INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS
(IRISH GROUP)

Presents

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SUDS and Groundwater Monitoring

PROCEEDINGS

OF THE

IAH (IRISH GROUP)

26th ANNUAL GROUNDWATER CONFERENCE

Tullamore Court Hotel, Co. Offaly

Tuesday 25th and Wednesday 26th April 2006

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TULLAMORE
Co. OFFALY

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The IAH (Irish Group) are grateful to White Young Green Ireland for providing assistance with the registration aspects of the conference

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International Association of Hydrogeologists (Irish Group)

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 2006 IAH (Irish Group) Committee:

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CONFERENCE OBJECTIVE

The conference will be of great value to local authority engineers, consultants, planning officials, environmental scientists, public health officials and hydrogeologists.

This year we have the benefit of several, very experienced key note speakers from the USA, Britain and Ireland to inform us of new techniques, perspectives and results that will broaden and stretch our understanding of the two main topics. We will have important lectures on concepts, legislation, latest research, and case studies. We will learn of successes and failures so that we can learn from the experience of others.

SUDS, or Sustainable Urban Drainage Systems, is an important new concept, at least here in Ireland, that relates to planning, engineering, hydrology, road design and hydrogeology. It is becoming an integral part of all infrastructural development, in both urban and rural areas. Controlling and discharging surface water in a safe manner is becoming complex and expensive. In the US it is sometimes referred to as 'Low Impact Development' or 'Best Management Practice'. In Ireland, SUDS is being used in city planning, new housing on the periphery of villages, major roads schemes, industrial estates and business parks and is also extending into agriculture with the safe disposal of farmyard runoff. So far it has been commonly presented from a landscape design and surface water runoff perspective, but this has overlooked the value, significance and role of groundwater. The conference will deal with the whole concept and practice of SUDS but will also stress how to use and not abuse groundwater.

The second theme of the conference, Groundwater Monitoring, links to SUDS. Monitoring is an established topic, but the purpose of the conference is to alert the delegates to new Directives and forthcoming legislation that will require a new paradigm for groundwater monitoring. The conference will provide the conceptual foundations for this new monitoring regime relating to groundwater recharge (e.g. SUDS recharge), groundwater flow, groundwater chemistry and microbiology, and the critical assessment of monitoring boreholes, illustrated by means of practical experience here and outside Ireland.

"It doesn't just go away, you know…" SUDS and Groundwater Monitoring

PROGRAMME DAY I - TUESDAY, 25th APRIL

10:00 Registration
Tea, Coffee, & Exhibits

11:00 Welcome and Introduction
David Ball, President IAH (Irish Group)
Monitoring in Real Time – New Orleans 29th August 2005

SESSION I

11:15 SUDS - the Current US Perspective: Integrated Storm Water Management in New Development Areas

Robert Pitt, Cudworth Professor of Urban Water Systems Department of Civil, Construction, and Environmental Engineering, The University of Alabama, Tuscaloosa, USA

11:55 The Draft EU Directive on the Protection of Groundwater Against Pollution: an Assessment of the Implications

Donal Daly, Geological Survey of Ireland,
Matthew Craig, Environmental Protection Agency,
Garrett Kilroy, EPA Research Fellow, Trinity College, Dublin.

12:30 Panel Discussion

12.45 Buffet Lunch & Exhibits

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SESSION II

14:00 A UK and European Perspective of Sustainable Urban Drainage (SUDS) with Particular Reference to Infiltration Systems and Groundwater Pollution

J Bryan Ellis, Professor, Urban Pollution Research Centre, Middlesex University, UK

14:35 Proven and Practical Methods of Groundwater Recharge in Built-up Areas in the USA

Gary Mercer, Senior Water Resources Engineer, CDM, Boston, USA

15:10 Panel Discussion

15:30 Tea, Coffee & Exhibits

SESSION III

15:50 SUDS – Principles and Drivers
Pádraig Doyle, Principal Engineer, Dublin City Council

Groundwater Issues Arising from Sustainable Urban Drainage Policies
Fionnuala Collins, Senior Hydrogeologist, RPS Group, Dublin

16:30 Road runoff in Ireland – Implications for groundwater and management in a SUD System
Paul Johnston and Míchael Bruen, Senior Lecturers, Trinity College, Dublin and University College, Dublin

17:00 Constructed Wetlands – Hydraulic Design – Sustainable Drainage & Treatment Systems
Pamela Bartley, Bartley & óSúilleabháin Environmental Engineering, Galway, Paul Johnston & Laurence Gill, Trinity College, Dublin

17:30 Panel Discussion. (Close at 18:00)

The panel discussion will be followed by ***a wine reception*** *in the Tullamore Court Hotel sponsored by City Analysts Ltd. (Dublin and Limerick)*

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PROGRAMME DAY 2 - WEDNESDAY, 26th APRIL

SESSION IV

09.00 Groundwater Monitoring: the Importance of Setting Clear Monitoring Objectives Based on an Appreciation of the Hydrogeology
Bruce Misstear, Senior Lecturer, Dept. Civil and Environmental Engineering, Trinity College, Dublin, David Banks, Holymoor Consultancy, Chesterfield, UK

09.30 Groundwater Monitoring and Sampling: New research and the importance of borehole construction
Peter Dumble, Principal Hydrogeologist, Waterra (UK) Ltd.

10:00 Managing Large Datasets
David Mclorinan, Senior Hydrogeologist, White Young Green Ltd., Belfast

10:30 Coffee, Tea and Exhibits

SESSION V

10:50 Defining baseline quality for nitrogen compounds in groundwater flow systems
Mike Edmunds Professor, Centre for Water Research, Oxford University

11:25 Screening Methodology for the WFD Water Quality Monitoring Network
Matthew Craig, Environmental Protection Agency, Dublin, Henning Moe, CDM Ireland Ltd., Dublin, Natalya Hunter Williams, Geological Survey of Ireland, Dublin

11:45 Establishing Natural Background Levels for Groundwater in Ireland
Gerry Baker, Donal Crean & Sean Moran, O'Callaghan Moran & Associates, Cork

12:05 Detailed and integrated monitoring of the positive and negative impacts of large scale water abstractions and treated waste water discharges on habitats and ecosystems
Paul Ashley, Chief Hydrogeologist, Mott MacDonald, Consulting Engineers, Cambridge, UK

12:35 Panel Discussion
13:00 Closing Address
Patrick Laffly, TMS Environment Ltd
13:05 Buffet Lunch
14:30 Sponsored Golf Outing

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Session I

SUDS - THE CURRENT US PERSPECTIVE: INTEGRATED STORM WATER MANAGEMENT IN NEW DEVELOPMENT AREAS

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Department of Civil, Construction, and Environmental Engineering
The University of Alabama, Tuscaloosa, AL 35487 USA

ABSTRACT

There are many stormwater control practices available to address an expanding list of surface and groundwater protection objectives. There is an emerging trend to use combinations of individual stormwater devices and approaches to better reduce the wide variety of problems that occur with urbanization. These combinations use complementary unit processes in order to remove both particulate and dissolved forms of pollutants, and to manage the complex urban hydrological cycle. These combinations of unit processes, termed treatment trains, can be applied at individual controls and throughout a developed site. This paper describes two such treatment trains, one that can be used at a critical source area, and another example using different complementary controls throughout a newly developing industrial site.

INTRODUCTION

Many urban runoff control practices are available. These include infiltration devices (such as subsurface infiltration trenches, surface percolation areas, and porous pavements), sedimentation devices (such as wet detention ponds), public works practices (such as grass drainage swales, street cleaning, and catchbasin cleaning), critical source area controls (media filters, chemical treatment, etc.). Many of these devices can be located at source areas and/or at outfalls. In most situations, combinations are needed to meet the broad needs of a comprehensive stormwater management program and receiving water objectives (Burton and Pitt 2002).

There are therefore many stormwater control options, but all are not suitable for every situation. It is important to understand which controls are suitable for the specific site conditions and can also achieve the required goals. This will assist in the realistic evaluation for each practice considering the technical feasibility, implementation costs, long-term maintenance requirements, and life-cycle costs. The most promising and best understood stormwater control practices are wet detention ponds. Less reliable in terms of predicting performance, but showing promise, are stormwater filters, treatment wetlands, and biofiltration devices.

An interesting study examined 11 types of stormwater quality and quantity control practices that were used in Prince George's County, Maryland (Shepp and Cole 1992). They concluded that several types of stormwater control practices had either commonly failed or were not performing as well as intended. Generally, wet ponds, treatment wetlands, sand filters, and infiltration trenches achieved moderate to high levels of removal for both particulate and soluble pollutants. However, only wet ponds and treatment wetlands demonstrated an ability to adequately function without frequent maintenance. Control practices which were found to perform poorly were infiltration basins, porous pavements, grass filters, small "pocket" wetlands, extended detention dry ponds, and oil/grit separators. Early designs of infiltration stormwater controls had high failure rates which could often be attributed to poor initial site selection and/or lack of proper maintenance. The poor performance of some of the controls was likely a function of poor design, improper installation, inadequate maintenance, and/or unsuitable placement of the control. Greater attention to these details would probably reduce the failure rate of these practices. The wet ponds and treatment wetlands were much more robust and functioned adequately under a wider range of marginal conditions. Other important design considerations include: safety for maintenance access and operations, hazards to the general public (e.g., drowning) or nuisance (e.g., mosquito breeding), acceptance by the public (e.g., enhance area aesthetics and property values).

The majority of the available stormwater treatment processes are more effective for the removal of particulates, especially the settleable solids fractions, than the dissolved pollutant fractions. Removal of dissolved, or colloidal, pollutants is minimal in most commonly used stormwater controls and therefore pollution prevention at the sources is usually a more effective way to control the dissolved pollutants. Fortunately, most toxic stormwater pollutants (heavy metals and organic compounds) are mostly association with stormwater particulates (Pitt, *et. al.* 1996). Therefore, the removal of the solids will also remove much of the pollutants of interest. Notable exceptions of potential concern include: nitrates, chlorides, zinc, pathogens, 1,3-dichlorobenzene, fluoranthene, and pyrene.

STORMWATER QUALITY

When local stormwater quality data is not available, the data collected as part of the US EPA's stormwater permit program, and summarized in the National Stormwater Quality Database (NSQD), can be used (Maestre and Pitt 2005). The NSQD project reviewed and statistically analyzed data collected by municipalities [municipal separate storm sewer systems or MS4s] at their stormwater outfalls under their National Pollutant Discharge Elimination System (NPDES) permits (summary data provided in Table 1; the full database, including tables showing concentrations for different land uses, is located at http://unix.eng.ua.edu/~rpitt/Research/ms4/mainms4.shtml, along with several published papers describing the database features and example evaluations). This database reflects outfall samples from throughout the United States. There were significant differences in concentrations associated with different land uses and geographical areas for most pollutants, while seasonal variations (excluding snowmelt) were much less. Higher concentrations were observed for some pollutants at the beginning of rains in some areas (the "first flush" effect), but only in land uses having large fractions of paved areas, and only for some pollutants. Prior summaries of source area data (Pitt, *et al.* 2005) indicated how some locations (critical source areas) were more contaminated than other areas. These more contaminated areas are mostly paved areas that are associated with high levels of automobile activity, storage of heavy equipment, or other material, etc. In most cases, special stormwater controls should be located at outfalls serving the most contaminated areas, and at critical source areas where the most contaminants originate.

Table 1. Summary of MS4 Stormwater Outfall Data from National Stormwater Quality Database.

Pollutant	Frequency of Detection, % (Filtered, %) - Overall	Median Unfiltered Concentration (Filtered Concentration) for Detected Values - Overall	Median Unfiltered Concentration (Filtered Concentration) for Detected Values - Residential Areas	Median Unfiltered Concentration (Filtered Concentration) for Detected Values - Commercial Areas	Median Unfiltered Concentration (Filtered Concentration) for Detected Values – Industrial Areas
TSS (mg/L)	98.8	59	49	43	81
COD (mg/L)	98.4	53	55	58	59
Fecal Coliforms (MPN/100mL)	91.2	5,090	7,000	4,600	2,400
Fecal Strep. (MPN/100 mL)	94.0	17,000	24,300	12,000	12,000
NO_2+NO_3 (mg/L)	97.3	0.60	0.60	0.60	0.69
Phosphorus (mg/L)	96.6 (85.1)	0.27 (0.13)	0.31 (0.18)	0.22 (0.11)	0.25 (0.10)
Cadmium (µg/L)	40.8	1.0	0.5	0.96 (0.30)	2.0 (0.6)
Chromium (µg/L)	70.2 (60.5)	7.0 (2.1)	4.5	6.0 (2.0)	12 (3.0)
Copper (µg/L)	87.4 (83)	16 (8.0)	12 (7.0)	17 (7.6)	21 (8.0)
Lead (µg/L)	77.7 (49.8)	17 (3.0)	12 (3.0)	18 (5.0)	25 (5.0)
Nickel (µg/L)	59.8 (64.2)	8.0 (4.0)	5.6 (2.0)	7.0 (3.0)	14 (5.0)
Zinc (µg/L)	96.6 (96.1)	116 (52)	73 (32)	150 (59)	200 (112)

CRITICAL SOURCE AREA CONTROLS

There are a number of controls that can be used at critical source areas within the drainage area. These include biofiltration, porous pavement, hydrodynamic devices, filtration devices, etc. The following briefly describes a newly developed device that incorporates several different unit processes in a unique combination that has been tested under EPA support at pilot and full-scale installations (Pitt and Khambhammettu 2006). The UpFlow Filter™ was developed to overcome a number of problems of existing source area treatment devices to allow high treatment flow rates with good to excellent levels of control, and reasonable maintenance requirements.

Recent research on filtration examined alternative media and ways to reduce clogging that is prevalent with typical stormwater filtration. Upflow filtration was examined as a way to reduce clogging, at the same time as providing a much higher treatment flow rate. The UpFlow™ Filter was conceived as a treatment device to allow many of the treatment train components of the multi-chambered treatment train (MCTT) (Pitt, *et al.* 1999) but that can be used in a smaller area by providing much faster unit area stormwater flow treatment rates. Pollutant removal mechanisms in the UpFlow™ filter include several unit processes:

- Coarse solids and litter removal in the sump and by screens
- Capture of intermediate solids by sedimentation in sumps by controlled discharge rates
- Capture of fine solids in primary filtration media
- Sorption and ion-exchange capture of dissolved pollutants in primary and secondary media

The basic removal of solids is therefore dependent on physical sedimentation in the sump, and by filtration in the media. Figure 1 is a drawing of the full-sized commercial unit showing the water treatment path during normal operation. The UpFlow™ Filter was designed to be placed in a standard 4 ft (1.2 m) diameter catchbasin inlet, having a sump. Up to six upflow filtration modules can be used in each UpFlow™ Filter, and the media can be selected to target specific treatment flow rates and pollutants of interest. Figure 2 shows the performance of the UpFlow™ Filter during controlled tests using finely graded silica particles representing typical stormwater particles, while Table 2 shows the results of the filter during actual rains. High removals of almost all particles were observed. The flow-weighted treatment level of the device was about 80% for particles.

Table 3 shows the needed treatment flow rates to treat specific levels of the annual runoff volume. The needed treatment flow rates are less than the corresponding flow rate distributions because portions of the largest events are treated, while the flows in excess of the treatment flow rate bypass the device. If an 80% control objective is desired (a relatively common objective for many U.S. locations), the device would need to have a flow-weighted pollutant removal rate of about 90% and the about 90% of the annual runoff volume would need to be treated at that level. With lower treatment objectives, there would be more combinations of removal rates and treatment volumes. The UpFlow™ Filter can provide about 25 to 35 gpm (95 to 130 L/min) treatment flow rates per module. Therefore, only about one module would be needed per acre (0.4 ha) of paved area in Seattle in order to treat about 90% of the annual runoff volume, while about four modules would be needed per acre (0.4 ha) of paved area in Atlanta to treat the same percentage of the annual flow. With an 80% flow-weighted pollutant removal rate, this would correspond to an annual pollutant control level of about 70 to 75%. Other media can be used having higher pollutant removal rates, but they typically have lower treatment flow rates, requiring more modules for the same drainage area.

TREATMENT APPROACH FOR NEW INDUSTRIAL DEVELOPMENT

Treatment train approaches for stormwater management should also be applied at larger scales. The following is a recent example for a new industrial development in Huntsville, AL. Being a new development, there were no physical restrictions that would typically be associated with a retro-fitting project. This was an unusual project in that we worked with the site planners and engineers, and the site owners, from the early stages of site planning in order to optimize the conservation design aspects of the site development project. In most cases, the site engineers would address stormwater issues well after major aspects of the site layout had been completed, severely restricting available conservation design options. During retro-fitting projects, only selected source area options may be available, along with outfall controls, if space allows.

The stormwater elements proposed for the new 250 acre (100 ha) Huntsville industrial park will result in a conservation design that minimizes both runoff water volume discharges and stormwater pollutant discharges. The stormwater management elements of the conservation design are included at several levels at this site. Deed restrictions will require some simple on-site controls, as needed, the drainage system will be constructed to encourage grass filter treatment and biofiltration, and the main drainage subareas will contain large grass swale conveyances and wet detention ponds. Much of the upland areas of the site will also remain in open space. There area numerous sink holes on the site and these will be isolated from the drainage system by berms and buffers to restrict surface runoff entry.

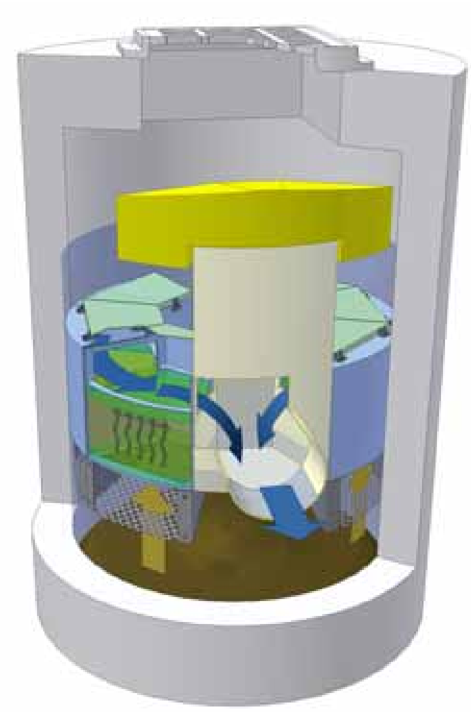

Figure 1. UpFlow™ filter drawing showing normal filtering operation (Hydro International, Ltd.).

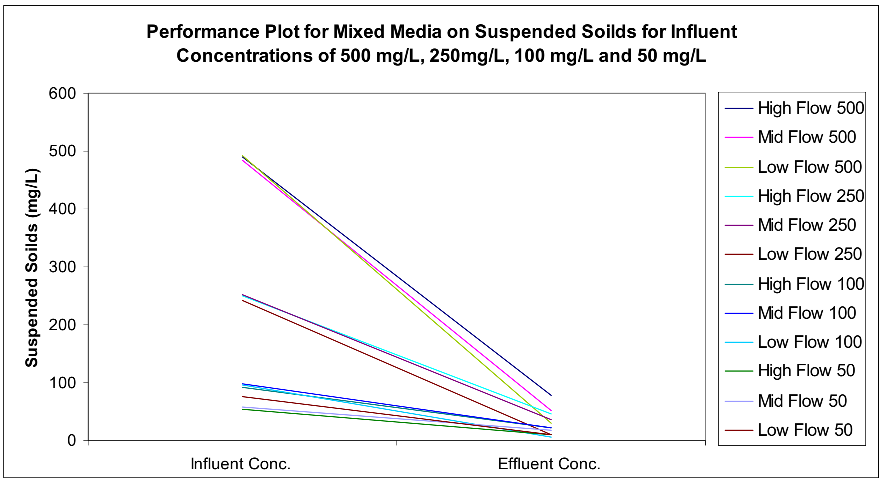

Figure 2. Performance plot for mixed media for suspended solids at influent concentrations of 500 mg/L, 250 mg/L, 100 mg/L and 50 mg/L.

Table 2. Calculated Mass Balance of Particulate Solids for Monitoring Period

particle size range (µm)	SS influent mass (kg)	SS effluent mass (kg)	SS removed (kg)	% reduction
0.45-3	9.3	2.8	6.6	70
3-12	18.7	6.4	12.3	66
12-30	22.4	7.7	14.7	66
30-60	26.7	6.8	19.9	74
60-120	4.6	1.8	2.9	61
120-250	19.8	4.3	15.5	78
250-425	11.5	0.0	11.5	100
425-850	17.1	0.0	17.1	100
850-2,000	10.5	0.0	10.5	100
2,000-4,750	4.8	0.0	4.8	100
>4,750	3.5	0.0	3.5	100
sum	148.9	29.8	119.2	80

Table 3. Treatment Flow Rates Needed for Different Treatment Objectives*

	Annual Flow Rate Distributions (gpm/acre pavement)			**Treatment Flow Rates Needed for Different Levels of Annual Runoff Volume Treatment (gpm/acre pavement)**		
Location	**50th Percentile**	**70th Percentile**	**90th Percentile**	**50%**	**70%**	**90%**
Seattle, WA	16	28	44	10	18	30
Portland, ME	31	52	80	18	30	53
Milwaukee, WI	35	60	83	20	35	65
Phoenix, AZ	38	60	150	20	35	90
Atlanta, GA	45	65	160	25	40	100

* multiply by 9.5 to obtain L/min/ha of pavement

These elements will work together to provide the most cost-effective set of stormwater controls for the site and provide high levels of control of both runoff volume and pollutant discharges. These site elements are all relatively common controls that have been applied at many locations throughout the U.S., and have been designed to take advantage of specific site characteristics and the desire to use this site as a demonstration of effective stormwater controls for the region.

Figure 3. Layout of North Huntsville Industrial Park Showing Conservation Design Elements

The stormwater controls include three main elements:

1) Critical source areas will need special attention. Industrial stormwater permits usually specify specific activities needing control. At industrial sites, these areas usually include material storage areas and truck loading bays. Most bulk material storage areas subject to rainfall exposure should preferably be covered, or the storage areas need to be bermed and the runoff treated with specialized controls (such as the Multi-Chambered Treatment Train). Heavy equipment yards (and public works yards) also need similar attention. Loading bays also need to be hydraulically isolated with the runoff treated with specialized controls (such as the UpFlow^{TM} Filter).

2) The building materials should be selected with pollution prevention in mind. The most serious problems normally associated with low and medium intensity industrial areas are the zinc concentrations in the runoff associated with the use of galvanized metal. In many areas, galvanized metal has been largely replaced by Zincalume or Galvalume (aluminum with zinc coatings), which still result in large zinc concentrations in the runoff. There has also been a shift from in-situ application of roofing paints to factory-painted paint to the metals. There have been considerable advances in coating technology, with increased durability and decreased breakdown of roof coatings and materials. The zinc concentrations from zinc-coated metal roofs is related to the degree of weathering and corrosion, with runoff from heavily weathered and corroded roofs having several times the zinc concentrations compared to runoff from roofs in good condition. Also, most of the zinc in runoff from metal roofs is in the dissolved state which is much harder to control and has more damaging environmental effects.

3) The building areas should have bioretention/grass swales for site runoff control. They will be located on the downslope side of the paved areas and roofs to direct the roof and lot runoff to the drainage systems. The bioretention/grass swales will be relatively small and mild sloped and can be easily maintained. They will be used in conjunction with other drainage way and pond stormwater controls as summarized below.

DRAINAGE WAY AND POND STORMWATER CONTROLS

The site was divided into four main drainage subareas, designated as subareas A, B, C, and D. The drainage way and pond stormwater elements, in conjunction with the lot-scale controls, will result in a conservation design that minimizes both runoff water volume discharges and stormwater pollutant discharges. The same stormwater elements are not recommended for each subarea due to different characteristics in each area. As an example, the industrial sites in subarea A are about evenly divided into an area that will be developed with conventional drainage having minimal on-site stormwater controls having conventional curbs and gutters, and an area with on-site stormwater controls. The conventionally developed area will discharge near the head of a wet pond with no regional swale treatment, while the other area will drain through a long natural grass drainage way before entering a wet pond. In addition, this area will incorporate on-site bioretention controls (site grass swales graded as linking rain gardens) to provide grass filtering pre-treatment and infiltration) to help compensate for the other area having minimal site controls.

Subarea B has extensive natural grass swales (two parallel swales) that will significantly reduce the runoff volume before another wet pond. Site bioretention controls can also be used to further reduce the volume, if desired. The pond will also be reduced in size to better fit the available area due to the reduced runoff volume. Site bioretention controls are not likely to be needed in this subarea due to the large amount of swales available. Subarea C is mostly developed with little open area, but with roadside grass swales that are suitable for runoff volume reductions. Site bioretention controls can also be used in this subarea. In this subarea, a relatively small pond (0.19 acres, or 0.08 ha) could be used due to the runoff volume reductions from use of grass swales. However, a full-sized pond (0.38 acres, or 0.15 ha, at normal pool elevation) is recommended to reduce the maintenance problems and make it more aesthetically pleasing. Subarea D will also utilize a roadside swale system along with on-site bioretention for runoff volume reductions. A wet pond will be located on adjacent city-owned land that may be developed in the future as a residential area. The large pond will also treat the runoff from that area.

These varying stormwater controls will provide an interesting and useful demonstration for the City of Huntsville. The drainage way and pond stormwater controls recommended for each subarea are listed in Table 4. The WinSLAMM model was used to predict reductions in runoff volume and particulate solids discharge vs. what would be expected with base conditions. These projections were based on 40 years of Huntsville rainfall data (1959-1999). Base conditions are defined as conventional development design with curb and gutter drainages and directly connected impervious surfaces. Other pollutants are expected to be reduced by similar percentages: those that are mostly associated with the dissolved fraction (nitrates and pesticides, for example) are expected to be reduced by about 50 percent and those mostly associated with particulates (phosphates and many heavy metals and PAHs, for example) are expected to be reduced by up to 90 percent. The percent reduction in runoff and sediment loss with the conservation design vs. base case increases as rain depth decreases. A 90 percent or greater reduction in sediment loss occurs with rain depths of approximately 2.5 inches or less. A 70 percent or greater reduction in runoff occurs with rain depths of approximately 1.5 inches or less.

Table 4. Summary of the conservation design stormwater components for each subarea and their projected reductions in runoff volume and particulate solids discharge[1]

Drainage Area	Drainage way and pond stormwater controls	Runoff Volume reduction[2]	Particulate solids reduction[2]
A	Pond, swale, and site bioretention	61	96
B	Small pond and swale	69	93
C	Pond and swale	68	94
D	Off-site pond, swale, and site bioremediation	50	92
Total Site Area		56	93

[1]Projections based on 40 years of rainfall data (1951-1999) using WinSLAMM model
[2]Base conditions are conventional development design with curb and gutter drainages and directly connected impervious surfaces.

COST OF STORMWATER CONSERVATION DESIGN

Garver Engineering of Huntsville provided a review and cost analysis of the conservation design for Phase 2 of the North Huntsville Industrial Park. This analysis established the base construction cost of the park using conventional engineering practices to which adjustments were made for the additional facilities required for conservation design. Credits were then applied for the value of practices replaced by the conservation design construction. The base bid to which these adjustments were made was $1,163,429. The net difference for this phase is approximately $33,750 in cost savings for the conservation design. Other intangible but nevertheless real advantages favor the conservation design plan. These include:

- Enhanced groundwater recharge through infiltration of treated stormwater through permeable soils, rather than the collection and conveyance of the runoff off the site.
- Reduction in offsite management costs of peak storm water volume. The discharge channel downstream from the wet ponds may have required concrete armoring if conventional stormwater facilities had been used. This cost savings alone could range from $400,000 to $500,000.
- Preservation of natural drainage areas by their incorporation into the master drainage system.

Significant improvements in post development groundwater and surface water quality.

Future cost savings are also anticipated through the use of conservation design practices in Phase 3. While this phase is more dense and conventional in layout, cost saving will accrue through the use of swales with curb outlet flumes in lieu of stormwater inlets with reinforced concrete piping. Phase 2 bid prices for inlets are $2,300 each while reinforced concrete pipe ranges from $34 to $53 per foot for 18-inch to 30-inch pipe. The unit costs for outlet flumes are $2,000 each while the grass swales cost approximately $15 per foot. Actual savings will depend on final engineering design and construction bids, but experience suggests that savings could range from $50,000 to $100,000.

CONCLUSIONS

This paper presented two case studies illustrating how treatment trains for stormwater management can be effectively used at different scales; one a critical source area treatment device, and the other a new industrial park. In both cases, different, but complementary, unit processes work together to result in an effective stormwater management process. This paper also briefly showed how using a continuous simulation model can be effectively used in sizing different types of controls, and how the different unit processes can function together. A recent paper (Pitt and Voorhees 2007) can be examined to illustrate how this type of information can be used in a decision analysis framework to guide in the selection of the most appropriate stormwater management program considering many conflicting objectives (costs, maintenance, pollutant control, runoff volume reduction, etc.).

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THE DRAFT EU GROUNDWATER DIRECTIVE ON THE PROTECTION OF GROUNDWATER AGAINST POLLUTION: A SUMMARY AND ASSESSMENT OF THE LIKELY IMPLICATIONS

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ABSTRACT

The proposed Groundwater Directive provides the detail on the means by which the Water Framework Directive requirements to prevent and control groundwater pollution are met. In particular, it sets out criteria and procedures for assessing groundwater chemical status, requires identification of significant and sustained upward trends and reversal of these trends where they are posing an environmental risk, and outlines the basis for establishing measures to prevent or limit the input of pollutants into groundwater. Account must be taken not only of the need to protect groundwater for environmental reasons, but also for human health reasons. Groundwater quality standards are set for two pollutants – nitrate and pesticides – by the EU. Member States must consider setting threshold values for 10 listed pollutants and any other substances regarded as putting groundwater at risk. A programme of measures is required which would reverse trends, and where necessary, prevent input of hazardous substances into groundwater, and limit input of other pollutants not considered hazardous.

