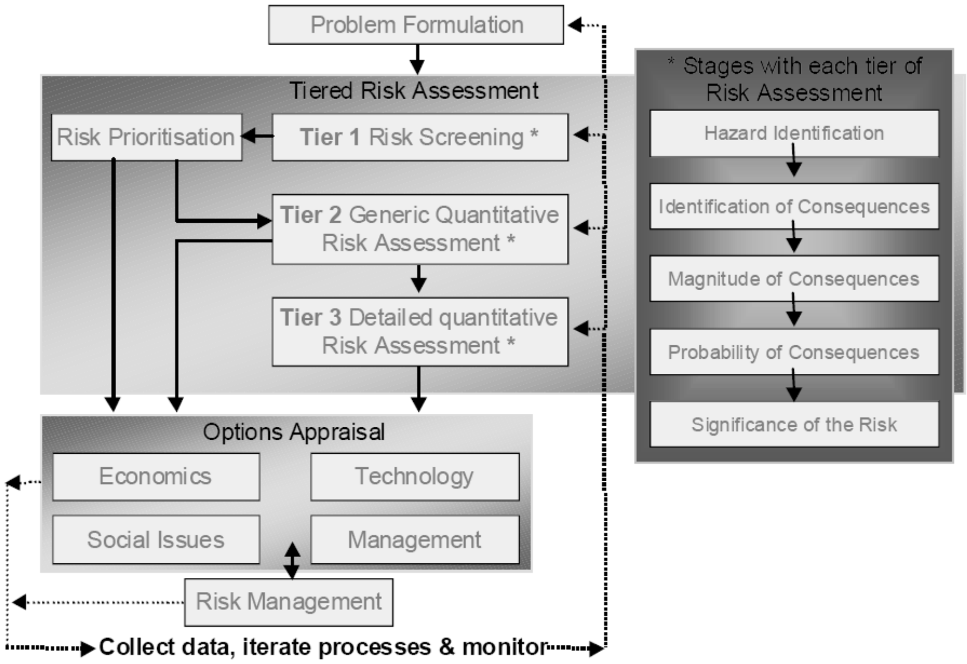


Figure 6.1: UK Environmental Risk Assessment Framework (DETR, 2000)

DEFRA and the EA have produced a clear methodology on risk assessment in the UK called Contaminated Land Exposure Assessment (CLEA). To accompany the methodology, a wealth of guidance has been published which sets out best practice procedures for the derivation of SVs known as soil guidance values (SGVs), which if exceeded, indicate that further assessment or remedial action may be required.

The Contaminated Land Research (CLR) reports (CLR 7 to 10) explain the methodology for deriving SGVs. At the time of writing, 10 SGV reports had been published by the EA and DEFRA. As outlined in the November 2006 DEFRA publication, *Soil Guideline Values: The Way Forward*, 'the SGVs are intended as helpful tools to local authorities to use in determining that land is contaminated on the basis that there is significant possibility of significant harm being caused (SPOSH)' (DEFRA, 2006).

The EA manages an on-going programme of research to develop SGVs. A priority list of 55 substances is currently underway (Carlon, 2007). Following completion of the priority list, a review process of all publications within the CLEA programme will be commenced. The most up to date EA software is 'CLEA UK beta version 1.0'. Given its beta status however, many regulators are reluctant to use it. The EA itself states on its website that the software 'is used cautiously when applying it to user created chemicals and scenarios until the final version is released'. In the meantime, risk assessors in the UK are either using the CLEA UK model or other commercially available modelling tools such as RISC for the derivation of human health screening values.

DEFRA and the EA have invited input from stakeholders in an effort to continually improve the development process. In some cases, practitioners have found SGVs overly stringent. Furthermore the number of published SGVs is currently limited and local authorities are therefore faced with decision making on the acceptability of SVs derived by risk assessors in the industry, based on the DEFRA and the EA guidance.

6.1 LAND QUALITY MANAGEMENT DERIVED VALUES

In July 2006, the Chartered Institute of Environmental Health (CIEH), in conjunction with Land Quality Management (LQM), facilitated a risk assessment workshop at the University of Nottingham. The CIEH is a professional and educational body for environmental health professionals working in both the public and private sector in England, Wales and Northern Ireland.

The workshop brought together a number of practitioners from both the public and private sector that sourced contaminant specific toxicological and physico-chemical input data for use in the derivation of the GAC using the CLEA-UK (beta) risk assessment tool. The delegates during the workshop subjected the outputs to rigorous review, resulting in the development of Generic Assessment Criteria (GAC) compliant with UK policy and guidance for use in generic quantitative human health risk assessment.

The GAC values derived as part of the assessment have been published in the LQM/CIEH publication *Generic Assessment Criteria for Human Health Risk Assessment* in 2007 (LQM and CIEH, 2007). The ten substances (mainly metals) for which DEFRA and the EA have already derived SGVs were not remodelled. The publication outlines GAC values for 31 substances together with the underlying basis for their derivation including the fate and transport and toxicity input values. The aim of the publication is to assist the wider contaminated land community in fulfilling their roles and responsibilities in their day-to-day work in the management of land contamination.

A comparison of Dutch Values, SGVs and CIEH/LQM screening values for a selection of compounds has been included in Table 6.1. The comparison illustrates the variability in approaches adopted within Ireland and the UK.

As is the case for the derivation of the SGVs for organic compounds, the GACs vary depending on the soil organic matter (SOM) content for each of the land-uses. For organic compounds it is widely accepted that there is a relationship between the bioavailability and the organic matter content of the soil and a correction factor is therefore recommended for this parameter (RIVM, 2001).

Soil Screening Values (SVs)		Dutch Guideline Values (VROM, 2000)		UK Soil Guideline Values (SGVs)			CIEH/LQM GAC								
	Units	Target Value	Intervention Value	Residential with plant uptake	Residential without plant uptake	Commercial/ industrial	Residential with plant uptake			Residential without plant uptake			Commercial/ industrial		
Metals															
Nickel	**mg/kg**	35	210	50	75	5000	-			-			-		
Mercury	**mg/kg**	0.3	10	8	15	480	-			-			-		
Copper	**mg/kg**	36	190	-	-	-	111			2080			45700		
Zinc	**mg/kg**	140	720	-	-	-	330			8250			188000		
							1% SOM	**2.5% SOM**	**5% SOM**	**1% SOM**	**2.5% SOM**	**5% SOM**	**1% SOM**	**2.5% SOM**	**5% SOM**
PAHs															
Benzo(a)pyrene	**mg/kg**	-	-	-	-	-	1.12	1.08	1.09	1.3	1.31	1.32	29.7	29.7	29.9
Dibenzo(a,h)anthracene	**mg/kg**	-	-	-	-	-	1.14	1.13	1.1	1.3	1.34	1.32	29.7	29.7	29.9
Fluorene	**mg/kg**	-	-	-	-	-	38.4	91.4	184	2770	2640	2700	59000	59400	59500
Naphthalene	**mg/kg**	-	-	-	-	-	3.47	8.47	17	6.94	17.1	33.7	290	720	1440
Total PAHs (sum 10)	**mg/kg**	1	40	-	-	-	3.47	8.47	17	6.94	17.1	33.7	290	720	1440
Petroleum hydrocarbons															
Benzene	**mg/kg**	0.01	1	-	-	-	0.575	1.33	2.57	0.613	1.41	2.75	26.9	62.1	121
Toluene	**mg/kg**	0.01	130	-	-	-	0.624	1.46	2.85	0.694	1.63	3.18	30.4	71.1	139
Ethylbenzene	**mg/kg**	0.03	50	-	-	-	0.624	1.46	2.85	0.694	1.63	3.18	30.4	71.1	139
Xylene	**mg/kg**	0.1	25	-	-	-	0.624	1.46	2.85	0.694	1.63	3.18	30.4	71.1	139
Mineral Oil	**mg/kg**	50	5000	-	-	-	0.624	1.46	2.85	0.694	1.63	3.18	30.4	71.1	139

Table 6.1: Comparison of screening values used in the UK and Ireland - Dutch, SGVs and CIEH/LQM

7.0 RECOMMENDED APPROACH IN IRELAND

It is clear from the above discussions that a consolidated approach in the EU regarding derivation and selection of generic values is in its infancy. There are many commonalities however in risk assessment approaches internationally. Risk assessment has been adopted by the EPA for groundwater assessment and the Irish EPA also supports the process for the management of contaminated land.

The use of soil screening standards in Ireland is currently inconsistent and many local authorities require guidance and training in the assessment of contaminated land. In the absence of national SVs there is a need for the EPA to give guidance on the appropriate SVs to use. Given that the UK has already published a plethora of guidance and has commenced a large scale process of studying a wide range of contaminants, the authors propose that Ireland formally acknowledge the UK approach and utilise this vast resource for the assessment of contaminated land. The authors further recommend that the LQM/CIEH SVs derived in 2006 are adopted in Ireland. The EPA should lead this process of change and issue guidance to local authorities and risk assessment practitioners in an effort to improve technical understanding of SVs and general understanding of the risk assessment process.

The consultation process between industry, risk assessors and other stakeholders, with respect to improvements in DEFRA and EA publications, is on-going and should lead to further advancements to insure that the final framework is fit for purpose. A contaminated land forum and working groups should be established by the EPA to keep apace of developments in the UK and to agree a means of harmonising the approach in Ireland. This would provide a mechanism by which information could be shared with local authority representatives to ensure that decision makers remain up to date with current best practice.

8.0 CONCLUSIONS

If appropriately used, SVs can reduce the cost of risk assessment and simplify decision making for local authorities and risk assessors. When used out of context however, guidance values can be incorrectly prescribed as a panacea by which some local authorities and risk assessors incorrectly adjudicate on requirements for remediation. Clear guidance is required to allow guideline values to be interpreted by non-specialists, however there is an onus on regulators to understand the appropriate use of adopted or derived generic screening values, so that consistent guidance and coherent decision making can take place.

9.0 REFERENCE LIST

- Carlon C (Ed), 2007, *Derivation Methods of Soil Screening Values in Europe. A Review and Evaluation of National Procedures Towards Harmonisation*, European Commission, Joint Research Centre, Ispra (EUR 22805 EN), 306 pp.

- DEFRA, 2008, *Environmental Protection: EU Thematic Strategy for Soil Protection, including proposals for a Soil Framework Directive*, Department for Environment, Food and Rural Affairs http://www.defra.gov.uk/ENVIRONMENT/land/soil/europe/index.htm (Accessed 28th March 2008)

- DEFRA, 2006, *Assessing Risks from Contaminated Land: A Proportionate Approach (Soil Guideline Values - The Way Forward),* Department for Environment, Food and Rural Affairs

- DETR, 2000, *Contaminated Land: Implementation of Part 2A of the Environmental Protection Act 1990, Circular 02/2000*, Department for the Environment, Transport and the Regions

- EA, 2004, *CLR 11 - Model Procedures for the Management of Land Contamination*, Environment Agency

- EA and DEFRA, 2002a, *CLR 10 - The Contaminated Land Exposure Assessment (CLEA) Model: Technical Basis and Algorithms*, Environment Agency and Department for Environment, Food and Rural Affairs

- EA and DEFRA, 2002b, *CLR 7 - Assessments of Risk to Human Health from Land Contamination,* Environment Agency and Department for Environment, Food and Rural Affairs

- EA and DEFRA, 2002c, *CLR 8 - Potential Contaminants for the Assessment of Land,* Environment Agency and Department for Environment, Food and Rural Affairs

- EA and DEFRA, 2002d, *CLR 9 - Contaminants in Soil: Collation of Toxicological Data and Intake Values for Humans,* Environment Agency and Department for Environment, Food and Rural Affairs

- Ferguson C, 1999, *Assessing Risks from Contaminated Sites: Policy and Practice in 16 European Countries,* Land Contamination & Reclamation, Volume 7, 33-54

- LQM and CIEH, 2007, *Generic Assessment Criteria for Human Health Risk Assessment,* Land Quality Management and Chartered Institute of Environmental Health, Land Quality Press

- Nathanail P and Earl N, 2001, *Human Health Risk Assessment: Guideline Values and Magic Numbers,* Assessment and Reclamation of Contaminated Land, Thomas Telford

- RIVM, 2001, *Technical Evaluation of the Intervention Values for Soil/Sediment and Groundwater (RIVM Report* 711701 023)*,* National Institute of Public Health and the Environment, The Netherlands

- Swartjes F and Walthaus H, 2007, *Risk-Based Assessment of Soil and Groundwater Quality in the Netherland (Dutch Soil Protection Act)*, European Commission, Derivation Methods of Soil Screening Values in Europe. A Review and Evaluation of National Procedures Towards Harmonisation

- VROM, 2000, *Circular on Target Values and Intervention Values for Soil Remediation*, Ministry of Housing, Spatial Planning and Environment (VROM) DBO/1999226863

REDUCTIVE DECHLORINATION OF CHLORINATED ALIPHATIC HYDROCARBONS IN SHALLOW GROUNDWATER BALLINASLOE, COUNTY GALWAY, IRELAND

Sean Moran (1), A. Tevfik Arguden, and Randall K. Sillan

(1) O'Callaghan Moran & Associates, Granary House, Rutland Street, Cork,

sean@ocallaghanmoran.com

ABSTRACT

This paper presents the findings of a remediation project at a former Integrated Pollution Control Licensed (IPC) manufacturing facility in Ballinasloe County Galway. Prior to this project the remediation methods described in this paper and applied at the site had not previously been applied in Ireland.

Elevated concentrations of several volatile organic compounds (VOCs), including trichloroethylene (TCE), trichloroethane (TCA) and their breakdown compounds, were first discovered in groundwater samples collected from the site in November 1999. From January 2000 to December 2003, work was completed to:

- *Delineate the extent of groundwater contamination,*
- *Characterize groundwater geochemistry,*
- *Assess soil-gas concentrations under the building slab to evaluate potential volatilization to the indoor air,*
- *Establish risk-based site-specific cleanup objectives, and*
- *Determine the most effective and economically feasible remediation option by performing field-scale pilot tests.*

Based on the favorable geochemical data gathered from the site from 1999 to 2002, it was concluded that enhancing the naturally occurring biological reductive dechlorination processes with hydrogen donors would accelerate degradation of volatile organic compounds into non-toxic end-products. Furthermore, this in-situ remedial alternative would minimise worker exposure, be the least expensive and eliminate the need for and cost of disposal of waste generated.

A total of 83,000 liters of a 1% ethanol-water solution was injected into 26 injection wells twice within a 12-month period. The second injection occurred when groundwater analytical results indicated that the ethanol was depleted. Following an unexpected hiatus (approximately 18 months) of field activities due to the sale of the facility and transfer of the IPC license to a new owner, the site activities resumed and a third and final injection was performed in November 2005.

Analytical data gathered from the site documented the decline of the contaminant concentrations from over 50,000 microgram per liter (µg/l) to well below the site specific cleanup goals established for the site. After three injection episodes, only vinyl chloride slightly exceeded its cleanup goal of 7 µg/l in two cross-gradient wells (C-11 and C-19). The remaining chlorinated ethenes and ethanes were either reduced to below their respective cleanup goals, or were transformed into non-toxic end products. The residual vinyl chloride that remained in the downgradient margins of the plume where more aerobic conditions prevailed in 2005 were eliminated using hydrogen peroxide, a chemical oxidant.

Site Setting

The Site is located in a mixed residential and industrial area approximately one mile to the west of Ballinasloe, County Galway. The Galway to Dublin railway line runs parallel to the Deer Park River along the northern Site boundary. Marshy undeveloped lands lie to the east and west of the Site. Residential properties are located to the south-east and south-west (hydraulically upgradient) of the Site.

The Site was first developed in 1972 and was utilised as a manufacturing facility until 2001. Operations at the site included metal plating and degreasing operations that used TCE and TCA. The former chemical storage areas, the plating room and former degreaser units were located along the northern portion of the manufacturing building. There are no public or private groundwater abstraction wells on the site. The facility is currently unoccupied.

Geology and Hydrogeology

Regionally peat deposits dominate the subsoil. Underlying the peat deposits are glacial tills derived from limestone bedrock which contain sand and gravel lenses. To the west of the site, drumlin and esker deposits composed of glacial tills and poorly draining silty clays dominate the topography.

Prior to site development activities in 1971, due the soft, marshy nature of the ground in the northern portion of the site, approximately 1.5 meters of hardcore was placed before the construction of the site buildings, parking lots and roads. As a result the site is approximately 1.5 meters higher then the adjoining vacant parcel to the east. The fill material beneath the northern portion of the site consists of medium to coarse sand with occasional gravel within a silty-clayey matrix. The natural overburden beneath the fill typically comprises silty sand with gravel, becoming sandy gravel toward the west of the site. Coarse limestone cobbles and boulders are present immediately above the bedrock which is approximately 3m below ground level. The upper 0.5- to 1.0-meter portion of the bedrock appears to be weathered and/or reworked. The groundwater beneath the site moves from the south southwest to the north northeast toward the Deer Park River. The hydraulic gradient across the site is relatively flat (approximately 0.0016).

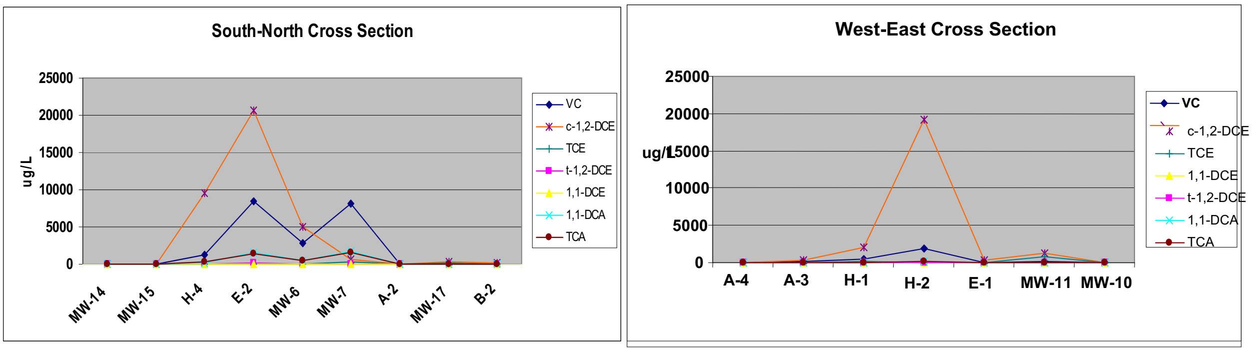
The subsoils tends to be dominated by glacial till comprising boulder-clays and silts with low hydraulic conductivities and are not defined as an aquifer by the Geological Survey of Ireland (GSI). The bedrock beneath the site is considered by the GSI to be a poor aquifer with yields typically between 15 to 20 m^3/day. There is insufficient yield for public supplies, though yields may support the needs of individual dwellings. Naturally occurring impurities in the groundwater such as manganese and iron make it unsuitable for potable use in some areas.

Initial Groundwater Chemistry

Several chlorinated ethenes and ethanes were identified in groundwater samples collected from the former courtyard area ("Source area") situated to the north of the manufacturing building. Geochemical data gathered from late 1999 to early 2002 indicated the shallow groundwater was anaerobic likely due to wide-spread marshy conditions with high organic content in the peaty subsoils and poorly productive aquifer.

During operation of the facility TCE and TCA were used and stored at the site over two decades. Although there were no known significant releases at the site, minor historical releases of TCE and TCA appeared to have impacted the shallow groundwater beneath the site. By 2002, the concentrations of these parent compounds in shallow groundwater were significantly reduced by natural attenuation processes. The highest concentrations of TCE and TCA in the source area in 2002 were 279 microgram per liter (µg/l), and 1,485 µg/l, respectively. The total VOC concentration in the same area was over 55,300 µg/l (Figure 1). During the same period, breakdown or daughter products of TCA and TCE (cis-1,2-dichloroethylene (DCE) ,11-dichloroethane, 1,2 dichlorethene and vinyl chloride) accounted for approximately 70% of the total VOCs in the source area. A similar trend, with significantly lower VOC concentrations, was detected in groundwater samples collected from the mid-plume and the distal end of the plume. These observations, coupled with the trace concentrations of chloride, ethane and ethene in groundwater samples from the source area suggested that reductive dechlorination was occurring and had produced significant quantities of breakdown products in the groundwater plume.

Figure 1 VOC levels in Longitudinal and Transverse Section through the Plume



Feasibility Study

Several ex-situ and in-situ remedial alternatives were evaluated based on technical feasibility, cost effectiveness, compliance with risk management, environmental legislation and regulations, and current and future use of the property. The ex-situ alternatives (i.e. pump-and treat and dual-phase extraction) were subsequently rejected due either to requirements of significant cost (both initial construction and continuing operation and maintenance expenses) and/or the potential worker exposure to hazardous substances as contaminated water and soil vapor are extracted, treated and disposed under applicable permit conditions.

Based on the favorable geochemical data gathered from the site from 1999 to 2002, it was concluded that enhancing the naturally occurring biological reductive dechlorination processes with hydrogen donors would accelerate biotransformation of volatile organic compounds into non-toxic end-products. Furthermore, this in-situ remedial alternative would minimise worker exposure, be the least expensive and eliminate the need for and costs of disposal of waste generated.

Anaerobic Bioremediation Processes

Biodegradation of organic compounds in groundwater occurs via three mechanisms:

- use of the organic compound as the primary growth substrate;
- use of the organic compound as an electron acceptor;
- co-metabolism

The first two mechanisms involve the microbial transfer of electrons from electron donors (primary growth substrate) to electron acceptors. This process can occur under aerobic or anaerobic conditions (USEPA, 1998). Electron donors include natural organic material, fuel hydrocarbons, and the less oxidized chlorinated ethenes and ethanes. Electron acceptors are elements or compounds that occur in relatively oxidized states. The most common naturally occurring electron acceptors in groundwater include oxygen, nitrate, manganese (IV), iron (III), sulfate, and carbon dioxide. In addition, the oxidized chlorinated solvents such as TCE, TCA, DCE and dichloroethane (DCA) can act as electron acceptors under favorable conditions. In the anaerobic groundwater conditions beneath the site nitrate, iron, sulfate and carbon dioxide may have acted as electron acceptors. By 2002, as an electron acceptor, the nitrate was reduced to NO_2^-, NH_4^+, or N_2 via denitrification, and nitrate concentrations in groundwater decreased.