The proposed Directive presents major challenges and opportunities. A defensible scientific basis will be critical in several areas: establishing thresholds; determining groundwater body status; deciding on the starting point for trend reversal; and choosing the list of pollutants posing a significant threat to groundwater. This will need to be based on a good conceptual understanding of the hydrogeology of the different aquifer types and, in particular, of the linkages to surface water and terrestrial ecosystems. In addition, comprehensive representative monitoring data will be required, together with a proper evaluation of these data. Undoubtedly, this proposed Directive will provide a good framework for protecting groundwater. However, difficulties may arise if the proposed programme of measures required to implement the Directive has some economic and social costs, as is likely to be the case. It is vital that the regulatory authorities in Ireland are aware of the challenges and the need for sufficient resources to implement the Directive effectively.

INTRODUCTION

THE LINK WITH OTHER DIRECTIVES

- The protection of groundwater against pollution is currently regulated under Directive 80/68/EEC (which will be repealed in 2013) and Directive 2000/60/EC, the Water Framework Directive (WFD). The WFD established a framework for Community action in the field of water policy and sets out general provisions for the protection and conservation of waters, including groundwater. Article 17 states that specific measures shall be adopted by the EU Commission to prevent and control groundwater pollution, so as to ensure achievement of the groundwater related environmental objectives of the WFD. These measures must include:
- criteria for assessing good groundwater chemical status; and
- criteria for the identification of significant and sustained upward trends and for the definition of starting points for trend reversals.

The proposed new Groundwater Directive (GWD) essentially provides the detail on the means by which Article 17 will be complied with. In addition, Articles 4, 11 and 17 of the WFD refer to preventing or limiting the inputs of pollutants into groundwater. The GWD elaborates on the measures required and provides a framework for making the WFD ‘prevent or limit’ objective operational, thereby enabling Directive 80/68/EEC to be repealed in 2013.

CURRENT STATUS OF THE NEW GROUNDWATER DIRECTIVE

In summary, the first reading opinion was adopted by the European Parliament in April 2005; political agreement was reached at Council in June 2005; a Common Position was adopted by Council in December 2005; and the second reading was launched in February 2006. If no significant unresolved issues arise between the Council and Parliament, the Directive is likely to be agreed at a plenary session of the European Parliament in June, with publication by the end of 2006. Clearly, if disagreements arise, the adoption will be delayed.

CONTEXT OF PAPER

This paper is based on the common position adopted by the Council of the European Union in December 2005. Changes may arise, or may even be arising as this paper is being written. Consequently, the GWD must be considered as a proposed Directive, and the current text as a draft. Indeed there is no guarantee that a GWD will be agreed. However, if not, each Member State must still classify groundwater, and the content of the draft GWD is likely to influence any default position adopted.

LAYOUT OF PAPER

This paper summarises the content of the draft GWD, starting with the recitals and then taking the more significant Articles and associated Annexes in turn, highlights certain implications and provides discussion on issues that are of particular relevance. The discussion sections are based on the current interpretation of the authors; this may change as understanding of the Directive develops and as EU guidance is received. Extracts from the draft GWD are shown in italics.

GROUNDWATER DIRECTIVE BACKGROUND STATEMENTS

GROUNDWATER DIRECTIVE RECITALS

The “whereases” or recitals before the Articles in the draft GWD highlight the link with the WFD, and also identify an important additional objective.

(1) *Groundwater is a valuable natural resource which should be protected from chemical pollution. This is particularly important for groundwater-dependent ecosystems and for the use of groundwater in water supply for human consumption.*

(2) *Decision No 1600/2002/EC of the European Parliament and of the Council of 22 July 2002 laying down the Sixth Community Environment Action Programme includes the objective to achieve water quality levels that do not give rise to significant impacts on, and risks to, human health and the environment.*

(3) *In order to protect the environment as a whole, and human health in particular, detrimental concentrations of harmful pollutants in groundwater should be avoided, prevented or reduced.*

KEY ISSUES AND IMPLICATIONS

While the draft GWD emphasises the relevance of protecting groundwater, so that the environment and groundwater-dependent ecosystems are not impacted detrimentally (as required by the WFD), it also mentions human health as an issue that must be considered. Human health is not specifically mentioned in the WFD, although Article 7.3 of the WFD refers to ‘drinking water’. This is a broadening of the remit required for river basin management, and is expanded upon in certain Articles of the GWD.

RELEVANT DEFINITIONS

Article 2 sets out important definitions, as follows:

(1) *"groundwater quality standard" means an environmental quality standard expressed as the concentration of a particular pollutant, group of pollutants or indicator of pollution in groundwater, which should not be exceeded in order to protect human health and the environment;*

(2) *"threshold value" means a groundwater quality standard set by Member States in accordance with Article 3;*

(3) *"significant and sustained upward trend" means any statistically significant increase of concentration of a pollutant, group of pollutants, or indicator of pollution, which presents an environmental risk for which trend reversal is identified as being necessary in accordance with Article 5;*

(4) *"input of pollutants into groundwater" means the direct or indirect introduction of pollutants into groundwater as a result of human activity.*

ASSESSING GOOD CHEMICAL STATUS

CRITERIA FOR ASSESSING STATUS (ARTICLE 3)

Groundwater Quality Standards

In assessing the chemical status of a groundwater body (GWB) or group of bodies, groundwater quality standards (GQSs) are set for two pollutants only – nitrates and pesticides. The standards are 50 mg/l for nitrate (as NO_3), 0.1 µg/l for an individual pesticide and 0.5 µg/l for the sum of all individual pesticides detected. However, more stringent values may be established where these standards are considered to result in failure to meet the WFD environmental objectives.

Threshold Values

Member States must consider establishing threshold values (TVs) for 10 listed pollutants, given in Annex II of the draft GWD; arsenic, cadmium, lead, mercury, ammonium, chloride, sulphate, trichloroethylene, tetrachloroethylene and conductivity (as an indicator of saline or other intrusions), and for any other pollutants, which have been identified as contributing to the characterisation of GWBs as being at risk of failing to achieve good groundwater chemical status. These must be established by December 2008. While the groundwater quality standards for nitrates and pesticides are set in the draft GWD, TVs are set by Member States. In addition, TVs may vary within a Member State, and may be set at national level, an individual River Basin District level, or at the level of an individual GWB or group of GWBs.

In establishing the TVs, the following factors are taken into account:

- *The extent of interactions between groundwater and associated aquatic and dependent terrestrial ecosystems (e.g. fens, turloughs);*
- *Hydrogeological characteristics including information on background values and water balance;*
- *The origins of the pollutants, their possible natural occurrence, their toxicology and dispersion tendency, their persistence and their bioaccumulation potential.*

Where feasible, Member States are requested to report relevant information on the establishment of the TVs, for example:

- The number of GWBs characterised as being at risk, and, more significantly, the pollutants that are considered to be the cause;
- A description of the GWBs at risk, including their relationship with associated surface waters and directly dependent terrestrial ecosystems, and, in the case of naturally-occurring substances, the natural background levels (NBLs);
- The scale at which a TV will apply, which can range from national to GWB scale;
- Relationships between TVs and:
  - Observed NBLs for naturally-occurring substances;

- Existing national, Community or international environmental quality objectives and standards; and
- Any relevant information concerning toxicology, persistence, bioaccumulation potential and dispersion potential of the pollutants.

The list of TVs may be amended, by adding or removing pollutants, or existing TVs may be adjusted, based on new information in the context of the periodic review of River Basin Management Plans. Protection of human health as well as the environment is explicitly required.

Where GWBs are shared across international boundaries, the establishment of threshold values must be co-ordinated between the Member States.

KEY ISSUES AND IMPLICATIONS

1. The proposed GQSs for nitrates and pesticides are the same as the drinking water maximum admissible concentrations; the scientific basis for using these as environmental standards is not clear. However, the GWD allows for more stringent values in circumstances where the GQSs are considered to be too lenient. For instance, if the dissolved inorganic nitrogen (DIN) median limit of 2.6 mg/l N (total NH4-N, NO2-N and NO3-N) set by the EPA for estuaries (EPA, 2001) is applied in circumstances where groundwater provides a significant proportion of river flow, the 50 mg/l standard for nitrate may have to be reduced. This could potentially have major implications for the implementation of measures in areas where nitrate concentrations in groundwater are relatively high due to the impact of agricultural activities. Consequently, a convincing scientific basis for any reduction in the value below 50 mg/l would be essential.

2. During the WFD Article 5 characterisation process completed in March 2005, a significant number of GWBs were categorised as ‘probably at risk’ on the basis of the risk assessment for mobile organics (e.g. certain pesticides and PAHs). However, this risk assessment methodology was quite rudimentary and conservative in nature, and there were no confirmatory monitoring data. Consequently, this is an aspect of groundwater quality that needs urgent investigation and assessment in Ireland.

3. The requirement to set TVs for substances considered to be putting a GWB at risk is an important step in providing a scientific basis for protecting groundwater and other associated receptors. It is a continuation of the process started in the 1990s by the Geological Survey of Ireland in producing threshold values for certain pollutants to aid groundwater quality assessments, which was then progressed by the setting of EPA Guideline Values (EPA, 2003).

4. The characterisation process showed that phosphorus entering groundwater in areas of extremely vulnerable (shallow soil/subsoil) karst aquifers is posing a threat to surface water ecosystems (WFD Groundwater Working Group, 2004; Kilroy and Coxon, 2005). Consequently, a TV will need to be derived for this pollutant for these particular areas. As the surface water and ecosystem EQSs are low (µg/l rather than mg/l), deciding on this TV will be challenging.

5. Deriving defensible TVs will pose a significant scientific challenge as they depend on a good understanding of the link between groundwater and associated receptors, and having adequate information of NBLs. They must take account of existing environmental quality objectives and issues such as toxicology and bioaccumulation potential. If relatively low values are considered necessary, it could result in a proposed stringent programme of measures, which may not be accepted readily. Similarly TVs set too high may not provide adequate protection of ecosystems.

6. The concept of TVs varying for different GWBs or groups of GWBs is a challenging one, and may result in complaints that some areas are being dealt with more severely than others. However, if high status surface water bodies and sensitive ecosystems are to be maintained and

protected, as required by the WFD, lower and more stringent TVs will be required relative to areas with surface water bodies classified as being of good status. Therefore, justification for the establishment and application of different TVs for different GWBs will need to be made on a sound scientific basis.

7. The establishment of TVs for cross border GWBs will require close cooperation between relevant regulatory bodies in each Member State.

PROCEDURE FOR ASSESSING GROUNDWATER CHEMICAL STATUS (ARTICLE 4)

Groundwater Directive Requirements

The chemical status of a GWB or a group of GWBs is determined as 'good' when:

1. Article 4.2 (a): The GQS or the TV is not exceeded at any monitoring point in the GWB or group of GWBs;

Or

2. Article 4.2 (b): The GQS or TV is exceeded at one or more monitoring points, but an appropriate investigation is undertaken which confirms that:
    - The concentration of pollutants exceeding the GQSs or TVs are not considered to present a significant environmental risk;
    - The other groundwater chemical status conditions are being met, such as no saline or other intrusions, and there is no significant impact on both associated surface waters and terrestrial ecosystems;
    - Drinking water is protected such that no additional purification treatment is required;
    - The ability of the GWB or group of GWBs to support human uses has not been significantly impaired.

Essentially, if any of the monitoring points have concentrations greater than the GQS and/or the TV, further investigations are necessary to show no significant risk or impact. In this situation, whereby a GWB is classified as being of good chemical status, Member States must take measures to ensure protection of ecosystems and human uses on the part of the GWB represented by the monitoring point(s) at which the GQS or TV has been exceeded.

Details on the procedure for assessing groundwater chemical status is outlined in Annex III of the GWD; this Annex is mainly concerned with Article 4.2 (b), i.e., circumstances where the GQS and/or TV is exceeded. The assessment must be undertaken on GWBs and groups of GWBs characterised as being at risk and must consider all contributing pollutants. Investigations must take account of:

- The information collected as part of characterisation;
- The water quality data provided by the groundwater monitoring network; and
- Any other relevant information, including a comparison of the annual arithmetic mean concentration of the relevant pollutants at a monitoring point with the GQSs and TVs.

Member States are required to estimate the extent of the body of groundwater having an annual arithmetic mean concentration of pollutant higher than a GQS or TV, using appropriate aggregations of the monitoring data and supported by concentration estimations based on a conceptual understanding of the GWB or group of GWBs.

In addition, Member States are required to assess:

- The amounts (e.g. pollutant loading) and the concentrations of pollutants being, or likely to be, transferred from the GWB to associated surface waters or directly dependent terrestrial ecosystems;
- The likely impacts on these ecosystems;
- The extent of saline or other intrusions into the GWB caused, for instance, by abstraction of groundwater; and

- ♦ The risk from pollutants in the GWB to the quality of water abstracted for human consumption.

Maps showing the status of the GWBs must be produced, and all monitoring points where the GQS and/or TV are exceeded must be shown.

Key Issues and Implications

1. Significant temporal variations in pollutant concentrations can occur in Irish bedrock aquifers (unlike in most other EU countries), due to low effective porosities, high velocities via conduit and fissure flow, and often rapid recharge. Consequently, it is probable that peak concentrations in monitoring wells in some GWBs may exceed, for instance, the GQS for nitrate or the TV for phosphate.

2. While peak concentrations for certain pollutants may exceed GQSs and/or TVs at certain monitoring points, it may be possible to show that these peaks do not pose a significant environmental risk or impact, as surface water EQSs are given as annual means. Also, the text of Annex III indicates that the statistical value that should be compared with the GQS and/or TV is the annual arithmetic mean. This may result in a misrepresentation of water quality problems because they are often seasonal in nature and their significance may be diminished through the reporting of annual means.

3. The establishment of GQSs or TVs based on drinking water maximum admissible concentrations (MACs) as mean values will not be sufficient to ensure protection of drinking water, as the MAC is literally a maximum value. Consequently, it may be necessary to choose a lower mean value as a threshold that would ensure that peak values do not exceed the MAC. This could vary depending on the aquifer type. For instance, taking nitrates as the pollutant, in porous aquifers such as sand/gravel, a threshold of 45 mg/l might be adequate, while in a fissured or karstic aquifer, a threshold of 37.5 mg/l might be required. Clearly, good hydrogeological understanding and adequate long term monitoring data would be required to provide the basis for this value.

4. Status determination requires an assessment of the aerial extent of a GWB exceeding a GQS or TV. In this instance the exceedence at the monitoring point is based on an annual arithmetic mean concentration using an appropriate aggregation (e.g. six year rolling average). However, delineating these areas will be challenging.

5. Estimating the amounts and concentrations of pollutants transferring from GWBs to associated surface waters and groundwater dependent terrestrial ecosystems will be difficult in Ireland in view of the complex hydrogeology and our current limited knowledge of the groundwater needs of these ecosystems. It will be critical that no GWBs are incorrectly classified as having ‘poor’ status, as this will result in unnecessary measures that are likely to have social and economic impacts.

6. As drinking water and human health are factors in determining GWB status, specific monitoring for parameters other than chemical substances will be required; in particular microbial pathogens and radiological parameters.

7. As saline intrusion is relevant to status determination, a brief assessment of the impact of groundwater abstraction in coastal areas will be required.

8. Clearly, the monitoring data are essential in assessing the groundwater status; therefore it is vital that the network is representative of the hydrogeological settings, pressures and impacts present in Ireland. Once again our complex hydrogeology and the heterogeneous fissured nature of our bedrock aquifers makes this problematical. Defensible decisions cannot be based on monitoring

data alone unless a vast network is installed, which could then be claimed to be representative. However, an expensive and onerous approach such as this is not necessary in the Irish context. The recommended approach for Ireland not only involves focused monitoring, but also risk assessments (source-pathway-receptor assessments) up-dated regularly to parallel and inform the assessments based on the monitoring data. It is also likely to involve regular (once in every river basin management reporting cycle is suggested), major reappraisals of the monitoring network.

SIGNIFICANT AND SUSTAINED UPWARD TRENDS (ARTICLE 5)

IDENTIFICATION OF TRENDS

A requirement of the GWD is that Member States must *identify significant and sustained upward trends* on all GWBs characterised as being at risk. Monitoring is critical to this activity. The monitoring programme must be designed to detect these trends in relevant pollutants. Monitoring frequencies and locations must be selected such that:

- upward trends can be *distinguished from natural variation with an adequate level of confidence and precision*;
- *upward trends can be identified in sufficient time to allow measures to be implemented in order to prevent or at least mitigate environmentally significant detrimental changes in groundwater quality*;
- *the physical and chemical temporal characteristics of the GWB, including groundwater flow conditions and recharge rates and percolation time through soil or subsoil*, can be taken into account.

Trend analysis in time series of individual monitoring points will be based on a statistical method, such as regression analysis.

TREND REVERSAL

Trends which present a significant risk of harm to the quality of aquatic ecosystems, to human health, or to actual or potential legitimate uses of the water environment must be reversed, using a programme of measures. Trend reversals must be demonstrated using the monitoring network.

In general, the starting point for implementing measures to reverse significant and sustained upward trends will be when the concentration of the pollutants reach 75% of the parametric values of the GQSs or TVs. There are some exceptions to this:

- The starting point may be either <u>earlier</u> or <u>later</u>, depending on whether it is considered and can be shown that the trend reversal measures will prevent any environmentally significant detrimental changes in groundwater quality from occurring;
- A <u>different starting point</u> is justified where the detection limit does not allow for establishing the presence of a trend at 75% of the parametric value.

KEY ISSUES AND IMPLICATIONS

1. A phrase and a concept, which is not present in the WFD or the draft GWD up to this point, are introduced in this Article; namely, '*actual or potential legitimate uses of the water environment'*. The precise meaning of this and the implications are not clear – presumably EU guidance will be drafted.
2. Identification of trends and trend reversal will require both adequate data and staff resources. Therefore, a significant increase in staff resources must be provided to successfully undertake this task.
3. Further guidance on trend assessment and reversal is planned by the EU when the GWD is finalised.

PREVENT OR LIMIT INPUT OF POLLUTANTS (ARTICLE 6)

GROUNDWATER DIRECTIVE MEASURES

In order to achieve the objective of preventing or limiting inputs of pollutants into groundwater, Member States must ensure that the programmes of measures include:

- *All measures necessary to* aim to prevent *inputs into groundwater of any hazardous substances*. These substances are listed in Annex VIII of the WFD.
- All measure necessary to limit input of pollutants not considered hazardous, *so as to ensure such inputs do not cause deterioration in good groundwater chemical status, significant or sustained upward trends on the concentrations of pollutants in groundwater, and do not otherwise cause pollution of groundwater. Such measures shall take account of established best practice, including Best Environmental Practice and Best Available Techniques specified in relevant Community legislation*.

Member States must consider whether the pollutants listed in Annex VIII of the WFD, in particular the metals, are hazardous or non-hazardous. In addition, the inputs of pollutants from diffuse sources of pollution must be taken into account whenever technically feasible.

Six circumstances are listed in the GWD where Member States may exempt from the measures associated with inputs of pollutants, three of which are particularly relevant to groundwater in Ireland:

- Where the quantity and concentration of pollutants are considered by the competent authorities to be *so small as to obviate any present or future deterioration in the quality of the receiving water*;
- Where the *consequences of accidents or exceptional circumstances of natural cause could not reasonably have been foreseen, avoided or mitigated*;
- Where it is *not technically feasible to prevent or limit the input of pollutants without using measures that would increase risks to human health or to the quality of the environment as a whole; or measures where it would be disproportionately expensive to remove quantities of pollutants from, or otherwise control their percolation in, contaminated ground or subsoil.*

KEY ISSUES AND IMPLICATIONS

1. The ‘prevent or limit’ issue has important implications for regulatory authorities. Guidance on the interpretation of this Article will be provided by the EU. Consequently, this issue is not dealt with in any detail in this paper.

2. The existing Groundwater Directive 80/68/EEC requires that Member States take the necessary measures to prevent “List 1” substances from entering groundwater. The draft GWD requires measures *to aim to prevent inputs into groundwater of any hazardous substance*. The ‘aim to’ objective seems to be a significant change, although a legal determination on its meaning may be necessary. In addition, there will no longer be a ‘list’ of substances classed as ‘hazardous’ by the EU; rather Member States must identify these substances, taking account of an indicative list in Annex VIII of the WFD.

3. The exemptions are important and, arguably, realistic, particularly the one that allows for ‘small’ inputs.

4. The risk to human health as well as the environment is mentioned in this Article.

CONCLUSIONS

The proposed GWD provides a coherent and comprehensive framework for protecting groundwater in Ireland. It presents major scientific challenges for hydrogeologists and water managers. An improved monitoring network is required, together with progress in understanding the complex hydrogeology of Irish aquifers. The implementation of the measures needed to comply with the GWD are likely to have some social and economic costs. Therefore a good scientific basis will be needed to justify these measures; the challenge of providing this justification must not be underestimated.

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SESSION II

A UK AND EUROPEAN PERSPECTIVE OF SUSTAINABLE URBAN DRAINAGE (SUDS) WITH PARTICULAR REFERENCE TO INFILTRATION SYSTEMS AND GROUNDWATER POLLUTION

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ABSTRACT

A review of UK and European infiltration systems for the control and treatment of runoff from impermeable urban surfaces indicates a variable performance and rather confused evidence as to whether they pose any immediate threat to the requirements of the EU Water Framework Directive. The principal concerns relate to their long term performance and issues of clogging, basal accumulations to potentially hazardous waste levels and breakthrough threats resulting from "facilitated transport" due to organic matter-pollutant complexation and double-porosity mechanisms. More careful design and location criteria, including the introduction of pre-treatment, alternative filtration media and positive drainage are recommended as well as more detailed consideration of sub-soil characteristics and chemical composition.

INTRODUCTION

The EU Water Framework Directive (WFD) requires that all sources of diffuse pollution should be identified and quantified with Article 11 requiring a Programme of Measures (PoMs) for control and monitoring of such diffuse sources within future River Basin Management Plans (RBMPs). The Directive establishes under Article 4 a general requirement that pollutant inputs to groundwater be prevented or limited and places a prohibition on direct polluting discharges to groundwater. In compliance with Article 5 of the WFD, the UK regulatory authorities have completed preliminary characterisation (or basic risk assessment) of all surface and groundwaters in order to determine the most significant pressures and impacts on the receiving water environment and to assess the likelihood that waterbodies (WBs) will achieve the relevant Environmental Quality Objectives (EQOs).

The England & Wales Environment Agency (EA, 2004) assessment framework for example, uses land use activity, source pressure, exposure pressure and impact data in its characterisation. WBs in England & Wales have been assessed with respect to the magnitude of the activity or pressure exerted on them and, where data is available, the susceptibility of the receiving water to failing their EQOs. The outcome of the assessment framework is expressed in a categorisation of high, moderate, low or no exposure pressure. This pressure category is then converted to a risk factor using impact data where available together with a measure of confidence. If impact data is not available, thresholds for risk category have been determined subjectively. Urbanisation is considered to constitute a prime source and exposure pressure category, although both land use activity and impact data are uncertain at the current preliminary risk assessment stage and will require further information to fully justify appropriate PoMs for the RBMP1 stage.

At the same time, the WFD is also serving as a driver for the review of UK stormwater management practices with government initiatives urging a more holistic approach to the management of flood risk (DEFRA, 2004). Official guidelines embodied in the Planning Policy Statement (PPS) on "***Pollution Control***" (PPS23) state that any adverse impacts on water quality must be considered including "effluent" discharges such as urban runoff with suitable provision being made for the drainage of surface water. PPS25 on "***Flood Risk***" promotes the use of structural SUstainable Drainage Systems i.e. SUDS as termed in the UK or BMPs in continental Europe and North America.

There are, therefore, considerable pressures to identify and quantify source pollutant loadings within urban catchments and to evaluate their potential risks to receiving waterbody and groundwater objectives.

URBAN DIFFUSE SOURCE DATA CHARACTERISATION

A considerable database already exists on the quality of drainage from impervious urban surfaces, although it is undoubtedly incomplete and variable in terms of quality control, land use coverage and receiving waterbody impact assessment (Wild *et al*., 2002; Ellis and Mitchell, 2006). Table 1 shows the outcome of the Article 5 EA (2004) risk assessment analysis which suggests that there is a likelihood of considerable pressure and exposure on both surface and groundwater resources resulting from existing non-point urban source pollution.

Table 1. Waterbody Pressures Resulting from Diffuse Urban Sources.

RIVERS				GROUNDWATERS		
Risk of Failing WFD EQOs	**No. of WBs**	**River Length** (km)	**% River Length**	**No of WBs**	**Area of WBs** (km^2)	**% Area of WBs** (km^2)
At Risk	176	3,441	5	12	1,513	1
Probably at Risk	895	14,103	20	65	16,662	13
Probably not at Risk	2222	20,087	29	190	89,803	69
Not at Risk	4523	32,587	46	89	22,732	17
Not Assessed	0	0	0	0	0	0
TOTAL	**7816**	**70,217**	**100**	**356**	**130,710**	**100**

About 25% of river lengths and 14% of groundwaters are considered to be "*at risk*" or "*probably at risk*" in England and Wales from such drainage sources with some 18,175 km^2 of aquifers being potentially impacted. A comparison of the overall pollutant event mean concentration (EMC) values recorded for urban surface water discharges within the UK with prevailing EU standards for drinking water has shown that, apart from NO_3-N, most urban runoff pollutants exceed the standard by large margins (Ellis, 1997 and 2000). The same situation has been observed in the United States by Pitt *et al* (1996) where average storm EMCs occur at levels that could pose threats to the potable and secondary re-use of urban groundwaters.

In terms of pollutants types, heavy metals and hydrocarbons (especially soluble and colloidal-associated species), solids, bacteria, pesticides/herbicides and other organics such as MTBE are the principal contaminants of concern for groundwater assessment. Non-agricultural use of herbicides for the control of vegetation/weeds in urban areas is one example which is known to be contributing to levels above the drinking water standard in adjacent groundwater within the UK. As much as 34% of applied herbicide can be lost in surface runoff and resulting groundwater triazine levels can frequently reach between 1.5 – 20 µg/l (Ellis et al., 1996; Ellis, 1997).

SUSTAINABLE DRAINAGE SYSTEMS

Direct infiltration of impermeable surface water runoff to ground is a traditional practice associated with fin drains and soakaways from both highway and roof drainage although sand filters, infiltration trenches and basins as well as grass swales, porous paving and filter strips are more recent infiltration devices associated with sustainable drainage systems (SUDS or BMPs). UK national advice incorporated into Planning & Policy Guideline Note, PPG25 (2001) on "***Development and Flood Risk***" advocates that all development plans should promote the use of SUDS drainage and also indicates that local authorities should work closely with the EA and Scottish Environment Protection Agency (SEPA), sewerage undertakers, navigation authorities and developers to coordinate surface water runoff control "*as near to the source as possible through the use of sustainable drainage systems*"

Most local councils in the UK are now incorporating sustainable development guidelines within their Local Development Documents which include reference to SUDS options for surface water drainage e.g. Cambridge City Council Sustainable Development Guidelines, 2002, where SUDS "*should be considered at the earliest stages of master planning*". Many district councils, e.g. S. Gloucestershire DC (www.southglos.gov.uk), have produced non-statutory Supplementary Planning Guidance (SPGs) covering SUDS techniques, although many SPGs carry caveats in respect of direct infiltration. Dundee and Dumfries city councils convene regular open stakeholder meetings to discuss development applications that have a surface water interest at which SUDS alternatives are considered.

The EA state they are "*keen to promote the SUDS concept, which has much to offer to reduce pollution and flood risks*" (www.environment-agency.gov.uk/waterquality/diffusewater pollution). SEPA has actively promoted SUDS since the mid-1990s and today over 800 sites and 4000 SUDS devices have been implemented (Wild *et al*., 2002) and the Scottish Executive Planning Advice Note, PAN 61 encourages developers and drainage engineers to consider SUDS as front-line flow and quality control structures for surface water runoff. Both regulatory agencies caution however, on the need to ensure that SUDS do not prejudice the integrity of adjacent groundwaters. However, they do not recommend that there should be some form of pre-treatment prior to infiltration or that some infiltration systems at particular locations might require under-drainage and/or positive drainage outlets.

SUDS have also been widely adopted in Europe with swales, infiltration trenches and detention systems being very popular in Germany whilst retention and infiltration basins and porous paving systems are popular BMP stormwater management measures in France. The German Water Association (ATV) technical standard (ATV2002) sets out permissible infiltration facilities for 14 differing urban land use/activity types based on presumed pollution load. Under this standard, the use of infiltration drainage for industrial and heavy goods parking areas is prohibited, whilst highways carrying more than an average daily traffic density of 15,000 vehicles/day are required to provide approved pre-treatment prior to surface infiltration. In Scandinavia, retention ponds and constructed wetlands are common stormwater drainage devices with swales and infiltration basins also being widely used; both offer additional advantages of winter snow storage. In southern European countries, such as Greece, Italy, Spain and Portugal the use of SUDS/BMPs is limited largely because of concerns relating to their ability to perform under short duration, high peak intensity flow conditions. Despite such concerns, they are being gradually introduced as evidenced by the detention basin (with pre-treatment settling) installed for the 2004 Olympic rowing basin and the oil separation and retention basins used at Athens airport. In Spain, infiltration basins and porous paving systems were widely used in the construction of the Barcelona 1992 Olympic village and were also promoted in the 1997 master drainage plan for the city. A review of the use of SUDS/BMPs within the European context is given in Revitt *et al*. (2003), and detailed descriptions of the various types of SUDS and their O&M requirements with examples can be obtained from the EU DayWater project website (www.daywater.org) accessed via Hydropolis and the BMP window. Full design guidance can be obtained from the CIRIA (2000a, 2000b and 2004) reports, with CIRIA Report R156 (Bettess, 1996) providing specific design advice for infiltration drainage.