Similarly, iron (III) was reduced to iron (II) via iron (III) reduction, and iron (II) concentrations in groundwater increased. The relative abundance of CH_4 in shallow groundwater was attributed to both natural organic decay and the use of CO_2 as an electron acceptor by methanogenic bacteria during methanogenesis. Geochemical evidence also suggested that chlorinated ethenes and ethanes were being used as electron acceptors, and were reduced to less chlorinated daughter products. Historical analytical trends suggested that the TCE and TCA concentrations would continue to decrease and the concentrations of DCE and vinyl chloride (VC) would increase and then gradually decrease as the daughter product is used as an electron acceptor or is oxidized. As each subsequent electron acceptor was utilized, the oxidation state of the groundwater became more reducing.

This process is called reductive dechlorination where the chlorinated hydrocarbon is used as an electron acceptor, and a chlorine atom is replaced with a hydrogen atom. An appropriate source of carbon (electron donor) for microbial growth must be available for reductive dechlorination to occur. The reductive dechlorination pathways for chlorinated ethanes and ethenes seen at this site originate with the source contaminants TCE and TCA which break down to either 1,1 DCA from TCA or to Cis-1,2-DCE, Trans-1,2-DCE or 1,1-DCE from TCE. These further breakdown to either Chloroethane for TCA or vinyl chloride for TCE. Ultimately they produce ethane or ethene respectively.

Pilot Test

A pilot study was conducted to evaluate the most effective and economically feasible treatment option for in-situ anaerobic bioremediation of chlorinated compounds in groundwater. Two hydrogen donors, HRC, a specially formulated product designed to time-release lactic acid upon contact with water, and food grade ethanol (1% by volume) were used to evaluate the effectiveness of the reductive dechlorination processes in the groundwater plume beneath the site. In this process, the hydrogen donors (i.e. lactic acid or ethanol) are metabolized by indigenous microbes in the subsurface to produce hydrogen which is then used as the electron donor during reductive dechlorination of TCE and TCA to the final end products ethene, and ethane, respectively.

Two test plots each containing at least two injections and three downgradient monitoring wells were constructed in the source area. The injection and monitoring wells were sampled to establish the baseline groundwater geochemistry. The pilot injection wells in each plot were injected with either HRC or ethanol. The post-injection groundwater geochemistry was monitored monthly from January 2002 to April 2002. Analytical results gathered from the both pilot test plots (Tables 1) revealed the following:

1. Increased mass fraction of degradation by-products and step-wise degradation,
2. Decreased molar ratios between parent and daughter compounds,
3. Increased concentrations of end-products such as chloride, ethene and ethene,
4. Decrease in sulfate and increasing methane concentrations indicating sulfate reduction and methanogenesis.

Table 1 Representative Wells from HRC and Ethanol Test Plots

		H1		MW7				H1		MW7	
VOCs	Units	Jan-02	Apr-02	Jan-02	Apr-02	Parameter	units	Feb-02	Apr-02	Feb-02	Apr-02
VC	ug/L	4296	8368	1008	2862	**CO2**	mg/l	27	26	22	50
t-DCE	ug/L	56	86	8	17	**Ethane**	ng/l	28000	40000	180	49000
DCE	ug/L	31	49	4	12	**Ethene**	ng/l	88000	120000	580	320000
DCA	ug/L	743	1547	167	484	**Hydrogen**	nM	1.5	0.52	1.2	6.2
c-DCE	ug/L	14720	20616	975	4990	**Methane**	µg/l	260	530	8.6	1400
TCA	ug/L	1522	1470	192	434	**Chloride**	mg/l	64	46	27	118
TCE	ug/L	49	0	6	1	**Fe Dissolved**	mg/l	20	0.32	19.5	0.36
		HRC Well		**Ethanol Well**		**sulfate**	mg/l	0.32	18	6.55	21

Although HRC and ethanol both performed well, in the pilot test ethanol was selected for the full-scale application due to its lower cost and ease of delivery into the contaminant plume.

Enhanced Bioremediation with Ethanol
Dechlorination is dependent upon the supply of hydrogen (H_2), which acts as the electron donor (Fennel et al., 1997; EPA, 1998). The hydrogen is produced as a result of the microbial degradation of a primary substrate (e.g., lactate, acetate, butyrate, ethanol, BTEX, or other such compounds).

Microorganisms that facilitate dechlorination compete with sulfate-reducers and methanogens for the H_2 produced in such a system. When degradation of the original substrate/electron donor occurs rapidly, the process yields high concentrations of H_2 and the sulfate-reducers and methanogens appear to be favored over the dechlorinators (Fennel et al., 1997). Conversely, when substrate degradation produces a steady supply of H_2 at low concentrations, the dechlorinators are favored. Thus, complete dechlorination is favored when a steady, low-concentration supply of H_2 is produced through microbial degradation of substrates. Therefore, the type of substrate/electron donor can also play a role in how thoroughly a natural system is able to dechlorinate solvents. Indigenous organisms utilize ethanol for energy, which results in the formation of an intermediate hydrogen pool. Via halorespiration, the microorganisms (dechlorinators) obtain energy for growth using the hydrogen as the electron donor and the chlorinated hydrocarbon as a terminal electron acceptor.

Depending upon the extent of ethanol oxidation, one mole of ethanol can produce two or six moles of hydrogen. Under anaerobic conditions, ethanol can be biodegraded to acetic acid by incomplete oxidation (Equation 1) or to carbon dioxide by complete oxidation (Equation 2):

(1) $CH_3CH_2OH + H_2O \rightarrow CH_3COOH + 2H_2$(1) (2) $CH_3CH_2OH + 3H_2O \rightarrow 2CO_2 + 6H_2$

The use of ethanol as a substrate to stimulate dechlorination has been successfully demonstrated at the lab scale (Gibson and Sewell, 1992; Pavlostathis and Zhuang, 1993; Fennell et al., 1997). Injection of ethanol at the field scale has been conducted at several sites during in situ co-solvent flushing activities to solubilise and extract a non-aqueous phase liquid (Rao et al., 1997; Jawitz et al., 2000,) and during enhanced bioremediation (LFR and OCM, 2002; Mravik et al., 2003),.

Since ethanol is miscible with water, high concentrations of ethanol can easily be introduced to the subsurface, however, high ethanol concentrations (typically greater than 2.5%) inhibits microbial growth and survival. Based on observations from field studies, the optimal ethanol concentration for enhancing reductive dechlorination at the site was approximately one percent by volume (10,000 parts per million). At one percent ethanol, the injected solution has the physical properties of water, which could easily be injected into the subsurface through well screens or direct push equipment. Thus, the quantity of ethanol solution delivered at an injection point is determined by the size of the treatment zone.

Design and Implementation

Pump and slug test data gathered from various wells at the site indicated that the likely groundwater migration rate is approximately 7 meters per year. Data gathered during the pilot study, however, showed that ethanol, HRC, and their byproducts migrated approximately 6 meters distance during the 90 days monitoring period. The reason for the apparent increase in the migration rate during the test was due to the significant displacement of groundwater from the injection wells during the injection of 3000 litres of ethanol groundwater solution. The injection volume created a locally greater hydraulic head thus increased hydraulic gradient and migration rates. Therefore using 6-meter minimum "radius of influence" around each injection point was believed to be more than adequate to evenly distribute the ethanol solution in the subsurface and treat the dissolved plume. In order to deliver the substrates to the overburden and transitional zone, the injection wells were screened from approximately one meter below ground level (mbgl) to the bottom of transition zone. During injection well installation, each borehole was extended into the bedrock (cored approximately 0.5 meter) to determine the bottom elevation of the transition zone.

A total of 26 injection wells were installed to cover the source area and the edges of the contaminant plume. The injection well network was designed to extend well beyond the source area, both horizontally and along the centerline of the plume. The injection wells were constructed with 50mm diameter PVC well screen and appropriate length of solid PVC casing. The annulus was back-filled with a gravel filter pack. The wells were developed by surging the well screen and pumping water until the extracted water is clear and the indicator parameters (pH, conductivity, temperature and dissolved oxygen) stabilized.

Baseline Sampling

Groundwater samples were collected from all 26 injection and 6 existing monitoring wells prior to ethanol injection. The samples were analyzed for volatile organic compounds, volatile fatty acids, inorganic compounds, and dissolved gases. Analytical results of the baseline sampling are presented in Table 1 above.

Ethanol Injection

The ethanol solution was injected from the upgradient (source area) toward the downgradient end of the plume using at least four injection wells simultaneously. The injection rate was determined by the hydraulic conductivity of the formation and groundwater mounding during injection.

Approximately 3,200 liters of ethanol-water solution was delivered to the subsurface at each injection well as a 1% by volume solution of ethanol. A total of 83,000 litres of ethanol-water solution were injected in the 26 injection wells twice within a 12-month period. The second injection occurred when groundwater analytical results indicated that the ethanol was depleted. Following an unexpected hiatus (approximately 18 months) of field activities due to the sale of the facility and transfer of the IPC license to a new owner, a third and final injection was completed in November 2005.

Results

Analytical results of the groundwater samples with the historical trends since 2002 are presented in graphical form in Figures 2 - 4. The analytical results of the groundwater samples were compared against the Site Specific Clean up Goals (SSCG) which were developed using human health based risk assessment and agreed with the EPA. The SSCGs are listed in Table 2 below.

Table 2 Site Specific Clean up Goals

Units	Parameter	Symbol	SSCG
µg/L	Vinyl Chloride	VC	7
µg/L	t-1,2-dichloroethene	t-DCE	600,000
µg/L	1,1-dichloroethene	DCE	2,600
µg/L	1,1-dichloroethane	DCA	720
µg/L	c-1,2-dichloroethene	c-DCE	800,000
µg/L	1,1,1-trichloroethane	TCA	130,000
µg/L	Trichloroethene	TCE	1,500

SSCG: *The Site Specific Cleanup Goal (SSCG) for target chemicals were developed by LFR and OCM in October 2001 using the most stringent U.S. Environmental Protection Agency (U.S. EPA) exposure factors (USEPA 1996). Toxicity factors were obtained from the U.S. EPA Integrated Risk Information System (IRIS). Target individual excess lifetime cancer risks were set at 1 in 100,000 (1 x 10^{-5}), and target non-cancer risks were set at a hazard quotient equal to 1. Conservative assumptions which generally tend to overestimate chemical extent, chemical migration and potential chemical exposure were made in developing the SSCGs. The Risk Based Corrective Action Assessment illustrated that VOCs detected in the groundwater at SSCG concentration are likely to degrade to insignificant levels before reaching downgradient, off-site receptors.*

The groundwater analytical data gathered from the site between 2002 and 2006 documented the contaminant concentrations decline below the site specific SSCGs. After three injection episodes, only VC slightly exceeded its SSCG in two cross-gradient wells (C-11 and C-19). The remaining chlorinated ethenes and ethanes were reduced below their respective SSCG, or were transformed into non-toxic end products.

Figure 2 Representative Source Area

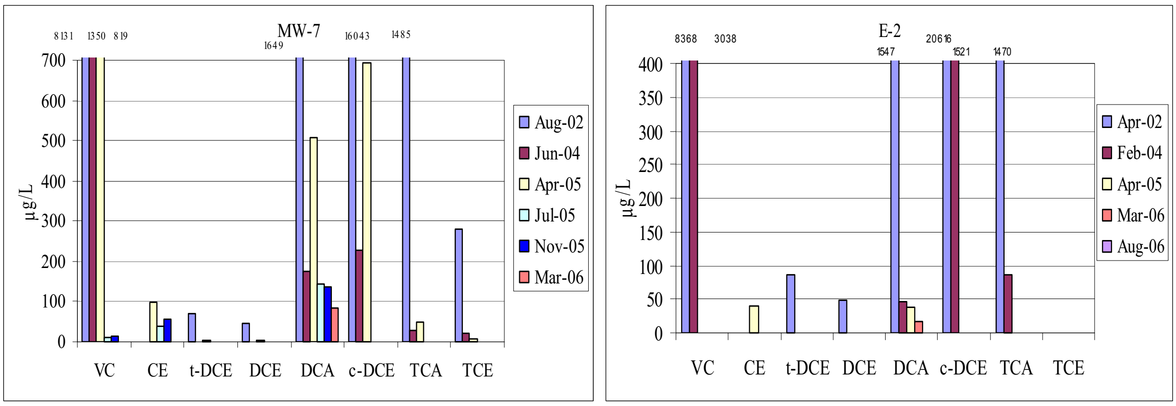
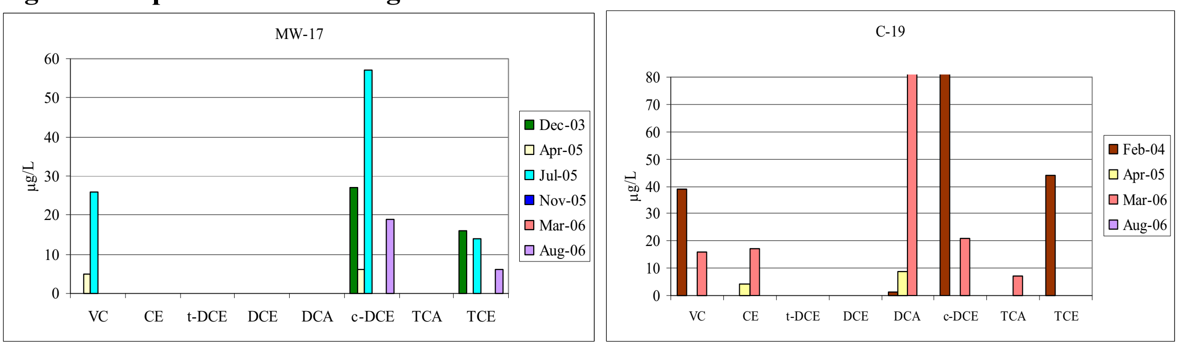


Figure 3 Representative Downgradient Wells



Chemical Oxidation

VC persisted above the SSCG of 7ugl in 6 of the down hydraulic gradient monitoring wells. Dissolved oxygen levels in the groundwater in this portion of the plume indicated aerobic conditions which may explain why complete reductive dechlorination did not occur in these areas. Chemical oxidation using hydrogen peroxide (H_2O_2) was therefore implemented for remediation in this area. The oxidant rapidly reacts with the naturally occurring oxidizable material and the target contaminants at contact producing innocuous substances such as carbon dioxide (CO_2), water (H_2O), and inorganic chloride. The rate and degree of degradation of compounds such as TCE, DCE and VC increase with increasing concentrations of oxidant above the natural oxidant demand. H_2O_2 is generally used with a catalyst to produce hydroxyl radicals (OH•). Ferrous salt (Fe^{+2}) was added to increase the oxidative strength of peroxide by producing (OH•). Ferrous salt was added to each well prior to the injection programme. 100 litres of 15% H_2O_2 solution was injected into the six wells in October 2005 and in February 2006. Monitoring in February 2006 and again in August 2006 confirmed that VC was no longer present in the groundwater.

Conclusions

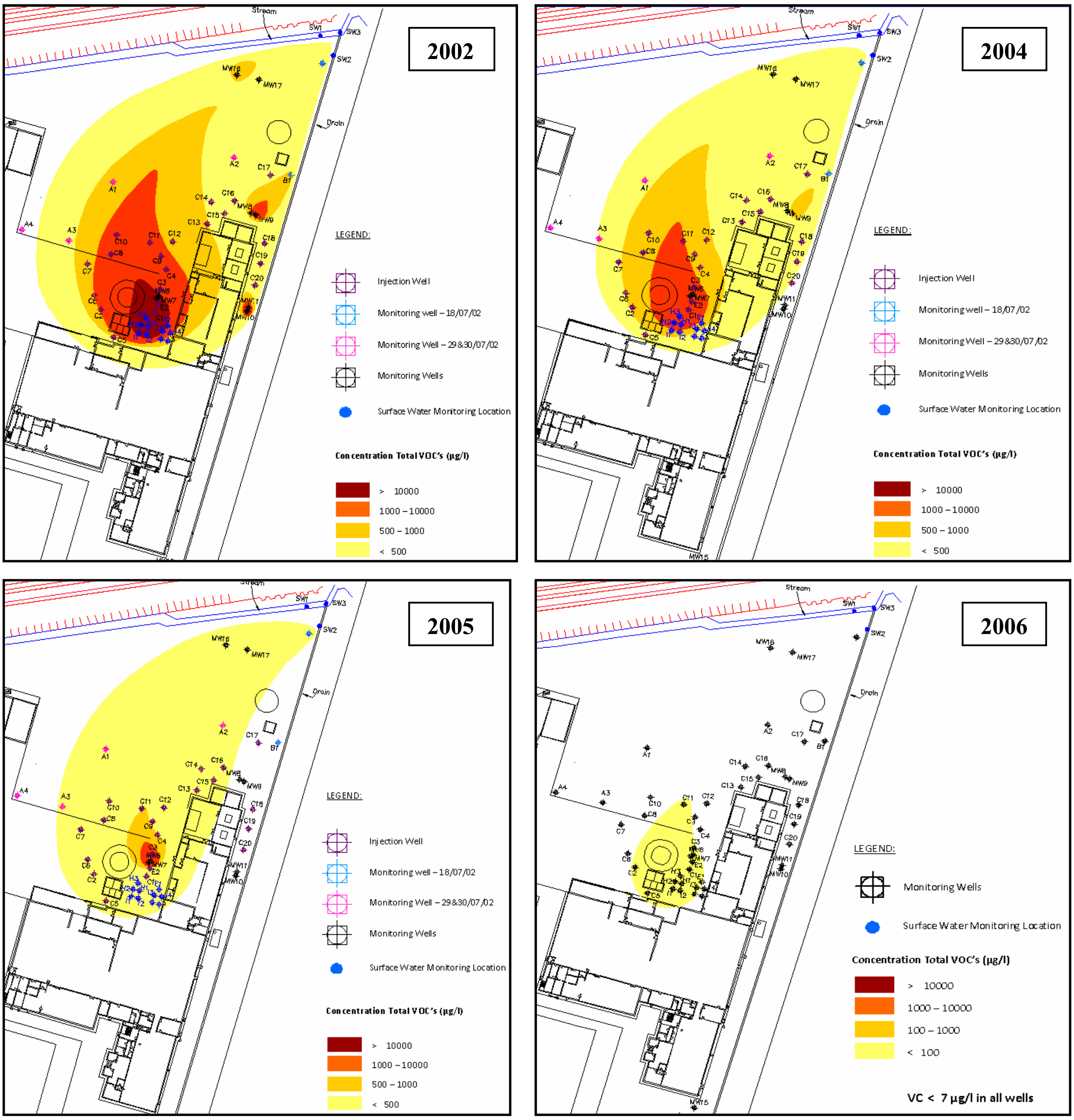
- Analytical data gathered from the site documented the decline of the contaminant concentrations from over 50,000 microgram per liter (µg/l) to well below the site specific cleanup goals established for the site.

- After three injection episodes, only vinyl chloride slightly exceeded its cleanup goal of 7 µg/l in two cross-gradient wells (C-11 and C-19).

- The remaining chlorinated ethenes and ethanes were either reduced to below their respective cleanup goals, or were transformed into non-toxic end products.

- The residual vinyl chloride that remained in the downgradient margins of the plume where more aerobic conditions prevailed in 2005 were eliminated using hydrogen peroxide, a chemical oxidant.

Figure 4 Plume Reduction 2002 - 2006



Acknowledgements

The authors would like to acknowledge the co-operation and guidance provided by the Environmental Protection Agency (EPA) in bringing this project to a successful conclusion.

The installation of the injection wells and one round of ethanol injection was supervised by TMS Environment Limited in accordance with the remedial action plan designed by LFR and OCM and approved by the EPA.

Alexandra Bel, OCM for assistance with data tables and figures included in the paper.

References

Fennell, D.E., Gossett, J.M. and Zinder, S.H., 1997. Comparison of butyric acid, ethanol, lactic acid, and proprionic acid as hydrogen donors for the reductive dechlorination of tetrachloroethene. Environmental Science and Technology, 31(3): 918-926.

Gavaskar, B.C. Alleman and V.S. Magar (Editors), Bioremediation and Phytoremediation of Chlorinated and Recalcitrant Compounds. Battelle Press, Columbus, OH, pp. 39-46.

Gavaskar, B.C. Alleman and V.S. Magar (Editors), Bioremediation and Phytoremediation of Chlorinated and Recalcitrant Compounds. Battelle Press, Columbus, OH, pp. 15-22.Sewell, G.W. 2001. personal communication. June 25.

Gibson, S.A. and Sewell, G.W., 1992. Stimulation of reductive dechlorination of tetrachloroethene in anaerobic aquifer microcosms by addition of short-chain organic acids or alcohols. Applied and Environmental Microbiology, 58(4): 1392-1393.

Jawitz, J.W., Sillan, R.K., Annable, M.D., Rao, P.S.C. and Warner, K., 2000. In-situ alcohol flushing of a DNAPL source zone at a dry cleaner site. Environmental Science and Technology, 34(17): 3722-3729.

Koenigsberg, S.S., Farone, W.A. and Sandefur, C.A., 2000. Time-release electron donor technology for accelerated biological reductive dechlorination. In: G.B. Wickramanayake, A.R.

LFR Levine·Fricke Inc. 2001. Remedial Action Plan. Former Sages Dry Cleaner, 5800 Fort Caroline Road, Jacksonville, Florida. FDEP Facility Identification Number: 169600614. September 6.

LFR Levine·Fricke Inc., O'Callaghan Moran & Associates 2002. Remedial Action Test Implementation Report: May 2002.

Mravik, S.C., R.K. Sillan, A. L. Wood, and G.W. Sewell, 2003. Field evaluation of the solvent extraction residual biotreatment technology, *Environmental Science and Technology*, 37 (21), pp. 5040-5049.

Pavlostathis, S.G. and Zhuang, P., 1993. Reductive dechlorination of chloroalkenes in microcosms developed with a field contaminated soil. Chemosphere, 27(4): 585-595.

Rao, P.S.C., Annable, M.D., Sillan, R.K., Dai, D., Hatfield, K.H., Graham, W.D., Wood, A.L. and Enfield, C.G., 1997. Field-scale evaluation of in-situ cosolvent flushing for enhanced aquifer remediation. Water Resources Research, 33(12): 2673-2686.