SUDS INFILTRATION SYSTEMS

Infiltration is the process whereby runoff volume is percolated to the subsurface through unlined grassed channels and surfaces, dry basins/trenches, soakaways and via porous surface media. Infiltration systems are best suited to soils with infiltration rates exceeding 12 - 15 mm/hour (e.g. sandy loams, sands, sandy gravels) with a minimum of 1.5 m separation to the seasonally high water table. An overall filtration rate of 5 $m^3/ha/m^2$ has been recommended for total solids removal efficiencies of between 64% - 90% (Revitt *et al*., 2003).

An average volumetric loss of between 33% and 80% has been recorded for grass swale channels with all the load reduction of dissolved pollutants occurring due to the infiltration mechanism (UKWIR, 2003). Outflow, as a percentage of total rainfall, averaged 22% and 46% from two porous paved car parks in Edinburgh (Schluter and Jefferies, 2001) with 0.4 mm equivalent depth outflow of a total 8.8 mm recorded for one rainfall event on one site. Total loss is typically recorded on porous paving for rainfall events of less than 5 mm and the majority of the discharge volume outflow is at rates below 1 litre/s/ha (Abbott *et al*., 2003). Such loss of volume can produce greater mass load reductions for many chemical constituents (and solids) than can be achieved by reduction of concentrations in other types of storage SUDS such as detention/retention basins, wetlands or settlement chambers. However, surface storage facilities such as extended detention basins have also recorded volume reductions varying between 8% to 60% (UKWIR, 2003), primarily related to infiltration mechanisms. Similar infiltration below permanent pool wet retention basins has been recorded from French studies with soluble and colloidal metal mobilisation (Zn, Cd and Cu) occurring to groundwater from saturated polluted benthal sludges (Bechet *et al*., 2004).

Soakaway practice may provide little if any protection to groundwater from pollutants carried in surface drainage, and especially for soluble contaminants such as herbicides, MTBE, metal species such as zinc, cadmium and platinum, as well as monocyclic aromatic hydrocarbons such as benzene or toluene or other organochlorine solvents. Whilst soakaways and infiltration basins/trenches are invariably above the water table, even the presence of a thick non-saturated zone may not guarantee sufficient aquifer protection. In chalk for example, the pore space in the unsaturated zone is normally totally filled with water, with only fissures draining under gravity. Beneath such infiltration systems, the fissure space can rapidly fill after rainfall and the transit time to the water table may be very rapid. Tracer tests undertaken on soakaway drainage on the M1/M25 junction in Hertfordshire, UK, indicated potential travel times of about 2 km/day via fissure flow within the underlying chalk; the maximum recorded speed was 100 m/hour (Price *et al*., 1992). On the basis of the tracer behaviour, it was estimated that pollutant concentrations reaching abstraction wells some 3 km away from the soakaway injection point, were likely to be about 4 μg/l for every tonne of pollutant reaching the soakaway structure.

Whilst rapid dilution and adsorption of stormwater pollutants will occur in receiving surface waters, direct infiltration to ground can cause problems because of the more limited dilution that occurs at the point where the infiltrated stormwater mixes with the aquifer water and because of the slow rate and extent of mixing within the aquifer. If cracking, macropores and fissures exist in the unsaturated zone, there will also be a likelihood of rapid-flow transport to the underlying aquifer. Once the infiltration rate exceeds the hydraulic conductivity of the infiltrating matrix, flow will occur directly down fissures. The potential for highway filter drains and fin drains infiltrating spillages into underlying fissured strata and leading to rapid groundwater pollution has been demonstrated in the UK by Price (1994) and Zimmerman *et al* (2004) in Germany. The same potential has been shown for highway soakaways (and particularly from multiple soakaways or drainfields) which are vulnerable to the release and transmission of large volumes of discrete, dense non-aqueous liquids (DNAPLs) into underlying strata (Barker *et al*., 1999). This potential threat is one reason for the standard EA policy not to allow direct infiltration of major highway and motorway discharges within designated Zone I (Inner Source Protection) regions.

Infiltration acceptability matrix approaches for the infiltration of various types of impermeable urban surfaces and associated land use activities have been proposed (Ellis, 2000; CIRIA, 2004; ATV2002). Whilst providing a useful screening function based on the likely magnitude and composition of the source discharge, such matrix approaches assume Darcian diffusion and overlook site gradients, relation to storm drain depth or connectivity or to space/cost suitability criteria.

SUDS INFILTRATION PERFORMANCE

Table 2 summarises a selection of European studies that have examined the potential of infiltration systems to remove urban runoff pollutants especially heavy metals and hydrocarbons. Full reference and detail for the listed studies can be obtained from the EU DayWater project website (www.daywater.org) under publications (Deliverable 5.1; Revitt *et al*., 2004).

Table 2. European Infiltration Performance Studies.

Source	Type of Study	Outcome
Mikkelsen *et al*., 1997. (Denmark). *Wat.Sci.Tech*.,29(1/2), 293-302	PAH, HM and AOX accumulation from highway runoff within surface (basin) and sub-surface infiltration systems	• Inverse relation between contaminant mass recharging groundwater and soil depth • Little risk of infiltration contamination to underlying groundwater; but high HM charge leads to binding • Potential for adsorbable pollutants to accumulate in soil surface at "risk" levels
Dierkes *et al*., 1999 (Germany) *Wat.Sci.Tech*.,39(2) 325-330	Distribution and fate of highway pollutants on adjacent verges and embankments	• Highest concentrations located in upper 5cm of soil and within 2m of highway • Mineral oil HCs degraded but PAHs accumulated within upper 10cm of soil • Recommend removal of upper soil layer as hazardous waste
Barraud *et al*., 1999 (France) *Wat.Sci.Tech*.,39(2), 185-192	Comparison of recent and 30 year old infiltration basin	• HM and HC concentrations high in top 5-10cm of soil but declined exponentially with depth • Only 31% of soluble zinc retained in older soakaway which could pose risk (cf 54-88% removal in new soakaway)
Pratt *et al*., 1999 (UK) *Wat.Sci.Tech*., 39(2), 103-109	Removal of oil/grease, COD in porous paving	• Effective degradation of HCs within substrate • Nutrient supply limiting factor on degradation
Newman *et al*., 2002 (UK) *Wat.Sci.Tech*., 45, 51-56	Removal of oil/grease by porous paving	• Effective HC degradation (up to 90% removal) over 4 year operation • Requires careful/appropriate design and O&M for successful and sustained operation
Bardin *et al*., 2001 (France) *Wat.Sci.Tech*.,43(5), 119-128	HM removal by sand filters	• Significant removal of solids, HCs, bacteria although pre-treatment separator ineffective • HM removal only 17% ; (Pb 59% removal) • Low removal rate for nitrate
Raimbault, 1999 (France) Revitt *et al*., 2003 Del5.1 DayWater	Clogging of porous surfaces having reservoir structures	• Infiltration depths decrease over time; average values reducing from 0.85 cm/s to 0.15 cm/s over 3 years. • Increasing compaction densities in top 2 cm of porous surface; clogging horizons above 2/3cm within 3/6 years
Wild *et al*., 2002 (UK) Report, SNIFFER, Edinburgh.UK	Highway filter drain performance	• Mean runoff value from drain of 42% ; ranging between 1-200% (over 100% events being related to snowmelt) • Highly variable pollutant removal rates but 75% removal of solids
Revitt et al., 2004 (UK) Deliverable 5.1, EU DayWater Project	Infiltration basin in residential area	• Accumulation of HM at all depths and exponential mobilisation of soluble species with depth • Average K value of 10-4-10-6 m/s required in underlying unsaturated zone.
Legret and Colandini, 1999. (France) Revitt *et al*, 2003 Del5.1 DayWater	Porous paving with reservoir structure	• Average 96% volume loss through surface to underlying subsoil • 60 – 80% reduction in solids and HMs within paving structure
Schluter and Jefferies, 2004 (UK) *Proc.NOVATECH04* Graie, Lyon.	Infiltration trenches on residential estates	• Mean peak flow reductions 45% - 75% and average inflow volume reductions between 25% - 90% • Occasional surcharging due to inlet chamber clogging
Larmet et al., 2004 (France)	Infiltration basins in industrial/commercial areas	• Bacterial facilitated removal of colloidal/soluble metal species to unsaturated zone • Clogging of basin with hazardous "waste level sludge"
Deschesne et al., 2004 (France) *J.Env.Mangt*.,71,371-380	Infiltration basins for highway, residential and industrial runoff	• High topsoil (10cm) retention of HMs and HCs with rapid concentration decrease with depth (to background at 50cm) • Substantial basal clogging of older basins • Average trapping efficiency is over 50% but HCs more mobile than HMs
Agence de L'Eau *et al*., 1999. (France) Report LROP (Also see Revitt *et al*., 2004) Del.5.1 EU DayWater project)	Porous asphalt surfacing with aggregate reservoir structures	• 35% - 80% solids removal; BOD/COD 45%-80% removal • Variable metal retention (Zn 16%-35%; Pb <75%) • Metal retention in top 50cm of surfacing/substrate; background levels maintained in subsoil
Balades *et al*., 1998 (France) Proc.NOVATECH04 Graie, Lyon	Porous highway and industrial surfacing with aggregate reservoir structure	• Average 4% discharge of total rainfall over 4 year period • Total solids reduction >60%; high reduction (60%-80%) for HMs • Mean HC outflows <0.02 mg/l
Puehmeier *et al*., 2004 (UK) Proc.NOVATECH04 Graie, Lyon	Porous paving using composite, floating geotextile and geocellular reservoir units	• Enhanced oil degradation but formation of emulsion flocs • Good HC spillage containment • Leaching of phosphate (>0.3 mg/l) from self-fertilising geotextile

The studies all indicate that infiltration systems give effective volume reduction but have rather more variable performance in terms of retaining and degrading urban runoff pollutants, especially soluble/colloidal species. This conclusion concurs with the observations of the Scottish SUDS database where there is a legacy of unsuccessful infiltration facilities (D'Arcy and Wild, 2002), although many early failures can be related to poor, inadequate design and/or lack of maintenance. Nevertheless, the Scottish SUDS Working Party have found that 50% of infiltration devices gave unsatisfactory performance with more than half being deemed to have failed (Schluter and Jefferies, 2005) and posing receiving waterbody risks. Lack of maintenance and post-construction runoff form unstabilised upstream areas were identified as the principal problems. Similar failures have been recorded for infiltration facilities in the US especially for trench and filter drain systems, and French experience of highway infiltration performance has also been impaired by clogging and toxic waste accumulation issues (Revitt *et al*., 2003).

Undoubtedly there is evidence that long term operation can be prejudiced by clogging and the accumulation of potentially hazardous basal sediments. In addition, there appears to be potential for metal and organic pollutant breakthrough, primarily associated with soluble and colloidal species as well as threats arising from limited buffering and dilution due to double-porosity effects in certain types of underlying strata. The infiltration of surface runoff from "hotspot" urban land use activities (garage forecourts, industrial yards, heavy goods vehicle parking areas, heavily-trafficked roundabouts etc.) present a particular issue and infiltration systems for such land uses should be avoided.

POLLUTANT COMPLEXATION AND LONG TERM PERFORMANCE

The concept that infiltration provides a "treatment " process is simplistic in that any mass removed from solution must either remain stored within the soil compartment or be leached out i.e. it really serves as a mass storage technology. Exchange capacity and sorption processes do not remain fixed over time or space, being highly dependent on local prevailing soil and solution conditions. Any changes in the water quality character infiltrating a site can potentially change the geochemical conditions, leading to the possible release of the sorbed mass in the subsoil.

The conventional design guidance for infiltration systems does not currently sufficiently consider the issue of "facilitated transport" within the unsaturated zone. Fundamental processes such as metal and hydrocarbon complexation (and subsequent transformation and transport) with dissolved and natural organic matter (DOM/NOM) remain highly speculative as do the unsteady hydraulics of repetitive, pulsed cycling in double-porosity strata. There is considerable geochemical evidence that non-ideal solute breakthrough (with long tailing and sharp initial wave fronts) is a normal consequence of natural porous media. NOM-metal complexation possesses retardation factors anything between 4 – 7 times lower than uncomplexed forms. Metal breakthrough can also lag the DOM breakthrough. This could suggest that the soil substrate "cleanses" the metal-DOM complex as it infiltrates, resulting in an initial breakthrough of metal-cleansed DOM followed by a later breakthrough of toxic metal-DOM complex as the soil becomes increasingly saturated with DOM. Size exclusion may also play some part in the co-transport of the metal with DOM. This metal complexation to soluble DOM can control metal mobility with partitioning to sorbed DOM playing only a relatively minor role.

If such processes are generally valid, then ultimate metal (and soluble hydrocarbon) breakthrough to the underlying groundwater is almost inevitable. It has been argued that the application of compost and other recycled materials as substrate media in porous paving and trench systems can be effective in removing monolayer soluble/colloidal metal species (Seelsaen *et al*., 2005) but they can also leach out high concentrations (>4mg/g) of DOC. In addition, existing colloid facilitated transport modelling assumes that the relevant partitioning mechanisms follow linear, equilibrium sorption kinetics. The use of such simple modelling assumptions would be inadequate to describe the NOM-metal complexation. Existing background metal concentrations in the underlying soil layers must be important considerations in infiltration design, since metal displacement will occur as a result of competitive adsorption/exchange, and/or dissolution effects posed by the multi-component system. Sub-soils containing concentrations in excess of 20 µg/g for copper and lead, 50 µg/g for zinc and 1 µg/g for cadmium should be avoided. In addition, organic carbon content should exceed 0.5% to improve metal attenuation, and minimum depth to any underlying unconfined aquifer should be at least 3m.

CONCLUSIONS

It is clear that infiltration devices intended for the treatment of urban surface water discharges should always have pre-treatment facilities e.g. settlement basins, swales, filter strips etc., in order to prevent basal clogging, and to reduce first-flush effects as well as strictly comply with the WFD not to direct polluting discharges to ground. The increased use of artificial media and cellular PPE units in substrate and reservoir structures of lined porous paving systems having perforated under-drainage may also help to reduce the threat to groundwater of soluble and complexed pollutant forms. However, the failure to detect potential pollutant concentrations in outflow drainage or in adjacent groundwaters does not necessarily imply long term operational acceptability over the full lifetime of an infiltration facility. Continued usage of direct infiltration may eventually lead to exceedance of the local sorption capacity, allowing pollutant plumes to move further down gradient and therefore potentially become a threat under the terms of the WFD. Where the groundwater table is subject to substantial variation of more than a metre or so, it may be necessary to use nested wells screened to various depths to reliably detect any pollution plume, which is likely in any case to be relatively thin in the region of infiltration.

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PROVEN AND PRACTICAL METHODS OF GROUNDWATER RECHARGE IN BUILT-UP AREAS IN THE USA

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Notes:

Session III

SUDS – PRINCIPLES AND DRIVERS

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ABSTRACT

The seven local authorities in the Dublin Region have now introduced new policies regarding drainage. These policies were produced under the Greater Dublin Strategic Drainage Study (GDSDS) and they are now also being applied outside the original study area. The policies include a commitment to the use of SuDS (Sustainable Drainage Systems) which are now required on all new developments both public and private. SuDS are a method of reducing or preventing excess stormwater runoff by mimicking natural hydrological behaviour. This brings benefits in terms of stormwater control, removal of pollutants and sometimes, the provision of amenity areas. Four new stormwater control criteria have been introduced under the GDSDS. These relate to water quality, flooding and the prevention of erosion. In achieving these objectives, the use of infiltration based techniques is critical, particularly in urban areas. There have been concerns expressed that the use of infiltration techniques might transfer pollutants to ground water but this will not happen in properly designed systems. The idea of SuDS is to prevent development from interfering with the natural hydrological cycle and this should benefit ground waters. While new techniques are always met with scepticism, the advent of SuDS will bring enormous benefits to all of our waters and the communities that depend on them.

SUSTAINABLE DRAINAGE SYSTEMS (SUDS)

To understand SuDS we must first consider the natural hydrological behaviour of a greenfield site. When rain falls on such a site, it normally soaks into the soil. Only where there is particularly heavy rainfall will some of it run off slowly over the ground surface to the nearest ditch or watercourse. Most pollutants are filtered through soils or broken down by bacteria.

When these sites are built on, much of the area becomes impermeable. With no soakage available, runoff is piped to the nearest watercourse or storm drain. Thus both the volume and rate of runoff can dramatically increase. In built up urban areas this runoff ends up in urban streams or existing pipes that were never designed for these loads. This may lead to flooding or increased overflows from combined sewers, neither of which is acceptable. Excess runoff also causes problems with increased surcharging of pipes. This causes pipe damage and maintenance problems and may lead to ex-filtration of foul sewage into the groundwater. Excess flows in combined systems cam also lead to increased costs for wastewater treatment. The potential to naturally remove pollutants is lost.

SuDS are defined by CIRIA as "a sequence of management practices and control structures designed to drain surface water in a more sustainable fashion than some conventional techniques". Using SuDS techniques, water is either infiltrated or conveyed more slowly to water courses via ponds, swales or other installations. A full list of SuDS techniques is included as an Appendix. The use of SuDS closely mimics natural catchment behaviour and results in attenuation of stormwater runoff and improved environmental performance.

STORMWATER CONTROL

Stormwater attenuation has been a requirement in Dublin since 1998 when Dublin City Council introduced it's Stormwater Management Policy. This limited runoff from developed sites to pre-development rates or, in simple terms, runoff from new development was not permitted to exceed 2l/s/ha with excess runoff being stored on site. In most cases, developers met their obligations by installing Hydrobrakes or similar devices to control runoff and underground concrete tanks to store excess water. This led to problems with maintenance, with cleaning of underground tanks being a particular health and safety issue. Building these tanks also had cost implications for developers who began looking at alternatives to underground tanks. Surface ponds were already being included in some developments for aesthetic or fire fighting reasons but the potential of these to provide stormwater control or environmental improvements was rarely exploited. SuDS devices offer alternative solutions while providing significant environmental benefits.

ENVIRONMENTAL ISSUES

Public interest in the quality of our surface water has never been higher and this is now being reflected in increasing political interest. There are significant problems to be addressed. The EPA Water Quality Classification for the Dublin region shows extensive pollution. 82% of waters in the Eastern River Basin District are at risk of not reaching "good status" by 2015. This is a requirement of the EU's Water Framework Directive so there is now significant pressure on local authorities to improve water quality.

Stormwater runoff from urban areas impacts on surface water quality. Firstly, storm runoff can carry pollutants such as oil, anti-freeze, animal and human waste, decaying leaves, grass or other waste matter to our surface waters. This is particularly critical in the case of the first flush, where material may have decayed for several weeks in dry weather before being washed into the watercourse. This is particularly serious when the baseflow in the watercourse is low, which would also be consistent with a long dry period. Secondly, surface water in older areas frequently drains into combined sewers. These were designed to carry dry weather flow and smaller storm events but to overflow to watercourses during more severe storms. In practice, many combined sewers now drain considerably greater areas than they were originally designed for and this extra load means that they can overflow in relatively minor rainfall events. The overflow of, admittedly dilute, foul sewage to a watercourse has obvious pollution implications.

The use of SuDS can help address both of the above issues by providing a control on stormwater volume and quality. Volume control reduces the number and severity of overflows from combined sewers. SuDS also have a direct bearing on water quality. Infiltration systems lead to pollutants being filtered out or broken down by bacteria. Swales encourage pollutants to settle out or be broken down naturally. Retention systems such as ponds also allow settlement and natural breakdown of pollutants via aquatic plants and other organisms.

AMENITIES

In addition to providing runoff and pollution control benefits, some SuDS can provide amenities to local communities. Ponds or wetlands can be visually attractive. They provide vital habitats for birds and other wildlife and recreational opportunities for local people. Local authorities are now obliged under the Biodiversity Convention to promote wildlife habitats and the introduction of wildlife into urban areas can only be positive.

GREATER DUBLIN STRATEGIC DRAINAGE STUDY (GDSDS)

As previously mentioned, Dublin City Council has had a stormwater control policy since 1998. This was also applied on an informal basis in the surrounding counties. This policy did succeed in reducing stormwater runoff but did nothing to address environmental concerns and it was believed that the runoff control process could be improved upon. The Greater Dublin Strategic Drainage Study (GDSDS) was established in 2001 to analyse the existing drainage system in the Greater Dublin Area and to make recommendations on future drainage policies. The study area included the functional areas of the four Dublin local authorities and parts of Meath, Kildare and Wicklow. The approximate extent and population involved in this area is 150,000ha and 1.2 million respectively. The value of the study was just over €10 Million. The study included hydraulic modelling of foul and storm drainage networks and of eight rivers in the area. A key outcome of the study was the recommendation of future regional drainage policies for the Greater Dublin Area. This included policies on New Development, Environmental Management, Climate Change, Infiltration/ Ex-filtration and Basements. The new drainage policies arising out of the Greater Dublin Strategic Drainage Study have now been included in the Development Plans for Dublin City Council and the surrounding local authorities. The use of SuDS is advocated under both the Environmental Management and the New Development policies. The Technical Document on New Development contains the following text on Sustainable Drainage Systems:

All new development shall incorporate SuDS facilities, unless the developer can demonstrate that SuDS is impractical due to site circumstances. Where SuDS cannot be provided, the developer shall provide alternative means of dealing with pollutants.

SuDS are now compulsory for all new development in the region. There are signs that this policy is also being implemented by local authorities outside the Dublin region. In the author's opinion, these policies, or something very similar, will be adopted countrywide in the near future.

The New Development document specifies four key stormwater control criteria that SuDS systems need to address.

River water quality protection: SuDS are required to protect water quality in our rivers and streams. This is done by either infiltrating the smaller rainfall events which are associated with most pollution problems or by treatment of runoff.

River regime protection: It is necessary to stop large sudden inflows of water into a stream as these can cause erosion and downstream deposition.

Level of service (flooding) for the site: To prevent uncontrolled flooding of the developers own site, particularly flooding of properties.

River flood protection: To prevent large additional volumes of water being discharged to the river and increasing the risk of downstream flooding.

SuDS techniques that use infiltration of surface water into the ground are the most suitable techniques for addressing all four of the above criteria. Infiltration techniques include the use of soakpits, swales, filter drains and permeable paving. Permeable paving is a system of laying individual bricks, either mechanically or by hand, on a compacted granular bed. Gaps between the bricks are filled with coarse gravel or stone such that any water falling on the surface can permeate through the gaps in the bricks to the layers below. Geotextiles or geogrids are used to give additional strength, to trap pollutants and to offer separation of layers. Generally the underlying layers contain either drainage stone or proprietary plastic systems offering high voids ratios to store water. Water can then infiltrate into the underlying ground or can be stored and allowed to runoff slowly to a watercourse. Even with no infiltration, there is commonly zero runoff for small rainfall events due to evaporation and soakage into hard surfaces. Runoff from roofs can also be stored/ infiltrated in the area beneath the permeable paving. Infiltration techniques are particularly useful in urban areas where ponds and similar SuDS systems can take up valuable development land. There is increasing pressure to maximise the development potential of all lands in urban areas particularly near public transport links so infiltration based techniques are seen as one of the most attractive SuDS systems for use in the Dublin area. Use of infiltration techniques also eliminates the perceived risk of drowning in ponds or wetlands but loses out on the amenity benefits these can provide.

Concerns have been expressed that during extreme rainfall events, storage features might become full and infiltration systems might become waterlogged, with the result that the system could no longer cope and surface flooding could occur. It is true that it would not be practically possible to design a SuDS system to protect against every possible flood event but conventional piped systems are not designed to do that in any case. Failure of a SuDS system should be more gradual than failure of a conventional system. The focus should thus be on flood routing to ensure that flooding is confined to green spaces or roads rather than properties.

The focus on infiltration of stormwater causes concern to some engineers who fear that raised ground water levels could effect foundations or road sub-bases. In reality filter drains have been used on roads projects for years without significant problems. What is important is that filter drains or soakways should be far enough from foundations to ensure that stability issues won't arise.

Infiltration techniques have the following benefits in relation to the four criteria mentioned above.

River water quality protection

Infiltration techniques prevent any runoff from small, short duration rainfall events. These make up the majority of rainfall events and are associated with most of the pollution entering our watercourses. The first benefit of infiltration is that it can prevent this pollution reaching the watercourse at all. In addition to this pollutants can be trapped on the surface of geotextiles and broken down by bacteria. There is extensive research to prove the effectiveness of this with regard to permeable paving. Infiltration techniques are particularly effective in dealing with the first flush effect referred to earlier in this paper.

River regime protection

Attenuation is required to prevent sudden large discharges of water into our rivers causing scouring and erosion of banks and subsequent downstream deposition. By intercepting runoff and infiltrating it, we can eliminate this effect.

Level of service (flooding) for the site

Infiltration systems are very effective in dealing with the short, sharp summer storms that cause most local flooding problems as they tend to have excess capacity at these times.

River flood protection

The key to river flood protection is to store large volumes of excess water generated by the new development. We can prevent runoff from arising by allowing it to infiltrate, or where runoff can't be prevented, infiltration systems with overflows can delay runoff to the extent that it is not released while the river is in spate, thus preventing downstream flooding.

SUDS AND GROUNDWATER

There is sometimes a perception that SuDS involves artificially draining surfaces into the ground water and that the effects of this on ground water have not been fully thought out. Generally speaking, this is very much a misconception. The idea of SuDS is to mimic the natural hydrological cycle. We are not trying to put extra water into the ground. We seek to ensure that development does not interfere with the natural pathway of land drainage. The ground will soak up whatever water it can and any excess will run off slowly to the local watercourse, just as happens on a greenfield site.

The benefits of allowing infiltration are numerous. We're all familiar with problems caused by ground shrinkage, especially where urban trees are taking up water that can't be replaced due to extensive loss of permeable ground. SuDS systems promote ground water recharge and can reduce the scale of this problem while maintaining the natural pore water pressure. Ground water levels are predicted to decrease due to global warming and extensive urbanisation could exacerbate this. Increased use of SuDS will help to reduce these impacts. This will benefit ground water and will also promote increased recharge of smaller urban streams. This is particularly important in Summer when low flows in urban streams can lead to serious environmental/ecological problems. The onus is now on local authorities to prevent these problems, especially with the advent of the Water Framework Directive.

While SuDS are promoted as an ideal solution to pollution of surface water, there are fears that pollutants could be introduced into ground water and that an 'out of sight out of mind' attitude might prevail. The quality of groundwater is often neglected in our cities where abstraction of ground water is rarely an issue. However, there is a growing recognition that pollution of ground water will eventually show up in our surface waters and the Water Framework Directive is increasing the pressure to tackle this issue.

While SuDS rightly seeks to increase the amount of water infiltrating into the ground, it must be remembered that water coming off developed sites is likely to carry more pollutants, particularly inorganic matter, than would be seen in the greenfield situation. If infiltration is carried out over a smaller area than before, then this increases the risk of either water volume or pollutant content being unacceptably high. Draining a newly developed area into a limestone aquifer via a sink hole is not SuDS. This would be wholly unacceptable and the pollutant content of surface waters and how we deal with those pollutants is a vital concern, particularly near aquifers that are used for abstraction.

It is important to realise that properly constructed SuDS can remove pollutants. Ponds and wetlands have a well recognised ability to do this but soakpits, swales and filter drains can also remove pollutants. Gross pollutants can be trapped on surface layers or in underlying clay before being broken down by bacterial action. Even artificial systems such as permeable paving will remove pollutants. Initially there is the obvious potential of the brick surface and underlying geotextile to trap silts and other pollutants. It should also be remembered that bacteria living on stone layers or geotextiles will organically break down oils and other hydrocarbons. The central message, then, is that SuDS can actually clean up surface water before discharging into the ground.

In some cases it may be felt that the pollution risk is too great or the aquifer too sensitive to allow infiltration of surface waters from developed areas. Impermeable clay or adjacent foundations may render infiltration systems unsuitable. This does not preclude the use of SuDS. Lined ponds or under-drained swales can be used. Permeable membranes can be laid beneath the storage area underneath brick paving. This will slow down the rate of runoff, clean up the surface water and still prevent any infiltration into the ground. Natural leakage and evaporation will prevent a significant runoff from a large number of rainfall events even with zero infiltration.

CONCLUSION

The use of SuDS is now compulsory in the Dublin region and this is spreading to other parts of the country. SuDS bring enormous benefits in terms of stormwater control, removal of pollutants and provision of amenities to local communities. It is unlikely that the requirements of the Water Framework Directive can be met without greatly increased use of SuDS systems. There are fears that SuDS may clean up our surface waters by transferring the pollutants to ground waters, but this should not be the case. Proper design is necessary to ensure that ground water, particularly aquifers, is protected. Overall, the philosophy is not to introduce new water into the groundwater, but rather to prevent development from interfering with the natural hydrological cycle. The ground will soak up whatever water it can and any excess will run off slowly to the local watercourse, just as happens on a greenfield site. The use of SuDS will provide for ground water recharge and help to mitigate the lowering of ground water levels that climate change experts are currently predicting. This will have benefits for all sections of society. I hope that hydrogeologists will see the advent of SuDS not as a threat to groundwater but as a great benefit to all our waters and the plant, animal and human communities that depend on them.