Regenesis. Distribution of HRC in the aquifer. HRC Technical Bulletin H-2.8.2., 8 pages.

Regenesis. Bedrock application of HRC. HRC Technical Bulletin H-2.8.3., 3 pages.

Schuhmacher, T.T., Bow, W.A. and Chitwood, J.P., 2000. A field demonstration showing enhanced reductive dechlorination using polymer injection. In: G.B. Wickramanayake, A.R.

U.S. Environmental Protection Agency (EPA), 1998. Technical protocol for evaluating natural attenuation of chlorinated solvents in ground water. EPA/600/R-98/128, Office of Research and Development, Washington, D.C.

Session IV

CONSTRUCTION DEWATERING FOR BASEMENTS IN GRAVELS

Dr Toby Roberts and Qingchao Ren, WJ Groundwater Limited

Bournehall House, Bushey, Hertfordshire WD23 3HP, UK

tobyroberts@wjgl.com

ABSTRACT

There is continuing demand for underground space in urban areas for basements, services and transport infrastructure. Where construction involves excavation in water bearing granular soils, control of groundwater levels will be required during excavation and construction. A side support earth retaining structure may be necessary, particularly in urban areas, and if appropriately constructed this can also act as a groundwater cut-off. Groundwater levels within the excavation may then be controlled by the retaining structure cut-off or by dewatering or often by a combination of these methods. Where the retaining structure only provides a partial cut-off then internal dewatering may result in external drawdown below neighbouring structures bringing a possible settlement risk if superficial soft soils are present which may be under-drained. This paper examines the interaction between physical cut-off depth, dewatering flows, hydrogeology and external drawdowns in high permeability soils using numerical modelling. The modelling is based on conditions in Cork City, Republic of Ireland, and the results from the modelling are compared with data from relevant basement excavations.

1.0 INTRODUCTION

Basement excavation, particularly in urban areas, in water bearing granular soils will typically require both an earth retaining structure for side support and control of groundwater levels during construction. The earth retaining structure may comprise sheetpiles, secant piles or diaphragm walls which provide a vertical hydraulic barrier in addition to side support. Other retaining structures include contiguous piles or soldier piles and lagging which do not provide a complete groundwater barrier. In circumstances where an aquiclude is present below the granular soils an earth retaining structure which provides a hydraulic barrier may toe into this stratum to provide a virtually sealed box. In this case the trapped groundwater will need to be removed from the box in the course of excavation, and both residual seepage and external drawdowns should be minimal. This paper examines the alternative scenario where the granular soils are sufficiently deep that it is either not feasible or not cost effective for the retaining structure to reach down to a suitable aquiclude. Under these circumstances a dewatering system will be required to control inflows below the retaining structure and consideration will have to be given to any possible consequences of external drawdowns below neighbouring structures.

Techniques are available for installing artificial horizontal barriers using jet grouting or permeation grouting techniques. There are considerable cost and programme penalties in using these techniques and they are usually only considered as a last resort in circumstances where minimisation of inflow is essential. This situation might arise where;

- Inflows would be excessive;
- There is no access to a suitable discharge point;
- Costs for discharge are excessive;
- External drawdown would be unacceptable due to adverse impact on water resources;

- External drawdown would be unacceptable due to a settlement issue in respect of neighbouring structures; or,
- The groundwater is contaminated and treatment or discharge costs would be excessive.

The use of horizontal barrier techniques is not considered further in this paper.

The paper focuses on relatively high permeability soils (> 10^{-3} m/s) where the need to limit inflow rates and external drawdowns is likely to prove most critical. The paper presents data from a simple numerical modelling study which examines the relationship between cut-off depth, inflow and external drawdown in both homogeneous soils and anisotropic soils (where the vertical permeability is lower than the horizontal permeability). The results from the modelling are compared with data from several basement excavation projects undertaken in Cork City, Republic of Ireland.

2.0 THE HYDROGEOLOGY OF CORK

Cork City is located in the River Lee flood plain which overlies a buried valley. The valley was formed in the Carboniferous and Devonian rock 20,000 to 16,000 years B.P. during the Pleistocene Glaciation, see Figure 1.

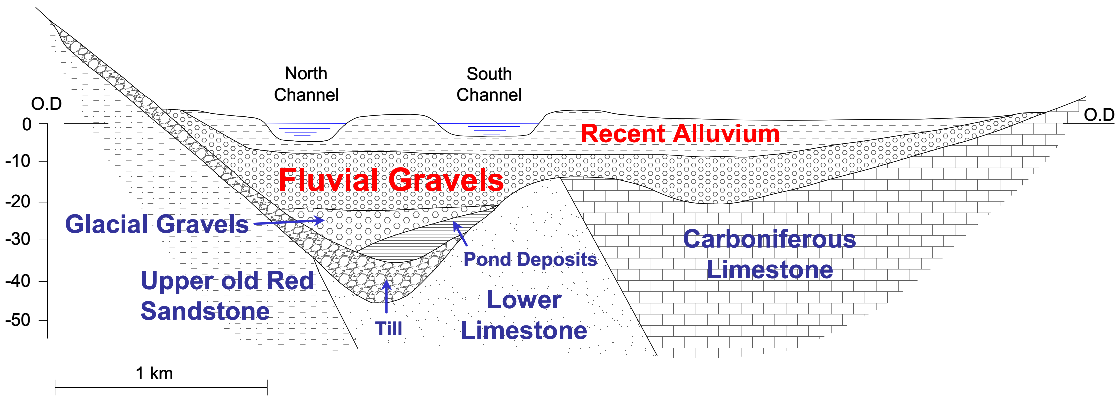


Figure 1: Diagrammatic geological cross section across the River Lee in Cork City centre, after Reilly and Sleeman (1977).

This occurred when sea level fell to about -130 mOD and glacier action together with melt water release cut down to the new base level to meet the lower sea-level. The Lee Buried Valley runs from Crookstown in the west towards Youghal in the east, a distance of at least 60 km. It is of the order of 0.5 to 0.75 km wide and is more or less directly underneath and in line with the central island of Cork City, Reilly and Sleeman (1977) and Milenic and Allen (2002). The Buried Valley is well exposed between Classes and Garryhesta in the Ballincollig/Ovens area to the west of the city where it is being quarried for sand and gravel. Here it contains a remarkably uniform succession of gravels. However beneath Cork City the details of the stratigraphy are less clear. The Buried Valley was infilled predominantly with sand/gravel. The deeper deposits comprise glaciofluvial outwash underlain in some areas by Till and Pond Deposits (likely to be predominantly silt and clay of low permeability), Scourse et al. (1992). At a shallower level the sand and gravel glaciofluvial outwash has been reworked within the valley system by the rising sea level at the end of glaciation about 10,000 years B.P. The boundary between these zones may be marked by a narrow band of organic material of Holocene age within the gravel deposits which was identified at about -6 mOD at a site near the east end of the central island, Long and Roberts (2008).

Later the valley was infilled with estuarine clays, silts and peats, typically 3 to 4 m thick. Marshes formed the final shape of the upper estuary after the sea level steadied near its present level about 6,000 years ago. At this time the river was braided, flowing in a large number of channels between a series of marshlands, (see Figure 2). These marshlands were progressively reclaimed, the channels culverted and the islands urbanised. Ultimately the majority of the Cork waterways were covered over to accommodate the spacious streets required of 18th century planners leaving today the North and South Channels of the Lee, Whittow (1974).

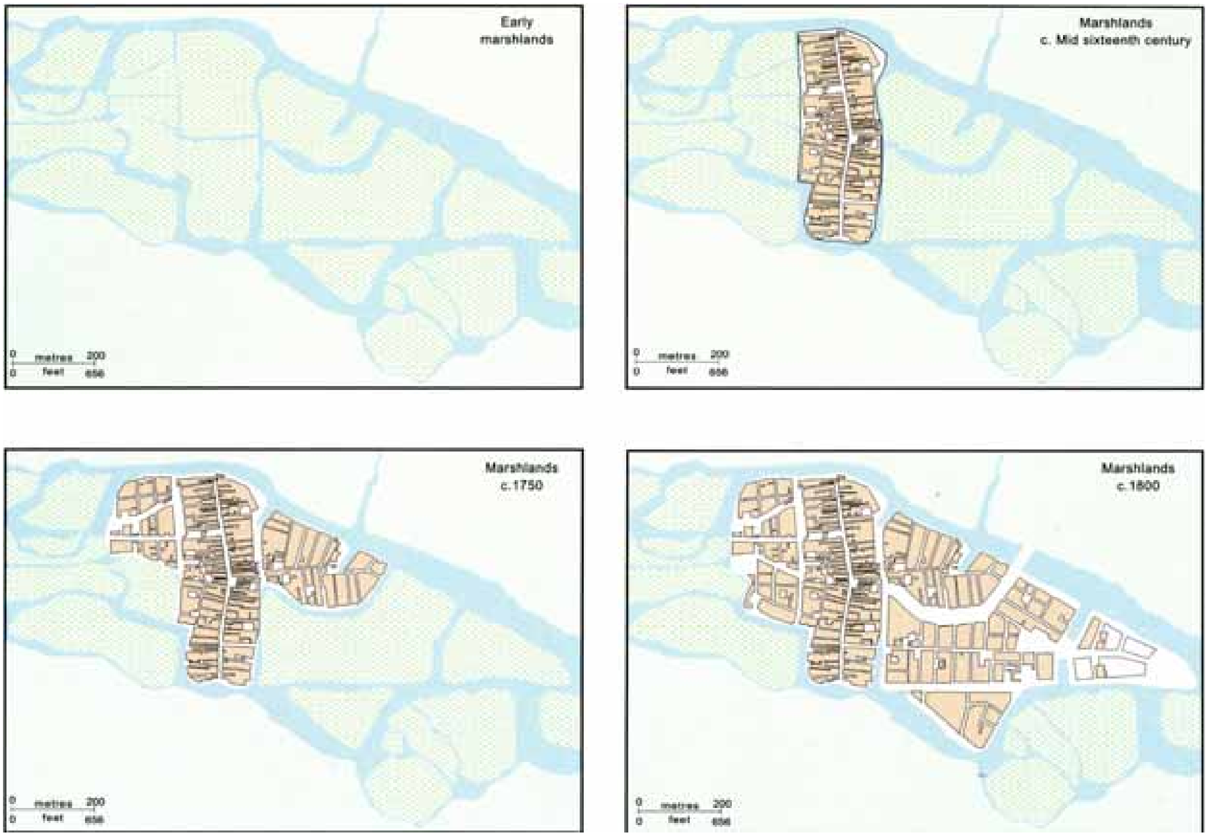


Figure 2: Development of marshlands Cork City, O'Flanagan (2005).

This history has resulted in the present sequence of ground conditions in Cork City which typically comprises;

Made Ground: Generally granular in nature, includes concrete, brick fragments and occasional organic lenses. Figure 2 suggests the made ground may be thickest below the streets where old river channels have been infilled.

Alluvium: Comprises mostly soft to firm sandy silt with lenses and pockets of organic clay and peat.

Fluvioglacial Gravels: Generally described as loose becoming dense fine to coarse sandy gravel with some cobbles and sandy zones at depth.

This sequence is underlain by Till, Pond Deposits and bedrock (Limestone and Sandstone) at 20 to over 35 m depth below the central island in Cork City.

The Lee is at least partly in hydraulic connection with the Fluvioglacial gravel aquifer. Recharge takes place by vertical percolation through the river bed and probably by bank infiltration. The Lee is tidal within Cork City in the range +1.5 to -1.5 mOD. The aquifer is generally under moderate sub-artesian pressure due to the alluvium above. Groundwater levels are at approximately 0 mOD but are subject to tidal fluctuations typically in the range +1 to -1 mOD. Permeability data from pumping tests is summarised by Long et al (2007) indicating that the bulk horizontal permeability of the gravels is high, being of the order of 1×10^{-3} to 3×10^{-3} m/s.

3.0 NUMERICAL MODEL

Excavations for basements have been undertaken for several developments in Cork City over the last 8 years. Required drawdown levels have typically ranged from -2 to -7 mOD for single up to triple basements. Numerical modelling for a typical scenario has been undertaken using the 2D finite element software package SEEP/W, produced by Geo Slope of Canada.

The basic assumptions made were as follows;

Excavation	Circular 30 m radius (radial flow assumed) Equivalent to a 47 m by 47 m square cofferdam
Dewatering wells	Located internally 3 m in from the cut-off with a response zone down to -11 mOD
Groundwater level	+1 mOD
Target internal drawdown	-5 mOD assumed
Model base	No flow boundary assumed at -25 mOD
Distance of influence	200 m from side of excavation assumed
Permeability	Horizontal permeability $k_h = 2 \times 10^{-3}$ m/s Vertical permeability, ; $k_v/k_h = 1.0$, $k_v/k_h = 0.1$ and $k_v/k_h = 0.01$
Cut-off depth	Varied from no cut-off to full cut-off

The model grid used is shown in Figure 3.

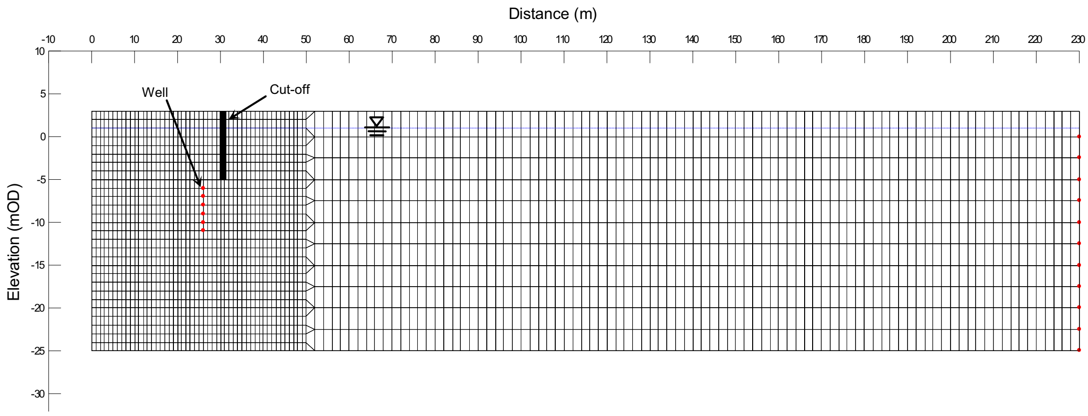


Figure 3: SEEP/W model grid

A series of model runs have been undertaken varying both the cut-off depth and the vertical permeability. For each run, steady state output has been obtained on the basis that experience has shown that steady state groundwater levels are established within a few days of the start of pumping for both pumping tests and dewatering schemes. External drawdown was determined 1 m outside the cut-off at a level of -5 mOD.

4.0 MODEL RESULTS

As would be expected in such high permeability conditions flow rates are high at 866 l/s for $k_v/k_h = 1.0$ with no cut-off. Inflows are reduced by extending the cut-off depth and by increased anisotropy. The model results have been plotted in a non dimensional form in Figures 4 and 5. Figure 4 shows the required abstraction flow expressed as a percentage of the flow with no cut-off (for $k_v/k_h = 1.0$) plotted against the percentage cut-off depth below standing groundwater level. Figure 5 shows the external drawdown expressed as a percentage of the internal drawdown again plotted against the percentage cut-off depth below standing groundwater level. In each case results have been shown for the three permeability cases; $k_v/k_h = 1.0$, $k_v/k_h = 0.1$ and $k_v/k_h = 0.01$.

The results of pumping tests are primarily controlled by horizontal flow and therefore give values for k_h only. Borehole logs in Cork City often show the presence of sand bands at various horizons, Long and Roberts (2008), and these might be expected to give anisotropic conditions so that $k_h > k_v$. However the scale of this anisotropy will depend on the lateral extent and interconnection of the sand horizons which cannot readily be determined from borehole logs. In short there is some evidence for anisotropic conditions but pumping tests and borehole logs cannot identify their significance and scale.

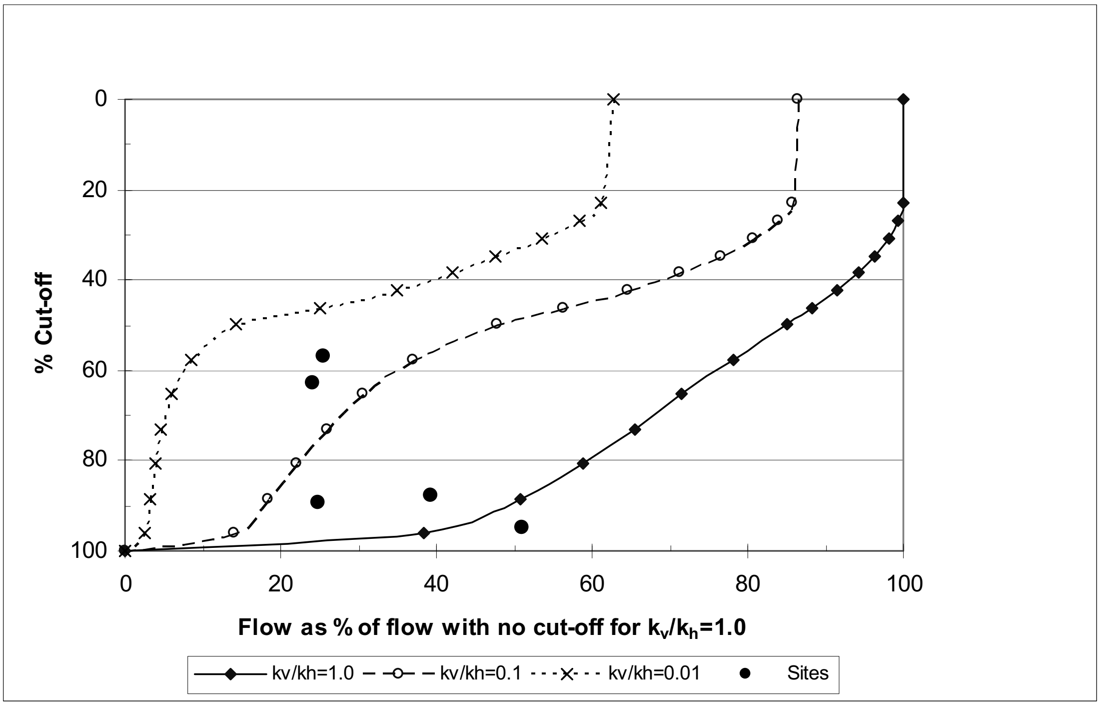


Figure 4: Model Results - Percentage cut-off verses percentage flow

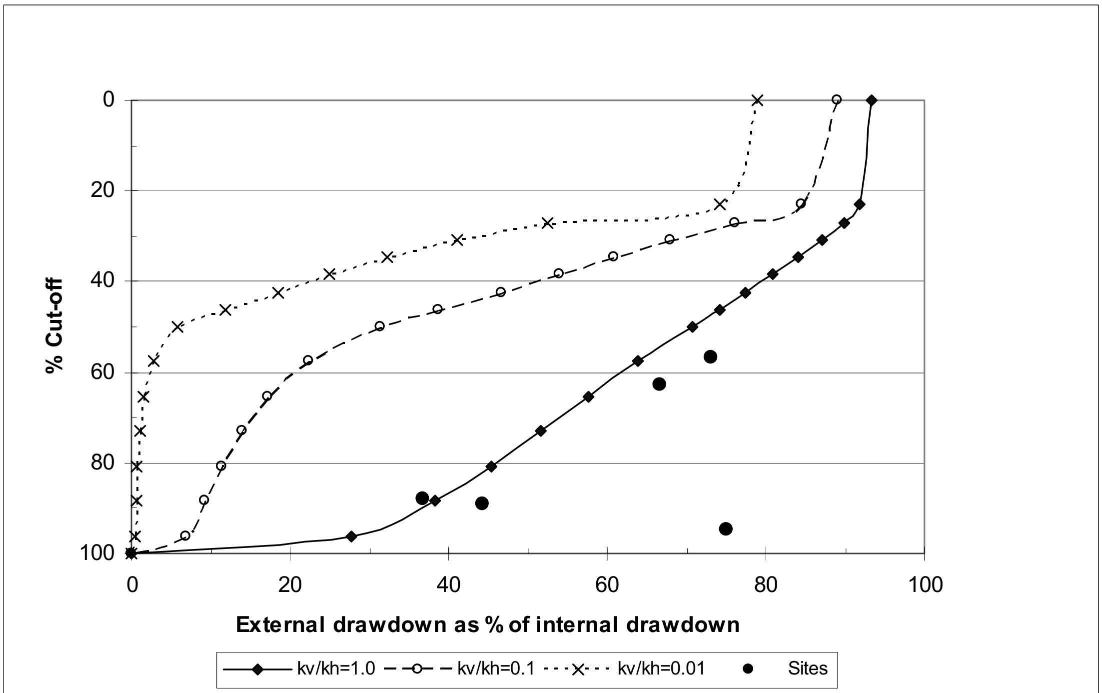


Figure 5: Model Results - Percentage cut-off verses percentage external drawdown

As summarised below the model results given in Figures 4 and 5 shows that the effectiveness of a cut-off is highly dependent on the ratio k_h / k_v.

Anisotropy	% Cut-off	% Flow	% External drawdown
k_v/k_h = 1.0	50	85	70
k_v/k_h = 0.1	50	47	31
k_v/k_h = 0.01	50	14	6

The assumption that k_v/k_h = 1.0 (isotropic conditions) would have to be the conservative assumption in the absence of other supporting data. The above table illustrates that for k_v/k_h = 1.0 a 50% depth cut-off results in minimal reduction in flow and external drawdown. Indeed for these isotropic conditions, flows and external drawdowns remain appreciable even for a 90% cut-off. k_v/k_h = 0.1 is commonly assumed to be the case by the geotechnical community due to the natural fabric/bedding of granular deposits. The presence of sand layers identified in some boreholes suggests that this is a perfectly possible scenario in the gravel aquifer in Cork City. For k_v/k_h = 0.01 there would be an expectation of clearer evidence of anisotropic conditions and bedding than has generally been found in Cork City. It is clear that some understanding of the relationship between horizontal and vertical permeability would assist in assessing the benefit to be gained from extending the depth of perimeter cut-offs in high permeability soils in terms of reduced flow and reduced external drawdown.