NOTE: Technical documents relating to the new policies can be downloaded at:
http://www.dublincity.ie/shaping_the_city/environment/drainage_services/greater_dublin_strategic_drainage_study/

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APPENDIX

TYPICAL SUDS INSTALLATIONS

Permeable Pavements	Use of porous asphalt, porous paving or similar concepts to reduce imperviousness thus minimising runoff. Runoff infiltrates to a stone reservoir where some breakdown of pollutants occurs before controlled discharge to a drain or watercourse or direct infiltration.
Filter Drains	A gravel filled trench, generally with a perforated pipe at the base which conveys runoff to a drain or watercourse. These provide attenuation and trap sediments.
Infiltration Trenches/ Soakways	Gravel or rock filled pits or trenches designed to store runoff while letting it infiltrate slowly to the ground. Provide treatment of runoff through filtration, absorption and microbial decomposition.
Bio-Retention	These devices are depressions back filled with sand and soil and planted with native vegetation. Provide filtration, settlement and some infiltration. Typically under drained with remaining runoff piped back to the drainage system or watercourse.
Swales	Grass lined channel designed to convey water to infiltration or a watercourse. Delays runoff and traps pollutants via infiltration for filtering effects of vegetation.
Detention Basins	Dry vegetated depressions which impound stormwater during an event and gradually release it. Mostly for volume control but some pollutant removal achieved via settlement of suspended solids and some infiltration.
Retention Ponds	Permanent water bodies which store excess water for long periods allowing particle settlement and biological treatment. Very effective for pollutant removal but limited to larger developments. Have high habitat and aesthetic benefits.
Stormwater Wetlands	Like retention ponds but with more vegetation and less open water area. Excellent for pollutant removal. Also provide aesthetic and habitat benefits.

GROUNDWATER ISSUES ARISING FROM SUSTAINABLE URBAN DRAINAGE POLICIES

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Notes:

ROAD RUNOFF IN IRELAND – IMPLICATIONS FOR GROUNDWATER AND MANAGEMENT IN A SUD SYSTEM

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ABSTRACT

The expanding economy in Ireland in recent years has resulted in extensive new development of roads together with a large increase in traffic densities. Impacts on the environment have also become of increasing concern and emissions from highways and transport are a primary source of potential contamination. Highway runoff has been investigated as one such emission, as part of a research project for the EPA and National Roads Authority. While the intended discharge from conventional drainage from motorways is conventionally surface water, the measured performance of drainage systems in use suggest that groundwater may be an equally important receptor in Irish conditions.

INTRODUCTION

The construction of motorway grade roads in Ireland has developed rapidly in recent years, along with the economy, and it has accelerated under the National Development Plan 1000-2006. At last report (NRA, 2005b), the road network in Ireland included 2740 km of national primary routes of which 192 km were classified as motorway (7% of total). If dual carriageways are included, some 17% of the national primary routes might be classified as 'highway'. Concomitant with the development of the road network has come an increase in traffic densities. With vehicle numbers reaching some 1.8 million in Ireland, traffic counts (AADT – Annual Average Daily Traffic) on busy routes have reached over 93000 per day on the M50 at the Red Cow interchange and around 24000 on the N6 Athlone bypass (NRA, 2005a). While these densities are still not large compared to many other European locations, they do carry environmental implications.

The combination of roads and traffic densities represent an ongoing potential hazard in terms of emissions to the environment, exacerbated by prevailing meteorological conditions. The vehicle engine, as a combustion system burning fuels, emits exhaust gases and liquids (water, unburned hydrocarbons). Frictional resistance between the tyres and the road surface, combined with wind and rain, results in sediment/runoff from the road. A variety of detritus may fall from wear of the vehicles themselves and their engines in transit. The road infrastructure may also deteriorate with time (e.g. crash barriers) and the application of de-icing salts in winter can result in significant emissions. Nevertheless, emissions are typically airborne (gases and aerosols) or water borne (via rainfall and runoff). Key indicator compounds for road and vehicle emissions include polyaromatic hydrocarbons (PAHs), heavy metals (Cr, Pb, Cu, Cd and Zn), phosphorous and chloride.

In this context, the Environmental Protection Agency (EPA) and the National Roads Authority (NRA) initiated a research project to investigate the nature of the water borne emissions (i.e. runoff) from highways in Ireland and their effect on principal receptors. The work has been undertaken by teams from UCD and TCD involving hydrologists, engineers and biologists. By conventional design, drainage from highways in Ireland has concentrated on removal of excess water with a view to maintaining both safety for vehicle movement as well as the geotechnical integrity of the road construction. Any improvement in the quality aspects of the drainage water have tended to be a beneficial side effect. However, under the increasing traffic densities on major new roads, the chemical quality of the runoff was seen to be a potential issue, especially as the intended discharge point for most highway drainage is a nearby surface water course. Measurement of road runoff and its quality at selected sites on new Irish highways was undertaken and the results compared with similar European studies. Specific attention was paid to the effects on the quality of receiving streams. In the light of the results of the analyses of runoff quantity and quality, specific treatment options were evaluated and a trial site initiated for a constructed wetland as an improved method of road drainage management, conforming to Sustainable Urban Drainage System (SUDS) practice. Moreover, under the Water Framework Directive (EU, 2000), a highway may constitute a possible contaminant source for both surface water and groundwater. The relevant pathways and their susceptibilities to migrating contamination needed to be assessed in an Irish context.

SOURCES OF CONTAMINATION IN ROAD RUNOFF

Over the period of the design life of a road, both the road itself and the traffic that use it are the primary sources of compounds with potential to contaminate the environment. The principal drivers for moving these potential contaminants to environmental receptors are the traffic densities involved and the accompanying climate (e.g. rainfall and hydrological regime).

An integrated study reported by TRL (2002) in the UK represented a first attempt in Europe to investigate the complete source-pathway-receptor framework in the context of potential pollution emanating from roads. On the basis of a mass balance approach, the study attempted to assess the quantities of unregulated compounds that are released, what proportion of these emissions enter the local roadside environment, the relative importance of each transport mechanism and where in the environment these compounds are likely to occur. The study was based on 14 roadside sites covering 7 countries including the UK, across a range of climates and traffic densities in Europe, over a period of some 30 months. While the emission rates from vehicles and from the deterioration of road infrastructure were estimated using a mass balance approach, the concentrations of contaminants in the roadside environment were sampled and measured. Typical traffic levels on highway sites in Ireland would fall towards the lower end of the ranges examined in this EU study. However, the results from the study, as shown below, indicated that ‘transfer rates’ for pollutants to the roadside environment varied widely. Nevertheless, typically, PAHs were around 10% of emissions although metal transfer rates were much higher, often exceeding 100%, indicating possible external airborne deposition. Chlorides were commonly associated with de-icing activity although commonly over 60% of applied mass was transferred to the roadside environment. Significantly, up to half of the pollutants that were deposited in the roadside environment arrived via the aerosol/airborne pathway rather than by conventional rainfall/runoff. Care is therefore needed in defining source contamination related to traffic and roads. What is deposited on the roadside environment becomes a secondary source for possible onward migration to a hydrological receptor. The contaminant load carried by road runoff itself may only represent a proportion of the total deposition in the roadside environment.

Table 1: Estimated emission rates from roads and vehicles, EU data

POLLUTANT	CALCULATED EMISSION RATES* (g/km road/yr)	TRANSFER RATES (G/KM ROAD/YR)*	
		HIGHWAY RUNOFF	AERIAL DISPERSAL
Total PAHs	65 – 721	<1 - 7	<1
Cd	1 – 10	<3 - 6	<1 - 35
Cr	14 – 162	<1 - 30	<1 - 156
Cu	9,248 – 108,893	<1 - 1,125	<26 - 539
Pb	7,391 – 110,984	14 - 1115	<10 - 541
Zn	2,479 – 61,369	111 - 8091	<98 - 2,447
Cl (kg/km/yr)	1,225 – 15,249	<1 - 9,261	<1 - 2,523

* one carriageway, downwind side only (adapted from TRL, 2002)

HIGHWAY RUNOFF AND RECEPTORS IN IRELAND

The focus of the present study has been on the effects of highway runoff on potential receptors. Conventional highway drainage design in Ireland involves collection of the runoff from the road surface for ultimate delivery to a surface water course. Such collection and discharge systems commonly take three forms: a French (filter) drain system along the margins of the carriageway, a kerbed system using gullies and piped drains and an 'over the edge' system in which surface water drains into the grass verge and is allowed to drain down the side slopes and into ditches at the base of an embankment. Petrol interceptors and sedimentation ponds are often incorporated as part of recent road design. The initial objective of the study was therefore to identify a series of highway-stream crossings where road runoff was being discharged through a collection system and to record any identifiable changes in biochemical streamwater and sediment quality which could affect aquatic ecology.

Fourteen stream crossing sites (Table 2) were selected from a total of 46 investigated and comprehensive analyses were conducted of sediment quality, vegetation, biota and overall stream quality upstream and downstream of the highway crossing. Overall, there were few significant effects that could be directly attributable to highway runoff discharge.

Table 2: Sites selected for water, sediment and biological sampling.
**Control sites **2004 Data*

Site No.	Site name	Monitoring period	.AADT/HGV%**	OS NGR	Drainage system
S1	Tributary of Slane River at Dunbauin Bridge (N7)	2002-2003 and 2005	50,729/12.8	N 970 242	Filter drain & over edge
S2	Hartwell River at Tobernavore Bridge(N7)	2002-2003 and 2005	50,729/12.8	N 925 220	Over the edge
S3	River near Rowanstown at Maynooth By-pass (M4)	2003 and 2005	39,088/7.5	N 934 363	Filter drain
S4	Lyreen River at Maynooth Bypass (M4)	2002-2003 and 2005	39,088/7.5	N 913 370	Filter drain
S5	White River at Dunleer Bypass (M1)	2003 and 2005	24,369/12	O 135 260	Filter drain
S6	Owendoher North East of Newtown (M50)	2003 and 2005	43,624/6.0	O 135 255	Filter drain
S7	Tributary of Painstown River near Bohrenphilip (N7)	2002-2003 and 2005	50,729/12.8	N 955 235	Over the edge
S8	Morell River near Johnstown (N7)	2002-2003 and 2005	50,729/12.8	N 919 216	Filter drain
S9	Lackan River, tributary of the Pollaphuca Reservoir*	2003	N/A	O 012 110	N/A
S10	Glen O' Downs Stream (N11)*	2005	34,540/6.8	O 266 105	Filter drain + kerb & linear drainage
S11	Glenview Stream (N11)*	2005	34,540/6.8	O 256 117	Filter drain + kerb & linear drainage
S12	Doonfin Lower Tributary (N59)*	2005	2,513/6.8	G 508 325	Over the edge
S13	Tributary of Ardnaglass Stream at Carrowree (N59)*	2005	2,513/6.8	G 544 330	Over the edge
S14	Spaddagh Tributary (N5)*	2005	5,875/11.3	M 370 979	Over the edge

Stream sediment was sampled at three locations, both upstream and downstream of the outfall at each site. Two sampling campaigns were undertaken, 2 years apart. For heavy metals (Cu, Cd, Pb and Zn) in the first sampling round, the means of all the downstream concentrations in μg/g were not significantly higher than the upstream sites at $p=0.05$. The sampling two years later showed only slightly higher variability between upstream and downstream with significant differences occurring on the M50 site and the N59 where a large road catchment was draining into a small stream. PAH content of sediment, however, did show large changes in concentration at sites on the M50 and N11.

At two sites on the N7 and M4, there was some evidence of accumulation of heavy metals in the roots of Apium nodiflorum (European marshwort) near the discharge points but not to anomalous levels. Macroinvertebrate sampling studies (Burns, 2004) showed that the mean number of taxa at a site ranged from 14 to 47 although, with one exception (M4) there were no significant differences in taxon richness between upstream and downstream sites. Moreover, there were no major differences in biological quality according to the EPA Q-value rating system. Finally, there was no noticeable difference in the condition of fish from the various sites. Similarly, the analyses of the fish did not reveal a negative impact of road-runoff on these biota. It was a common experience in this study that most sites were already impacted by nutrient/organic pollution making it extremely difficult to isolate possible effects of the road-runoff from other pollution effects.

HYDROLOGICAL PROCESSES IN HIGHWAY RUNOFF

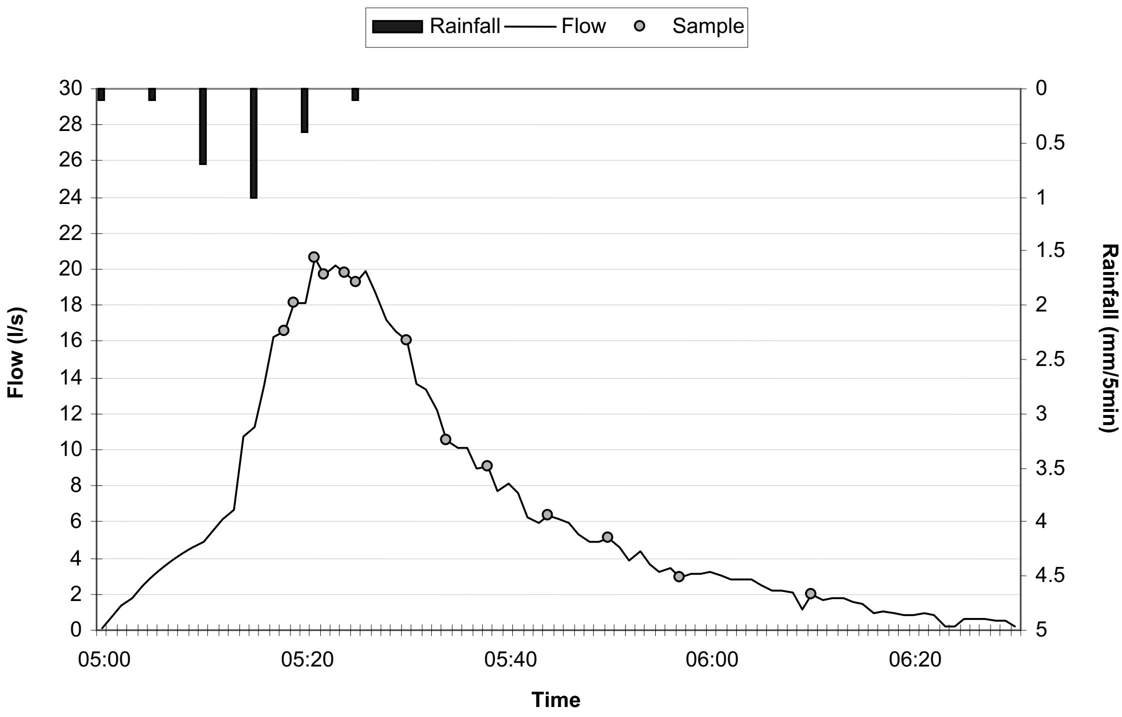
A parallel part of the study was to analyse the processes underlying the runoff from a highway pavement and its pathway to a potential receptor. In this context, 5 sites were chosen on motorways with reasonably high traffic densities and incorporating the two main drainage types: a kerbed system and filter drains (Table 3).

Table 3: Road runoff monitoring sites

Site No.	Site Description	Road	Drainage Area, m^2	Type of Drainage System	OS NGR
1	Kildare Bypass	M7	14184	Kerb and gully	N 667 113
2	Maynooth Bypass	M4	9760	Filter drain/Over the edge	N 913 370
3	Maynooth Bypass	M4	1100	Over the edge	N 913 370
4	Monasterevin Bypass	M7	9600	Filter drain	N 569 059
5	Monasterevin Bypass	M7	11368	Kerb & gully; wetland	N 569 059

Flow monitoring was undertaken for each site along with runoff sampling for chemical analysis. Automatic samplers and flow velocity sensors were deployed within the pipework draining the carriageway at each site. Rainfall was also measured and logged for each site using automatic tipping bucket raingauges. A conventional measure of the hydrological behaviour of an urban drainage site is the ‘runoff coefficient’ (RC): the proportion of incoming rainfall that is caught in direct runoff in the drainage system.

Figure 1: Kildare, N7: 30 June 2005 Typical storm event of 110 from this site



The consistency of the results at each site with such a large difference in runoff coefficient indicated that with filter drainage, a significant part of the runoff was not reaching the surface water receptor. Moreover, both sites, on the N7, 15km apart, were under the same rainfall regime. The Maynooth site (N4) gave a runoff coefficient of 0.46 but the drainage system was more complex, with indications of groundwater flow entering the pipe network upgradient of the outfall and measurement point. The implications of these differences in runoff coefficient are that a significant part of the highway runoff is being 'lost' in roadside percolation to the subsurface. The design of filter drains would support this hypothesis as concrete pipe sections are laid at the base of a trench beside the roadway and coarse aggregate is infilled above. The whole trench and pipework is encased in a wrapping of 'Terram' filter geotextile which is typically folded over at the top below a top dressing of aggregate. This geotextile has the effect of acting as an effective filter which can clog quickly with the sediment load from the road runoff, having the net effect of diverting a portion of the drainage water away from the intended concrete drain. Excavation of a ten year old filter drain system on the N7 dual carriageway near Naas provided strong evidence of this process of drain clogging. Under such drainage design, the implication is that the unintended receptor for much of the drainage is groundwater. Currently, it is estimated that some 65% of motorway drainage in Ireland uses the 'French' filter drain system.

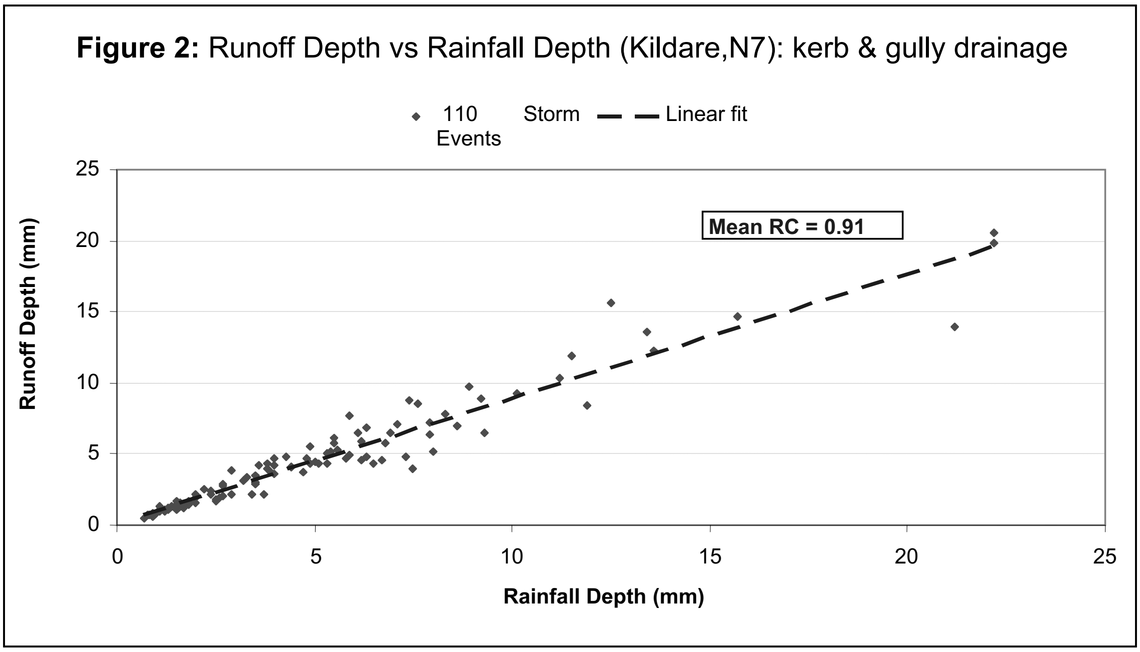


Figure 2: Runoff Depth vs Rainfall Depth (Kildare,N7): kerb & gully drainage
110 Storm Events
Linear fit
Mean RC = 0.91
Runoff Depth (mm)
Rainfall Depth (mm)
0
5
10
15
20
25

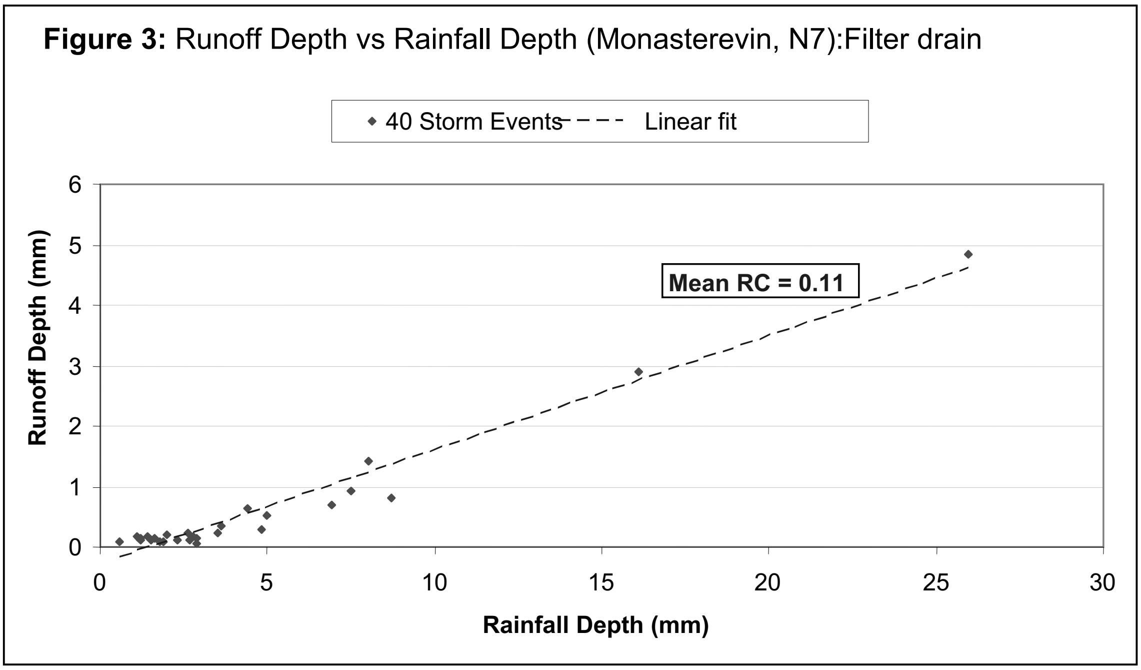


Figure 3: Runoff Depth vs Rainfall Depth (Monasterevin, N7):Filter drain
40 Storm Events
Linear fit
Mean RC = 0.11
Runoff Depth (mm)
Rainfall Depth (mm)
0
1
2
3
4
5
6
10
15
20
25
30

RUNOFF QUALITY

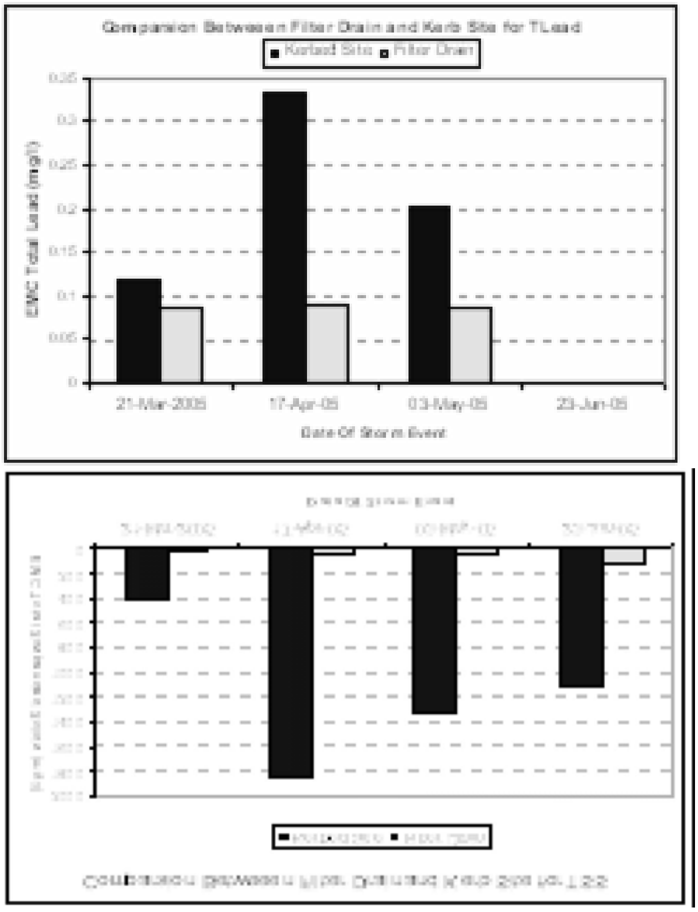
The quality of the runoff was measured at discrete intervals during storm events and ‘event mean concentrations’ (i.e. flow weighted concentrations) determined for a range of typical contaminants.
As for the runoff itself, the drainage system was found to have a strong effect on the quality of the discharged drainage water. The ‘true’ quality of the runoff is represented by concentrations measured at the point of discharge from the road surface and Table 4 gives a range of such concentrations measured at the Kildare kerbed site on the N7.

Table 4: Maximum, Minimum and Mean contaminant values
(30 storm events at Kildare site, N7, kerb and gully drainage)

Substances	Minimum Value	Maximum Value	Mean Value
Total Suspended Solids (mg/l)	4	3325	425.48
Total Organic Carbon (mg/l)	0.75	47.93	5.77
Chloride (mg/l)	0.87	17.44	4.25
Total Phosphate (mg/l)	0.029	3.00	0.46
Total Copper (mg/l)	0.008	0.393	0.0895
Dissolved Copper (mg/l)	ND	0.031	0.011
Total Zinc (mg/l)	0.048	2.36	0.461
Dissolved Zinc (mg/l)	ND	0.045	0.035
Total Cadmium (mg/l)	ND	0.02	0.008
Dissolved Cadmium (mg/l)	ND	0.006	0.0017
Total Lead (mg/l)	0.041	0.485	0.098
Dissolved Lead (mg/l)	ND	0.05	0.024
Total PAH (µg/l)	<0.01	84.79	5.29
Acenapthene(µg/l)	<0.01	1.183	0.038
Acenaphthylene(µg/l)	<0.01	0.205	0.035
Anthracene(µg/l)	<0.01	1.749	0.158
Benzo(a)anthracene(µg/l)	<0.01	8.147	0.376
Benzo(b)+(k)fluoranthene(µg/l)	<0.01	7.029	0.343
Benzo(ghi)pyerylene(µg/l)	<0.01	2.936	0.141
Benzo(a)pyrene(µg/l)	<0.01	4.789	0.233
Chrysene(µg/l)	<0.01	10.727	0.544
Dibenzo(ah)anthracene(µg/l)	<0.01	0.975	0.073
Fluoranthen(µg/l)	<0.01	20.57	1.123
Fluorene(µg/l)	<0.01	0.644	0.030
Indeno(123cd)pyrene(µg/l)	<0.01	2.545	0.122
Naphthalene(µg/l)	<0.01	3.048	0.479
Phenanthnene(µg/l)	<0.01	15.082	0.729
Pyrene(µg/l)	<0.01	13.319	0.861

These results as measured and as determined in terms of event mean concentrations are similar to those of other European studies for comparable traffic densities. However, significant attenuation is provided by the filter drain system in common use in Ireland. Two examples of the comparison between kerbed and filter drain sites for suspended solids and lead are given in Figure 4. The implication is that not only may the runoff be diverted to the subsurface by the drainage system but a significant part of the load carried by the runoff is also being diverted from the intended surface water receptor.

FIGURE 4: Comparison of total lead concentrations and suspended solids on N7 in runoff between kerbed site and filter drain discharge, 3 storm events.



The nature of the load carried by the runoff, principally heavy metals and hydrocarbons, implies that there is likely to be a strong correlation between suspended sediments and these concentrations. There is strong likelihood that the metals will be adsorbed by sediment and/or soil and to a lesser extent the same fate applies to the hydrocarbons although the latter may also be subject to biodegradation once trapped. Again the excavated ten-year old filter drainage system supported this hypothesis with high values of hydrocarbon concentrations being measured in soil adjacent to the filter drainage system. However, measurements of metal concentrations in runoff at the Kildare site indicated that significant proportions of the concentrations were in dissolved form rather than adsorbed on particulates at the point of sampling. The EU study lead by TRL (2002) included direct investigation of groundwater in conjunction with road runoff and found that metals, when discharged into the subsurface tended to be trapped, depending on soil conditions but it was more likely that hydrocarbons would reach groundwater as a receptor, occasionally in high concentrations on the sites studied. The present study was not able to investigate groundwater directly, although the evidence indicates that it remains a probable receptor, depending on subsoil and drainage conditions in the vicinity of the road.

SUDS

The aim of sustainable drainage systems is to mimic natural drainage systems and to manage the quality aspects of runoff before discharge to the environment. In this respect, a final stage in the project was to suggest and implement alternative drainage systems which might avoid the possible impacts, in this case, likely to be on groundwater. In this context a wetland for the receipt of direct, kerbed runoff was built on the N7 Monasterevin bypass, approximately 30m x 15m and approximately 0.5m depth, planted with Typha and Phragmites. It is constructed on the Monasterevin site for which there is baseline data on both runoff quantity and quality. Data so far collected indicates that the wetland is performing to design and will act as both a flow regulating device/detention pond as well as a quality attenuation mechanism. Removal rates for both suspended sediments and metals are over 80%. Nevertheless, such a mechanism remains essentially a detention mechanism for such contamination, and in the long term there will be maintenance requirements. Nevertheless, as a sustainable drainage system, it is clearly more advantageous than the current filter drain systems and is likely to have a more pragmatic mitigation effect in terms of groundwater protection under the Water Framework Directive.

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CONSTRUCTED WETLANDS – HYDRAULIC DESIGN – SUSTAINABLE DRAINAGE & TREATMENT SYSTEMS

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ABSTRACT

Constructed wetlands can be employed for flood retention or wastewater treatment functions. This paper provides information on the development of a procedure for the design of constructed wetlands principally for the treatment of agricultural wastewater generated on dairy farms. Equations were derived based on international models. Field data were provided by Waterford County Council for 13 existing wastewater treatment surface flow constructed wetland systems in the Annes Valley. The performance data show that a degree of wastewater treatment is achieved. These data were used for derivation of sizing equations. The proposed design equations define an area required to treat a characteristic agricultural wastewater to a standard that conforms to urban wastewater (UWW) discharge criteria (DELG, 2001). It is proposed that wastewater wetlands are preceded by a balancing pond. Mass balances relating to hydrological and phosphorus dynamics were investigated. The applicability of a constructed wetland is site dependant and discharge options and groundwater protection responses must be considered.