5.0 CORK CITY EXPERIENCE

Conventional wisdom is that it is more cost effective to install a cut-off to the depth required for structural support than to extend the cut-off in order to reduce pumping. This view is generally the case for soils of medium permeability (10^{-5} to 10^{-3} m/s) where there is little or no restriction on external drawdowns or discharge flow rates. The situation for excavations in Cork City may deviate from this norm in two important respects; the permeability (at least the shallow horizontal permeability) is greater than 10^{-3} m/s and there is a concern that appreciable external drawdown may under-drain the superficial alluvium resulting in a possible settlement risk to neighbouring structures founded on this stratum. Figure 6 is reproduced from CIRIA (2000) and shows the range of application of dewatering systems as a plot of drawdown against permeability.

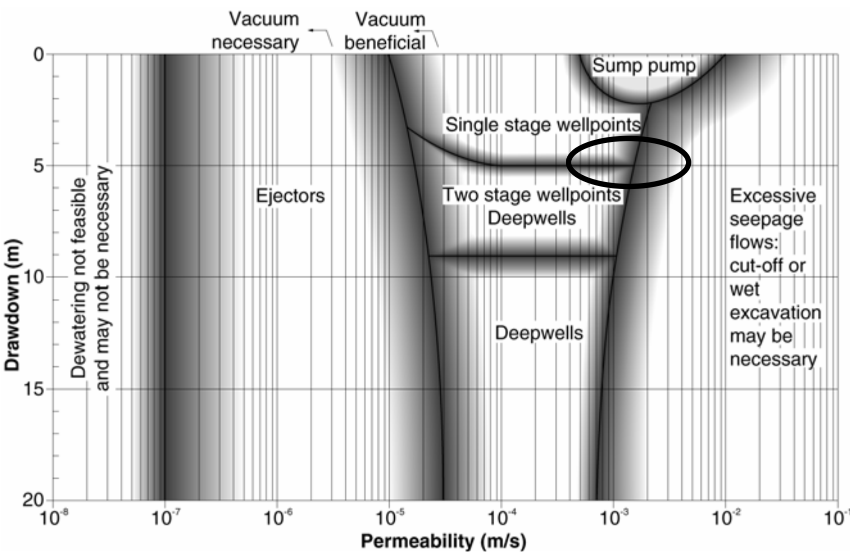


Figure 6: Range of application of pumped well groundwater control techniques, from CIRIA (2000).

Oval shows typical conditions for basement excavations in Cork City.

The oval shown in Figure 6 highlights the likely conditions for a basement excavation in Cork City; 4 to 6 m drawdown and permeability of the order of 10^{-3} m/s. It can be seen that this is straying into the area where seepage flows may be excessive so that a cut-off may be necessary to limit inflows. This is consistent with experience reported by Allen and Milenic (2003) who give data for an excavation in Cork City carried out in open cut (*i.e.* with no cut-off) with a 110 m perimeter where required abstraction flows were high, in the range 250 to 290 l/s, for a required drawdown of 4 m. These high flow rates may well lead to discharge difficulties unless direct access is available for discharge to a large capacity receptor such as the river. These flow rates would certainly be excessive for most sewer networks.

Data from a range of projects undertaken in Cork City is summarised in the table below,

	Cut-off	**Flow as % of flow with no cut-off**	**External Drawdown as % of internal Drawdown**	**Source of information**
Site Ref No.	%	%	%	
Site A	88	39	37	Long et al (2007)
Site B	57	26	73	Author's files
Site C	95	51	75	Author's files
Site D	89	25	44	Author's files
Site E	63	24	67	Author's files

These results have been plotted on Figures 4 and 5. There are serious limitations in deriving this table for several reasons as follows,

- Where rock level has been established it often varies appreciably across the site. For the purposes of the table the level assumed is the average level where proven.
- For some sites the cut-off depth varied around the perimeter. For the purposes of the above table the average cut-off depth has been used.
- The rock may not provide a low permeability base layer as assumed in the model. This particularly applies to the Limestone present to the south which underlies Sites C (this may explain the high external drawdown relative to the apparently high cut-off which is out of step with the rest of the data, see Figure 5).
- In order to estimate the flow with no cut-off a permeability and distance of influence must be assumed. The values used for the modelling work have been used in this case. Comparison with sites where no cut-off was installed suggests that this may be an underestimate.
- The eExternal drawdown varies around the site perimeter. For the purposes of the above table the average external drawdown has been used.

The site data shown on Figure 5 is most consistent with $k_v/k_h = 1.0$ (particularly if the data for Site C is ignored, see comments above). The site data shown on Figure 4 is a lot less consistent. This may reflect genuine variations at the different sites but could also be partly due to the uncertainty in estimating the flow with no cut-off present, see comments above. Also this study has concentrated on anisotropic conditions, an alternative scenario is that isotropic conditions prevail but the permeability reduces below a certain horizon. In this case flow rates would be sensitive to whether or not the cut-off reached the reduced permeability horizon rather than the percentage cut-off achieved.

6.0 CONCLUSIONS

This paper sets out to examine the relationship between physical cut-off depth, flow rates, external drawdowns and anisotropic conditions in high permeability granular soils. The modelling work has shown that the efficiency of a partial cut-off is sensitive to anisotropic conditions where the vertical permeability k_v is less than the horizontal permeability k_h. For $k_v/k_h = 0.1$, which is considered to be common in granular soils, partial cut-off efficiency is significantly enhanced over the isotropic case where $k_v/k_h = 1.0$. The model results have been compared with data from several basement excavation projects undertaken in Cork City in high permeability gravels. The data is not conclusive but seems to imply that isotropic conditions, where $k_v/k_h = 1.0$, prevail.

The model results show, as would be expected, that increasing the cut-off depth clearly reduces inflow rates and minimises external drawdowns. However the results also show that it is extremely difficult to predict the precise benefits that will accrue in any particular situation. In particular a conventional single well pumping test undertaken at design stage, in advance of the partial cut-off installation, is likely to be of much less value than experience of dewatering from a nearby site with similar geometry and hydrogeological conditions. This conclusion is the cause of significant challenges to basement project design teams because the cost of extending cut-offs is significant and must be off-set against the possible risk of settlement associated with external drawdowns. A conventional single well pumping test with a single level piezometer array is mainly influenced by the horizontal permeability rather than the vertical permeability and as a result sheds only modest light on these issues. Consideration should be given to more elaborate pumping test arrangements with multi level piezometer installations that could be analysed by modelling to evaluate both the vertical and the horizontal permeability.

7.0 REFERENCES

Allen and Milenic (2003). *Drainage problems during construction operations within a buried valley gravel aquifer*. A Allen and D Milenic. RMZ – Materials and Geoenvironment, Vol. 50 (1), p1-4, 2003.

CIRIA (2000). *Groundwater control – design and practice*. M Preene, T O L Roberts, W Powrie and M R Dyer. Construction Industry Research and Information Association, CIRIA C515, London 2000.

Long et al (2007). *Soil characterisation and construction of deep basements in high permeability gravels in Cork, Ireland*. M Long, T Roberts and M Creed. Proc. 14th European Conference on Soil Mechanics and Geotechnical Engineering (ECSMGE), Madrid, September 2007.

Long and Roberts (2008). *Engineering Characteristics of the Glaciofluvial Gravels of Cork City.* M Long and T Roberts. Paper presented at joint meeting of the Geotechnical Society of Ireland and the Irish Association of Hydrogeologists at the Geological Survey of Ireland, Beggars Bush, Dublin on 8 January 2008.

Milenic and Allen (2002). *Lee Buried Valley – A significant groundwater resource*. D Milenic and A Allen. Proceedings XXXII IAH Congress, Mar del Plata, Argentina, Groundwater and Human Development, Bocanegra et al (eds.) 8 p, 2002.

O'Flanagan (2005). *Transformation, a minor port town becomes a major Atlantic port city*. Part 2, in *Atlas of Cork City*. Edited by Crowley, Devoy, Lineham and O'Flanagan, Cork University Press, (2005).

Reilly and Sleeman (1977). *Geology of Ringaskiddy Basin.* Unpublished report for Cork Harbour Commissioners. T Reilly and A Sleeman. Geological Survey of Ireland 1977.

Scourse et al. (1992). *New evidence on the age and significance of the Gortian temperate stage: a preliminary report on the Cork Harbour site*. J D Scourse, J M R Allen, W E N Austin, R J N Devoy, P Coxon and H P Sejrup. Proceedings Royal Irish Academy, Vol. 92B, pp21 – 43, 1992.

Whittow (1974). *Geology and Scenery in Ireland.* J B Whittow. Pelican, Harmonsworth, Bungay Suffolk, 1974.

THE SYMBIOTIC RELATIONSHIP BETWEEN GROUNDWATER AND GEOTECHNICAL ENGINEERING

Gordon Clarke (1), David Hughes (2), Lee Barbour (3)

(1)Fritsch, Chiari & Partner ZT GmbH - Diesterweggasse 3, 1140 Vienna, Austria, Clarke@privat.fcp.at

(2) School of Planning, Architecture and Civil Engineering, Queen's University Belfast, Belfast, UK

(3)Department of Civil & Geological Engineering, 57 Campus Drive, University of Saskatchewan Saskatoon, Saskatchewan, SK S7N 5A9, Canada

ABSTRACT

Geotechnical engineering works often interact with hydrogeologic systems. In some cases these interactions are detrimental to the performance of both temporary and permanent works. In many cases, the existing hydrogeologic system can also be negatively affected either by the geotechnical construction or the control systems used to mitigate the potential negative effects of groundwater inflow or pressures on the performance of the geotechnical works. Groundwater is viewed quite differently by water resources managers and construction engineers. To the hydrogeologist, groundwater is a potential resource, valued for abstraction or for its contribution to springs, stream flow and wetlands. In contrast, the construction engineer often perceives groundwater as a potential problem requiring a solution. Projects founded on or penetrating into water-bearing soils are often more difficult to construct than those entirely in the vadose zone or in very low permeability strata. Engineers will adopt methods to mitigate the effect of groundwater on construction. This might include temporary dewatering pumping or the construction of physical cut-off walls into the aquifer.

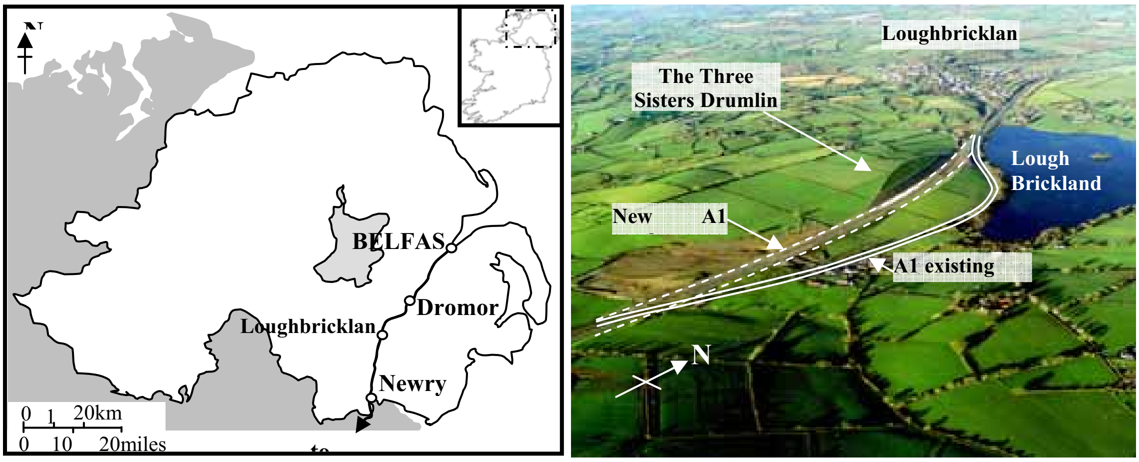
This paper provides a case history which highlights the types of interactions that occurred between groundwater and geotechnical construction associated with a new 18m deep cutting in a drumlin formed in overconsolidated till at Loughbrickland, Co. Down, Northern Ireland. The paper focuses on the effects of both cutting excavation and climate variability on the hydrogeology of the drumlin

INTRODUCTION

Roads Service (NI) commenced a construction project to upgrade a section of the dual carriageway from Loughbrickland to Beechill, located on the Belfast – Dublin road, in 2004. The improvement in horizontal road alignment required a major cutting through a drumlin. The Roads Service (NI) and Queen's University of Belfast (QUB) recognised that the Loughbrickland cutting provided an excellent research opportunity to better understand the mechanisms that govern the long-term stability of cuttings. They also realised the importance of understanding the influence of drumlin composition, till formation and hydrogeology on slope stability.

SITE LOCATION

The highway cutting is located on a new dualling of the A1 near Loughbrickland, Co. Down and is part of the main Belfast to Dublin Euroroute 1 (Figure 1a). The cutting is situated on a drumlin known as The Three Sisters (Figure 1b), approximately 125 metres above mean sea level (drumlin hollow approximately 80mAOD).



(a) (b)

Figure 1 (a) Location Map: Belfast – Dublin Euroroute 1. **(b)** Photograph of drumlin landscape: Three Sisters Drumlin (courtesy of Road Service NI)

GEOLOGY

The Loughbrickland area is covered with Late Midlandian till underlain with weathered Silurian gritstone, deposited by ice that moved southwest from a Scottish source and southwards across the area from the Lough Neagh basin during the Drumlin re-advance (McCabe et al, 1999). This ice flow is reflected in the numerous drumlins that dominate the topography of the area. Figure 2 shows a cross-section of the drumlin pre-excavation.

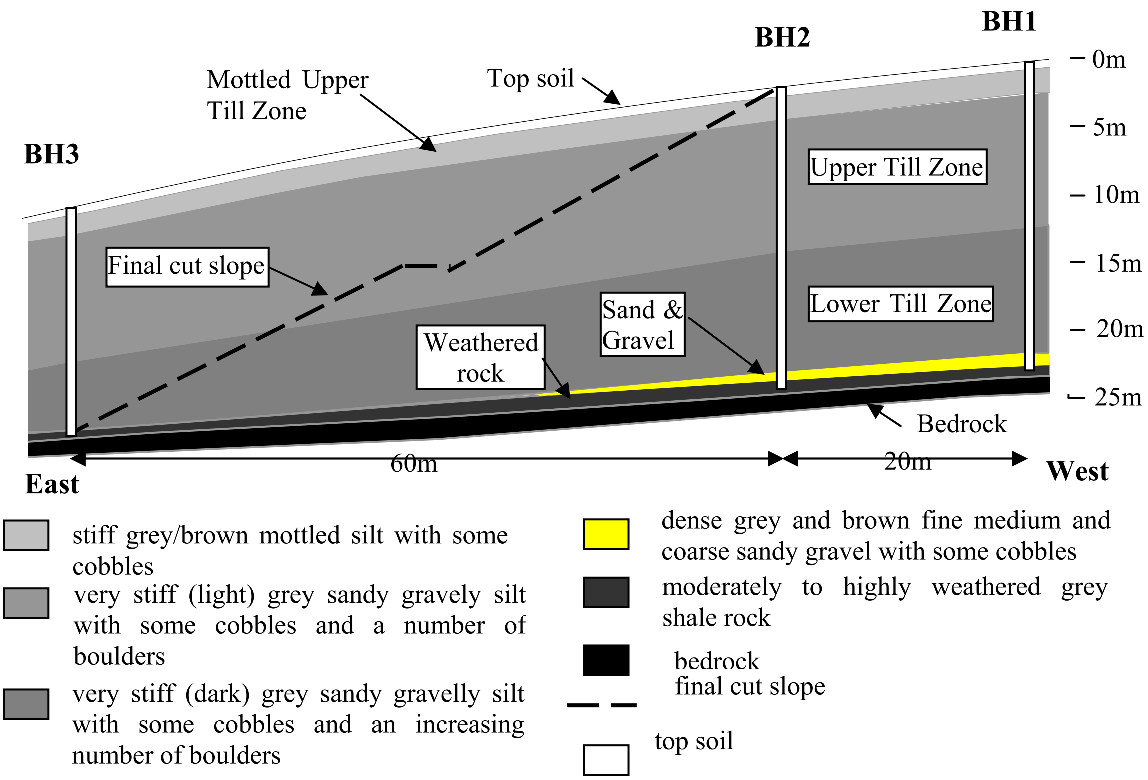


Figure 2 - Geologic E-W cross-section of drumlin

HYDROGEOLOGY OF A SOUTH DOWN DRUMLIN

CLIMATE

The climate at Loughbrickland, like the rest of Northern Ireland, is a temperate maritime climate. The weather is unpredictable at all times of the year and although the seasons are distinct they are considerably less pronounced than in continental Europe. Average daytime maximums in Loughbrickland are 6.5 °C (43.7 °F) in January and 17.5 °C (63.5 °F) in July. The Loughbrickland area is one of the driest parts of Northern Ireland. However, it has a mean annual precipitation of approximately 900 mm/year. The seasonal variation of rainfall in Loughbrickland is not large, but the wettest months are between August and January

Mean annual values for precipitation, evapotranspiration and run-off for Loughbrickland are 825 mm, 335mm and 455mm respectively for the period 2004 - 2006. Precipitation varies both annually and inter-annually. Typically the summer months are drier than the winter months. In contrast, potential evapotranspiration (PET) varies substantially between summer (~80 mm/month) and winter (~10mm/month). Elevated rates of summer PET contribute significantly to the development of a soil moisture deficit of 50-70 mm during the summer months. Actual Evapotranspiration (AET) equals Potential Evapotranspiration (PET) throughout most of the year with the exception being the period between May and September.

FIELD HYDROGEOLOGIC INVESTIGATION

A field hydrogeological instrumentation programme (2003-2007) was carried out following consultation with Roads Service and a preliminary site reconnaissance visit (Clarke, 2007). Figure 3 shows the location of the hydrogeological monitoring boreholes. Each borehole contained 2 – 3 standpipes which recorded head levels at independent elevations. The programme monitored soil and hydrogeological conditions at the Loughbrickland cutting and consisted of detailed monitoring of pore water pressure.

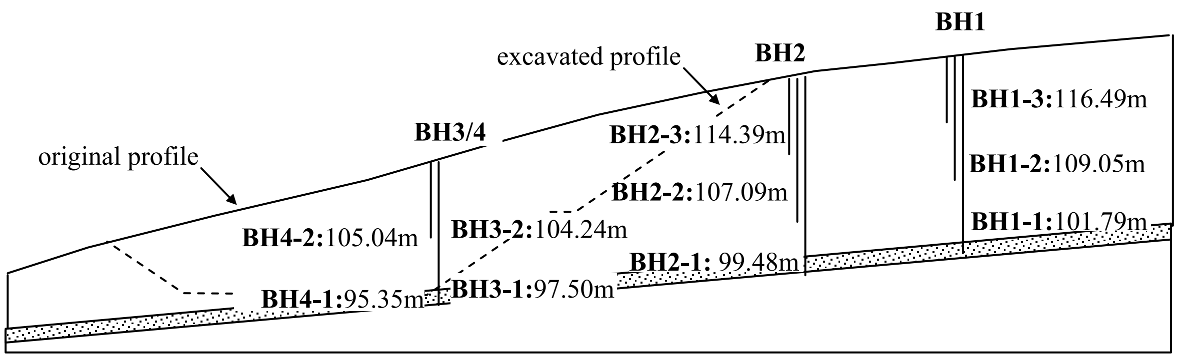


Figure 3 Loughbrickland drumlin cross-section.

Field based tests (e.g. falling head) indicated that the glacial till has a low hydraulic conductivity (6.5×10^{-10} - 1.9×10^{-8} m/s), which is typical of glacial till soils in the region and the UK. The results of the field permeability tests and initial head levels clearly show a contrast in hydraulic conductivity between a postulated upper and lower till. This change in hydraulic conductivity is in the range of 1 or 2 orders of magnitude and provides evidence of a conceptual model postulated by Clarke *et al.* (2007) that there are two till zones within the drumlin (Figure 2). There was no conclusive textural evidence for the variation in permeability however it is assumed that factors related to drumlin formation such as shearing or presence of a basal till unit are responsible for these differences.

The phreatic surface appears to vary seasonally between ground surface and a depth of 2.5 m, based on observations of the depth of mottled clay. It is important to note that although the water table fluctuates seasonally, the clay till above the water table likely remains near saturation through the year. The slope, in which the cutting was made, drains eastward to Lough Brickland.

The hydrogeological results also show the significant influence of the bedrock contact zone in providing an underdrain to the glacial till zones. The head level (106.9 mAOD) within the bedrock contact zone is significantly lower than till head levels, developing large vertical gradients within the lower till (Figure 4). The bedrock zone may extend to the Lough (Lough Brickland mean water level 83.27m) but is likely to be confined by the lower K till primarily over the area of the drumlin.