INTRODUCTION

Natural wetland systems are known to provide beneficial functions in terms of renewing natural resources and protecting aquatic and terrestrial ecosystems (EC, 2003). Wetlands retain waters and associated pollutant loads. Constructed wetlands aim to mimic the functioning of natural wetland systems. In Ireland, constructed wetlands have been employed for both flood retention components of SUDS projects (e.g. GDSDS – Tolka Valley) and wastewater treatment facilities for both agricultural (e.g. Teagasc, Duchas/OPW) and domestic wastewaters (e.g. Healy & Cawley, 2002). A catalogue of some Irish constructed wetland systems is provided by Otte (2005). The principal contaminant treatment process in a wetland is related to solids and the physical settlement of particulates. It is beyond the scope of this paper to provide literature review on the treatment functions and general issues related to constructed wetlands. There is a wealth of information available relating to the constructed systems (e.g. Cooper, 1990; Hammer, & Knight, 1994; Kadlec et al., 2000). In Ireland, much work has been done by a team that promotes a specific type of constructed wetland termed 'ICW's' (Integrated Constructed Wetlands) under the auspices of the NPWS, DEHLG. The NPWS team stress the importance of ecological and habitat functions as well as wastewater treatment functions. ICWs are surface flow wetland systems. Trinity College was commissioned by this group to develop a procedure for the design of constructed wetlands principally for the treatment of agricultural wastewater generated on dairy farms. Field data were provided by Waterford County Council for 13 existing ICWs in the Annes Valley.

This paper presents a snapshot of TCD the design equations developed, hydrological and mass balance determinations. Phosphorus retention characteristics were analysed by mass balance on the performance of some monitored wetlands in order to understand the dominant processes which might be expected in each pond and, in particular to assess the likely accumulation rates for P and their implications for maintenance. It is recognised that the field data used to develop the design equation are not ideal – many hydrological components are missing and the spot sampling technique was employed. In addition data pertains to a specific region of Ireland and in this regard, the design protocol remains to be validated with data from ICWs in other regions before being adopted nationally. Another facet of the TCD work was to study design and protocol concepts that should be applied. This work is based on international experience, as detailed in Kadlec et al. (2000) and with respect to dairy wastewater treatment wetlands, in particular, as provided by Knight et al. (2000), regarding the design of constructed wetlands for treatment of agricultural wastewaters. A protocol document was generated by the TCD team. Issues of import such as site assessment, primary treatment requirements, sizing equations, containment of wastewater, inlet and outlet structures, planting, commissioning, maintenance and monitoring were considered.

SITE SUITABILITY & SITE ASSESSMENT

With regard to Site Assessment the NPWS ICW team support the application of current site assessment techniques and subsoil classification (British Standards) as employed for on site wastewater treatment systems (i.e. EPA, 2000). It is acknowledged that the number of trial pits and percolation tests must be increased as a function of the area employed for the treatment system.

The TCD design team also highlight the following:

- Aim: to test for suitability for land disposal of effluent and/or need for pond lining/ reworking.
- First pond will be detention/settlement lagoon, $K=10^{-9}$ m/s and up to 2m deep with stepped berms for safety
- T-test, if necessary, serves to indicate the need for upgrading substrate to requisite 10^{-8} m/s.
- General ecological suitability (e.g. exposure, appropriate plant species).

Site Assessment should be completed in two stages

1. Outline suitability – what is likely discharge point and is area for ponds available and geology generally appropriate?
2. Gather data for design purposes including subsoil profile descriptions, percolation characteristics, hydrological and meteorological data

These issues are set in the context of the risk model that is familiar:

RISK = PROBABILITY OF AN EVENT X DAMAGE
PROBABILITY = fn (K,t) and the nature of the hazard (effluent)
$K = 10^{-8}$ m/s, thickness >0.6m
DAMAGE = fn (aquifer category)

CONTAINMENT OF WASTEWATERS

European Union guidelines (Cooper, 1990) suggest that treatment wetlands must be artificially lined unless in-situ soil hydraulic conductivity of 10^{-8} m/s can be proven. If one considers that hydraulic conductivity measurements should be tested in-situ the issue then arises on how does one test for 10^{-8} m/s? One solution currently being proposed is to employ laboratory particle size analysis to relate to hydraulic conductivity. In the case of an in-situ soil hydraulic conductivity of 10^{-8} m/s it may be sufficient to 'puddle' the soil to seal. In cases where in-situ soils natural hydraulic conductivity is less than 10^{-8} m/s, either an artificial membrane liner or an engineered clay liner must provide groundwater protection. The engineered clay liner should be a soil of appropriate lining composition. The Earth Bank Tanks design team (Gleeson & Skully, *pers. comm.*) suggest the following methodology for installing a clay liner: total clay depth should be 0.5m deep, laid in 5 layers of 100mm each, each layer must be traversed four times, twice in each compass direction by equipment of greater than or equal to 22 tonnes.

LEAKAGE

The chosen speciality of each professional leads to consideration of different aspects of constructed wetlands as of greater or lesser degrees of importance. For some, the ecological and habitat issues are paramount. For others, the degree of engineering that must be applied to render these systems appropriate as wastewater treatment facilities is crucial. For the hydrogeologist and environmental engineer/scientist, perhaps the focus is the issue of recharge of treated wastewater to groundwater.

Consider the following:

- The average size of constructed wetland in the Anne Valley was 9000 m^2, approximately.
- Consider a 10^{-8} m/s leakage rate.
- The calculated infiltration volume is 7.8 m^3/d.
- The issue then arises – what is the hazard posed?

A biofilm may form and act to improve quality but flow rate likely to be significant if subsoil allows. cf 'T test' value of 50 is equivalent to vertical velocity of 260 m/yr.

HYDROLOGICAL BALANCE

An average water balance was completed for one wetland system in Anne Valley that was appropriately instrumented to collect automatic flow data. This is presented in Table 1. These data relate to a wetland system that is perhaps not contained as per methods proposed.

Table 1 Average annual water balance for an Anne Valley constructed wetland 2003- 2004.

	cell 1	cell 2	cell 3	cell 4
Measured inflow (m^3/year)	3889	6954	7061	5667
Individual cell area (m^2)	1208.2	1906.4	2125.7	2435.4
Rainfall contribution @ 1.109m (m^3/year)	1339.9	2114.2	2357.3	2700.8
Evaporation @ 0.443m (m^3/year)	535.2	844.5	941.7	1078.9
Measured outflow (m^3/year)	6954	7061	5667	420
Calculated infiltration volume (m^3/year)	-2260.4	1162.7	2809.7	6868.9
Calculated infiltration rate (mm/day)	-5.126	1.671	3.621	7.727
Volume contributed to groundwater (m^3/day)	-6.193	3.185	7.698	18.819

GROUNDWATER PROTECTION RESPONSE

Groundwater protection schemes must be considered. The generic groundwater response matrix (DEHLG, 1998) is a familiar tool to all hydrogeologists and those working within the sphere of environmental engineering. How does one build on this for the specific question of site suitability for constructed wetlands? One proposed methodology was developed by Limerick County Council in consultation with GSI and TCD, as shown in Table 2. The 'Priority' data were developed for a situation in a particular area and are therefore not to be taken as generic or universally accepted.

Table 2 One particular proposed groundwater response matrix (Limerick County Council)

VULNERABILITY RATING	SOURCE PROTECTION AREA		RESOURCE PROTECTION AREA Aquifer Category					
			Regionally Important (R)		*Locally Important (L)*		*Poor Aquifers (P)*	
	Inner (SI)	*Outer (SO)*	*Rk*	*Rf/Rg*	*Lm/Lg*	*Ll*	*Pl*	*Pu*
Extreme (E)	1	1	3	3	3	3	3	3
High (H)	2	2	3	3	3	3	3	3
Moderate (M)	2	2	3	4	4	4	4	4
Low (L)	2	2	4	4	4	4	4	4

Priority 1: Not appropriate site for an ICW, unless it can be shown by a detailed site investigation (usually including drilling of monitoring wells) that there is no significant impact and that the likelihood of future impact is minimal.
Priority 2: Requires a) a walk-over survey, b) trial pits to a minimum depth of 2.0 m below the depth of the ponds and c) an assessment of existing water quality, particularly nitrogen and microbial pathogens. Monitoring boreholes might be required.
Priority 3: Requires a) a walk-over survey and b) trial pits to a minimum depth of 2.0 m below the depth of the ponds.
Priority 4: Requires a walk-over survey.

TCD CONSTRUCTED WETLAND DESIGN METHODOLOGY

The TCD proposed design equation defines an area required to treat a characteristic agricultural wastewater, that from dairy farm washings and yard runoff, to a standard that conforms to urban wastewater (UWW) discharge criteria (DELG, 2001). In this regard, the design is only valid for dairy farms and does not cater for any other influent source, such as silage effluent or slurry. The influent concentration data provided for the farms in the Annes Valley, in association with data available for dairy wastewater strengths from farms in county Cork, suggested a characteristic dairy farming influent wastewater strength shown in Table 1 on which the ICW design equations are based. The design UWW effluent standards (DELG, 2001) were employed as discharge effluent constraint concentrations (also shown in Table 3).

Table 3 Characteristic dairy wastewater influent (County Waterford and Cork farm data) and effluent strengths employed in TCD designs.

Parameter	**Influent (mg/l) Characteristic Dairy Wastewater**	**Effluent (mg/l) (UWW standards: DELG, 2001)**
MRP	22	1
NH4-N	185	1
SS	354	15
BOD	1129	10
COD	2607	70

Authorities must assess the relevance of UWW discharge criteria to ICW effluent discharge. Designing an ICW for an effluent strength according to UWW discharge criteria dictates that the receiving surface water body must have the appropriate dilution capacity to receive the ICW effluent, otherwise stricter effluent standards must be adopted and these in turn will increase the footprint area of the ICW. Discharges to groundwater that exceed $5m^3$/d also require licence.

The TCD design for constructed wetland treating dairy farm wastewaters consists of an initial "balancing" pond followed by at least three further ponds in sequence culminating in an exit flow at the discharge point.

PRETREATMENT BALANCING POND DESIGN

The success of a treatment wetland depends on an upstream primary lagoon to balance the flow and loads. An efficient initial pond treatment for such higher strength agricultural pond effluents is generally required for efficient wetland treatment (Tanner and Sukias, 2003). Such lagoons are normal for other systems treating agricultural effluents for example waste stabilisation ponds (facultative ponds systems), primarily to remove suspended solids and balance flow etc. Typical depths are 3-5m and average hydraulic retention times ~100 days (Sukias et al., 2003, Craggs et al., 2003). Hence, an empirical design equation was formulated as:

$$AL = [(0.36*AY) + (3.2*CN)]/20 \qquad \textit{(Equation 1)}$$

where:
AL = *Lagoon surface area (m^2)*
AY = *Yard Area (m^2)*
CN = *Number of milking cows*

The design equation for the balancing pond was arrived by consideration of the worst-case hydraulic loading to the system in a one-hour period. A one in one year storm event of 15mm/hr gave an instantaneous flow rate from the storm of 0.36*AY (m^3/d). Additional hydraulic loading could occur, simultaneous to the storm rainfall event, from milking parlour-washings. A design volume from cow washings of 30m3/cow/year1 was employed, which yielded an hourly loading of 0.05m3/cow/hour that in turn is equivalent to 3.2m3/cow/day (hence 3.2*CN). Designs for a primary lagoon use a conservative literature value of 20m3/m2/day hydraulic surface loading rate. It is proposed that the subsoil lining this balancing pond would require a hydraulic conductivity of at least 10^{-9} m/s.

CONSTRUCTED WETLAND DESIGN AREA

$$A_{ICW} = (1.5*A_Y) + (105*C_N) \qquad \textit{(Equation 2)}$$

where:
A_{ICW} = Total pond surface area (m^2)
A_Y = Yard Area (m^2)
C_N = Number of milking cows

The constructed wetland design equation proposed by TCD (Equation 2), although apparently simple, is based on complex iterations of the design area model/equation presented in numerous treatment wetlands design manuals (e.g. Kadlec et al., 2000), as follows:

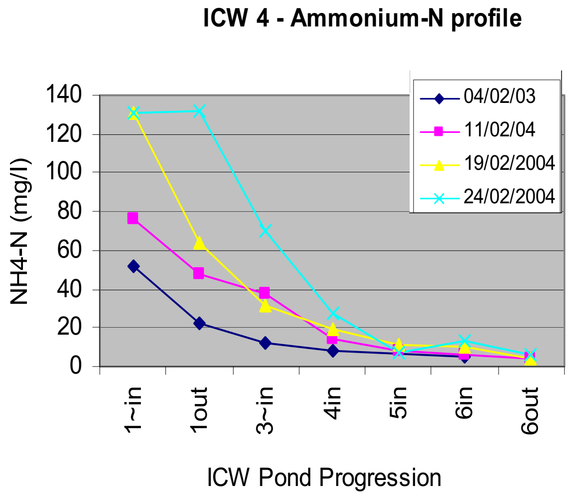
$$A = \frac{Q}{K}\left[\ln\left(\frac{C_0 - C^*}{C_{in} - C^*}\right)\right] \qquad \textit{(Equation 3)}$$

where:
A = Required area of constructed wetland (m^2)
Q = Design inflow volume to the wetland (m^3/s)
K = Rate constant
C_o = Design discharge concentration (mg/l)
C_{in} = Influent concentration (mg/l)
C^* = Background Concentration (mg/l)

Data supplied by the NPWS team to TCD initially facilitated determination of 'K' coefficients for six Waterford ICWs (Equation 3) on an annual time step. 'K' is a global rate constant that is proportional to the amount of active area (e.g. biofilms, plants and algae) per unit wetland area (Kadlec et al., 2000). The actual 'K' values were observed to be much lower than those presented in the literature for dairy wastewater treatment for dairy livestock operations (e.g. as suggested by Knight et al., 2000). In January 2004 the TCD team requested Waterford County Council to initiate an intensive monitoring programme for each individual pond of the six selected ICWs. This monitoring programme yielded input and output concentrations for each individual pond. These data were employed in the model (Equation 3) to determine the K value for each individual pond of the existing ICWs. This new data suggested that pollutant removal efficiency was much better in the initial ponds.

[1] This figure is based on actual flow data supplied from the instrumented ICW on Milo Murphy's farm situated in the Annes Valley, cognisant of climatic inputs, and validated with actual irrigated volumes on Teagasc experimental farms (with the known consideration that the experimental farms use more wash water than typical dairy farms). The 30m^3/cow/year value conforms to literature values (Brewer et al., 1999), with a factor of safety.

Indeed, the K values returned for the first two ponds of the ICWs located in County Waterford were in agreement with the literature values. However, the final ponds of the County Waterford ICWs had much lower K values. These findings can be observed graphically in Figure 1, which represents typical concentration profile for a typical ICW in County Waterford. Initial simulations were returning an average K value for each ICW.



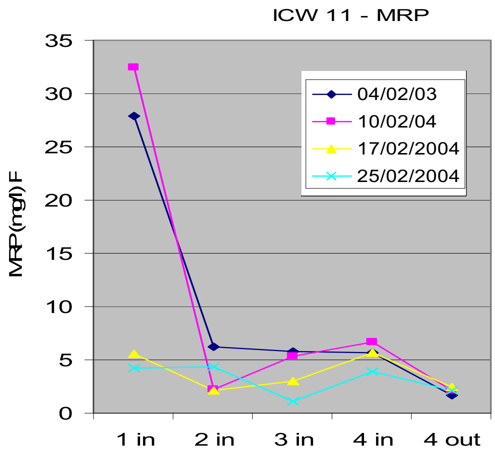


Figure 1 Example of concentration profiles through particular ICWs.

Therefore, there is a clear change in the reaction rate as the effluent moves through the ICW system, which has been rationalised in the design protocol into two distinct reaction rates (i.e. two-K design). The two-K system identified is most probably due to the heavier loading at the beginning of the system promoting a higher concentration gradient between the wastewater and the biofilm, whereby the kinetics of the system is typically operating at zero to a half order. As the effluent concentration weakens with distance through the ICW system, the concentration gradient decreases and thus the biological kinetics move towards first order.

The kinetics of the system thus understood, new simulations were carried out in two steps for each ICW to identify the required total area for a characteristic influent and required discharge standard (shown previously in Table 2). In the first step, the design area model (Equation 3) was employed with the characteristic influent, background and effluent wastewater concentrations2, actual Annes Valley ICW systems K values for each hydrochemical parameter, and the hydraulic loading rates for each farm to determine the area required for the first, more efficient portion, of the wetland. Model simulation to determine the required ICW area were firstly based on flow rates (Q) generated by dairy washings based on cow numbers and rainfall runoff from the yard only. However, the effect of rain falling on the open ponds must also be considered. Rainfall contributions increase hydraulic loading, influence concentrations and affect the processes described by the rate constant (K) of the design area model (Equation 3). Model simulations return a required higher ICW area to compensate for rain falling on the pond. The contribution of rain falling on pond's of increasing area was continuously reiterated until a stable ICW area was returned. In the second step of the total area simulations, the lower K value was used with an influent strength delivered from the first portion of the ICW (the observed breakthrough point from the intensive monitoring data) and an effluent strength as required by the discharge licence. The effect of rainfall on the ponds was considered through successive iterations of the design area model (Equation 3). The total required area was determined for each of the selected ICWs by summing the areas for each of the two steps of the simulation. Obviously, the area of the second step is much larger because of the low K values and the stringency of the effluent standard.

[2] It had been observed from the intensive monitoring data that there was a breakthrough concentration from the first part of the system and this was used as the required effluent standard from the more efficient first portion of the ICW system.

Simulations for MRP, NH4N, TN, SS, BOD and COD revealed that MRP and NH4N were the limiting parameters and they required equally large areas in order to achieve an effluent standard of 1mg/l MRP and 1mg/l NH4-N.

The design equation [AICW = (αAY) + (βCN)] was then obtained using the 'solver' function in EXCEL whereby the actual yard areas and cow numbers for each ICW were used in combination with the simulated required areas, yielded by the two-K simulations, to obtain the α and β factors for yard area and cow numbers. It was originally envisaged that the equation should contain some reference to effective rainfall but the Q component of the basic design equation considered rainfall. However, national validation may demonstrate that some revision is necessary. Perhaps effective rainfall may be included in future developments of the ICW equation.

ICW PERFORMANCE DATA

It is clear that a degree of wastewater treatment is achieved. Inlet and outlet data for successive ponds of two ICWs in Waterford are shown in Table 4.

Table 4 Hydrochemical data for successive ponds for two dairy wastewater treatment wetlands (Annual Averages 2004).

	Influents				
	pond 2	pond 3	pond 4	pond 5	effluent
ICW 9 - COD	613.57	245.00	37.83	34.80	42.43
ICW 9 - NH4-N	37.82	16.65	3.52	1.16	0.86
ICW 9 - MRP	10.82	5.02	1.50	0.68	0.48
ICW 9 - BOD	149.60	28.79	10.24	5.89	4.68

	Influents				
	pond 1	pond 2	pond 3	pond 4	effluent
ICW 9 - COD	1848.00	167.13	117.86	57.25	57.22
ICW 9 - NH4-N	34.31	25.01	11.52	5.21	0.41
ICW 9 - MRP	13.98	8.50	5.15	3.94	1.22
ICW 9 - BOD	698.28	73.55	35.00	10.50	13.10

PHOSPHORUS RETENTION

The dynamics of phosphorus in the treatment wetlands was investigated by mass balance analysis. In summary, the following was determined for the particular wetlands under investigation:

- *The average net annual phosphorous accumulation rate for the six ICWs is 414 g-MRP/d. This equates to an expected generic accumulation rate of 7.5 g-MRP/m^2/yr in terms of the area of any newly designed wetland according to the protocol equation. This equates to 13.1 g-MRP MRP/m^2/yr according to the actual size of the wetlands constructed.*
- *The annual average final effluent concentrations from three out of the six ICWs studied is in excess of the UWW discharge standard of 1 mg/l. This indicates the under-sizing of the ICWs according to the recommendation of the design protocol equation.*
- *There is little evidence based on the limited data set to reveal any significant migration of phosphorus down through the wetland systems.*
- *In general, the initial ponds in each wetland system show a higher accumulation of phosphorus in the sediments than the downstream ponds although this pattern is not as definitive as might have been expected.*
- *The annual accumulation rate of phosphorus in the wetland is on average 24% the annual fertiliser requirement of the farms. This stored phosphorus can eventually be used to reduce the fertiliser demand of the farm and improve its net metabolism.*
- *An initial balancing pond will greatly facilitate such phosphorous harvesting helping to optimise the maintenance period for the rest of downstream wetland ponds. If the phosphorous concentrations discharged from any ICW to receiving surface water bodies do start to become critical, an appropriate medium term solution would be to pass the final effluent through a sacrificial filter located at the effluent point from the ponds.*

Research data on phosphorus dynamics and constructed wetland soils is also presented by Dunne et al. (2005).

CONCLUSIONS

1. TCD design equations were derived using Kadlec et al. (2000) design equation, constructed wetland performance data and UWW discharge criteria.
2. The issues associated with discharge of treated wastewater to natural water features, be they surface or groundwaters, require careful consideration. Generation and retention of waters infers a need for eventual discharge; even though open water evaporation and evapotranspiration will play a part in the hydrological balance.
3. The applicability of constructing a wetland is completely dependent upon site assessment in the context of providing the appropriate groundwater protection response.
4. In terms of their use within SUDS, their efficiency is unquestionable provided the following points are considered:
    (a) They are placed appropriately in the 'management train' i.e. pre-treatment of wastewaters ensures the hazard and associated risks are reduced
    (b) The site is appropriate in the context of providing sustainable environmental protection – i.e. in poorly permeable areas (where one might expect a natural wetland to form) and containment is provided. However, leakage rates and consequent recharge to groundwater is a concern in areas of high permeability subsoil. Containment can be engineered with artificial liners.
    (c) An appropriate point of discharge is available.

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Session IV

GROUNDWATER MONITORING: THE IMPORTANCE OF SETTING CLEAR MONITORING OBJECTIVES BASED ON AN APPRECIATION OF THE HYDROGEOLOGY

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ABSTRACT

An effective groundwater monitoring programme must be based on having clear monitoring objectives and a good conceptual model of the hydrogeology. For many monitoring situations, an appreciation of the three-dimensional distribution of groundwater head is important, not only for observing water levels, but also for sampling water quality. Observation boreholes with long open sections are not appropriate, especially for investigating groundwater quality, since the samples will not be representative of a particular aquifer horizon, and such boreholes may also lead to cross-contamination between aquifers. In deciding what parameters to monitor, it is often valuable to include parameters additional to those required by regulations. An aspect of monitoring that is often neglected is the monitoring of well performance, and this aspect is also dealt with in this paper. It is very important that such monitoring covers the entire well system – which includes the aquifer, headworks, pumping plant and distribution system, in addition to the well itself.

1.0 INTRODUCTION

Monitoring is part of a *process* in groundwater resources assessment and management. To be effective, a monitoring programme should have clear objectives. As expressed by Gunston (1998) - in the wider context of hydrological data collection:

> "Hydrological data collection is a means to an end (the better management of water resources) not an end in itself (simply collecting numbers for the sake of it)".

The monitoring objectives must be based on a good understanding of the hydrogeology: there must be a conceptual model of the groundwater system, which can subsequently be refined when additional data have been collected and interpreted. The link between data collection and interpretation is important: if the data analysis process is divorced from the data collection, then it is not possible to refine the monitoring programme to meet the needs of the investigation. Also, if the hydrogeologist carrying out the analysis has not been involved in the data collection, then he/she may be unaware of some significant limitations in the accuracy of the data, or of how representative the data are of the situation being evaluated.

The initial conceptual model should include the occurrence of aquifers and aquicludes, the distribution of groundwater head and its relation to groundwater flow, and the linkages to surface water, both in terms of recharge and discharge from the system.

The paper will focus on the principles behind monitoring, not the detailed procedures or equipment. Many people tend to think of groundwater monitoring in terms of collecting data on groundwater level (head) and groundwater quality. A third important aspect will also be addressed in this paper: monitoring the performance of production wells.

2.0 GROUNDWATER HEAD

2.1 REMINDER ABOUT GROUNDWATER HEAD

The groundwater head (h) at any particular point in an aquifer is the sum of the pressure head and elevation head:

$$h = \frac{P}{\rho g} + z$$

where P is pressure, ρ is the density of the water, g the acceleration due to gravity and z is the elevation above an arbitrary datum. Head has units of length (metres).

In an unconfined aquifer, the water table represents the surface at which the pressure is equal to atmospheric. In a confined aquifer, the *piezometric surface* or *potentiometric surface* represents the level to which water will rise in wells. (The water table is a particular potentiometric surface for an unconfined aquifer). The slope of this surface defines the hydraulic gradient, which in turn controls the direction of groundwater flow.

2.2 HEAD AS A 3-D CONCEPT

In planning a monitoring exercise, it is important to remember that head varies in three dimensions. Whereas it is widely known that artesian boreholes occur in confined aquifers where the potentiometric surface is higher than ground level, artesian boreholes *can* also occur in unconfined aquifers. This can be illustrated with reference to the two aquifer situations depicted in Figure 1. Figure 1(i) shows a relatively high permeability aquifer where the gradient of the water table is shallow and groundwater flow is predominantly horizontal. The head contours are therefore approximately vertical and the head at any depth in the aquifer at a given horizontal (x,y) coordinate is approximately equal to the elevation of the water table. Hence wells exhibit similar static water levels, irrespective of depth (wells A and B in Figure 1(i)). Groundwater flow thus approximately follows the gradient of the water table.

Compare this with the case illustrated in Figure 1(ii). This shows groundwater flow in a low permeability aquifer in an area of high topography. Here, head is truly three-dimensional, varying with elevation (z) as well as horizontally (x,y). Head contours are complex and *not* necessarily vertical. Groundwater flow has upwards and downwards components. In recharge areas, head typically decreases with increasing depth, and groundwater flow has a downward component. A deep well here (well C) will have a lower static water level than a shallow one (well D). In discharge areas, head increases with increasing depth and groundwater flow has an upward component. A deep well here (well E) will have a static water level higher than a shallow one (well F). In exceptional cases, deep wells in discharge areas in *unconfined* aquifers may even have artesian heads, and overflow at the ground surface (as shown by well E in Figure 1(ii)). Aquifers with strongly three-dimensional head distributions will typically either have a strong topography or have relatively low permeability (or both).

The contrasting situations illustrated in Figure 1 should lead to different conclusions about the design of boreholes for monitoring groundwater levels. Whereas a two-dimensional network of observation boreholes with long well screens may be adequate for monitoring the head distribution in aquifers of the type illustrated in Figure 1(i), this network design would not be suitable for monitoring three-dimensional head distributions of the type in Figure 1(ii). For the latter type, a 3-D network of piezometers to varying depths would be required, with each piezometer having a very short open section so as to give a reading of head (h) at a specific point *(x,y,z)*.

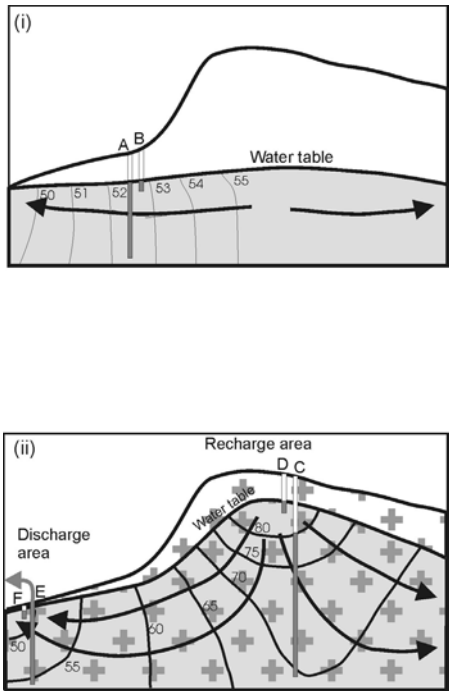

Figure 1. Examples of head distribution in (i) high and (ii) low permeability aquifers

2.3 CONTINUOUS VERSUS INTERMITTENT MONITORING

The traditional means of measuring water level is by an electric contact gauge, or 'dipper'. Although dippers are simple and reliable, labour costs can be considerable if a high frequency of monitoring is required. In this situation, continuous (or near-continuous) measurements can be made by installing pressure transducer systems connected to data loggers. Small, self-contained transducer and logger systems are now widely available, to fit inside the narrowest of observation boreholes. The data are downloaded to portable computers at regular periods, for example monthly (the download intervals will depend on the chosen measurement frequency and data storage capacity of the logger). Water level pressure readings need to be adjusted for variations in atmospheric pressure, and so a monitoring network will normally include a separate barometric unit located within the monitoring area. Automated systems are particularly valuable for e.g.: monitoring of water levels at short time intervals during a pumping test, monitoring short-term cyclic fluctuations (such as tidal effects), detecting small changes in water level due to the impacts of nearby abstractions.

2.4 SURFACE WATER – GROUNDWATER INTERACTIONS

River baseflow analysis is one of the principal approaches for estimating recharge in Ireland (Misstear, 2000). One of the challenges in baseflow separation is to identify what is actually being included in the baseflow component of the stream hydrograph. Depending on the analysis, the baseflow might include: releases from bank storage, releases from peat deposits, interflow from subsoils, shallow groundwater discharge and deep groundwater discharge. If the objective is to try and understand aquifer recharge, then it is the latter two components that are of most interest.

In recent times there has been an increase in the use of automated systems for hydrograph analysis. With such systems, the baseflow predictions are sensitive to the length of time base or other 'recession parameter'. Inspection of well hydrographs may help in selecting the appropriate parameters for a particular stream gauge record. However, suitable well hydrographs - suitable both in terms of well location and monitoring frequency - are scarce in Ireland and so this approach will only be possible in a small number of situations.

3.0 GROUNDWATER QUALITY

The objective of groundwater quality monitoring must be clearly defined, as this will influence the design of the monitoring installations, the choice of parameters to be analysed and the frequency of sampling. The purpose may be for initial hydrochemical characterisation, contamination investigation, regulatory compliance, operational monitoring or for research.