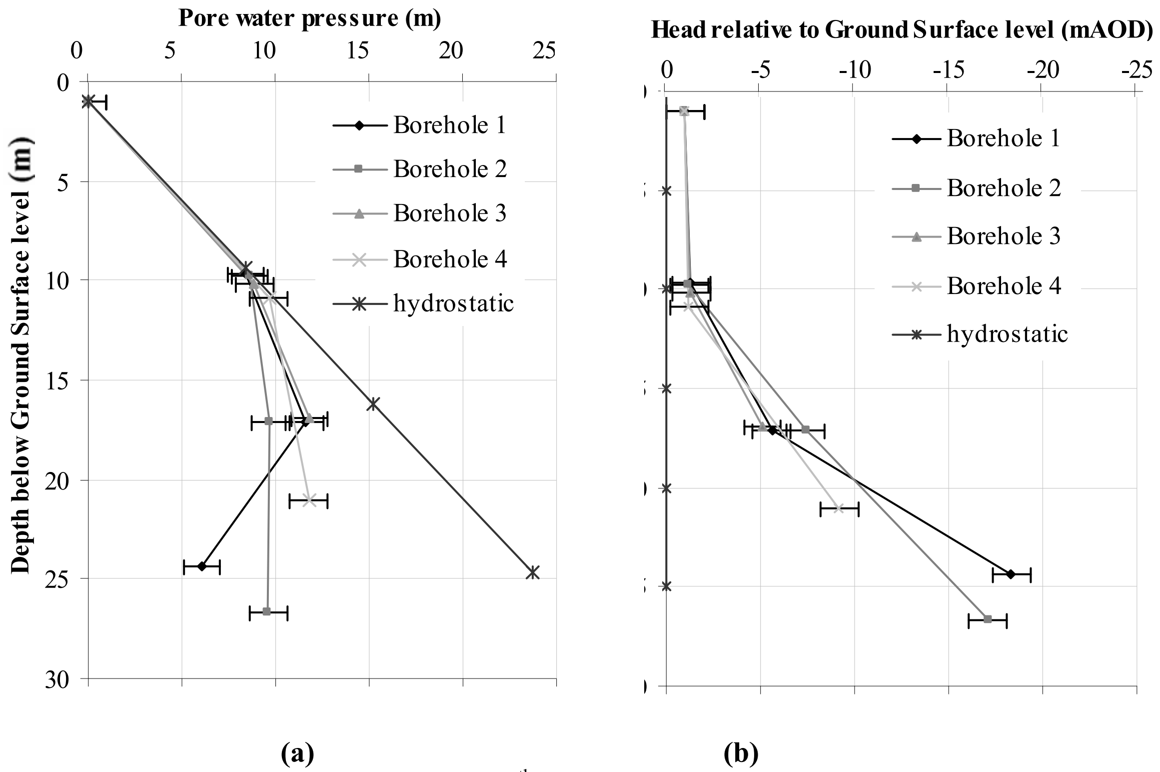


Figure 4 - Boreholes 1-4 pre-excavation (20th April 2004) **(a)** pore water pressure distribution and **(b)** relative head level distribution (assumption water table 1m below surface).

STEADY STATE HYDROGEOLOGICAL MODELING

Steady state modeling, using the SEEPW model (Geo-slope International, 2004) was used to determine the hydrogeological regime within the drumlin prior to excavation. The model was verified using field pore-water pressure data collected from the 10 installed standpipes. Figure 5 illustrates the 2-D seepage model that was created based on drumlin boundary conditions to simulate typical winter conditions - water table at surface (Lough Brickland head level 83 m AOD, no flow boundary at the centroid of drumlin and a ground surface water level P=0) and soil parameters. The modelled hydrogeology provided a good correlation with field pore-water pressure data. The drumlin hydrogeologic regime is a typical example of upslope recharge and downslope discharge. The hydrogeologic system is driven by topography and stratigraphy of the drumlin and the under drainage that exists in the bedrock contact zone. A hinge point has been drawn in Figure 5 representing the point where the gradients across the 'water table' change from downward flow (recharge zone) to upward flow (discharge zone). This point is equivalent to a hinge point between recharge and discharge (Freeze and Cherry, 1979). The subtle variation in hydraulic conductivity between upper and lower till zones has a significant effect on the seepage regime in the drumlin and head levels.

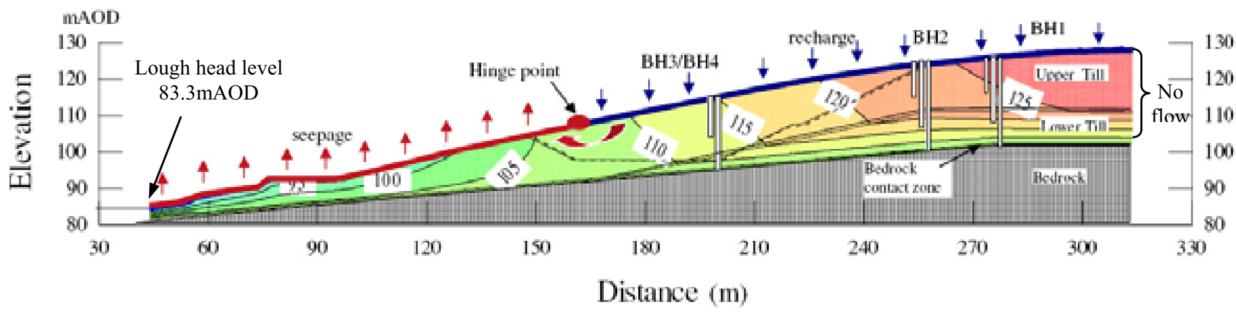


Figure 5 – Total head distribution in initial steady state model of pre-excavation conditions.

A study was performed using the seepage model to estimate potential recharge rates by varying q (unit flux boundary) and assessing the steady state phreatic surface level at BH1 and BH2. Figure 6 summarises the results from the recharge sensitivity analysis. One of the significant conclusions of this study was how sensitive this water table depth is to the recharge rates. The model estimates that in pre-excavation and post-excavation conditions a reduction in annual recharge of 20 mm/year could contribute to a reduction in water level of approximately 10m. Prior to excavation, a minimum annual recharge of 30-35 mm/year is required to maintain the phreatic surface within 2.5 m of the surface prior to excavation. This estimated recharge is consistent other Irish and UK tills that estimate annual recharges varying from 22 to 35 mm/year (Fitzsimons, 2006). The estimated annual recharge is approximately 3% of the average annual precipitation (1107 mm). This is comparable to the Irish and UK tills data (recharge = 1.7 - 6% of annual precipitation), which were reviewed by Fitzsimons (2006). In contrast, post-excavation 45-50 mm/year of recharge is required to maintain a water table within 2.5 m of the surface. The cut slope provided a significant discharge area and increased the potential for recharge in the drumlin above the crest of the cut slope. The figure illustrates how sensitive the water table/head levels are to changes in recharge rates and geometry.

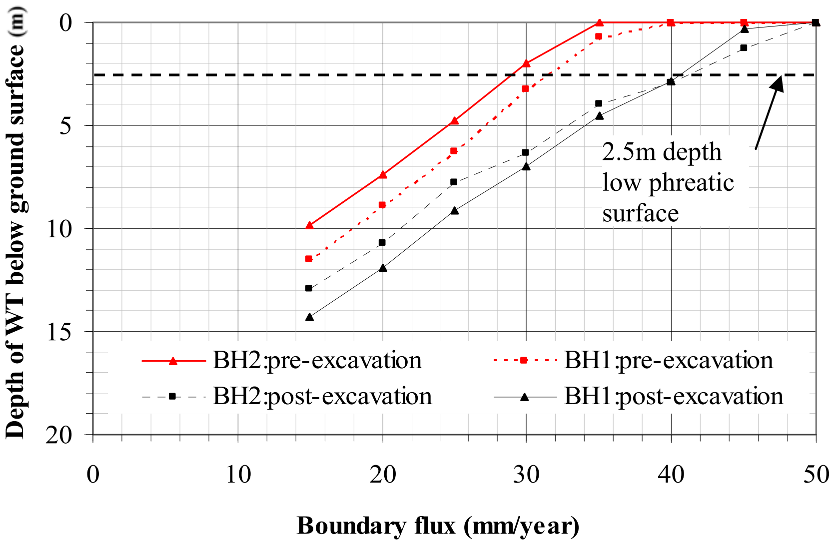


Figure 6 - Variation in the depth to water table for various rates of recharge pre and post-excavation.

WATER BALANCE ESTIMATION

A field measurement of run-off was not available. However, an annual estimate can be determined based on a simple water balance using PPT, AET (assumed to be similar to PET) and estimates of net percolation or recharge (Table 1). Run-off is assumed to occur when daily values of PPT exceed combined AET and Recharge (or net percolation). Run-off is estimated to be 49 - 63% of PPT during the research period. This estimate compares well with national and UK (% of PPT) of 60% (1961 - 1990, DEFRA, 2004) for long-term annual average run-off. Figure 7 shows the annual cumulative water balance parameters assuming run-off is 55% of precipitation. The study also provided an estimate of the recharge varying from zero percolation in winter to approximately 50 mm/year in summer prior to excavation.

Table 1 - Annual PPT, AET, recharge and estimated runoff (Meteorological Office, 2006).

Description	Annual Total (mm)		
	2004	**2005**	**2006**
Precipitation (PPT)	756	772	948
Actual Evapo-transpiration (AET)	347	337	321
Recharge*	35	35	35
Estimated Run-off**	374	400	592
Runoff (% of PPT)	49%	52%	63%

* Recharge based on initial steady state modelling using field permeability
** Estimated run-off = PPT-AET-recharge*

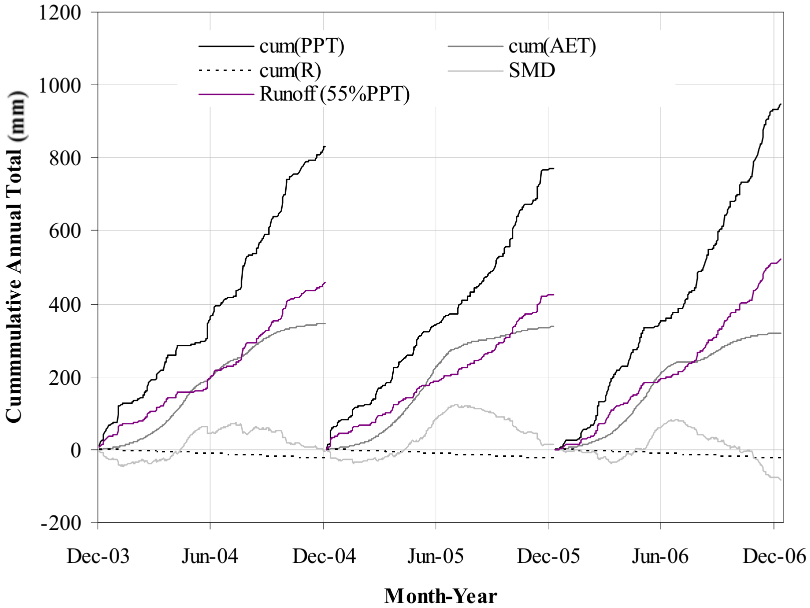


Figure 7 – Estimated annual cumulative water balance: assuming run-off = 55% PPT (Jan 2004 - Dec 2006).

THE EFFECTS OF EXCAVATION ON HYDROGEOLOGY AND STABILITY

The primary purpose of the research at Loughbrickland was to evaluate the effect of climatic change and variability on the long-term stability of cuttings in tills. As it turned out, the flowing artesian conditions that were created during the excavation provided the greatest risk to groundwater and slope stability in the short term. It should be noted that little information was available on the groundwater conditions from the original site investigation undertaken for the design of the cutting. The presence of the confined aquifer at the base of the drumlin was also not identified in the original site investigation. The following paragraphs illustrate the threat posed by a lack of a thorough understanding of the hydrogeology to the short term stability of the road infrastructure at Loughbrickland.

The final excavation of the cutting resulted in critical changes in the hydrogeology of the drumlin. Relatively minor decreases in head occurred within the underlying weathered bedrock and the drumlin during the excavation. Figure 8 illustrates the modeled hydrogeological regime within the drumlin immediately after excavation. The head level within the weathered rock aquifer remained relatively constant at 105 m AOD relative to the excavated surface level 98 m AOD. Critical uplift conditions developed during the final stages of excavation as a result of artesian heads, despite pore water pressure dissipation through the installed standpipes (Figure 9a). The confined aquifer and upward hydraulic gradient caused low effective stress (σ') conditions in the toe of the slope which resulted in failure of the toe (Figure 9b).

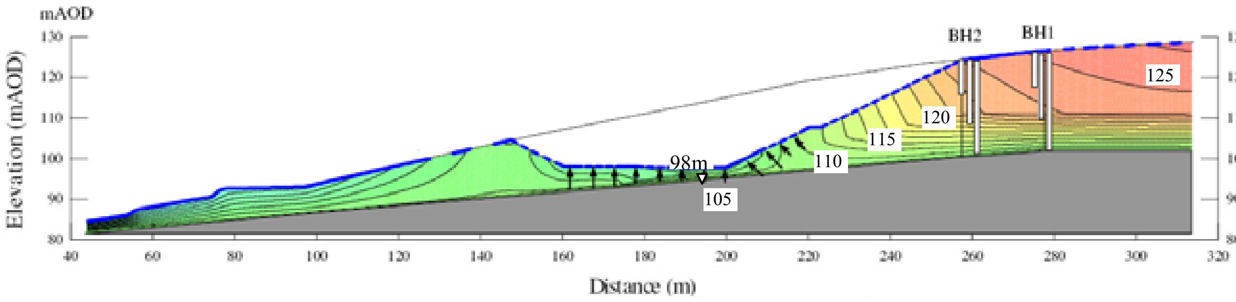


Figure 8 – Immediate post-excavation steady state model verification: total head contours (3rd Sept. 2004)

(a) **(b)** **(c)**

Figure 9 – (a) Photograph of water (artesian) flow caused and **(b)** toe failure as a result of the developed artesian conditions Sept. 2004. **(c)** Remediation work to shallow slope failures – granular fill replacement.

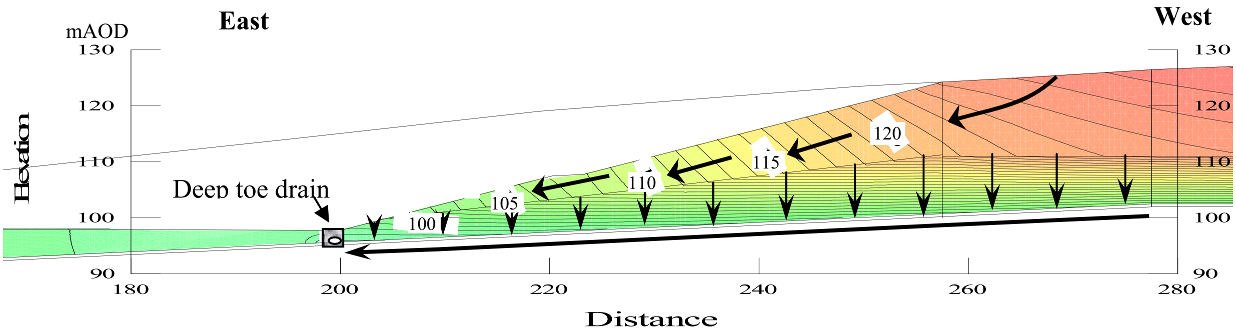


Figure 10 - Total head contour diagram (Head/mAOD) of steady state seepage analysis post-excavation and following the installation of deep intercept toe drain.

Shallow failures were also observed on the cut slopes at Loughbrickland soon after the completion of excavation (Figure 9c). These are likely to be the most common cause of failure throughout the service life of the cutting as described by Perry (1989). The problems are mainly confined to overconsolidated clays and particularly high sections of cuttings similar to Loughbrickland. However, the ongoing cost of repairs to the slope failures constitutes a significant amount of maintenance expenditure. The maintenance component of the whole life cost may be up to 3.7 times the capital cost for construction with a high rate of slope failure (Reid & Clarke, 2000).

In response to the critical conditions a deep toe drainage was installed which completely dissipated the pressure in the weathered rock layer and stabilised the excavation (Figure 10). The dissipation of the aquifer pressure helped to stabilise the slope by creating a free draining layer at the base of the drumlin. The hydrogeology reverted back to the original downward hydraulic gradient within the lower till layer.

CONCLUSION

Civil engineering works that involve excavation into the vadose and saturated zones may create significant impacts on the groundwater environment and subsequent geotechnical stability issues. In the majority of cases measures can be adopted to mitigate the effects of these impacts. The artesian conditions that were induced at Loughbrickland highlighted that a clear understanding of the hydrogeology of an area is essential in the geotechnical design of infrastructure.

Ideally, the potential for these impacts to occur needs to be assessed for a given site and project at an early stage in investigation and planning. Once this has been done the project design can be varied, and mitigation measures adopted, to control or avoid these impacts. Toe failures and flooding of the site could have been avoided by controlled depressurisation of the aquifer.

Obtaining an accurate water balance was one of the key elements in ensuring understanding and accurate modeling of the field conditions. The hydrogeological modeling emphasized the sensitivity of groundwater levels to recharge rates and hydraulic conductivities of the soil.

Monitoring of appropriate meteorological and hydrogeological parameters is an essential part of managing potential impacts. The research at Lough Brickland has established a good hydrogeological base line study of the drumlin. Consideration should be given to utilising this study site along with existing and future monitoring as a field site for further hydrogeological investigations including *in situ* studies of contaminant transport, geochemical processes, hydraulic transients and dynamics, and climate/groundwater interactions. Future monitoring of the instrumentation and model development on this slope could also provide insight into the effect on the hydrogeology on the long-term stability of the slope.

REFERENCES

CLARKE, G.R.T., 2007. The impact of climate on the hydrogeology and stability of a large excavation in a glacial till. PhD dissertation, Queen's University, Belfast.

DEPARTMENT FOR ENVIRONMENT, FOOD AND RURAL AFFAIRS (DEFRA), 2004. Source publication: e-Digest of Environmental Statistics, Published February 2004 Department for Environment, Food and Rural Affairs http://www.defra.gov.uk/environment/statistics/index.htm.

FREEZE, R. A. AND CHERRY, J. A, 1979. Groundwater. Prentice Hall, Inc.

McCABE, A.M., KNIGHT, J. and McCARRON, S.G., 1999. Ice-flow stages and glacial bedforms in north central Ireland: a record of rapid environmental change during the last glacial termination. Journal of the Geological Society, London, Vol. 156 pp. 63-72.

METEOROLOGICAL OFFICE, 2006. Historical Climate Data. HMSO, London.

PERRY, J. 1989. Transport And Road Research Laboratory Department Of Transport Research Report 199 A Survey Of Slope Condition On Motorway Earthworks In England And Wales.

GROUNDWATER IMPACTS FROM ENGINEERING PROJECTS

Martin Preene, Technical Development Director – Hydrogeology, Golder Associates (UK) Ltd., Golder House, Tadcaster Enterprise Park, Station Road, Tadcaster, North Yorkshire, LS24 9JF, UK

Martin_Preene@golder.com

ABSTRACT

A wide range of engineering works have the potential to cause detrimental impacts on the groundwater environment. Traditionally, the primary impacts that were of concern were the effects on groundwater levels and the derogation of groundwater sources as a result of dewatering abstractions. However, there is increasing recognition that there are risks of significant groundwater impacts even where dewatering pumping is not a major factor, for example if the project results in the creation of artificial barriers or pathways for groundwater flow. This paper summarises the major types of impact that can potentially occur when engineering projects interact with the groundwater regime.

INTRODUCTION

Water resource specialists such as hydrogeologists typically view groundwater as a 'good thing'. From their point of view, groundwater is a potential *resource* to be used. Traditionally, the primary use of water was for drinking water or industrial process use. In recent years other beneficial uses of groundwater have emerged, such as so-called ground source energy systems, where groundwater can be used as a source of heating and cooling for buildings (Banks, 2007).

In stark contrast, construction engineers involved in major below-ground engineering works (deep basements, tunnels, metro systems, etc) traditionally view groundwater as a 'bad thing'. On these projects groundwater is a *problem*. If an engineering project penetrates down to water-bearing strata this will require the use of groundwater control measures such as dewatering pumping and low permeability cut-off walls (Preene *et al.*, 2000).

Where engineering projects interact with the groundwater regime, there is the risk that detrimental impacts may result. Partly as a result of the requirements of the Water Framework Directive, there is currently increased interest in such impacts. For example, in the UK, the Environment Agency has recently developed guidelines on carrying out hydrogeological impact appraisals (HIA) for dewatering projects (Boak *et al.*, 2007). However, the Environment Agency methodology concentrates on the impacts resulting from groundwater abstraction. In reality, these impacts are only a sub-set of the groundwater impacts that can potentially result from engineering projects. This paper will summarise the major types of impact that can potentially occur when engineering projects interact with the groundwater regime.

POTENTIAL GROUNDWATER IMPACTS

The major potential groundwater impacts from civil engineering works were categorised by Preene and Brassington (2003), reproduced as Table 1. These impacts are grouped into five main categories:

1. Abstraction from aquifers.
2. Physical disturbance of aquifers creating pathways for groundwater flow.
3. Physical disturbance of aquifers creating barriers to groundwater flow.
4. Discharges to groundwaters.
5. Discharges to surface waters.

ABSTRACTION FROM AQUIFERS – TEMPORARY

Groundwater abstraction, in the form of temporary dewatering pumping can be used to allow construction below groundwater level. The methods available include pumping from sumps, wells or wellpoints (Preene *et al*., 2000).

A number of groundwater impacts may result. These include:

i. ground settlement
ii. depletion of groundwater-dependent features
iii. effects on water levels and water quality in the aquifer as a whole
iv. derogation of individual borehole or spring sources.

Ground Settlement

Ground settlement will occur whenever groundwater levels are lowered by abstraction. However, for the great majority of sites in Ireland, settlements from dewatering abstraction are so small that no distortion or damage is apparent in nearby buildings. Ground settlements large enough to cause consequential damage are most likely to occur at sites where significant thicknesses of soft peat and alluvial soils are present that are underlain by permeable strata that require dewatering.

Depletion of Groundwater Dependent Features

The degradation of groundwater groundwater-dependent features by groundwater lowering caused by abstraction for water supply is an issue that is widely recognised in water resource planning. However, Acreman *et al*. (2000) noted that, in some cases, degradation of the aquatic environment believed to be linked to groundwater abstraction may be due, at least in part, to other factors such as changes in land drainage, river channelisation and climate change.

For most construction projects, it is likely that many dewatering abstractions will be sufficiently short term and small in volume to avoid significant effects on groundwater-dependent surface features (unless they are immediately adjacent to the dewatering works). If significant impacts are predicted, possible mitigation measures include:

i. Installation of a groundwater cut-off barrier (although the cut-off wall may itself detrimentally affect groundwater flow).
ii. Artificial recharge of groundwater or surface water. The temperature, chemistry and sediment content of the water must be assessed to ensure this will not itself cause adverse impacts.

Effects on Water Levels and Water Quality in the Aquifer as a Whole

Only the largest and longest duration temporary dewatering systems are likely to have a significant effect on groundwater resources in an aquifer as a whole. Indeed, dewatering operations are often carried out in strata of low to moderate permeability, classified as non-aquifers in terms of their potential for supply, where the effects on groundwater supply are, by definition, minimal.

Concerns are also sometimes raised that prolonged dewatering abstractions may affect aquifer water quality by drawing in contaminated water from nearby sites. This includes: lateral migration of leachate contaminated plumes beneath non-engineered landfills; or, vertical downward migration of pollutants from near surface contamination from current or historic industrial activity. In such cases extensive datasets of baseline water quality are needed to allow the risk of the impact to be assessed. Numerical modelling of the dewatering system could be used at project design stage to specify location, depth, screen intervals, pumping regimes, etc. of dewatering boreholes to reduce the threat to aquifer water quality.

Derogation of Individual Borehole or Spring Sources

Any construction projects planned near public or private water supply boreholes have the potential to cause a reduction in yield associated with lowering of groundwater levels for the duration of dewatering works. These impacts may require mitigation in the form of replacement of lost yield with tanker or bottled water supplies or modification of the borehole or spring source itself.

ABSTRACTION FROM AQUIFERS – PERMANENT

It is not widely recognised that many structures and engineered features that extend below groundwater level involve some form of permanent drainage system. For basements and tunnels a pumping system may be involved, or for road and rail cuttings discharge may be by gravity if the topography allows. These drainage systems are effectively long-term abstractions.

These abstractions can cause the same types of groundwater impacts as for temporary abstractions. In reality, drainage systems for discrete structures of limited extent such as basements are unlikely to have a significant effect on groundwater levels apart from very locally. In contrast, more extensive structures such as tunnels, pipelines and deep road and rail cuttings with associated drainage may cause greater impacts. Their linear extent can allow them to intercept and discharge considerable groundwater flow. This can result in derogation of supply boreholes and depletion of springs which may be used for supply or which support groundwater dependent features. This impact can be mitigated by designing the structure to be watertight, without the need for groundwater drainage. If this cannot be done, replacement or upgraded water supplies may be required in the affected area, together with compensation flows to groundwater dependent features.

PHYSICAL DISTURBANCE OF AQUIFERS – PATHWAYS FOR GROUNDWATER FLOW

Engineering projects may inadvertently form permeable pathways along which groundwater may flow. Pathways may be temporary (such as investigation and dewatering boreholes) and can be sealed on completion. Other pathways could be formed by parts of the structure or works and may exist in perpetuity. Examples of permanent pathways include the granular bedding of pipelines (which may allow horizontal flow) or some types of piling or ground improvement processes (which can form vertical pathways). Open excavations such as road or rail cuttings may themselves form vertical pathways.

The consequential impacts of these pathways include (Figure 1):

1. Loss of yield if horizontal pathways act to divert water away from springs or supply boreholes.
2. Increased risk of aquifer pollution from land use or near-surface activities. This is of particular concern if the confining bed above an aquifer is punctured by the works, especially if the near-surface strata have been contaminated by historic or ongoing polluting activities.
3. Changes in groundwater quality if pathways are formed between different aquifer units. For example, poorly sealed investigation boreholes could allow mixing of fresh and more saline water in aquifers where groundwater quality is stratified, or polluted groundwater at shallow depth may be able to flow into deeper aquifers.
4. Uncontrolled flowing artesian discharges through inadequately sealed site investigation or dewatering boreholes.

Awareness of these impacts is important when designing boreholes. For example, all site investigation boreholes and dewatering boreholes must be adequately sealed on completion. Similarly, dewatering boreholes should ideally not be screened in more than one aquifer unit and should have grout seals at suitable levels to prevent the gravel pack acting as a pathway for vertical flow.
Deep structures such as shafts or basements should be designed to limit the potential for creation of vertical flow paths – for example by using raft foundations in preference to piles that would puncture

low permeability aquitard layers. If piling or ground improvement methods have to be used, methods should minimise the formation of vertical flow paths. Horizontal structures such as pipelines should have anti-seepage collars (known as 'stanks') at regular intervals along their route.

PHYSICAL DISTURBANCE OF AQUIFERS – BARRIERS TO GROUNDWATER FLOW

Closely spaced heavy-duty foundations may interrupt horizontal groundwater flow, causing a damming effect (Figure 2). Groundwater levels may rise on the upstream side of the structure, and be lowered on the downstream side. These effects may not be significant unless large structures fully penetrate significant aquifer horizons.

If these impacts are of concern the designer could consider using raft foundations or limiting the depth of piles or cut-off walls, to reduce aquifer penetration. Any continuous impermeable cut-off walls used for groundwater control during construction could be designed not to form permanent barriers to groundwater flow once construction is completed.

DISCHARGES TO GROUNDWATERS

Construction activities can create the potential for discharges to groundwaters, with the consequent risk of pollution and degradation of groundwater quality. The main sources of potentially polluting discharges are: leakages and spills of fuels and lubricants from plant and vehicles; run-off from operations such as concrete placement; and run-off of turbid surface water as a result of topsoil removal and excavation. Normally, the risk of polluting discharges can be reduced by the adoption of good practice, based on guidance from the environmental regulators for the site locality.

The risk of pollution is increased if pathways for groundwater flow are associated with the works. Often, open excavations form a ready pathway for inadvertent discharges to groundwater. Good site practice should include prohibiting refuelling of plant (and storage of fuels) in or near excavations. Surface water drainage should be arranged to reduce the risk of spills or run-off entering the excavation.

Structures with deep basements or below-ground spaces may also provide potential for discharges to groundwater in the longer term. If the structures are not watertight and penetrate confining beds over aquifers, leaks, spillages or surface water flooding may be able to percolate more freely into groundwater. Individually, such leakages may be small but their combined effect may lead to significant groundwater contamination.

DISCHARGES TO SURFACE WATERS

Groundwater flows from temporary dewatering or longer-term drainage must be disposed of. There may be detrimental impacts on the receiving water body, including:

i. Erosion of river banks or water courses by poorly arranged discharges. This can block or change flow as scoured material is re-deposited downstream. Impacts can be reduced by the use of gabion baskets, geotextile mattresses or straw bales to dissipate the energy of the water at the point of discharge.

ii. Suspended solids (clay, silt and sand sized particles) in the discharge water are a highly visible aesthetic problem, but are also harmful to aquatic plant, fish and insect life in surface waters. Any abstraction system should have adequate treatment to avoid suspended solids in the discharge water.

iii. Oil and petroleum products may appear in discharge water as a result of spills or leaks from plant, vehicles or storage areas. These are often light non-aqueous phase liquids (LNAPLs) and will not mix easily with water, appearing as floating films or layers on the surface of lagoons or watercourses and may be present in solution. Water may have to be passed through proprietary 'petrol interceptors'; collecting the oil products for separate disposal.

iv. Contaminated groundwater. When abstracting from or near a contaminated site, the discharge water may be contaminated. Unless discharged via sewers to a wastewater treatment works capable of dealing with the contaminants, the flow will require treatment prior to discharge. The scale of treatment can vary greatly. If the abstraction and discharge are to continue in the long term, the ongoing need for treatment can be a major constraint on the feasibility of a construction project.

MONITORING

Monitoring is an essential part of managing groundwater impacts from engineering projects. Typical parameters to be monitored could include:

i. Groundwater levels in wells and boreholes.
ii. Surface water levels in wetlands, streams, etc.
iii. Flow from springs and in associated watercourses.
iv. Water quality parameters at springs or boreholes, including the use of geophysical fluid logging in boreholes with stratified water quality.

The natural variability of the groundwater regime can make it difficult to establish baseline conditions against which to assess impacts such as changes in groundwater level. For major or sensitive projects it may be appropriate to install 'control' monitoring points, beyond the area influenced by the project.

Location of monitoring points should be determined by the conceptual model of the anticipated impacts. The majority of monitoring points should be located in aquifer units where impacts are expected. However, it is also prudent to carry out monitoring in aquifer units where no impacts are expected (e.g. horizons that are hydraulically isolated from the works by very low permeability strata).

CONCLUSION

Engineering projects have the potential to cause significant impacts on the groundwater environment. Provided the potential impacts are identified early enough in a project, mitigation measures and associated monitoring can often be adopted to control these impacts. However, it is important that the full range of impacts is addressed, not merely those impacts directly associated with dewatering pumping.

The principal groundwater impacts from engineering projects can be categorised as:

1. Abstraction from aquifers.
2. Physical disturbance of aquifers creating pathways for groundwater flow.
3. Physical disturbance of aquifers creating barriers to groundwater flow.
4. Discharges to groundwaters.
5. Discharges to surface waters.

The risk and significance of impacts on each site and project must be assessed individually, taking into account, for example, the nature of the works, the presence and vulnerability of aquifers, and the proximity and sensitivity of nearby water sources, etc.

REFERENCES

ACREMAN, M C, ADAMS, B, BIRCHALL, P and CONNORTON, B. (2000). Does groundwater abstraction cause degradation of rivers and wetlands? *Journal of the Chartered Institution of Water & Environmental Management*, 14, June, pp200–206.

BANKS, D (2007). Thermogeological assessment of open loop well doublet schemes – an analytical approach. *Groundwater Pressures and Opportunities*. Proceedings of the 27th Annual Conference, International Association of Hydrogeologists (Irish Group), Tullamore, 4.3–4.15.

BOAK, R, BELLIS, L, LOW, R, MITCHELL, R, HAYES, P, McKELVEY, P and NEALE, S (2007). *Hydrogeological Impact Appraisal for Dewatering Abstractions*. Science Report SC040020/SR1. Environment Agency, Bristol.

PREENE, M and BRASSINGTON, F C (2003). Potential groundwater impacts from civil engineering works. *Water and Environmental Management Journal* 17, No. 1, March, 59–64.

PREENE, M, ROBERTS, T O L, POWRIE, W and DYER, M R (2000). *Groundwater Control – Design and Practice*. Construction Industry Research and Information Association, CIRIA Report C515, London.

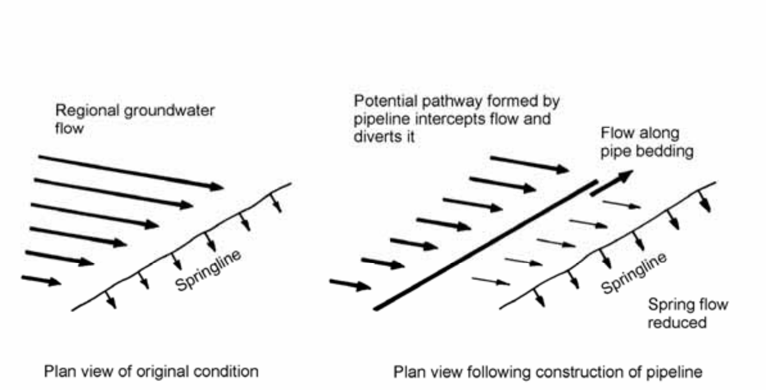


Figure 1: Pathways for groundwater flow

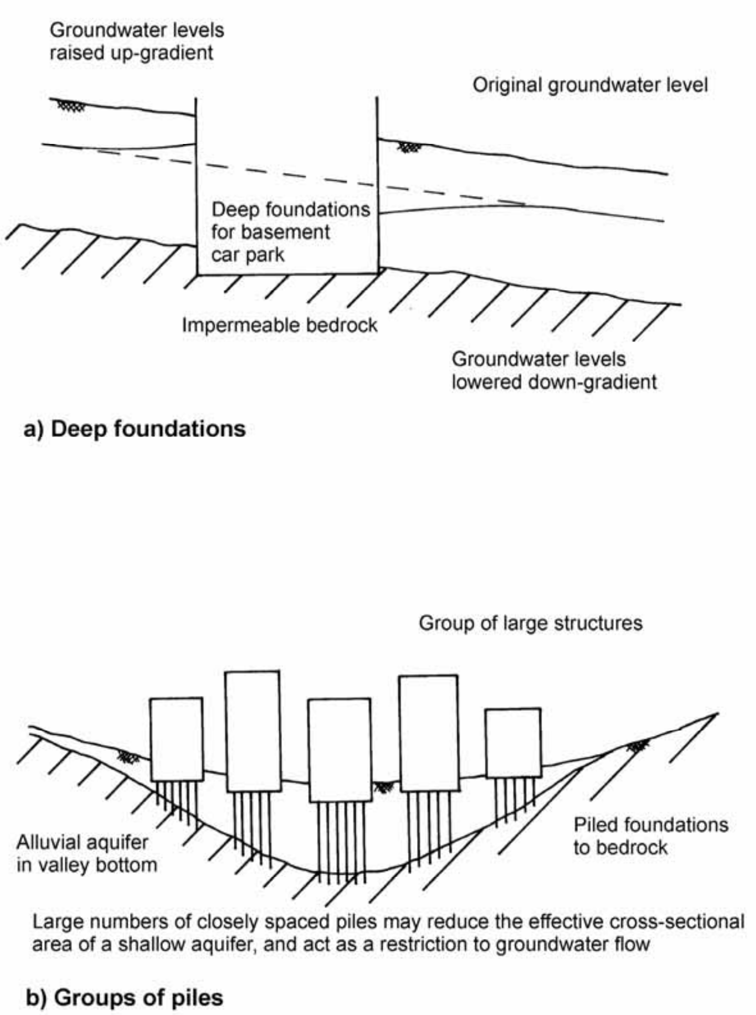


Figure 2: Barriers to groundwater flow

Table 1: Impacts on groundwater conditions from civil engineering works (from Preene and Brassington, 2003)

	Category	Potential impacts	Duration	Relevant construction activities
1	Abstraction	Ground settlement Derogation of individual sources Effect on aquifer – groundwater levels Effect on aquifer – groundwater quality Depletion of groundwater dependent features	Temporary	Dewatering of excavations and tunnels using wells, wellpoints and sumps Drainage of shallow excavations or waterlogged land by gravity flow
			Permanent	Permanent drainage of basements, tunnels, road and rail cuttings, both from pumping and from gravity flow
2	Pathways for groundwater flow	Risk of pollution from near surface activities Change in groundwater levels and quality	Temporary	Vertical pathways created by site investigation and dewatering boreholes, open excavations, trench drains, etc. Horizontal pathways created by trenches, tunnels and excavations
			Permanent	Vertical pathways created by inadequate backfilling and sealing of site investigation and dewatering boreholes and excavations and by permanent foundations, piles and ground improvement processes Horizontal pathways created by trenches, tunnels and excavations
3	Barriers to groundwater flow	Change in groundwater levels and quality	Temporary	Barriers created by temporary or removable physical cut-off walls such as sheet-piles or artificial ground freezing
			Permanent	Barriers created by permanent physical cut-off walls or groups of piles forming part of the foundation or structure or by linear constructions such as tunnels and pipelines Barriers created by reduction in aquifer hydraulic conductivity (e.g. by grouting or compaction)
4	Discharge to groundwaters	Discharge of polluting substances from construction activities	Temporary	Leakage and run-off from construction activities (e.g. fuelling of plant) Artificial recharge (if used as part of the dewatering works)
			Permanent	Leakage and run-off from permanent structures Discharge via drainage soakaways
5	Discharge to surface waters	Effect on surface waters due to discharge water chemistry, temperature or sediment load	Temporary	Discharge from dewatering systems
			Permanent	Discharge from permanent drainage systems

Session V

THE IMPORTANCE OF UNDERSTANDING RECHARGE WHEN UNDERTAKING GROUNDWATER RISK ASSESSMENT

[1] Shane HERLIHY, [2] Louise BURDEN

RPS Group, West Pier Business Campus, Dun Laoghaire, Co. Dublin, Ireland

e-mail: [1]shane.herlihy@rpsgroup.com; [2] louise.burden@rpsgroup.com

ABSTRACT

Groundwater recharge should be considered when developing conceptual site models (CSM) for contaminated land and environmental impact assessment projects. This paper presents three projects where an understanding of recharge was fundamental in assessing the risks posed to groundwater from proposed development projects and when developing a contaminated land remediation strategy. Time series groundwater level monitoring can validate understanding of recharge in CSM and in some cases indicate that revision of the CSM is required.

1.0 INTRODUCTION

In order to assess risks to groundwater a thorough understanding is required of the hydrogeological regime in question, fundamental to this is an understanding of not only what the receptors are but also understanding how recharge affects the overall hydrogeological system. A thorough understanding of recharge has been recognised in the Water Framework Directive (2000/60/EC) as being inherent to the process of groundwater resource protection through the characterisation and management of River Basin Districts. A combination of techniques should be employed whenever possible to assess recharge and can include water balance calculations, subsoil characterisation and mapping, surface water baseflow measurement and groundwater hydrographs. Recharge can have a positive or negative influence when considering groundwater risk and this paper outlines three practical case studies involving development and contaminated land projects in which an understanding of recharge was fundamental in assessing risks to groundwater.

The Water Framework Directive as implemented by the characterisation of River Basin Districts considers risks to groundwater from pollution pressures (groundwater quality) and abstraction pressures (groundwater quantity). In addition the Water Framework Directive and the RBD management schemes recognise the importance of groundwater and surface water dependent ecosystems which have the potential to be directly or indirectly affected by changes in the hydrological and hyrogeological regime of a particular catchment.

The importance of recharge in River Basin Management is discussed by Misstear in a paper for the National Hydrology Seminar 2000, which states that *"Reliable estimates of groundwater recharge are needed for a number of reasons including:*

- *Quantifying groundwater resources within River Basin Districts;*
- *Issuing of Abstraction Licences;*
- *Assessing the groundwater contributions to rivers (base flow) and to sensitive wetland habitats and hence the protection of these resources;*
- *Assessing groundwater vulnerability (high recharge implies high vulnerability);*
- *Delineating Source Protection Areas around major wells and springs (the size of the zone of contribution depends on the recharge)*

- *Delineating Nitrate vulnerable zones*
- *Identifying implications of changes in landuse and/or climate on water resources."*

Misstear (2000) acknowledges the difficulty of reliably estimating recharge and briefly reviews a number of methods for the calculation of recharge, in particular:

- Inflow estimation, including soil moisture budgets, infiltration coefficients, soil moisture flux approaches, hypsometers, tracers and direct observations;
- Aquifer Response analysis;
- Outflow estimation, in particular stream base flow analysis;
- Catchment Water Balance and Modelling.

Fitzsimons and Misstear (2006) examine the influence of the geological properties of tills in estimating recharge coefficients for fractured bedrock aquifers in Ireland. A literature review of recharge rates through tills indicated a range in recharge coefficients of 60% to 90% for thinner more permeable tills and 4% to 30% for thicker less permeable tills. A sensitivity analysis was undertaken using soil moisture budgeting techniques and modelling of vertical flow through tills and indicated that the recharge estimate is more sensitive to changes in geological properties of till than the soil moisture budgeting parameters. In particular, varying vertical hydraulic conductivity and thickness of tills were found to produce ranges of recharge estimates similar to those in the literature. The influence of scale of study was also noted, where till thickness and vertical conductivity varies across a catchment it is possible for effective precipitation in areas of thick low permeability till to run-off overland and infiltrate as indirect recharge in areas of thinner more permeable tills thus indicating the importance of a thorough conceptual understanding of geological conditions and how they may influence recharge at a particular site.

As a result of the sensitivity analysis carried out by Fitzsimmons and Misstear (2006), the Water Framework Directive Groundwater Working Group on Groundwater (2005) has derived a range of recharge coefficients for a selection of generically defined subsoil conditions.

This paper presents three case studies where the understanding of recharge conditions has been fundamental in assessing the risks posed to groundwater from a range of developments and from an area of contaminated land.

2.0 CASE STUDY 1. RECHARGE AS A PATHWAY: CONFIDENTIAL MANUFACTURING SITE

The subject site has been in operation for over 30 years and historic waste disposal practices at the facility resulted in the landfilling of construction and chemical process waste into dug pits 25-30 years ago. The groundwater contaminants of concern are toluene, dichloromethane (DCM) and tetrahydrofuran (THF).

The waste disposal area lies within a river valley with higher ground to the west and north. The geology beneath the site comprises low permeability glacial till and alluvial clays, which vary in thickness, overlying limestone bedrock. Groundwater flow within the limestone occurs predominantly within an upper weathered horizon of the limestone although some flow also occurs within deeper fractures. The groundwater flow direction within the weathered limestone unit generally follows the topography across the facility and discharges to the adjacent river.

Although the presence of low permeability glacial till provides some protection to the underlying limestone, this layer has been breached in places during excavation and early partial remediation attempts. Partial removal of the waste was undertaken in the mid 1990's and the site has been covered with a gravely clay fill of varying thickness. Extensive investigations within the waste body, including trial pits and boreholes, have shown that the waste body is largely dry with localised pockets of groundwater perched on lower permeability horizons within the waste body. Groundwater contamination has been observed within the underlying limestone aquifer unit. In order to assess the risks to the river and design a suitable remedial strategy for the site, it was necessary to have a thorough understanding of the hydrogelogical regime beneath the site and the pathways for contaminant migration in groundwater.

An extensive groundwater monitoring network has been developed on the site over the course of several years which has enabled a thorough understanding of lateral groundwater flow direction and contaminant fate and transport. Risk to the river receptor was concluded to be negligible based on this information however a source removal (dig and dump) remedial option was previously recommended to the client by other advisors.

Re-examination of the information and collection of additional hydrogeological site investigation data in the disposal area focusing on continuous groundwater level monitoring, vertical groundwater flow and seasonal effects demonstrated that a more sustainable remedial solution could be obtained. The hydrogeological regime beneath the disposal area is separated from the rest of the facility, to the north, by the presence of a zone of higher permeability to the north which effectively acts as a groundwater divide separating the higher ground, and steeper hydraulic gradients in the north from the low lying ground and much flatter gradients in the south beneath the disposal area. A topographic divide is present to the west, and the river is present to the south and east thus recharge from rainfall is the only input to the groundwater system immediately beneath the disposal area. Water balance calculations and groundwater throughflow calculations were consistent and used to validate this conceptual model.

The results of the groundwater monitoring have shown that groundwater within the limestone is partially confined by the overlying clay deposits. Groundwater levels within the waste body vary but are generally elevated above the piezometric surface in the limestone by 1.5 m. A hydrograph compiled from the automated monitoring data is presented as **Figure 1** and shows a clear response to rainfall within both the waste body and the underlying limestone. Infiltrating rainfall is therefore acting as a pathway for contaminant migration, with the source as the waste material and the receptors as the underlying groundwater and ultimately the river.