3.1 DESIGN OF MONITORING BOREHOLES

We saw above how the 3-D distribution of groundwater head can affect the design of a network for monitoring groundwater levels. This is also the case with groundwater quality: because groundwater flows vertically as well as horizontally in response to head variations (see Figure 1(ii)), the chemistry of groundwater also varies in three dimensions.

Monitoring installations should therefore be designed so as to enable samples to be collected at discrete depth intervals. Alternative designs for achieving this objective are illustrated in Figure 2. Installations with long open sections are not recommended. Not only will the resulting water samples represent some ill-defined mixture of water from different depths, the installations themselves may permit the movement of pollutants from one aquifer to another i.e. allow cross-contaminant to occur.

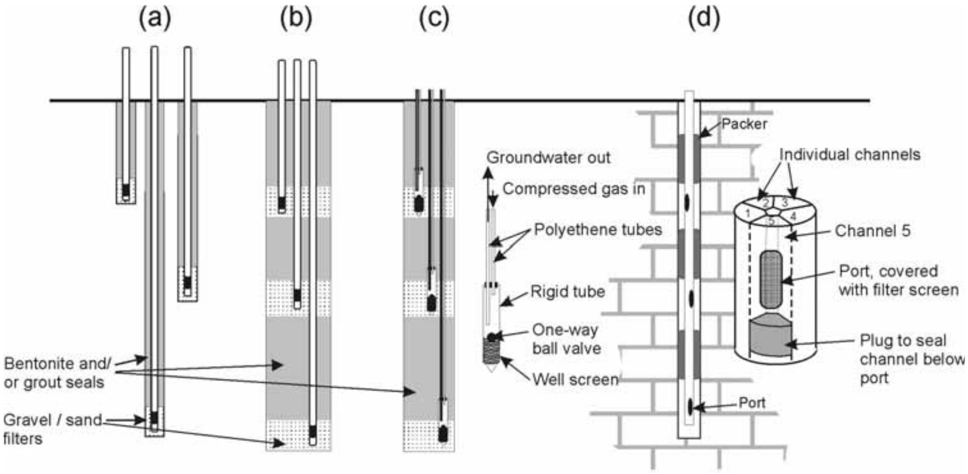

Figure 2. Monitoring installations

When installing observation boreholes for monitoring groundwater quality, the borehole construction materials should be non-reactive chemically. Following the terminology of Aller *et al*. (1991), the objective is to avoid obtaining 'false positive' results from contaminants that have leached into the groundwater from the borehole materials, and also to avoid 'false negative' results by failing to detect contaminants in the groundwater because these contaminants have been removed through sorption by the casing. Full chemical inertness with respect to all contaminants is probably impossible to achieve and in any case may not be necessary because, provided that the borehole is purged properly before sampling, the contact time between the construction materials and the groundwater being sampled is limited.

3.2 DECIDING WHICH PARAMETERS TO MONITOR

When sampling a groundwater, there is often a tendency to sample only those parameters required by the relevant regulations, or those which are directly relevant to a specific pollution problem. There are advantages in broadening the scope of the sampling programme to include the main physicochemical parameters (pH, redox potential, temperature, electrical conductivity) and the major ions (calcium, magnesium, sodium, potassium, (bi)carbonate, sulphate, chloride and nitrate) on a representative selection of samples from a site. The reasons for this are:

a) These parameters dominate the water chemistry and may influence how other elements or species behave. For example, it is of limited use to know the iron concentration in a water sample, if nothing is known about the redox and pH conditions.
b) They will also provide information on how the water's chemistry has evolved and may give some clues as to its residence time.
c) The major variables allow quality controls and reality checks to be made on the analytical results e.g. by calculating ion balance errors.

For drinking water, the chemical, radiological, microbiological, physicochemical and aesthetic parameters that should be analysed are defined by national drinking water regulations based on the EU drinking water directive. However, drinking water guidelines may not include every parameter of health significance. Examples of chemical parameters that should be considered for analysis in certain geological environments, even though they are not contained in some national regulations, include uranium, radon, thallium and beryllium, especially in crystalline rock terrains.

Because the potential number of groundwater quality parameters to be monitored in a new abstraction is enormous, it is not uncommon to measure:

- certain parameters indicative of good water quality, either continuously (at major abstractions) or at frequent intervals;
- a fuller (and much more costly) suite of determinands at less frequent intervals to verify the conclusions drawn from the indicator parameters.

In terms of water chemistry, the most valuable indicator parameters include pH, electrical conductivity (EC), temperature, dissolved oxygen, colour and turbidity. These can all be measured in the field using portable meters and comparators. Rapid variations in these parameters may indicate that the groundwater quality is unstable and responding to sporadic recharge events or seasonal events. This can be important with respect to detecting microbial pollution.

The term microbe applies to a variety of organisms. Many occur naturally in the groundwater environment. Others may be related to faecal or other anthropogenic pollution, and a few of these may be pathogenic to humans. It is normal practice to analyse for characteristic indicator parameters of faecal contamination. Two of the most common indicators are (i) thermotolerant (or faecal) coliforms and (ii) *E. coli.* However, much more work is required in Ireland on the occurrence of other harmful microbes in groundwater, including viruses and protozoa, and how these relate to the presence of faecal bacteria.

3.3 FIELD DETERMINATIONS

Field determinations are important for two main reasons:

1. The results may be needed immediately.
2. Some hydrochemical parameters are unstable and may change during storage and transport to a laboratory.

Parameters which are unstable following sampling include obvious examples such as temperature, dissolved oxygen and redox potential, which will tend to change rapidly when exposed to ambient atmospheric conditions. Under some circumstances, pH and alkalinity can also change during storage, due to possible degassing of carbon dioxide or precipitation of calcite.

The geosphere (relatively constant temperature, basic and reducing) is a very different geochemical environment from the atmosphere (fluctuating temperature, acidic and oxidising). Thus, when groundwater samples are brought to the surface, some parameters tend to change very rapidly on exposure to the atmosphere and good readings can be difficult to obtain. Therefore, it is important to try and minimise exposure to the atmosphere either by (i) monitoring certain parameters downhole or

(ii) pumping water through a sealed throughflow cell, within which the appropriate measurements are taken.

4.0 WELL PERFORMANCE

A well is often forgotten about after it has been put into operation. However, a water well does need some maintenance as it will deteriorate over the years. For maintenance to be effective, the causes of the deterioration in well performance must be identified through monitoring and diagnosis. It is not just the well itself that should be monitored: monitoring should encompass the whole system for abstracting groundwater, which includes the aquifer, pumping plant and any water treatment and distribution system.

4.1 MONITORING PARAMETERS

The main parameters that should be monitored to help identify the presence, location and cause of a potential problem are summarised in Table 1. Each abstraction well should be equipped with facilities to allow monitoring of water level, discharge and water quality.

The frequency of monitoring will depend to some extent on the use of the well and the monitoring facilities / capabilities available locally. Where automated systems for continuous monitoring of water level and discharge rate are not available, manual measurements should be taken at least weekly, and more frequently if possible. Water quality should be monitored at least monthly during the initial period of well operation after commissioning, and then the frequency can be reduced to quarterly if conditions do not appear to be changing after the first year.

Other useful methods for monitoring and diagnosis include direct observation of the condition of the well and pumping plant, downhole CCTV and geophysical logging surveys, regular well pumping tests, and pump efficiency measurements.

4.2 ASSESSING WELL PERFORMANCE

The hydraulic performance of a well can be assessed by a step drawdown test, which can be analysed to determine the proportions of well drawdown at different pumping rates due to 'aquifer loss' and 'well loss'. For example, according to the well known Jacob equation, the drawdown in the pumping well s_w is given by:

$$s_w = BQ + CQ^2$$

where Q the discharge rate and B and C are the coefficients of aquifer and well loss, respectively.

Step tests can be carried out at regular intervals during the lifetime of the well to determine if there has been a change in well performance. Figure 3 shows results of three step tests carried out at different times on the same well. Test 2 indicates an increase in coefficient B compared to the original test 1, which could, for example, be due to a decline in aquifer transmissivity owing to a fall in regional water levels. Test 3, on the other hand, also shows that coefficient C has increased (i.e. steeper slope) compared to the original test, suggesting that a reduction in well condition has also occurred. (However, we need to be cautious in interpreting the test data simply in terms of aquifer loss and well loss since, for example, the well loss term CQ^2 may include some turbulence effects in the aquifer).

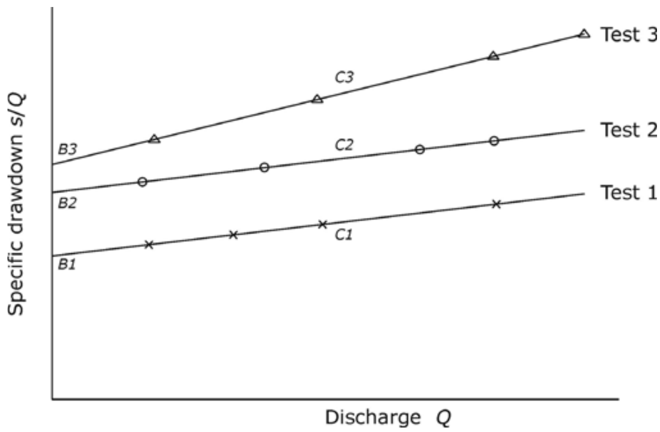

Figure 3. Step test results indicating changes in well performance (schematic)

Changes in well performance can also be assessed by monitoring the specific capacity of the well, and comparing these data with a set of normalised specific capacity results derived from the original step test. Changes in well efficiency can be assessed from (Helweg *et al*., 1983):

$$Efficiency = \frac{SC_C}{SC_O} \times 100\%$$

where SC_C is the current specific capacity of the well and SC_O the original specific capacity determined from the step drawdown test. For comparisons to be realistic, the specific capacity measurements during well operation should be carried out under similar conditions to the original step test – including similar rest water levels, pumping rates and pumping periods.

4.3 ESTABLISHING LONG-TERM WELL PERFORMANCE

The data from the original pumping tests can be used to predict the 'long-term' drawdown in the well for a range of pumping rates, against which operational pumping water level data can be compared. The method is described in Misstear and Beeson (2000), and essentially involves the extrapolation of short-term drawdown data from step tests to longer periods of pumping using the Cooper-Jacob equation. The interference effects of other pumping wells can also be taken into account in these predictions.

For wells that are pumped continuously, or nearly so, it may not be possible to shut down the pump for long enough to carry out controlled pumping tests or to measure rest water levels. In these situations, operational data can be used to establish the reliable yield of the well (Figure 4).

In addition to monitoring water levels inside a production well, it can also be useful to monitor water levels in the gravel pack between the borehole wall and the screen, as this will help in identifying the location of any clogging problems.

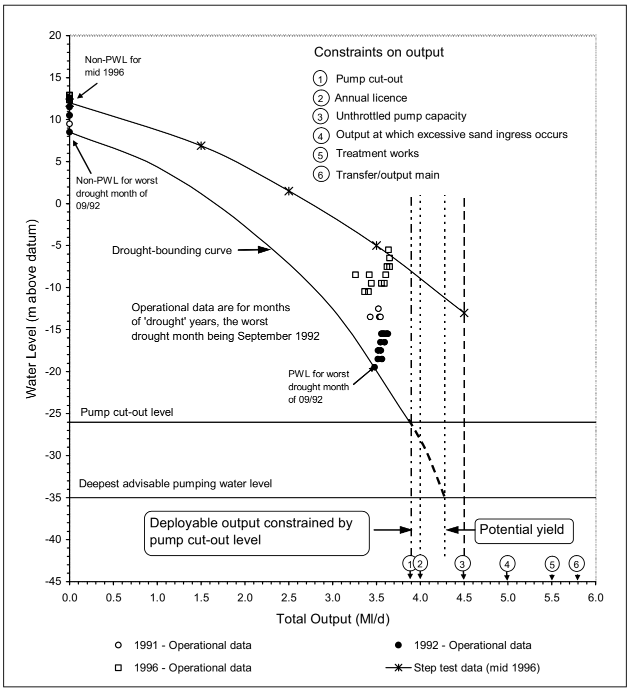

Figure 4 Example of well yield predictions based on operational data (after Misstear and Beeson, 2000)

4.4 WATER QUALITY MONITORING

In terms of well performance, the key chemical parameters to monitor include pH, electrical conductivity, Eh, iron (Fe^{2+}) and the dissolved gases carbon dioxide (CO_2), hydrogen sulphide and dissolved oxygen. It is essential that these analyses are made at the wellhead, using a flow-through cell to avoid the exposure of the sample to air. In addition to monitoring of water chemistry, samples can be collected for biofouling analysis using membrane filters, metal coupons and other techniques described in Borch *et al*. (1993), Howsam *et al*. (1995) and McLaughlin (2002).

5.0 CONCLUSIONS

a) For monitoring to be effective, the objectives of the monitoring exercise must be clearly established, and must be based on an initial conceptual model of the groundwater system. Otherwise, it is likely that the data collected will not be sufficient, or fully relevant, or representative of the groundwater conditions being investigated.

b) The design of a regional monitoring programme or pollution investigation must take account of the fact that groundwater flow occurs in three dimensions. Observation boreholes with long open sections are not suitable for determining the distribution of either groundwater head or groundwater quality, and can lead to cross-contamination of pollutants between aquifers. Observation boreholes should be designed with short screen sections to permit observations or sampling at discrete depths.

c) The operational performance of production wells should be monitored so that appropriate maintenance measures can be implemented.

ACKNOWLEDGEMENTS

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Table 1. Well system monitoring (adapted from Howsam et al., 1995)

	AQUIFER	*Well*	*Pumping system*	*Wellhead works*
PERFORMANCE	Abstraction/ Recharge	Discharge rate	Discharge rate	Flow meter and instrument accuracy
	Regional water level	Pumping water level	Discharge head	
	River base flows	Rest water level	Energy consumption	
	Regional water quality	Water quality		
		Specific capacity		
CONDITION	Abstraction/ Recharge	Appearance	Pump appearance	Appearance
	Regional water level	Hydraulic efficiency	Noise and vibration	Leakage
	River base flows		Rising main appearance	
	Regional water quality		Earthing	
PROCESS				
Physical	Formation grain size distribution	Gravel pack grain size distribution	Sand content	
	Flow rate/velocity	Gravel pack level		
		Flow rate/ velocity		
Chemical	Water chemistry	Water chemistry	Water chemistry	Water chemistry
	Geochemistry	Materials	Materials	Materials
Microbial	Nutrient status	Microbial activity	Microbial activity	Microbial activity
	Recharge water quality	Nutrient status	Nutrient status	Nutrient status
		Flow rate/velocity	Flow rate/velocity	Flow rate/velocity
		Oxygenation	Oxygenation	Oxygenation
		Materials	Materials	Materials
Structural/ Mechanical		Depth of infill or collapse	Failure	Failure
Operational	Aquifer status	Operating hours	Operating hours	

GROUNDWATER MONITORING AND SAMPLING – NEW RESEARCH AND THE IMPORTANCE OF BOREHOLE CONSTRUCTION

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ABSTRACT

The principal objective of most groundwater sampling programmes is to obtain a sample which is "representative" of the water quality in the surrounding geological formation. To achieve this, and to overcome the deficiencies of borehole design, various purging strategies have been proposed including "3 to 5 well volumes", "low flow", and, more recently, "passive" or "no-purge" sampling. Whilst all of these have their place, none can sensibly be used without an understanding of the local hydrogeology, the contaminants being sampled and the construction detail of the borehole. This presentation looks primarily at the last issue which is frequently overlooked, but is arguably the most important consideration in sampling. The importance of designing boreholes with shorter screened intervals is emphasised.

Recent research is presented illustrating how pumping from boreholes results in samples which should more correctly be considered as a "flow weighted average" across the screened interval. Sample concentrations can be significantly lower than the maximum concentration in the surrounding formation and will be biased toward inflows from more permeable strata. Other recent research on mixing mechanisms within a borehole question some of our basic assumptions, and provide important considerations for how we should carry out and interpret data from sampling programmes.

INTRODUCTION

Guidance for obtaining water quality samples from groundwater monitoring boreholes in the UK and Ireland has to date developed around a default requirement to purge "3 well volumes" from the borehole before taking a water sample for analysis (e.g. Environment Agency 2003). There are several other alternative purging strategies identified in the Environment Agency Guidance, such as "low flow" and "no purge" (or passive sampling) which in many cases could be more appropriate. Outside the hydrogeological community in these countries alternative methods are rarely adopted, primarily because of the requirement to carry out comparative tests against a 3 well volume purge strategy. There is however considerable research on low flow sampling methodology which has been widely adopted as guidance in the USA and elsewhere (see for example Puls and Barcelona 1996), whilst more recently, passive sampling techniques have gained credence in the USA (e.g. Parsons 2005).

The debate between proponents of different sampling methodologies in the USA (see for example Barcelona et al 2005) has increased research effort into flow mechanisms within the water column and is yielding some fascinating insights into the influence of borehole construction on the sampling process. Some of this research is presented below in the context of borehole design.

BOREHOLE CONSTRUCTIONS

Figure 1 illustrates five different possible constructions for boreholes which are defined further in Table 1. It is not uncommon in many investigations to install boreholes with long screened intervals. It is clear from the conceptual graphic provided by Figure 1 that the water quality of the sample collected will be very dependent on the vertical and horizontal positioning of the borehole well screen. Shorter screened or multilevel boreholes will provide greater certainty on the vertical interval sampled in the aquifer than will longer screened boreholes. This is further demonstrated by Figure 2 in which samples taken from multilevel boreholes and long-screened boreholes around the perimeter of a landfill site are compared (Dumble et al 2006). The vertical concentration gradients apparent in the multi-level boreholes give a very different interpretation to contaminant flow paths than is possible from using data solely from the long-screened boreholes.

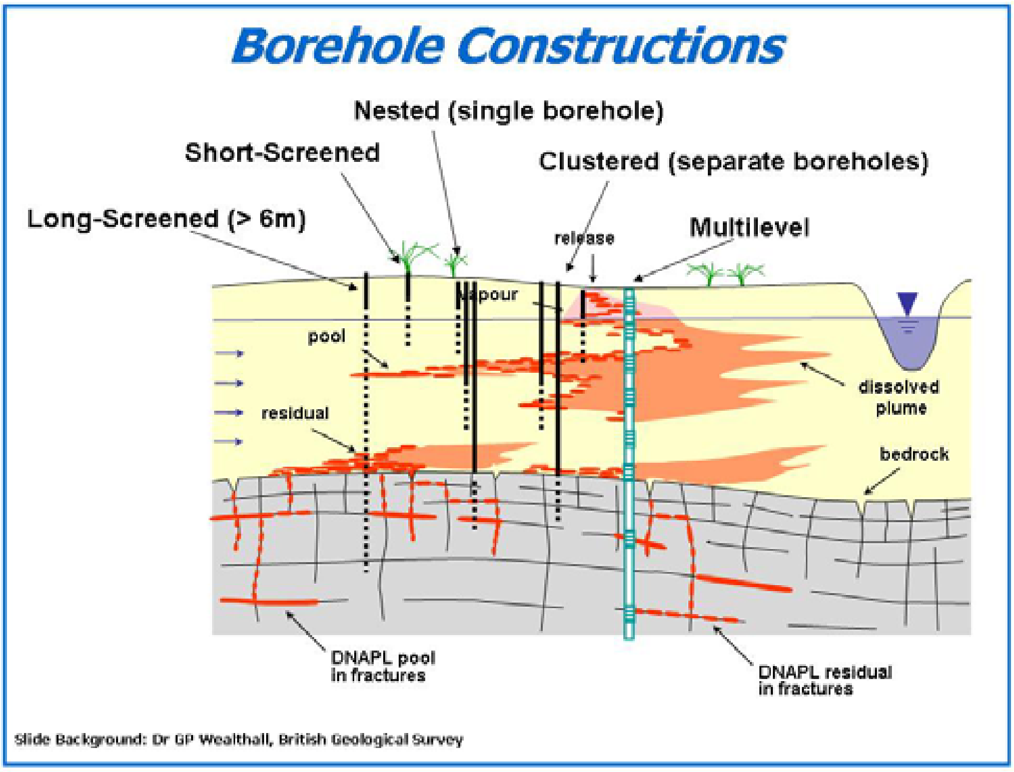

Figure 1: Alternative borehole constructions illustrating the importance of the positioning and length of screened intervals for sampling groundwater (conceptual section provided by Dr. GP Wealthall of the British Geological Survey).

Type of Construction	**Description**
Long-Screened Boreholes	Defined by USEPA as greater than 20 feet and by the UK Environment Agency as greater than 6 metres. These are arbitrary definitions.
Short Screened Boreholes	Screens are less than 6 metres in length.
Nested Boreholes	Two or more screened linings installed within the same drilled borehole.
Clustered Boreholes	Two or more short screened boreholes completed in close proximity to each other. Screened sections are at different vertical intervals.
Multi-level Boreholes	Usually a single casing in a borehole with ports isolated from each other at different depths.

Table 1: Types of borehole construction

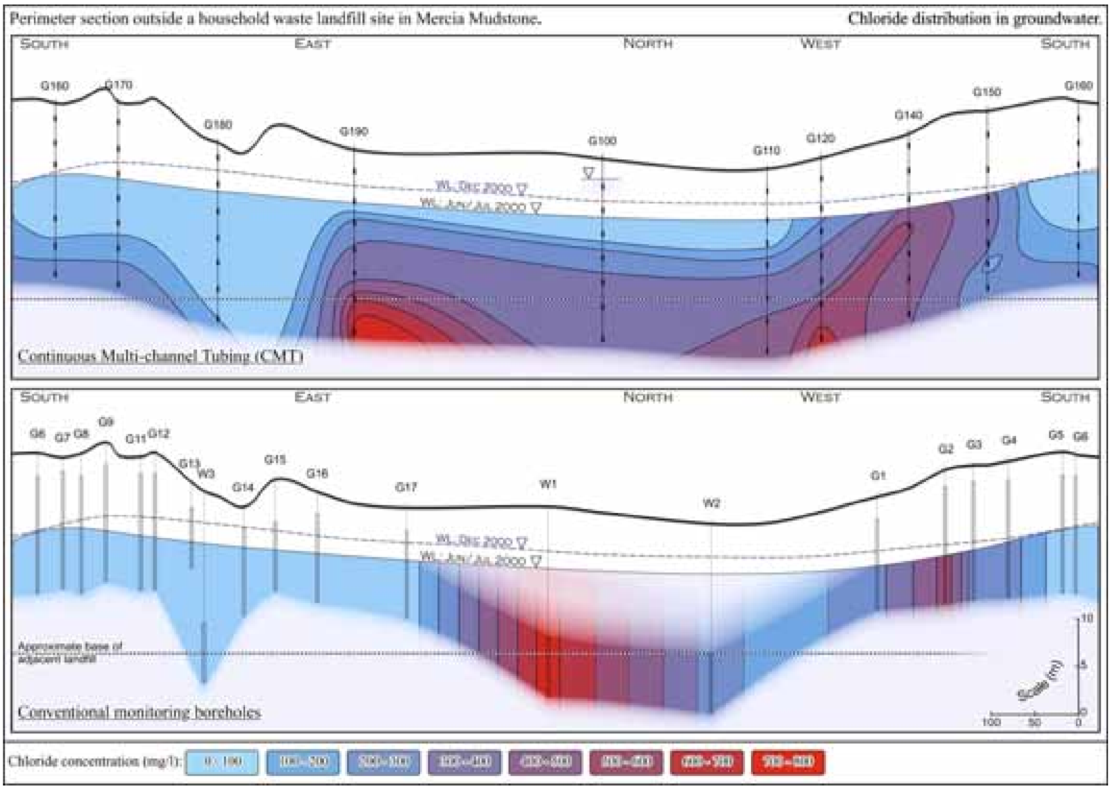

Figure 2: Comparison of chloride concentrations in groundwater based on samples collected from long-screened boreholes and multilevel systems around the perimeter of the same landfill site (Dumble et al 2006).

The use of long-screened monitoring boreholes can significantly mask hydraulic and chemical variations that occur naturally over short vertical distances with measurements and samples becoming averaged or biased toward the dominant condition (e.g. Martin-Hayden and Robbins 1997, Martin-Hayden 2000a,b; Gibs et al 2000; Sevee et al 2000; Britt 2005). Vertical flows induced by vertical hydraulic gradients, can cause the re-distribution of contaminants from one vertical zone to another, or can mask thin zones of contamination which become diluted, sometimes to below detection (e.g. Martin-Hayden and Britt 2006, Elci et al 2001, Hutchins and Acree 2000, Church and Granato 1996).
A new UK guidance document on borehole construction (Environment Agency 2005) provides an improved appreciation of screen length for taking groundwater quality samples (Table 2). In this document a short screened borehole is defined as less than 3 metres in length and ideally less than 2 metres, and this concurs with the trend in US EPA guidance towards "10 foot" well screens. For even shorter intervals multi-level systems can be used (e.g. Einarson and Cherry 2002, CL:AIRE 2002a,b, Dumble et al 2006). Multilevel systems can be used to target vertical intervals as short as 150 mm in length, but even these can occasionally yield false positive results as a consequence of contaminant redistribution during drilling of the borehole (Parker 2006).

SCREEN LENGTH AND SAMPLING OBJECTIVES

Borehole sampling objectives (Figure 3) have generally focussed on whether to collect a "composite" sample (i.e. a sample perceived to be representative of the average concentration across the entire screened interval of a borehole) or a "spot" or "discrete interval" sample (perceived to be representative of the inflow to the borehole at the depth of sampling). In simple terms, the composite objective would then be achieved by a well volume purging method whilst the discrete sample objective could be attempted using low flow or passive sampling methods. This guidance has by and large been presented in the absence of research on flow mechanisms that occur within the borehole itself. It is becoming clearer in the light of some of the research presented below that these sampling objectives may not be as straightforward as once thought particularly when sampling from long screened boreholes.

Aquifer Conditions / Monitoring Objectives	Response Zone/Screen Length			
	Multi-level	**Very short (<1 m)**	**Short (1 to 2 m)**	**Long (3+ m)**
Monitor general background water quality in thick aquifer	✓✓	x	✓✓	✓✓✓
Monitor general background water quality in thin aquifer	x	✓	✓✓✓	✓
Monitor LNAPL	x	x	✓✓	✓✓✓
Monitor DNAPL	x	✓	✓✓✓	✓✓
Detailed examination of contaminant distribution	✓✓✓	✓✓	✓✓	x

Key: x Not appropriate ✓✓ Appropriate
✓ Appropriate but not ideal ✓✓✓ Most appropriate

Table 2: Environment Agency guidance on screen lengths (Environment Agency 2005, Table 2.4)

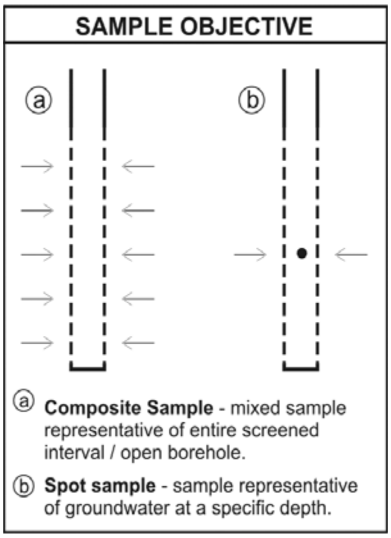

Figure 3: Sampling objective guidance (Environment Agency 2002, Figure 9.5).

MIXING MECHANISMS IN BOREHOLES

Thermal instabilities within monitoring boreholes will cause mixing due to the creation of convection cells. Gradients as small 0.01°C per metre are sufficient to induce complete mixing in a borehole water column in a relatively short time period (Martin-Hayden and Britt, 2006). A simple experiment in which the addition of a denser liquid dye in the top of a water column 40 cm in length, demonstrates complete mixing in less than 5 minutes (Figure 4). This is analogous to cooler (denser) water in the top of a borehole water column sinking and mixing into the well screen and influencing the sample quality taken from the borehole. Conversely if water in the top of the water column is warmer than in the well screen, water above the screen may become "trapped" and "stagnant" whilst thermally driven convection may still be occurring at greater depths in the borehole.

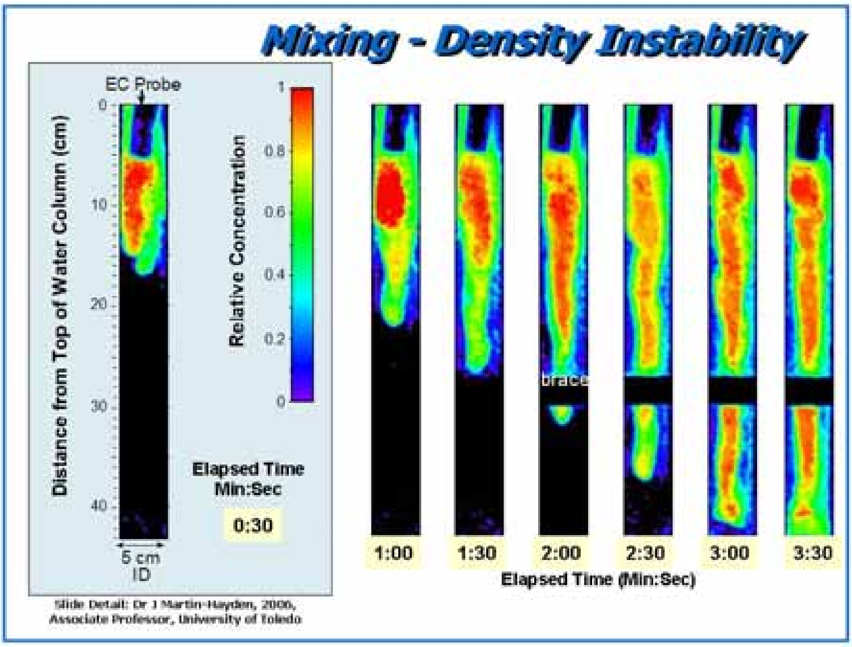

Figure 4: Dye test illustrating how mixing rapidly occurs in a water column by the addition of a higher density fluid. Similar situations can naturally occur in boreholes as a result of geothermal gradients (Martin-Hayden and Britt, 2006)

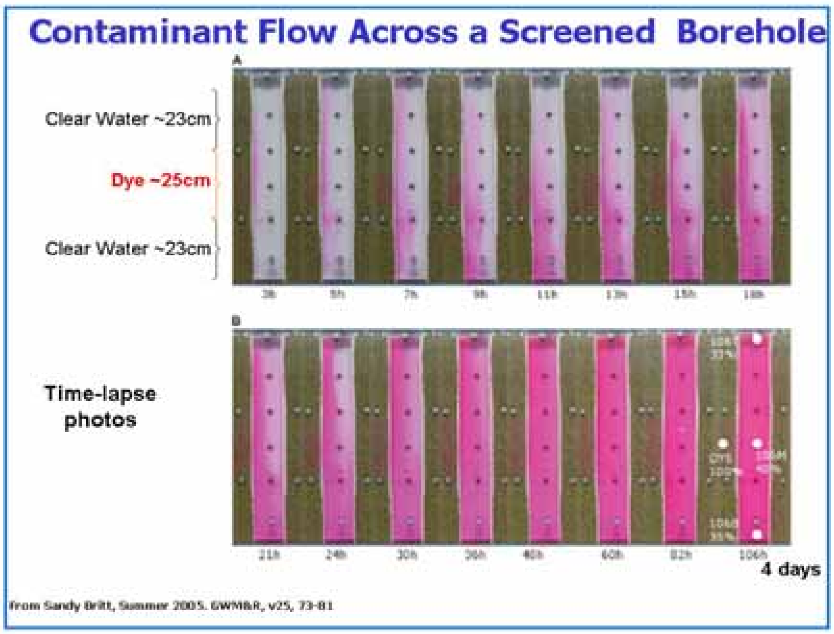

Figure 5: Time lapse photographs showing contaminant flow across a simulated borehole in a laboratory sand tank illustrating mixing (Britt, 2005, Martin-Hayden and Britt, 2006).

Martin-Hayden and Britt (2006) record the following observations on mixing in boreholes in the absence of thermal instability:

> *"Ambient mixing effects during background (non-pumping) conditions may also occur without overt effects of thermal instability. These ambient through-flow effects were investigated using a sand-filled flow tank with a two dimensional cross-section of a well built and tested at the California Department of Toxic Substances Control and detailed by Britt (2005). A uniform gradient and flow toward the right of the model were produced by constant head reservoirs at either end of the tank. Dyed water adjusted to the same density as the clear water was introduced over the middle 1/3 of the model and entered the well from the left (Figure 5). Ambient mixing of the stratified tracer (contaminant proxy) was prevalent in these experiments, regardless of small density differences or flow rate changes. When neutrally buoyant "contaminant" stringers flowed into the simulated monitoring well, some degree of flow-weighted dilution and mixing throughout the well was evident in all tests. For tests where small density differences (equivalent to only 10's of ppm total dissolved solids) were introduced to the well, muted stratification occurred. As noted in all of the experiments, the entry point of the contaminant proxy was not reflected in the concentrations present at the same interval within the well, except when dense dye was introduced at the bottom of the well. Furthermore, because the concentrations are redistributed in the well, the distribution down-gradient from the well will also be altered (i.e., the "shadow effect")."*

LOW FLOW PUMPING AND MIXING IN BOREHOLES

As a monitoring borehole is pumped the reduced head within the borehole is distributed along the screen and groundwater inflow along the entire screen begins to move toward the pump intake (Martin-Hayden and Wolfe 2000, Varljen et al 2006). This is illustrated by Figure 6 which is a simulated dye test. It has been estimated that it could take the removal of between 3 to 5 well volumes of water before a flow-weighted average sample unaffected by borehole mixing can be collected (Martin-Hayden and Wolfe 2000). If the pumping rate is changed mixing effects may shift and alter the weighting of the partial mixing. Most low flow purging samples are frequently taken after pumping significantly less volume of water and researchers have questioned whether these samples are the best possible sample from the borehole (Martin-Hayden and Britt 2006). Varljen at al (2006) argue otherwise but accept short screens are essential where low flow purging is used.

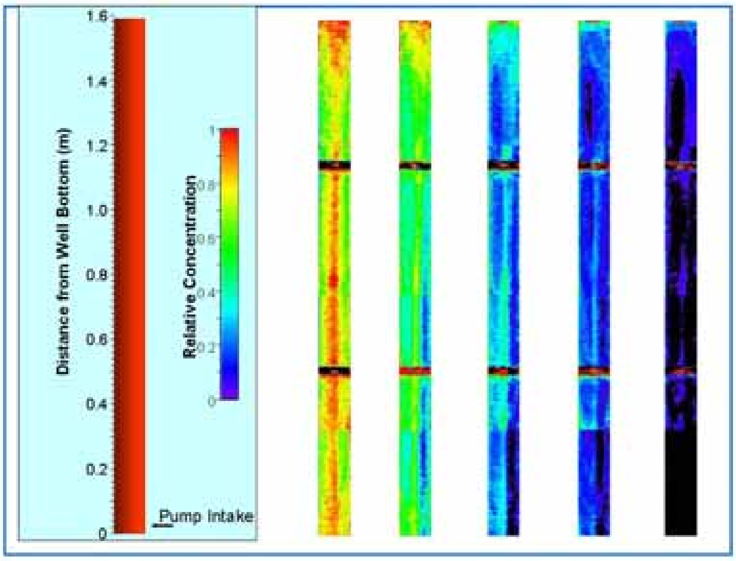

Figure 6: Dye test illustrating how low flow pumping from the base of a well screen induces inflow from the whole of the screened interval (Martin-Hayden and Britt, 2006)

CASE HISTORIES

MIXING UNDER A THERMAL GRADIENT?

Figure 7 is data collected by the author recording temperature and electrical conductivity varying with depth below water level in a borehole in the Lower Greensand aquifer of south-east England. There is a very obvious thermal gradient in the borehole column. Cooler (denser?) water is present nearer to the water surface in the borehole, warming with depth at a rate of 0.5 °C per m. Conversely the conductivity profile shows lower conductivity (less dense?) water nearer to the top of the water column rapidly increasing in conductivity into the top of the well screen, where the gradient shallows. If convection were fully established due to the obviously strong thermal gradient a greater degree of uniformity in conductivity due to mixing in the water column might be expected than actually occurs. This perhaps demonstrates that lab scale experimentation under controlled conditions may not quite so easily translate into the complexities of real world field conditions. However, data of this nature does clearly identify the potential for mixing and layering in boreholes due to influences of thermal gradients and screen positioning. If nothing else this data demonstrates that the water quality of the sample collected from this particular borehole will depend greatly on the placement of the sampling device and the purging strategy used.

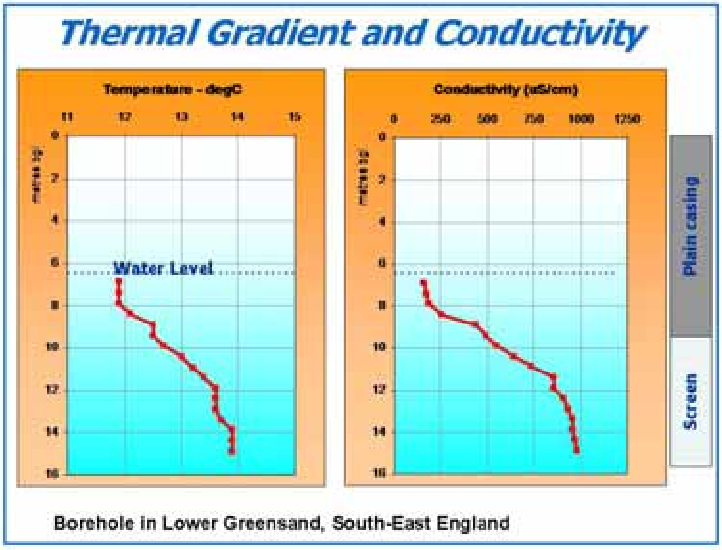

Figure 7: Temperature and conductivity profile of a borehole illustrating the presence of a natural geothermal gradient. The conductivity gradient appears to be contrary to convection mixing (data collected by author).

COMPARISON OF SAMPLE DATA FROM A SHORT-SCREENED BOREHOLE

Data selected from a controlled field sampling project by Paul (2006) is presented in Figure 8. In this work groundwater samples were obtained from 50 mm diameter monitoring boreholes with screened sections of 3.3 metre (10 feet) length from a relatively simple sand and gravel aquifer. Samples were collected using three different low flow pumping systems (bladder, peristaltic, and submersible) and two different passive diffusion samplers (labelled DMLS and PDB). Analytical results were then compared to samples obtained from a direct-push drilling system completed adjacent to the monitoring wells to simulate a multi-level sampling system.

There are some anomalies in the data (reflecting real field conditions), but the overall conclusion of this work is that the samples obtained with low flow pumps generally provide an average concentration over the entire screened interval whilst samples using discrete interval samplers provide more accurate vertical profiling information. The contaminant distribution in the passive discrete interval samplers compared closely to results from the multi level direct-push samples. In this field example there was no evidence of mixing effects impacting on water quality at specific depths.

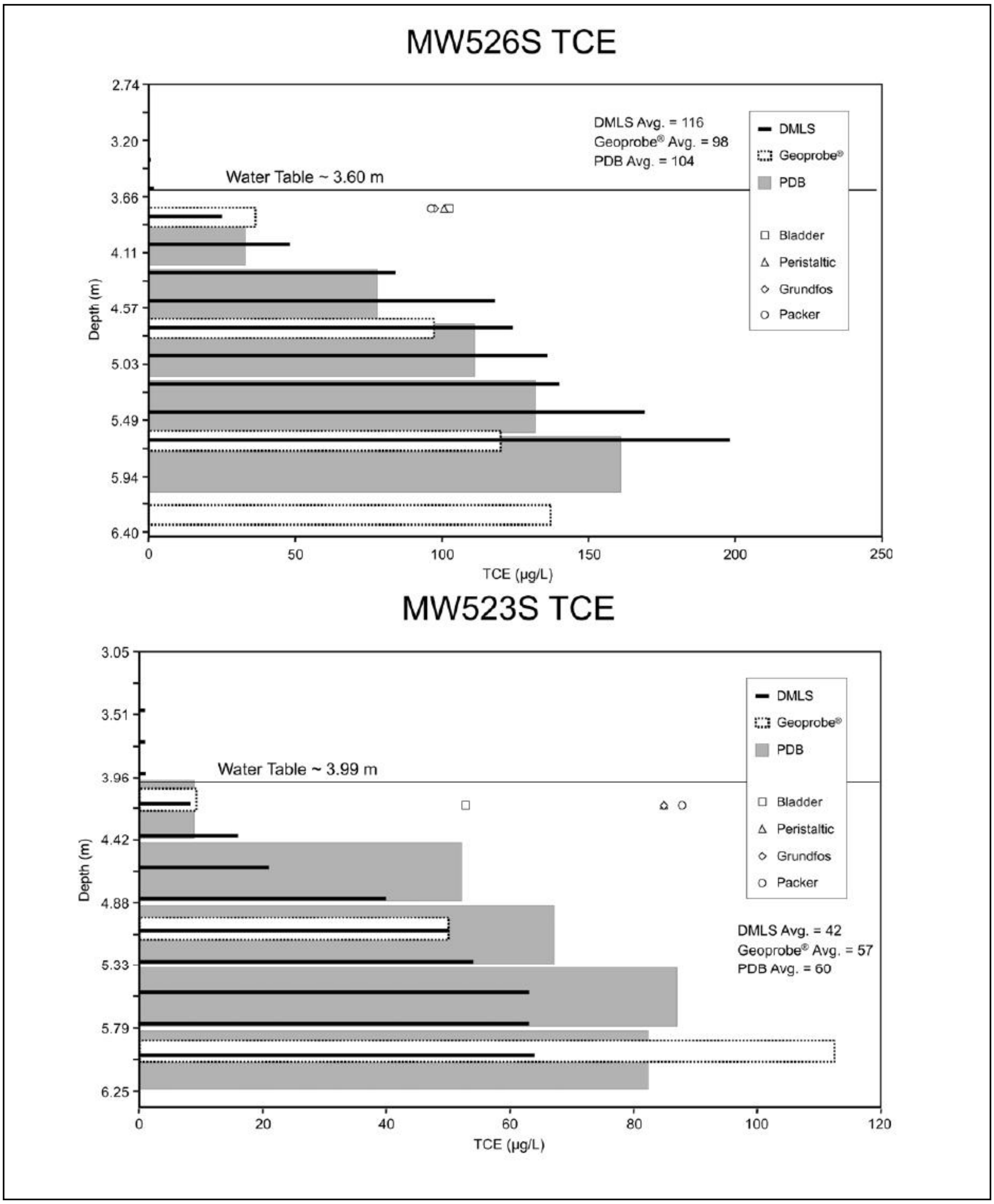

Figure 8. Comparison of TCE measurements with depth using different sampling devices and methods (Paul 2006, Figures 1 and 2)

CONCLUSIONS

There is a fascinating debate in progress on just how much impact mixing and flow within boreholes has on the quality of the groundwater sample being collected. Researchers now acknowledge that the best achievable sample obtainable from a borehole by pumping is a "flow-weighted average" of the total inflow across the entire screened interval. This will always be biased toward dominant flows from higher permeability strata. The objective of obtaining a discrete interval sample from a specified depth in a borehole using any pumped system is unrealistic, particularly in long-screened boreholes. Similarly, there is uncertainty on the efficacy of using passive samplers where ambient mixing effects could be present. On the other hand, field trials in short screened boreholes demonstrate that water quality stratification can occur in the borehole screen corresponding to the quality of groundwater in the adjacent aquifer.

Mixing effects in boreholes due to convection, hydraulic gradients and storage changes have been demonstrated at laboratory scale but have yet to be convincingly applied to field scale tests. The inference from the laboratory data is that water quality results from discrete interval sampling could be misinterpreted as a result of complex mixing processes within boreholes. Researchers have even challenged the efficacy of low flow purging, stating that sufficient water volume may not be removed to fully eliminate vertical mixing within the borehole column or to mobilise flows from all parts of the screen toward the pump.

The importance of designing boreholes with shorter screens has not fully permeated across the Atlantic, though new guidance is beginning to reinforce this message. It is worth noting that nearly all of the published research on sampling methodology has been carried out using data from short-screened shallow monitoring boreholes in the USA. The author is unaware of any good case histories evaluating the effectiveness of different purge strategies or passive sampling in long screened monitoring boreholes. Whilst long-screened boreholes are clearly not technically desirable, they are present in large numbers in Irish and UK monitoring networks. There is much to be learnt from the US debate, and a real need for research on this side of the Atlantic to harness some of the good science from the USA and put this into field practice and guidance over here.

ACKNOWLEDGEMENTS

This paper has been prepared in great haste to meet the deadlines of the IAH conference. Some of the ideas and research presented has by necessity been greatly simplified and the author strongly recommends others to read the original research papers for themselves.

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MANAGING LARGE DATASETS

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1. INTRODUCTION

Monitoring of groundwater quality and quantity is a key component of establishing and reporting on the status of groundwater bodies for Water Framework Directive compliance.

Development of suitable and workable monitoring programmes is a large undertaking which requires considerable forethought. Groundwater practitioners must not only ensure that monitoring networks are designed on a robust scientific basis, but also have an understanding of:-

- the planning and controls required to establish a suitable network;
- how the data is to be presented for interpretation;
- the volume of information which is likely to be collected;
- how the inevitably large datasets are to be managed;
- what controls are necessary to have in place to ensure the accuracy of the data obtained;

It is easy to underestimate the degree of management required to ensure that such systems are providing suitable information and to allow proper storage and manipulation of the data for scientific review.

This presentation attempts to highlight a few of the potential issues with the collection and management of large groundwater monitoring datasets and offers some solutions to these problems. This is placed in the context of a monitoring database and ancillary software used to manage quality and level data as a case study example. The monitoring comprises the collection of quality and level data from a large borehole network installed to monitor the effects of large-scale dewatering for basement construction in Central Belfast.

2. MONITORING NETWORKS

Qualitative and quantitative monitoring programmes will each have different emphasis. Water quality programmes may give special consideration to areas of already depleted quality or at risk areas such as coastal regions where pumping could induce saline intrusion and areas of known high aquifer vulnerability. Qualitative monitoring will probably be designed both, to help develop groundwater flow maps and also monitor areas of intensive groundwater abstraction to ensure sustainable use of groundwater resources.

The approach to designing monitoring networks will differ depending on the project scale and requirements, but generally, monitoring networks developed for WFD compliance are likely to comprise fairly large numbers of monitoring points, both for quality and quantity (level) monitoring. These may comprise a mixture of new and existing boreholes, plus natural springs, with the monitoring targeted at sensitive areas of aquifer but also providing good spatial coverage. While sources which are continually pumping should provide good quality data, level monitoring should be a sufficient distance from pumping sources not to be directly affected by pumping. It is therefore likely, spring monitoring points aside, that separate networks will be required.

3. MONITORING DATA

Quality data collected from monitoring programmes is likely to comprise field testing for unstable parameters (such as pH, temperature electrical conductivity and redox) along with microbiological and chemical water quality testing. This data will therefore arrive from different sources in various formats. If the sampling regime includes a large number of sites and fairly detailed analytical testing (e.g. including pesticides) is undertaken routinely, say on a quarterly basis, then before long, large quantities of data will accumulate.

The volumes of data which could potentially be generated from water level monitoring are truly immense given the modern utilisation to electronic data logging. These instruments, when managed correctly, undoubtedly provide extremely useful and detailed information on hydraulic responses, lag times and water level fluctuations, but the flip side is the generation of very large datasets.

For instance, a single water level data logger, set to log on a 15 minute interval, (a reasonable logging interval for groundwater monitoring) generates 35,000 level values per annum, excluding the associated date and time readings for each level value. If this is scaled up for regional monitoring programmes, it is not hard to imagine the average desktop computer grinding to a halt very quickly. In addition, the most commonly-used data management software, Microsoft Excel, can only hold 65,000 readings per spreadsheet and would not cope well for management of large datasets.

In addition to the need for large capacity data storage, the hydrogeologist requires great flexibility in terms of data presentation for review and analysis. To make the most of the data collected, it must be readily accessed in various presentation formats including tables, temporal plots for quality and hydrographs for water level / spring flow and there must be flexibility to adjust output so results from multiple sites can be compared against each other. Also, there may be a requirement for specific data to be exported from the dataset for use in other modelling or interpretative software.

Much of the review process requires examination of the entire dataset for each individual monitoring site, since the hydrogeologist will be looking for data trends, such as a slowly declining quality, increasing salinity, or gradually declining water level. There will therefore be little capacity to shelve older data into an archive, to keep the ‘cogs’ of the computer server wound.

4. CASE EXAMPLES

4.1 ENSURING ACCURACY OF LEVEL DATA FROM LOGGER SYSTEMS

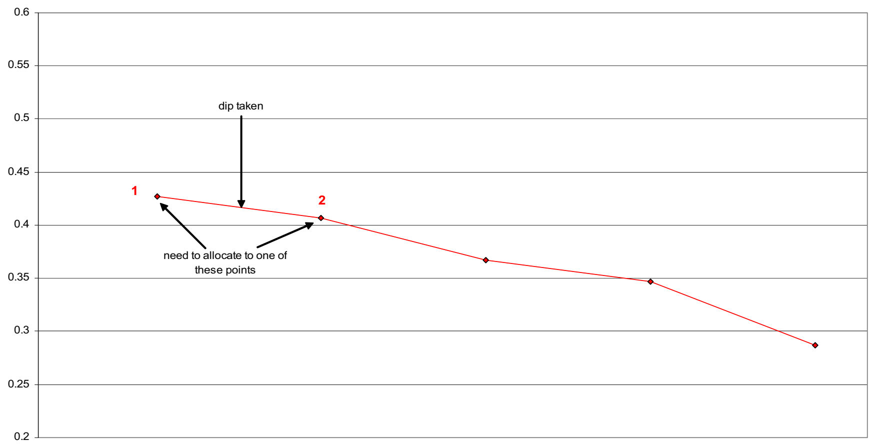
It is easy to underestimate the complexities of management of level data obtained from loggers.

Although most modern systems are, in a broad sense, easy to install and operate, it is even easier to introduce quite significant data error, at various stages, even when apparent good management practices are employed. It can be extremely difficult to detect these errors, even when examining small datasets, but arriving at an irretrievable situation could arise quickly when dealing with large monitoring datasets. Strict, but easy to follow, management practices, can prevent many errors occurring.

An example of one type of common error often overlooked relates to the timing of dip readings used to calibrate the logger, usually retrospectively once the data has been retrieved and returned to the processing office.

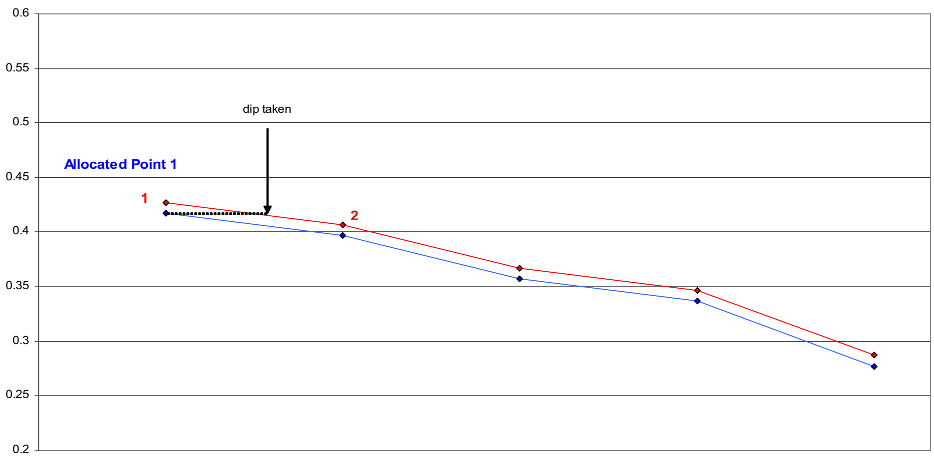
It must be remembered that loggers take readings periodically, with no readings in-between. The field technician should be aware of the logger interval, including exactly when (past each hour) the logger takes its readings and have their watch calibrated exactly to the logger time to obtain an accurate calibration dip.

If the field technician obtains a dip reading at a time between the logger readings, this must be applied/allocated to one of the data points closest to the time. Water levels calculated for the rest of the dataset rely on this allocation.

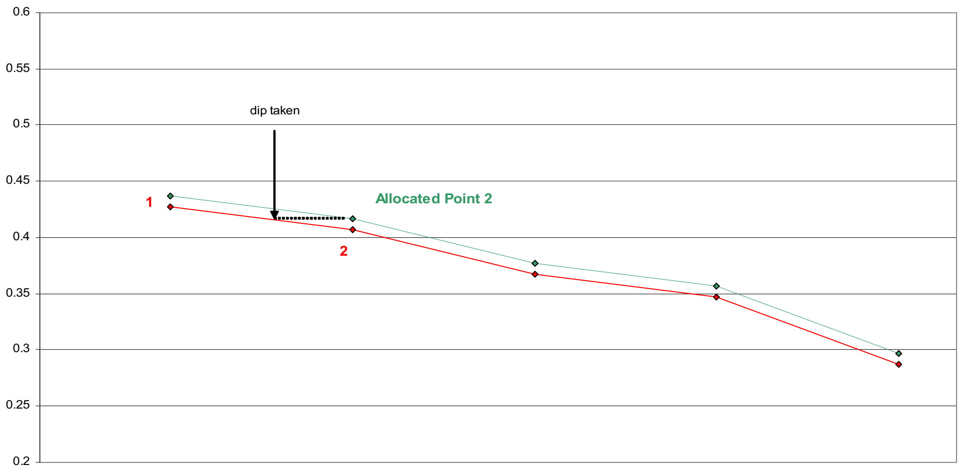


If water levels are fluctuating, neither data point will deliver an accurate hydrograph.

Assuming the red trace on the graph below represents accurate data, allocating the mis-timed dip reading to Data Point 1, results in a negative error, demonstrated by the blue trace, with the resulting hydrograph underestimating the groundwater head.



Conversely, allocation to Data Point 2, below, results in an overestimate.



These errors may be significant when analysing groundwater systems, especially for groundwater regime demonstrating small regional hydraulic gradients or when try to determine if over abstraction is occurring.

There are many more instances when simple and not obvious errors in the field or in the data processing procedure which impart error into the data. It will be important to develop good protocol for the management of logger data so that the various errors are avoided.

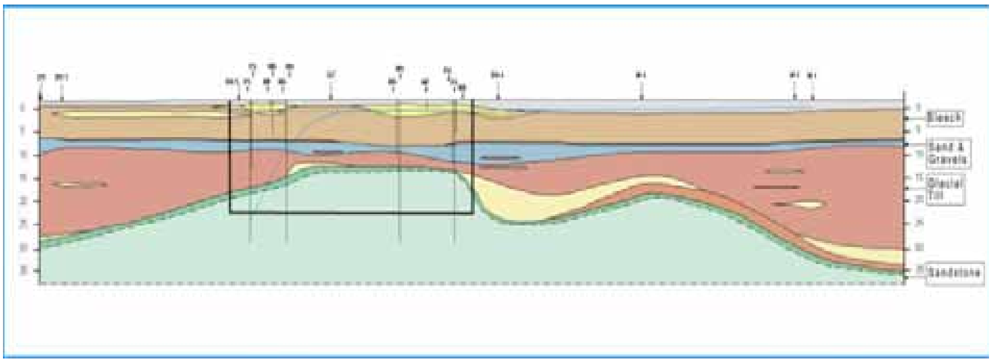
4.2 THE VICTORIA SQUARE MONITORING DATABASE

This case study is presented in the hope that it is to some degree useful for anyone involved in developing and implementing reasonably long term monitoring programmes, especially using multiple monitoring points and / or combining water quality and level monitoring. The example provides an indication of the degree of control required to manage large and assess datasets efficiently.

Rather than dwelling on the complexities of the hydrogeology observed and the difficult task of interpretation, (which I am sure will be the topic of many future hydrogeological presentations, at least in NI), this piece will concentrate on the data management element of the project.

The Victoria Square is a flagstone retail development in the centre of Belfast and represents the largest urban regeneration project the city has seen to date. The development includes the excavation of a 3-storey basement over a 1.2 hectare site area. The initial environmental / engineering impact assessment of the site undertaken as part of the planning process highlighted that groundwater quality and quantity impact would potentially be a significant risk of the development. The other high risk element identified was the potential settlement and structural impacts from the earthworks and dew-watering on the surrounding buildings, however this element will not be discussed further in the paper. In order to manage and mitigate the risk of drawdown and quality issues a groundwater monitoring programme was conditioned within the planning approval at the request of the E&HS Water Management Unit.

Diagrammatic Geological Cross Section across Victoria Square Basement



The general geological sequence comprises fill over unconsolidated estuarine alluvial deposits (locally known as sleech) over confined sands and gravels. These overlie glacial till, which blankets Sherwood Sandstone bedrock, a major aquifer. A ridge of Sherwood Sandstone occurs directly under the site and excavations into the bedrock are required.

A sheet pile ring is installed around the excavation area into bedrock. The sandstone bedrock is being actively dewatered (at a rate of around 20 litres / second) using a network of deep pumping boreholes installed around the site.

A groundwater level and quality monitoring programme was developed to monitor the effects of dewatering on the local groundwater regime in the various aquifer and aquitard units. The programme includes:-

- electronic monitoring of groundwater levels, in a total of fifty (50) locations around the city;
- level monitoring in the Lagan Estuary both upstream and downstream of the Lagan Weir;
- electronic logging of Electrical Conductivity between the site and the Lagan Estuary (to detect saline intrusion);
- on-site rainfall monitoring;
- monthly quality monitoring at 45 monitoring points;
- monthly hydrogeological report.

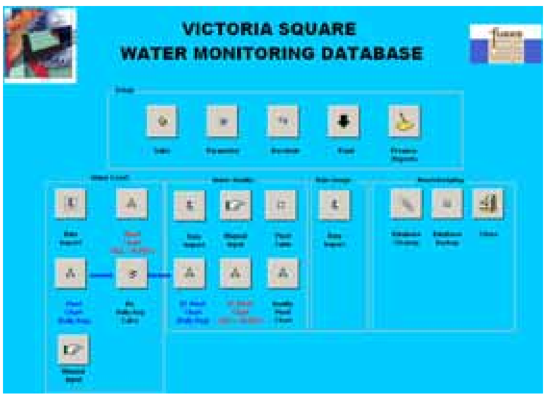
Site Location, with Monitoring Borehole Locations

Baseline monitoring was established prior to activation of the dewatering system. The dewatering period is estimated to be around 2 years, and has been underway for 1 year to date.

With the understanding that data management could potentially be the largest management issue with the monitoring programme, a database was designed in Microsoft Access to include all hydrogeological information and monitoring data for the programme. The Microsoft database was written by WYG specifically for this monitoring programme.

The database operates from a simple click screen, so no understanding of Access is required to use the system. Currently, the database holds over 20,000 water quality results and 1.75 million groundwater level readings.

Main Click-Screen



Quality Data

Field chemistry data is input from the click-screen via a data entry command. Incoming spreadsheet results from the chemical laboratory are accepted in the laboratory reporting format and allocated to the relevant boreholes using standardised sample IDs.