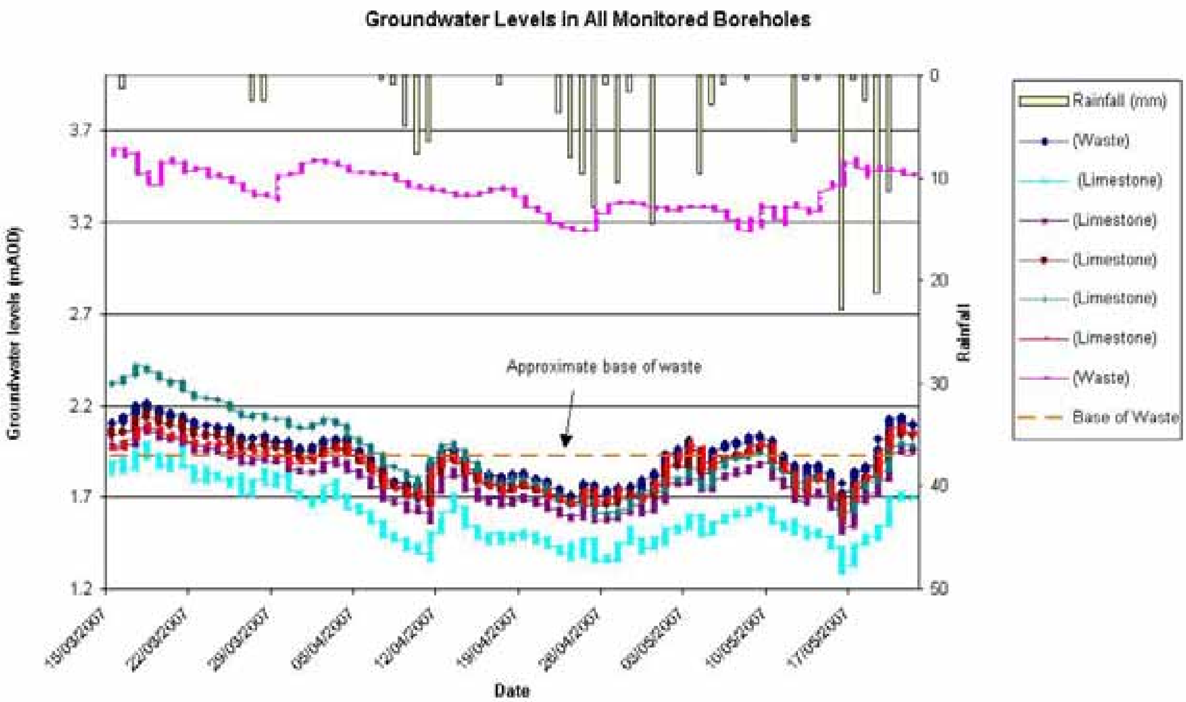


Figure 1 Hydrograph showing response to rainfall within waste body and underlying limestone

By placing a low permeability engineered cap across the landfill cell, the infiltration of rainwater through the waste body will be prevented and the pathway for contaminant migration effectively removed thus breaking the source-pathway-receptor linkage and removing the risk to underlying groundwater. Groundwater levels in the limestone will also be lowered by the reduction in recharge. Groundwater monitoring down gradient of the site has demonstrated a significant potential for natural attenuation, due to the aerobic nature of the groundwater and the low hydraulic gradient leading to high residence times. As a precautionary measure, groundwater is currently intercepted by a network of hydraulic containment wells before discharging to the river.

The case study highlights how a detailed understanding of recharge and vertical groundwater movement correlated with groundwater level monitoring can facilitate a more sustainable contaminated land remediation solution.

3.0 CASE STUDY 2. RECHARGE AS AN INDICATION OF NATURAL PROTECTION: PROPOSED FINGAL LANDFILL, COUNTY DUBLIN

The site for the proposed Fingal Landfill is located in North County Dublin, approximately 20 km north of the city centre and is bounded to the east by the M1 motorway. The site is located on the east facing slope of a valley with higher ground to the northwest and east.

Following an initial site selection study, an Environmental Impact Statement (EIS) was undertaken to assess the potential impacts of the development of an engineered landfill on the receiving environment including the underlying locally important bedrock aquifer. As part of the EIS, an extensive hydrogeological investigation was undertaken to establish the groundwater flow regime beneath the site and to identify potential groundwater receptors.

The investigations showed that the locality is underlain by Glacial Deposits comprising low permeability clays underlain by localised deposits of sand and gravel. The clay varies in thickness across the study area from 3 m to 27 m and measured permeability ranges from 10^{-6} m/s to 10^{-11} m/s. The bedrock beneath the site comprises fractured siltstone, mudstone and limestone of the Lucan, Naul and Loughshinny Formations. The bedrock units are classified by the Geological Survey of Ireland (GSI) as a Locally Important bedrock aquifer. The vulnerability classification of the bedrock aquifer, based on the GSI scheme (DoELG, EPA, GSI, 1999) ranges from high to low across the study area depending upon the thickness of the clays.

The piezometric surface within the bedrock aquifer generally reflects the topography and artesian conditions are locally present. Beneath the study area the groundwater flow within the bedrock is towards the south-east towards a north-south trending fault zone which runs approximately parallel to the M1 motorway. To the north is the Bog of the Ring public water abstraction. A groundwater divide is present to the north of the site which effectively separates the flow beneath the proposed landfill and the zone of contribution of the public water supply. Shallow perched groundwater is locally present within relatively higher permeability zones within the glacial deposits.

Monitoring wells were installed within the clay, sand and gravel and bedrock horizons across the site and surrounding area. Manual and automated monitoring of groundwater levels has been carried out at the site since June 2005 in order to understand the interactions between each of the horizons.

The study area is generally poor draining with the presence of the low permeability glacial clay impeding groundwater infiltration through subsoils. A study of surface water courses across the study area showed that flows were not sustained during dry periods indicating minimal base flow. To the north of the study area at Bog of the Ring the recharge coefficient has been calculated as 16% which equates to 57 mm/year (GSI, 2005).

The proposed landfill footprint has been located within the area with the greatest thickness of clays and therefore the lowest vulnerability rating. A qualitative analysis, using hydrographs, compiled from manual and automated monitoring data and using rainfall data from the meteorological station at Dublin Airport, was used to establish the variations in observed recharge across the study area.

Figures 2 and 3 shows hydrographs from different areas of the site demonstrating different responses to rainfall.

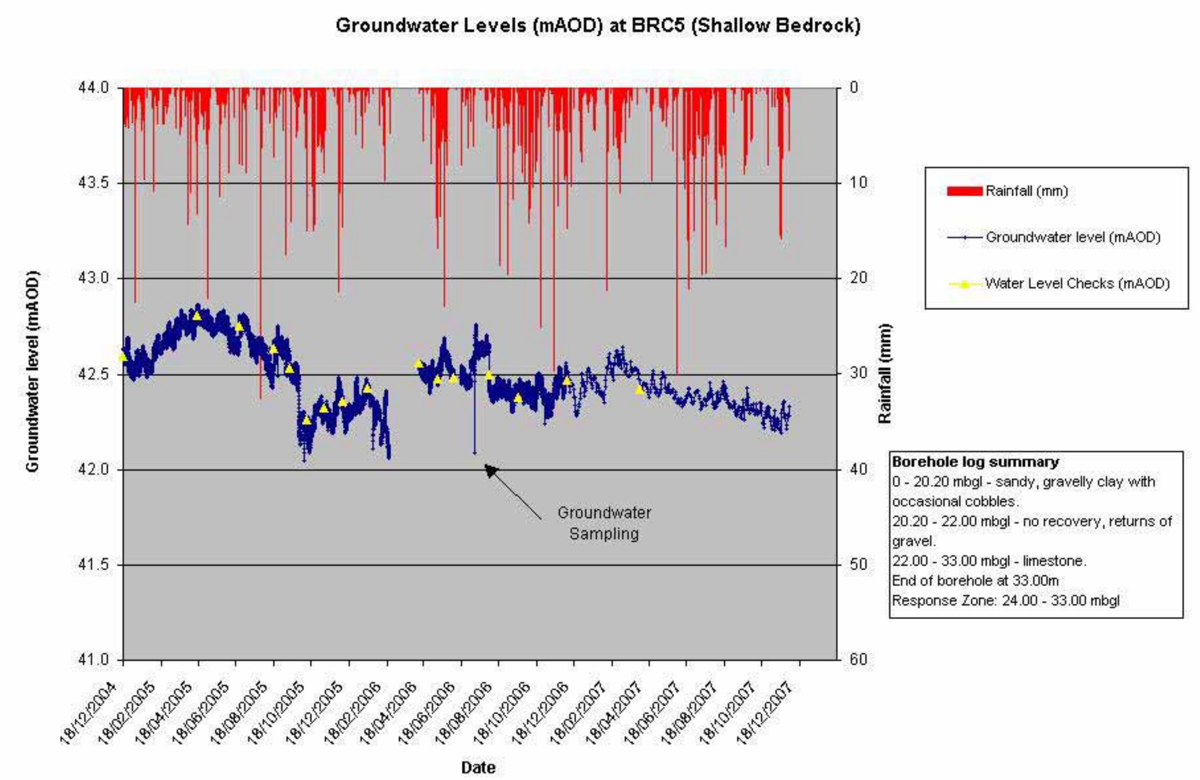


Figure 2 Hydrograph located within proposed landfill footprint showing low recharge conditions

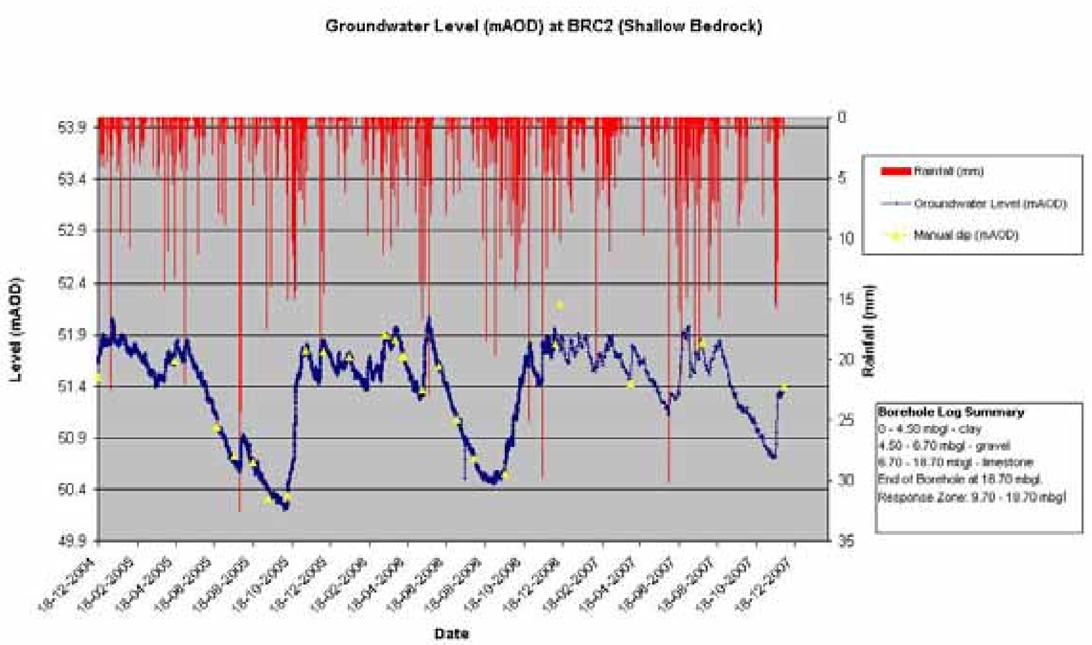


Figure 3 Hydrograph located outside proposed landfill footprint showing high recharge conditions

The hydrographs show that in areas with thicker clay (20 m thick), Figure 2, recharge to groundwater is minimal with the low permeability clays acting as a barrier to downward migration of infiltrating groundwater. Figure 3 from a well located in an area of thinner clay (4.5 m thick) illustrates a seasonal effect to recharge. By collecting detailed monitoring data over a prolonged period of time it has been possible to qualitatively assess recharge patterns across the site and confirm the proposed landfill footprint has been located within an area offering a high degree of natural protection and give added confidence to the Low Vulnerability designation beneath the proposed landfill footprint.

4.0 CASE STUDY 3. RECHARGE AS A RECEPTOR: NEW DUN LAOGHAIRE GOLF COURSE, COUNTY DUBLIN

This project was the subject of a paper on ‘Utilising Drainage Systems to Protect a Sensitive Groundwater Ecosystem’, (Herlihy *et al.*, 2007). The new Dun Laoghaire Golf Course is situated on a site of approximately 120 hectares in south County Dublin on the County Wicklow border.

The proposed development included changing the landuse from predominantly dairy agriculture to a 27-hole golf course with associated practice grounds, clubhouse, maintenance compound, storage reservoir and artificial lake features. The construction of drainage systems across the golf course to improve the drainage of the shallow soil and to improve the playability of the course during wetter conditions was a significant aspect of the overall development.

A Groundwater Dependent Ecosystem (GDE), the Ballyman Glen is located to the south of the site and comprises a steep incised valley with the Ballyman River flowing from east to west to the south of the study area. Within the Glen, groundwater seeps into the river causing ‘tuffa’ springs, a calcium carbonate precipitate, to form at some locations. A small area of alkaline fen is also present on the southern slope of the valley. A mosaic of habitats is present within the Glen area, of which the ecology of the ‘tuffa’ formation is particularly specialised being rare in both County Dublin and County Wicklow. The tuffa habitat is listed with priority status, on Annex 1 of the E.U. Habitats Directive (92/43/EEC). Consequently, Ballyman Glen is a proposed candidate Special Area of Conservation (pcSAC) under the Habitats Directive. The Glen is also proposed for designation as a Natural Heritage Area (pNHA) under the Irish Wildlife (Amendment) Act, 2000.

An Environmental Impact Assessment was completed to identify any potential impacts of the proposed development on the Glen which included a detailed hydrological and hydrogeological investigation.

The site is underlain by variable subsoils comprising brown sandy gravely silt in the north of the site and sand and gravel in the south of the site. The till overlies shale bedrock belonging to the Maulin formation. The bedrock is classified by the GSI as a locally important bedrock aquifer that is generally productive only in local zones.

River baseflow analysis (Figure 4) indicated that baseflow discharge to the Ballyman River amounts to approximately 55% of total flow across the catchment. Consequently, groundwater recharge was considered to be significant and potentially at risk from the development’s proposed drainage system. Subsequent groundwater level monitoring on the site confirmed that recharge was significant as illustrated in Figure 5.

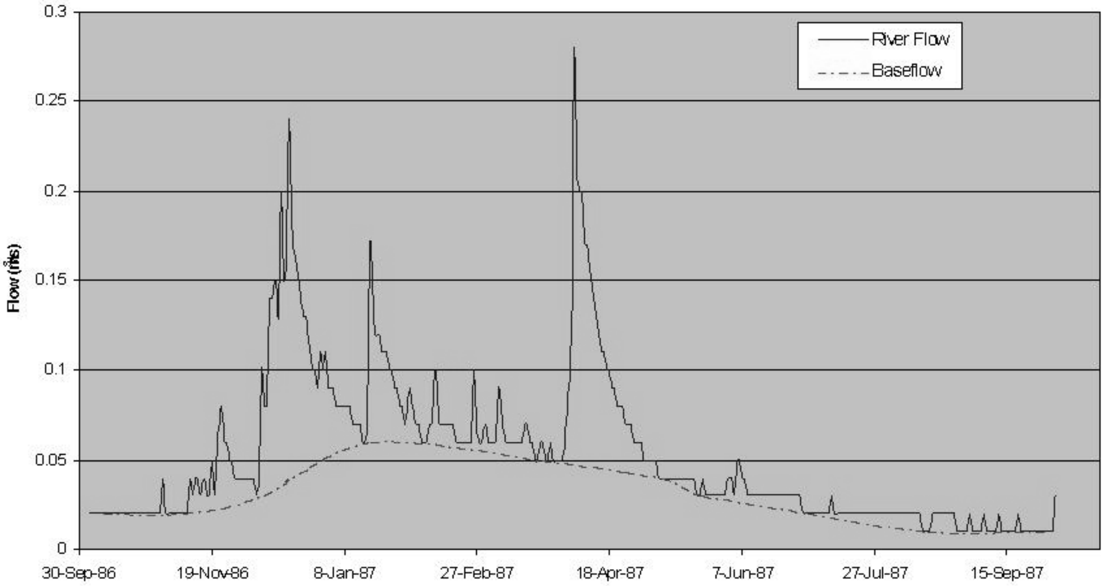


Figure 4 Ballyman River Baseflow Analysis

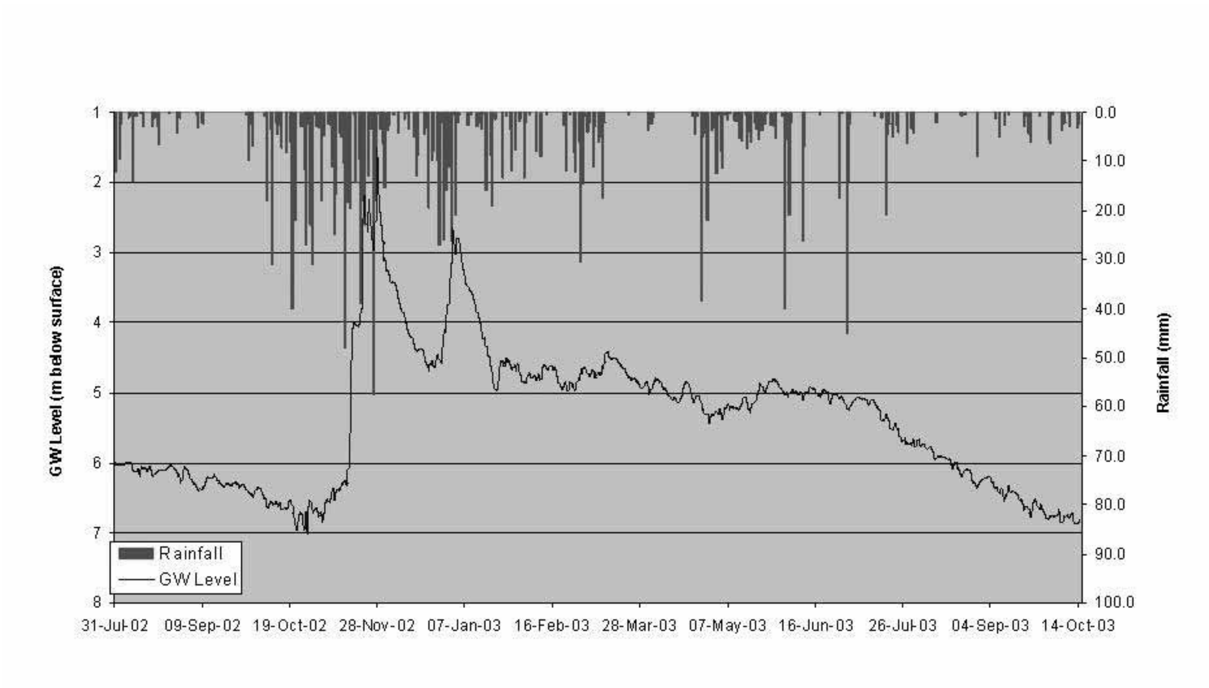


Figure 5 Aquifer Response to Recharge

The construction of hard standing areas, lined surface water features and a drainage system across the playing areas of the proposed golf club development had the potential to increase surface water runoff and decrease groundwater recharge thereby reducing base flow to the Ballyman Glen.

By understanding the hydrology and using a water balance for the catchment, the potential impacts of the golf course development and in particular the drainage system could be assessed by applying recharge coefficients to discrete areas of the development and calculating the overall effect on recharge.

The assessment concluded the development without mitigation would result in a 9% decrease in annual groundwater recharge. Due to the sensitivity of the Groundwater Dependent Ecosystem (GDE) in the Ballyman Glen, artificial recharge in the form of infiltration trenches and soakaways was included in the development to mitigate against the reduction of natural recharge.

5.0 CONCLUSIONS

The Water Framework Directive aims to protect both the quality and quantity of water resources. Understanding groundwater recharge is a fundamental prerequisite in achieving this objective, and Conceptual Site Models (CSM) should use a combination of techniques to assess its significance when determining risk to groundwater.

Land developments that significantly alter surface drainage characteristics can have potentially significant impacts upon sensitive receptors such as Groundwater Dependent Ecosystems. Changes to the overall water balance from such projects that reduce recharge will ultimately reduce groundwater discharge. In such cases, inclusion of artificial recharge and sustainable drainage systems to mitigate negative impacts should be incorporated into the overall design of the project.

Understanding the significance of recharge can aid in locating higher risk developments. Areas which exhibit low recharge potential offer a high degree of natural protection and long term groundwater level monitoring data can support conclusions regarding vulnerability classification.

Equally, recharge can act as a pathway for contaminants to enter groundwater and cause a risk to down gradient receptors. Sustainable remediation measures for historical contamination sources can be designed to specifically break the source, pathway receptor linkage by reducing and controlling recharge where this is the significant contaminant transport pathway into an aquifer.

Acknowledgements

The authors would like to acknowledge and thank the following:

Fingal County Council
Cosgrave Property Group
Anonymous client
Yvonne Cannon for review

References

Fitzsimons, V.P & Misstear, BDR. (2006) Estimating groundwater recharge through tills: a sensitivity analysis of soil moisture budgets and till properties in Ireland. *Hydrogeology Journal Volume. 14 Number 4*. April 2006

GSI (2005) Bog of the Ring Groundwater Source Protection Zones.

Herlihy, S. Cannon, C & Schluter, W. (2007) Utilising Sustainable Drainage Systems to Protect a Sensitive Groundwater Dependant Ecosystem: The Ballyman Glen. *IAH Congress Proceedings*, September 2007

Misstear, BDR. (2000) Groundwater Recharge Assessment: A key component of River Basin Management. *National Hydrology Seminar*, 2000.

WFD Working Group on Groundwater (2005) Guidance on the Assessment of the Impacts of Groundwater Abstractions, *Guidance Document no. GW5*. Approved March 2005.

OVERVIEW OF THE INTERFACE BETWEEN LAND USE PLANNING, DEVELOPMENT, AND HYDROGEOLOGY

Mary Cunneen

DPO: An Bord Pleanála[1]*,*

Dublin, Ireland

ABSTRACT

Two of the most common reasons for attachment of conditions to a planning permission by An Bord Pleanála are "in the interest of the proper planning and sustainable development of the area" and "to protect the environmental amenities of the area"; such reasons are commonly attached to conditions associated with extractive industry and landfill waste developments and recognise the importance of the protection of groundwater as a resource in itself, and as a link between environmental and residential amenities:

This paper traces the legal framework which facilitates sustainable development in terms of groundwater impacts, examines cases where such impacts are likely to occur in a significant manner, identifies deficiencies in the presentation of those impacts and makes a number of recommendations in that regard.

INTRODUCTION

The Boards mandate for consideration and assessment of hydrogeological issues in Section 37 appeal cases under the Planning & Development Act (2000) arises from (i) the concept of sustainability which informs Irish environmental policy and which underpins the National Sustainable Development Strategy of 1997; and (ii) the Board's requirement to have regard to the objectives of County Development Plans which must include the provision or facilitation of water supplies and which may include the regulation, promotion or control of the exploitation of natural resources – groundwater being generally regarded as such a resource. In pursuance of the latter objective a number of current Development Plans, as a result of the Water Framework Directive, include a policy to implement water quality management plans for protection of "inter alia" ground and surface waters.

The National Sustainable Development Strategy states that the sustainability concept "requires development to be within the capacity of the environment to support it without suffering lasting damage or depletion" and that where "there is uncertainty in regard to the limits or thresholds of the carrying capacity of the environment the precautionary principle must be applied". The Strategy identifies a number of strategic objectives for key environmental media – including "water resources". The Strategy acknowledges:

---

[1] An Bord Pleanála was established in 1977 under the Local Government (Planning and Development) Act, 1976 and is responsible for the determination of appeals and certain other matters under the Planning and Development Acts, 2000 to 2006, and with appeals under the Building Control Act, 1990, the Local Government (Water Pollution) Acts, 1977 and 1990 and the Air Pollution Act, 1987. Under continuity arrangements under the 2000 Planning Act, the Board continues to deal with appeals and other cases under the Local Government (Planning and Development) Acts 1963 to 1999 where such cases were initiated under those Acts. (Source www.pleanala.ie)

- That "a sustainable water policy must be based on protection, management and prudent use of water resources in the interests of optimised environmental quality and economic performance and efficiency".

- It is therefore an objective of the Strategy to protect and improve the quality of Irish water resources so as "*inter alia*" to:

    - Ensure that groundwaters may be used as required, as sources of drinking water supplies and for other beneficial uses;

    - Manage water resources effectively allowing beneficial development and use compatible with preservation of good quality.

REGULATION

The National Sustainable Development Strategy acknowledges that one of the approaches towards ensuring sustainable development is through regulation and in that regard the Strategy acknowledges the link between Irish environmental policy and legislation, particularly EU legislation. EU legislation in the form of Directives has generated and influenced much of Irish regulatory legislation which impacts on the hydrological and hydrogeological aspects of development. Chief among the Directives are:

- The Water Framework Directive (2000/60/EC) whose objectives as cited by the EPA are the protection of all high status waters, prevention of further deterioration of all waters and restoration of degraded surface and groundwaters to good status by 2015.

- The Groundwater Directive (80/68/EEC), as amended by the Groundwater Directive (2006/118/EC), which establishes a regime setting underground water quality standards and introduces measures to prevent or limit inputs of pollution into groundwater.

- The Nitrates Directive (91/676/EEC), which aims to reduce and prevent water pollution.

- The Urban Waste Water Treatment Directive (91/271/EEC) which aims to protect the environment from the adverse effect of discharges of urban waste water and waste water from certain industries.

- The IPPC Directive (96/61/EC) which applies to mainly industrial activities with a high pollution potential and which sets out measures designed to prevent or reduce air, water or ground pollution mainly by way of permit compliance.

- The 1975 Framework Directive on Waste, and amendments thereto, which have now been consolidated and codified under the new Waste Framework Directive (2006/12/EC), which aims to provide for the safe disposal of waste without significant adverse impacts on the surrounding environment.

The above Directives as transposed into National legislation give rise to applications that come within the remit of the Board other than under Section 37 of the 2000 Planning & Development Act e.g.:

- Under the 1992 Water Pollution Regulations, which implemented the 1977 and 1990 Water Pollution Acts, an appeal may be brought to the Board against the decision of a Local Authority to grant or refuse a licence for the discharge of trade or sewage effluent to any water including aquifers.

- Applications for Local Authority development under S.175 and 226 of the Planning and Development Act 2000.

- Applications to the Board under the Strategic Infrastructure Act 2006.

ENVIRONMENTAL IMPACT ASSESSMENT & THE BOARD

In the case of Section 37 appeals, the Directive which impacts probably most directly on the Board's function in the assessment and protection of groundwater is the EIA Directive of 1985 (85/337/EEC) as amended by Directives 97/11/EC and 2003/35/EC. The primary objective of those Directives is to ensure that projects, which are likely to have significant effects on the environment, are subject to an assessment of their likely impacts. The Directive has been transposed into National legislation under the Planning and Development Act 2000 and its associated regulations as well as the 1989 European Communities (EIA) Regulations and the 1999 Amendment thereof. The significance of those regulations lies in their attached schedules. These schedules to the regulations:

- Set out a list of development projects and threshold limits which are likely to have a significant effect on the environment and for which an EIS will be required; even if the development project falls below the threshold limit, but is likely to have a significant effect on the environment, an EIS will be required. From examination of the list of projects it is obvious that a number of these will have implications both directly and indirectly for groundwater.

- Refer specifically to the information required including "an estimate by type and quality of expected residues and emissions (including water, air, and soil pollution) …resulting from the operation of the proposed development", and a description of the aspects of the environment likely to be significantly affected by the proposed development including in particular "soil, water, air".

- Require the development location to be described in terms of the environmental sensitivity of the surrounding area and the schedule here refers specifically *inter alia* to "wetlands", coastal zones, nature reserves and parks, areas classified or protected under legislation and areas in which environmental quality standards laid down in the EU legislation have already been exceeded.

From the above it can be seen that the regulations not only deal with developments which are likely to have consequences per se in the type/quality of effluent discharges to groundwater but also to developments which while they may not be characterised by any significant effluent discharges themselves may affect the groundwater regimen of surrounding environmentally sensitive areas.

In regard to developments that require an EIA I wish to focus particularly on mineral/aggregates extraction development and landfill development and to identify a number of problems that have arisen there in regard to description and assessment of groundwater issues as set out in the EIS. In commenting on these cases I like to refer to the very useful and excellent Institute of Geologists of Ireland (IGI) Guidance on "Geology in Environmental Impact Statements" published in September 2002.

MINERAL & AGGREGATES EXTRACTION DEVELOPMENT

Mineral/aggregates extraction development presented to the Board in the majority of cases as quarry developments whose extraction area exceeds 5 ha. This category of development may be further subdivided into:

- Aggregates extraction which takes place above the water table.

- Aggregates extraction which takes place below the water table and involves groundwater pumping.

In the case of aggregates extraction which takes place above the water table major issues which arise are the exposure of groundwater to pollution usually via direct leachate of contaminants from ground level or indirectly via uncontrolled run off of contaminated surface waters. The Board in such cases seeks to pre-empt pollution of ground waters by measures such as the following:

(i) Prohibiting any quarrying below the water table.

(ii) Requiring a buffer layer to be provided between the lowest level of excavation works on site and the water table – usually a 1 metre deep layer.

(iii) Bunding

(iv) Provision of adequately sized and functionally efficient settlement lagoons which incorporate controlled discharge flows.

(v) Establishment of groundwater monitoring wells around the site boundary.

Informational deficiencies in regard to the above are largely characterised by failure of the developer to provide adequate baseline information, such as:

(i) Seasonal variations in the water table, usually because data has not been collected and collated over a sufficient period of time.

(ii) Inadequate no. of investigative monitoring points resulting in unsound conclusions being drawn as to the behaviour of groundwater below and in the vicinity of the site.

It is no surprise therefore that in a number of sand and gravel quarry decisions by the Board in 2006 analysis shows that potential contamination of groundwater is one of the more common reasons for refusal of permission.

Finally in the cases of recent S.261 cases involving sand and gravel extraction where the quarry operator has appealed a condition imposed by the Planning Authority requiring the quarry operator to undertake a groundwater monitoring programme which will inter alia identify the groundwater flow regime operating in the vicinity of the facility, the Board has, in the interests of environmental protection endorsed the attachment of that condition agreeing generally that "comprehensive protection measures for groundwater at the site can only be given their fullest expression when the groundwater flow regime operating below and within the vicinity of the site is identified" and that such identification will in turn facilitate the provision of a more comprehensive environmental management system at the site.

In regard to aggregate extraction developments below the water table common hydrogeological and associated hydrological informational deficiencies in the EIS may be identified as follows:

- Inadequate description of dewatering impacts due to failure to adequately explore and predict the extent of the resultant cone of depression and hence to assess the resultant consequences for groundwater associated features in the area e.g. natural conservation areas and wells.

- Inadequate data in regard to both time line and locational monitoring.

- Failure to address potentially cumulative groundwater impacts in regard to other development, unsound conclusions and hence

- Inadequate mitigation measures.

LANDFILL DEVELOPMENTS

In regard to engineered landfill developments for the disposal of waste, potential hydrogeological impacts as set out in the EIS focus largely on contamination by pollutants via leachate through the liner, via leachate from leachate lagoons and via uncontrolled contaminated and surface water run off; such a focus is appropriate particularly in view of the link between groundwater and surface water quality. Informational deficiencies in regard to such developments include:

(i) Inadequate clarification of the link between groundwater and surface water features and sites of ecological importance dependant on groundwater feed especially those which have been designated as worthy of protection.

(ii) Inadequate investigation of the groundwater regime below the site and in the surrounding area.

(iii) Inadequate description of remediation measures – usually restricted to stating that these will be put in place.

Assessment problems arising from the informational deficiencies in the EIS identified above are further compounded by a conflict between the conclusions of authoritative professionals e.g. between consultants engaged by the developer and those engaged by third parties. Often the conflict stems from variations in data input to models, variations in data weighting, variations in weighting of mitigation factors/proposals and even simply variations in the time line for which certain conclusions are deemed to be valid.

Consequences resulting from the provision of inadequate EIS hydrogeological information by the developer are:

- A request by An Bord Pleanála for additional information or clarification of same; on receipt of that information, the Board is required to circulate it to interested parties for comment; that results in a delay in assessing the development and a decision being arrived at.

- Undermining of other sections of the EIS and more intensive scrutiny of those sections, with potentially attendant consequences.

- In those areas where adequate information or clarification of emissions to groundwater arising from the operation of the activity is not satisfactorily provided the Board may, where a Waste Licence or an IPPC licence is required for that development, have no alternative but to refuse permission on the basis that the development is environmentally unacceptable – the Board being prohibited from granting permission for a development with attached conditions controlling emissions from the operation of that activity. (Cf. Sections 256 and 257 of the Planning and Development Act 2000). That scenario has arisen on a number of occasions despite a Waste Licence having been granted prior to the Board's decision or subsequently. The Board's decision however may be referenced to the fact that the objectives and range of issues which it takes into consideration in coming to its decision may differ from those which inform the EPA's assessment of the same development i.e. the role of the planning system is to assess whether the development itself is an acceptable use of the land rather than assessing the control of the processes or substances themselves; in fact any assessment by the Board assumes that the pollution control regime as approved by the EPA will operate efficiently and effectively.

IMPROVING EIS INFORMATION

The range and quality of information provided in an EIS can be improved through ensuring, *inter alia*:

- That Desk studies are not unduly relied on or supplant site investigative work.
- That the degree of investigative work undertaken both on site and in its vicinity can provide comprehensive and qualitative data over a sufficient period of time to justify any conclusions reached in regard to impacts; in that regard therefore additional sampling and mo0nitoring proposed to be undertaken prior to commencement of development should be unnecessary.
- That mitigation and remediation measures should be assessed in terms of degree of success – it is unrealistic to assume that a 100% success rate is achievable always and everywhere.
- That all significant impacts be described – positive as well as negative.

CONCLUSIONS

In conclusion, it must be stated that:

- Groundwater is one of our most important natural resources, being a building block in our natural environment and part of the link between our aquatic and terrestrial ecosystems.
- That the legacy of groundwater contamination can still be seen and experienced today and that in the interests of proper planning and sustainable development it behoves us to ensure that the legacy does not become an enduring scenario.
- That development which has potential hydrogeological impacts must be robustly assessed in order to prevent and control groundwater pollution.
- That the Board will, where it deems necessary to do so, engage consultative advice in order to provide robust objective assessments of hydrogeological impacts, particularly in the case of competing authoritative opinions.
- That the single greatest factor in facilitating an assessment of development impacts on groundwater is the quality of data presented in terms of adequacy and accuracy.
- That while time constraints may be instrumental in governing the quality of data presented, inadequate investigation work in regard to the site itself and its surrounding environment is a false economy which can delay a final decision by the Board pending the submission and analysis of further information.

THE IMPORTANCE OF HYDROGEOLOGY INFORMATION IN PLANNING APPLICATIONS AND APPEALS A BRIEF OVERVIEW WITH PARTICULAR REFERENCE TO QUARRIES

David M. Ball

Hydrogeologist

28 New Bride Street, Dublin 8

This contribution is a short comment on the role of hydrogeology and hydrogeologists in relation to Planning Applications EIS documents and Appeals. It is an adjunct to Mary Cunneen's paper. Mary is a full time Senior Inspector in An Bord Pleanala, whereas I, and other hydrogeologists, are consultants to the Board of An Bord Pleanala. We are issued with, what is called, a warrant that gives us the authority to go onto lands that are the subject of an appeal, and also conduct oral hearings, if the Board decide it is appropriate. We don't see all appeals that contain a hydrogeology element. The hydrogeology component in some of them is short, straightforward and easily understood by someone outside our discipline.

From my experience, we seem to get the appeals that are complex, or where the hydrogeology component is contentious or unclear. My comments are addressed primarily to the next generation of hydrogeologists; those of you in your twenties and thirties. I see you as the generation that can change things for the better. The better for your clients and the better for you and your careers. I have listened to many of you over the years express dissatisfaction with the way that you are asked to churn out EIS documents almost to a formula, without being able to gain a real understanding of the hydrogeology. I suggest that if you take heed you will appreciate the need to do better work, gain a sense of satisfaction in your work, and probably prevent your EIS and an appeal file ending up on my desk.

1. EIS Preparation and Appeals

Judging by what I have seen there is an insidious trend to produce, what I term, 'Token' EIS documents. These are EIS's that ticks all the boxes in terms of the topics and areas listed in the EPA Guidelines for an EIS. Therefore, they cannot be faulted for a major omission. The strategy appears to be focussed on obtaining planning permission, rather than actually producing an objective environmental impact statement for your client (the developer), the planning system and the public. The underlying strategy appears to be that a rather shallow superficial document is submitted to the County Council or planning authority. The County Council points out the areas that need further work, and draw up a request for additional information. In effect, the consultants are not preparing a full coherent EIS but are asking the County Council to be prescriptive and define the areas where work is required. In essence, the consultants are asking the County Council to take responsibility for scoping the EIS and setting the target to be achieved in order to obtain planning approval. The County Council's request for additional information can then be used to justify the further work and the further fees for the consultants. This strategy means that the consultant's staff do not get the opportunity to really get to grips with the hydrogeology and the onus is placed on the County Council to define the main issues for the EIS. This may seem to be a very 'smart move' by both the consultants and the developer, but it is short sighted. The County Council may not have the staff capable of assessing all the components in the EIS, or if they do, they may not have the time to do a

thorough assessment. Eventually, it is likely that after a period of gathering and submitting further information, and further discussions with the planning authority, permission is granted with a series of conditions. Sometimes these conditions are seen as too onerous and the client appeals the conditions to An Bord Pleanala, but usually the permission is appealed by third parties.
The appeals reach An Bord Pleanala, and provided that they are received by the due date and with the requisite fees, the appeal process begins. The inspector for the appeal assesses all the information 'de novo' (starting from the beginning). The inspector may submit an interim report containing a recommendation to the Board to request additional information. Again, the work required to do the job properly is being defined by an outside agency, the Board and its inspectors and consultants, rather than the original consultants. The request for additional information is sometimes interpreted as a willingness to grant permission. This can be a misinterpretation. Sometimes a major flaw in the original EIS and is revealed in full by the further information. The inspector's assessment of the new information may make it very clear that the board should refuse permission for the development. Therefore, the outcome of a process that started with a token, superficial EIS, ends two or perhaps three years later, with the Board refusing permission for the development. This outcome is very unsatisfactory for the developer and for the consultants employed by the developer, and usually the Board is blamed. However, what is often lost in this process and the recriminations at the end, is the responsibility of the firm of consultants to their employer (the developer), who probably paid a lot of money for consultants work, and to their staff. The consultants are paid by the developer and therefore have a responsibility to objectively inform the developer of the full potential environmental impact of the development and the long term consequences for the developer, and not just tell the developer what they want to hear.

2. Quarries

It is interesting to observe changes in the location and development of new quarries and extensions to existing quarries.

Quarry operators want rock, they don't want water. Developers of a new quarry usually carry out drilling programmes to prove the rock resources. They seldom site, design and drill boreholes in order to locate and investigate groundwater.

The water table is generally deep under the summits of mountains or the crests of ridges. Therefore, in the past it was attractive to quarry operators to develop sites in these locations particularly if the highland area is near a major market. For example, there are many quarries in the Dublin mountains. However, it is no longer possible to remove or damage the top of mountains or hills. It would be visually intrusive and there would be many appeals regarding loss of amenity and or damage to upland habitats. Quarry operators have had to come off the tops and down the valley sides. Therefore, new quarries are more likely to encounter the water table.

The recent spread of 'one-off' rural housing along the lanes and minor roads means that it is more difficult to find a quarry site that is away from housing on the lower slopes of our hills. Available sites are becoming smaller.

Quarry operators need to maximise the amount of rock they can get from these smaller sites, that are difficult to find, and expensive to develop. Therefore, they need to go deeper. As a result they have to go below the water table, and they have to de-water. Many quarries are in limestone, therefore there will be karst features. It is important to recognise that almost all limestones have been subject to karst weathering. Karst features are the norm. Karst may be restricted to relatively recent epikarst, but the real concern is deep palaeo-karst that will be encountered in a deep modern quarry.

Exploration drilling to prove the rock resources for a new quarry are not aimed at finding water bearing conduits. Many quarry operators do not understand or agree to an expensive and systematic drilling and testing programme to locate groundwater in their potential new site. A common fundamental weakness in the hydrogeology component of many quarry EISs has been the poor investigation of karst groundwater systems under, through and around the site. The unrealistic assessment of groundwater flow in karst in the EIS is a major flaw that leads to refusal of planning permission. But, it is also an abrogation of the consultant's responsibility to objectively inform the developer of the true risks, problems, costs of a thorough investigation and consequences of the proposed development. If a developer is informed of the real conditions under the site, then the developer has the opportunity to objectively assess the long term consequences and costs. Just focussing on getting over the hurdles to obtain planning permission and not carrying out a realistic assessment of the groundwater flow system is misleading, if not deceiving, the client, to whom the consultant is ultimately responsible. If the poor or inadequate hydrogeological assessment is finally revealed in the planning appeal process then the consultants client is often dismayed. I use these strong terms because, in several oral hearings, I have seen quarry operators lose confidence in their consultants as their information, evidence and work is exposed as seriously inadequate, or flawed.

I have several reasons for addressing my comments particularly at the younger members of my peer group. Older members of our profession should, by now, know how to do this work properly, whereas you are still learning. You are vulnerable to the pressures from your directors and managers who want you to do the minimum necessary to obtain the agreed fees in order to maximise the end of year profit centre figures. My comments are aimed at encouraging you to resist these pressures. There is only one way to do a job and that is to do it properly. We all make mistakes, but at least if you set off trying to do it properly, then you know that you have done your best. If you try to do it properly, you will also stretch yourselves and gain experience and satisfaction. If you don't do it properly, you will feel disillusioned with yourself, your managers and your company, and, of course, you will have let down your client and diminished your personal reputation.

I finish by giving some examples, where a superficial or inadequate EIS and further investigations, have ultimately lead to the failure to obtain planning permission. A common thread through these examples appears to be a reliance on information obtained from modern maps and web based GIS systems, rather than coherent hydrogeological investigations on the ground.

Information Sources

Many consultants seem to adhere to a view that anything produced and published by the OSI, GSI, EPA, Department of Environment, OPW etc is the last word on the subject. In other words, if the GSI publish an opinion (in the form of a map or report) on the geology or vulnerability or aquifer characteristics, then you need to do little further work. This attitude demonstrates either a naïve, or pragmatic, lack of understanding of the work of the GSI or other state agencies. The GSI and others publish information on the basis of the available staff and the available information, but usually the interpretation is not specific to your site or your proposed development. As I showed in a paper last year, there are many places where my detailed investigations reveal that the bedrock geology description and boundaries are not correct. This does not mean that the GSI has made a mistake. The GSI could not know otherwise, because no one before had investigated the area in detail. The examples also show that serious errors arise when the OSI 1:50,000 scale Discovery Series maps are used as the base maps for hydrogeological and hydrological investigations. I would strongly advise consultants, hydrogeologists, engineers and planners never to accept the drainage shown on these maps at face value. It is essential, throughout the country to refer to older maps, and, most emphatically, carry out good, basic, investigative field work. "If there is no river shown in a valley, ask yourself why, and then go and look for it." I draw attention to the fact that many state agencies are representing their freely available GIS information as overlays on these 1:50,000 scale maps. For example the EPA Envision overlays showing water features, omit many first order streams. These streams can be very significant in understanding water resources and surface water groundwater flow systems in karst limestone lowland areas. There are also major turloughs that are not shown or labelled. However to be fair to the EPA they have corrected the OSI omissions in some areas. The presentation of information gathered from on-line GIS systems in an EIS, without further investigation, is interpreted as a sign that the consultants compiling the EIS do not know what they are doing. They have not attempted to fully understand the basic hydrology and hydrogeology, and therefore cannot credibly present an assessment or understanding of the environmental impacts of the proposed development.

Conclusion

Token EISs for any major development, with no solid, coherent understanding of water resources, are unlikely to gain planning permission when they get to An Bord Pleanala.