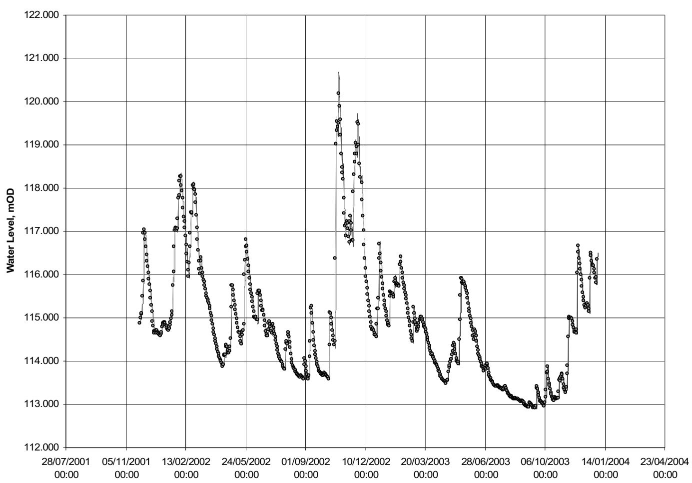
Data can be examined through live querying of the quality dataset, with outputs in the form of multi-parameter tables and temporal plots of individual parameters.

Level Monitoring

Manual dip readings are input from the main click screen via a data entry command. Loggers are compensated outside the database and raw compensated logger files are accepted as input. The database assigns each logger file to the relevant borehole, based on logger serial number (which appears in the logger files) avoiding any confusion over which data belongs to which borehole. Data is converted to Ordnance Datum, but can be plotted in raw formal also.

The database updates each borehole record, without overwriting previous data and will generate live individual or combined hydrographs on demand. At the initial stage of the project it became apparent that the processing of live spreadsheets from 15 minute interval data was extremely slow. The database was therefore improved to calculate daily average figures which can be used for faster hydrograph production. Daily averages and 15 minute data can be overlaid for Quality Assurance.

Overlay of Daily Averages and 15 Minute Data



The database also generates monthly minima data which is used to produce spatial drawdown contour maps using other software packages.

Positives and Negatives

The database can handle very large datasets and the inbuilt querying capabilities of the software allow a great deal of flexibility to be written into allowing the data to be presented in numerous ways. The ability to code the database to accept data input from loggers and laboratories and the automated allocation of the data to the correct monitoring points has reduced processing time by over 90%.

However, the standard Access package has limited graphing capabilities, and data spikes / erroneous results are not easily removed. These capabilities could, however, be added to systems to overcome these limitations.

5. SUMMARY

The development of regional groundwater monitoring networks to comply with WFD monitoring requirements will be a significant undertaking, and require a great deal of planning to design, implement and run. Strict field protocols for sampling and logger downloading and for data processing will limit errors in datasets. The volume of data collected must not be under-estimated. Easy-to-use databases, with built in automation of data processing tasks and output formats appear to be a good way of managing large hydrogeological datasets, but these must also be managed and updated regularly if problems are to be avoided.

Paper prepared by David McLorinan, an Associate within White Young Green Environmental Ltd. David is the manager of the water consultancy section within the Belfast office.

Session V

DEFINING BASELINE QUALITY FOR NITROGEN COMPOUNDS IN GROUNDWATER FLOW SYSTEMS

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ABSTRACT

Nitrate contamination of groundwaters is the largest water quality issue facing the water industry. Definition of the natural baseline concentrations are necessary to establish if pollution is taking place, as well as considering trend reversals within the context of the EC Groundwater Directive. The controls on N-species occurrence in aquifers are reviewed using examples from UK aquifers. Methods for determining natural baselines are then given, especially the trends in palaeowaters and unsaturated zone records. Baseline concentrations of around 1 mg l^{-1} NO_3-N are established for most areas, although dependent strongly on vegetation and land use. Examples from north Africa show that high natural baselines may occur, relating to leguminous vegetation cover, and illustrate the need for local studies. High nitrate and ammonium concentrations do not always signify that anthropogenic pollution is taking place.

INTRODUCTION

With the widespread application of NO_3 and NH_4 fertilisers in the mid-20th century, nitrate contamination of groundwaters has become a widespread and global problem (Parker et al. 1991). Most studies on the behaviour and fate of nitrogen focus on nitrification and denitrification processes in contaminated settings, not in pristine environments. Recently with the new European legislation (Water Framework Directive and Groundwater Directive) there has been a need to consider upward trend reversals and to what levels these reversals need to return. The need then arises to consider natural background levels since it is impossible to determine what constitutes pollution without first considering what is the natural baseline (Edmunds et al. 2003).

Nitrogen has become the focus of attention as the most significant contaminant of groundwater and the short turnover times of many groundwater systems (decades to century scale) means that nitrate concentrations have and still are rising to levels of concern for public water supplies across Europe. Treatment is costly and alternative supplies for primary supplies or for blending are limited. Nitrate concentrations in drinking water are limited by current legislation (EU **Council Directive 98/83/EC)** to 11.3 NO_3-N since above this they are believed to pose a health issue (the standard is intended to ensure that drinking water will not cause methaemoglobinaemia but other health impacts are still uncertain and not proven). Nitrate Vulnerable Zones (NVZ) have now been implemented around most water supply sources in European Countries in response to The EU Directive on Diffuse Pollution by Nitrates (91/676/EEC).

This paper will focus on the natural occurrence of nitrogen species (specifically NO_3 and NH_4) in groundwater. It will examine key processes in the N-cycle in relation to baseline occurrence and then proceed to look at methods for identifying and measuring natural background concentrations. The occurrence and trends in representative reference aquifers in the UK and Europe will be examined, especially in the light of recent studies as part of the European BaSeLiNe project (EVK1-CT1999-0006). High natural concentrations of nitrate are also found in some of the large aquifers of semi arid regions especially the Sahara and Sahel and these also are examined, both their occurrence in large sedimentary basins and in the unsaturated zone. Implications for groundwater management are then considered.

THE NITROGEN CYCLE AND BACKGROUND LEVELS

Nitrogen may occur in natural waters as NO_3^-, NO_2^-, N_2, N_2O, NH_4^+, and NH_3 and the mobility of the various nitrogen species in groundwater strongly depends on redox conditions. Under reducing conditions, the dominant species are N_2 and NH_4^+, and under aerobic (oxic) conditions nitrogen occurs stably as nitrate (NO_3^-) or rarely as nitrite (NO_2^-). Nitrite occurs as a metastable, microbiologically mediated, species in groundwater representing a transition state between nitrate and N_2. Under very alkaline and reducing conditions, nitrogen occurs as NH_3^O instead of NH_4^+.

Nitrate is a conservative species and is highly mobile in the presence of oxygen. Except at very low pH, NH_4 may be retarded during hydrochemical migration due to adsorption onto negatively charged surfaces of clay minerals and organic matter. In contrast, the negatively charged nitrogen species are not adsorbed. Apart from adsorption and redox conversions, direct uptake by plants plays an important role in the mobility of N-species, especially in the root zone and mediates entry into groundwaters.

It should be noted that atmospheric nitrate and ammonium inputs to soils and groundwaters are not negligible and although low in the past may nowadays form a significant part of the global geochemical input to aqueous systems. Although vegetation is usually a sink for nitrogen species it is noted that some plants (*leguminosae*) may release nitrogen to soil and groundwater (Sprent 1987) and this has an importance in many regions. Natural vegetation and land use therefore are prime factors in consideration of natural background levels and control of nitrogen. Outputs from soils are generally as NO_3.

Nitrate is stable in the presence of O_2 and this explains why nitrate persists in the shallow groundwater environment and in many unconfined aquifers. Understanding of redox controls are therefore important. Nitrate reduction takes place rapidly under anaerobic conditions and once oxygen is removed nitrate becomes the next electron acceptor. The rate of nitrate reduction is generally dependent on the presence of DOC in the aquifer. Although DOC is present in many groundwaters (at around 1 mg l^{-1}) it may not occur in sufficient amount or in a readily assimilable form and so microbiological (heterotrophic) catalysis may be limited in the natural groundwater environments. The presence of nitrogen reducing micro-organisms in major aquifers has been well demonstrated but the extent of their activity and efficiency in nitrate reduction is uncertain (Clark et al 1991). This is in contrast to environments polluted by organic wastes where ample reactive DOC is present.

Ferrous iron occurs as a trace element in many sedimentary environments and may be important as an electron donor for (autotrophic) nitrate reduction (Postma 1990, Oxley et al 1996) according to the reaction:

$$10Fe^{2+} + 2NO_3^- + 14H_2O \rightarrow 10FeOOH + N_2 + 18H^+ \qquad 1)$$

This reaction may be common in young sediments undergoing active diagenesis but also in major groundwater systems such as the Chalk where small amounts of Fe^{2+} are released incongruently on reaction of the impure carbonate (Edmunds et al. 1987) and may well account for the bulk of O_2 and NO_3 reaction. The stable end product in most groundwaters is N_2 gas.

DETERMINING NATURAL BASELINE CONCENTRATIONS

Three methods are given in this paper and illustrated for the estimation of natural baseline concentrations of nitrate. 1) Long term analytical records often contain nitrate data, although it is often difficult to locate these since old records have been discarded. Most groundwater agencies and utilities have good data for the past decade or so, but lack long term information that is valuable for establishing long term trends. Both in UK and in Africa, 2) the unsaturated zone records obtained from interstitial waters have provided detailed depth information over the decadal scale back to the late 19th century. In addition coring of deep aquifers has sometimes provided information on the interface between the modern and pre-industrial eras (ref). Some of the best information is derived from downgradient profiles in groundwater where the modern interface with palaeowaters can be identified and also where redox relations can be clearly seen.

NITROGEN SPECIES IN UK AND EUROPEAN REFERENCE AQUIFERS

Three UK aquifers may be compared (Edmunds et al 1984) – the Chalk, the Jurassic Limestones and the Triassic Sandstones to demonstrate the controls on N occurrence and N baselines.

In the Chalk of Berkshire, typical of other areas of European Chalk, nitrate was present at concentrations between 4.1 and 7.9 mg l^{-1} NO_3-N in the mid 1980s in samples from the unconfined aquifer. These groundwaters have clearly been influenced by diffuse anthropogenic sources. However nitrate concentrations occur below detection limits (around 0.1 mg l^{-1} NO_3-N) in the confined section and the abrupt change coincides with the redox boundary and the removal of oxygen. In the confined aquifer, ammonium is present at concentrations between 0.01 and l.0 mg l^{-1} NH_4-N and is likely to be derived from clay minerals, remaining stable under reducing conditions.

In the Lincolnshire (Jurassic) Limestone, the high nitrate concentrations found in the unconfined aquifer are rapidly reduced with the onset of reducing conditions. This aquifer undergoes loss of oxygen (and hence nitrate) at a more rapid rate than in the Chalk. The Jurassic limestones are impure carbonates and also contain low levels of organic carbon and as mentioned above it is uncertain how much microbially mediated denitrification may occur (Parker et al. 1991). The parent rock, however, contains considerable trace amounts of iron sulphide and oxidation of the rock causes a change in colour from grey to yellow brown. This suggests that reaction (1) above is an important control of the rapid reduction of both the O_2 and NO_3. The release of Fe^{2+} is also considered to be the cause of oxygen and nitrate removal in the Chalk as traces of Fe^{2+} are released during the freshwater diagenesis.

Nitrate concentrations are also high in the youngest unconfined groundwaters in the East Midlands Triassic Sandstone as a result of diffuse pollution from agrichemicals and oxidised organic wastes. Concentrations decrease markedly with increasing groundwater residence time in the unconfined zone. Decrease in NO_3-N concentrations occur upgradient of the redox boundary and precedes the loss of dissolved oxygen. The very low but detectable concentrations of nitrate in some of the older aerobic waters represent pre-industrial concentrations, which are stable in the presence of dissolved oxygen. Aerobic conditions have persisted in the red-bed sandstones (devoid of organic carbon and Fe^{2+}) for thousands of years (Edmunds and Smedley 2001; Smedley and Edmunds 2003) and conserve baseline nitrate concentrations of around 1 mg l^{-1}. As the nitrate front approaches the redox boundary however, the degree to which the aquifer would be capable of denitrification of such enhanced nitrate loadings is questionable, given the low concentrations of electron donors available in the system. The nitrate-reducing capability of the aquifer is therefore considered to be extremely limited. The concentrations of NH_4-N are below detection limit across the aquifer but detectable values (up to 0.9 mg l^{-1}) are found in the deepest, most evolved anaerobic groundwaters.

AQUIFER PROFILES

Downgradient profiles in the East Midlands aerobic aquifer indicate a pre-industrial baseline of around 1 mg l^{-1} NO_3-N. A further approach to defining the pristine groundwater quality is through interstitial water profiles – water extracted from core samples by centrifugation. Such a profile exists for the Chalk at Lulworth (Dorset UK). This 160m research borehole passed through Chalk which had been penetrated by diffuse pollution in the past half century, into pore waters (still aerobic) which contained background nitrate (Figure 2) . These baseline values also indicate an original concentration beneath Chalk grassland of around 1 mg l^{-1} NO_3-N. Comparisons may also be made from old analyses taken from records in the British Geological Survey (Table 1) for groundwater prior to the First World War which also indicate a baseline figure of 1 mg l^{-1} NO_3-N.

Site	Date	Nitrate NO_3-N (mg l^{-1})	Cl (mg l-1)
Corfe Mullen	1908	1.0	30
Durweston	1911	1.3	17
Upwey	1910	0.88	23
Sutton Poyntz	1913	0.73	19
Alton Pancras	1946	1.5	ND

Table 1. Summary nitrate information from archive data taken from British Geological Survey records.

BASELINE NITRATE IN NORTH AFRICAN GROUNDWATER

In contrast with European aquifers several studies of groundwaters of the large sedimentary basins in the Sahara/Sahel region have been conducted which record high concentrations of nitrate, apparently derived from natural processes and away from sources of human activity (Edmunds 2001; Edmunds and Gaye 1997).

One example concerns the huge Continental Intercalaire (CI) aquifer which has a recharge area in the Atlas Mountains of Algeria and discharges in the Chotts of Tunisia (Edmunds et al. 2003). In the CI aquifer oxidising conditions, as indicated by nitrate, persist for some 300 km along the section (Figure 3) and a distinct redox boundary can be recognised using the relationships between redox-sensitive species: NO_3-N, Fe (total), Mn, U, V and Cr. Thus the concentrations of total iron in solution climb from below 0.2 mg l^{-1} to values generally in the range 0.5 to >10 mg l^{-1}. Uranium, chromium and vanadium also confirm the presence of the redox boundary: uranium as the uranyl carbonate species and Cr as anionic species (e.g. CrO_4^{2-}) are generally mobile under oxidising alkaline conditions. Manganese is stable (as Mn^{2+}) over a much wider pH and Eh range than Fe^{2+} and this is reflected in the results from the CI where there is a progressive increase along the flow lines, unrelated to the NO_3/Fe-defined redox boundary.

In the CI aquifer the baseline concentrations lie between 1 and 8 mg l^{-1} NO_3-N and in the overlying aquifer, which is entirely aerobic (Guendouz et al 2003), nitrate concentrations in palaeowaters are also high (5 and 8 mg l^{-1}). The explanation of the high nitrate concentrations lies in the former vegetation cover of the region with leguminous plants predominating in the Sudan-Guinean vegetation which covered the region until the mid-Holocene (and traces of which still exist).

The high nitrate baseline concentrations have been confirmed by studies of the interstitial water studies of unsaturated zones of sandy aquifers in the region (e.g. in Senegal, Edmunds and Gaye 1997). Moisture profiles record water moving towards the aquifer at the decade to century scale in modern times. The high nitrate concentrations (also high NO_3/Cl ratios) are found beneath modern landscapes where leguminous vegetation (*e.g. Acacia spp*) still occurs.

Even in anaerobic groundwater sections the former presence of high nitrate may be deduced from the use of N_2/Ar ratios. Groundwaters from the Continental Intercalaire of the Azaouad depression in Mali contain mainly aerobic waters and contain nitrate concentrations up to 7.3 mg l^{-1} NO_3-N (Fontes et al 1991). The confined groundwaters contain a significant excess of dissolved nitrogen with respect to air saturation which is considered the product of denitrification.

The N_2/Ar ratios have been corrected for excess air using the noble gas ratios and then used to calculate the amount of NO_3 which was converted to N_2 gas. An equivalent of up to 10.2 mg l^{-1} NO_3-N has been reduced in this way.

CONCLUSIONS

Various studies are reported here which indicate that baseline concentrations for nitrate in pristine (pre-industrial or strictly pre-intensive agricultural) conditions were around 1 mg l^{-1} NO_3-N for areas of grasslands, although beneath temperate zone forests and woodlands the concentrations may have been even lower. Nitrate remains stable under aerobic conditions but is rapidly consumed following the removal of oxygen at redox boundaries, present in many aquifers. Nitrogen gas is the usual (harmless) end product of this process, although baseline concentrations of NH_4 may be stable under reducing conditions; this is important since quite often the presence of NH_4 is assumed to be an indicator of the presence of groundwater pollution.

Recent studies of groundwaters in UK and at the European scale reveal that the vast majority of groundwaters show some evidence of pollution. Rising trends towards the potable limits are commonplace and this poses a serious question for management, since the storage in the unsaturated zone may take decades still to move through to the aquifer. One possible intervention would be to apply natural attenuation by making use of the denitrification properties of reducing groundwaters – (re)siting boreholes downgradient in anaerobic aquifer sections.

It is shown from studies in semi-arid regions of Africa that it is always necessary to determine local baseline values. Nitrate concentrations approach or exceed accepted international drinking water limits in some areas. This raises intriguing questions as to human adaptability to nitrate.

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Figure 1. Downgradient trends for redox-related parameters in the Chalk aquifer of Berkshire UK. The solid line marks the redox boundary.

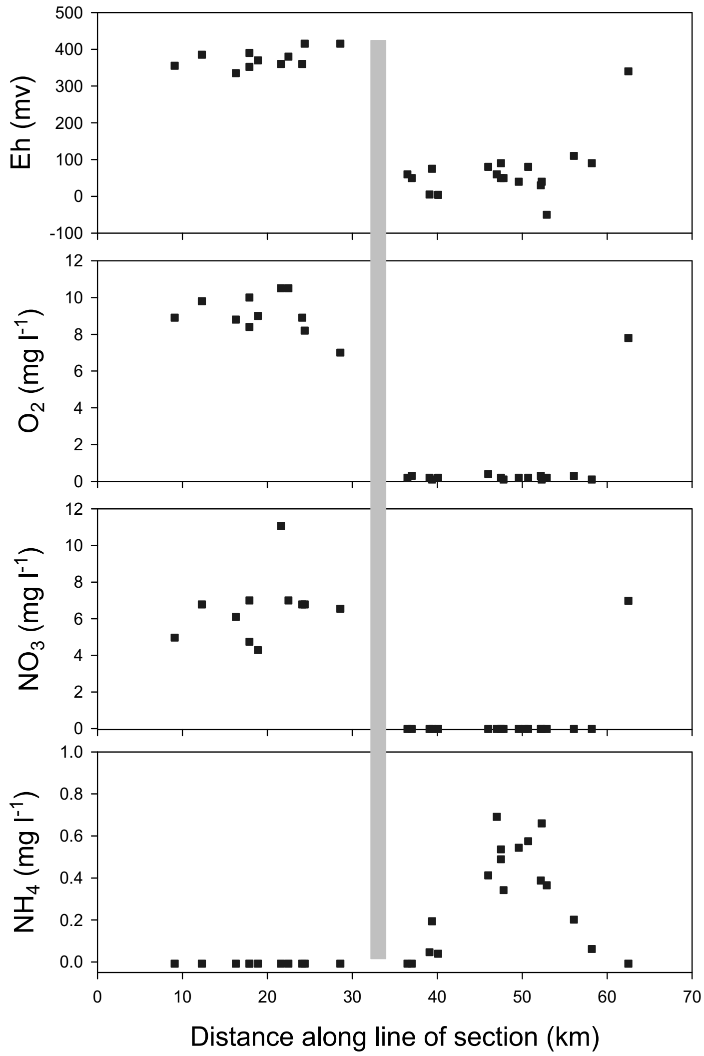


Figure 2. Interstitial water profile from a cored borehole in the Chalk aquifer at Lulworth, Dorset, UK. Nitrate concentrations are shown against other indicators of water quality. The temperature profile marks the interface at around 65m below ground level between groundwater circulation at the present day and older fresh palaeowaters emplaced during the late Pleistocene.

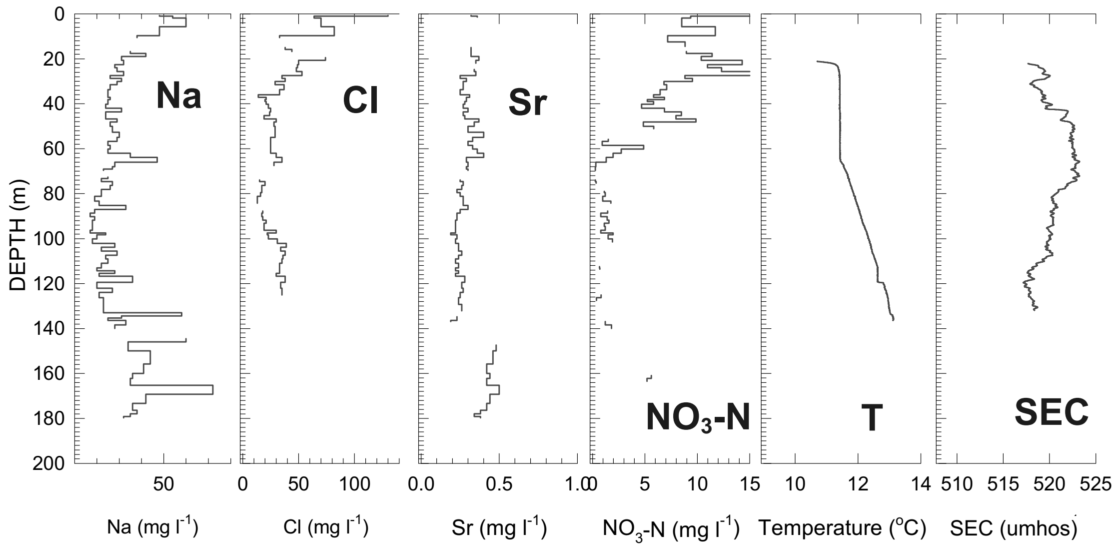
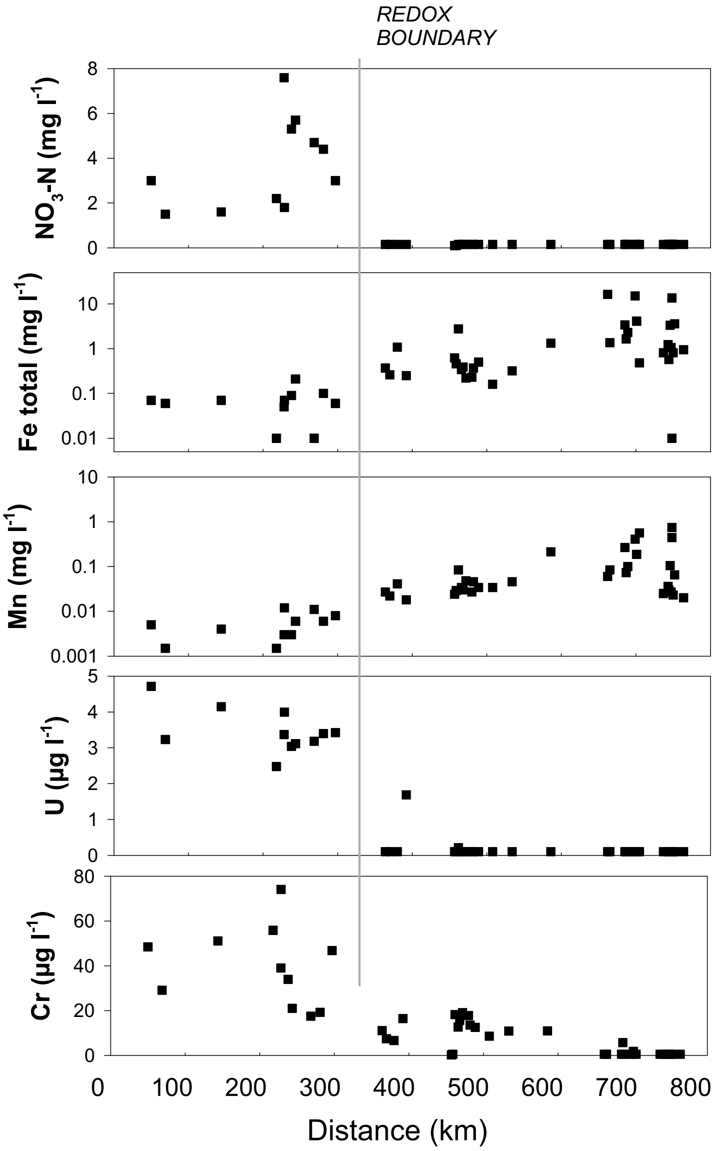


Figure 3. Downgradient profile of redox related elements and species along an 800km profile in the Continental Intercalaire aquifer (Algeria – Tunisia)



SCREENING METHODOLOGY FOR THE WATER FRAMEWORK DIRECTIVE GROUNDWATER QUALITY MONITORING NETWORK

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This paper is given on behalf of the National Groundwater Working Group (GWG)[1]

ABSTRACT

Historically, a select number of groundwater abstractions have been monitored in Ireland, ranging from public supply wells and springs to private domestic and industrial wells. In the majority of cases, the water quality data from each of these monitoring locations have been treated in isolation, although general water quality problems and trends have been reported.

A groundwater monitoring programme is required for the Water Framework Directive that provides an overview of groundwater chemical status, and of the impacts of groundwater chemistry on associated surface water and other ecological receptors in Ireland. The monitoring network design is based on a conceptual understanding of the hydrogeological system and pressures, with monitoring data used to test or validate this understanding. The groundwater quality monitoring network will consist of a surveillance monitoring network that is used to validate the Article 5 risk assessments, classify not at risk groundwater bodies and assess long-term trends, and an operational monitoring network that focuses on the pressures that are placing a groundwater body at risk.

The Groundwater Working Group has developed a national approach to designing the monitoring system that will ensure consistency across the different River Basin Districts, and this paper outlines the approach and its components. The initial monitoring network conceptualisation is complete and information on the final monitoring network will be provided to the Department of the Environment, Heritage and Local Government in June 2006, with monitoring beginning in December 2006.

1. INTRODUCTION

The Water Framework Directive (WFD) was established to create a framework for the protection of all waters through an initial characterisation of existing problems, the development of a monitoring programme to assess status, trends and the impacts of groundwater on associated receptors, and future Programmes of Measures (POMs) that will address management of identified problems.

Historically, groundwater quality monitoring in Ireland has focused on the protection of drinking water resources and investigating the impacts of point source pollution. The WFD adopts a holistic view of water resources, establishing links between groundwater and associated surface water and ecosystem receptors. Therefore, groundwater monitoring networks must be developed to improve knowledge of, and links between, groundwater and the ecological health of associated receptors.

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The WFD Article 5 Characterisation Report (2005) identified groundwater bodies that potentially have water quality problems, and monitoring data are required to verify the risk assessment carried out for this report. The amount of monitoring required depends largely on the confidence associated with the risk assessment, with more monitoring required to validate the risk assessment where confidence levels are lower.

2. WFD MONITORING REQUIREMENTS

Article 8 of the WFD requires the establishment of monitoring programmes for groundwater. The purpose of the monitoring programmes is to provide a coherent and comprehensive overview of water status within River Basin Districts in each Member State, and these programmes must be operational by 22nd December 2006. Monitoring will also support the overall water management objectives within the River Basin District and help to achieve the overall environmental objectives of the WFD. The WFD groundwater quality monitoring programmes will include:

- a surveillance monitoring network to supplement and validate the Article 5 risk assessment with respect to the risks of failing to achieve good chemical status and natural and anthropogenic trend assessments;
- an operational monitoring network to establish the status of *at risk* groundwater bodies and establish the presence of significant upward trends in the concentration of pollutants;
- appropriate monitoring to support the objectives of the Drinking Water Protected Areas and Protected Areas for habitats and species.

For a groundwater body to achieve *good chemical status*, the monitoring data will need to demonstrate that:

- the concentrations of pollutants do not exhibit the effects of saline intrusion by changes in conductivity;
- the concentrations of pollutants do not exceed the quality standards established by Member States in accordance with Article 17 of the WFD;
- the concentrations of pollutants do not result in a failure to meet the environmental objectives of Article 4 of the WFD for associated surface water and ecological receptors.

The WFD stipulates a core suite of determinands that must be sampled at groundwater quality monitoring locations. These determinands are dissolved oxygen, conductivity, pH, ammonium and nitrate. The core determinand list will be supplemented by water quality parameters that are indicative of the impact pressures identified as putting the groundwater body *at risk*. Although not formally required by the WFD, additional parameters such as temperature and a suite of major and trace ions will also be monitored to aid conceptualisation and help validate the Article 5 risk assessment. A selection of heavy metals may also be monitored to determine natural background concentrations and potential impact from anthropogenic activities.

The WFD allows flexibility in the frequency of monitoring, reflecting the variability associated with some water quality determinands. Indeed, the monitoring network locations, frequency of sampling, and determinands analysed will evolve with time as the conceptual understanding improves and POMs take effect. Where there is inadequate knowledge of the groundwater system and historical data are unavailable, monitoring frequencies will be higher until such a time has been reached when a satisfactory understanding has been achieved. In less dynamic systems, surveillance groundwater monitoring may only require two samples per year, with quarterly or even monthly samples initially taken in the more dynamic systems such as the karst (UKTAG, 2004).

Additional operational groundwater monitoring is required where the groundwater body is *at risk* from pollution or there is lower confidence in the Article 5 risk assessment. These samples will be taken between periods of surveillance monitoring, i.e. increasing the number of samples taken each year at certain locations. This frequency of monitoring will continue until there are adequate data to demonstrate improvements in water quality, and this indicates that the groundwater body is no longer at poor chemical status or is no longer *at risk*.

3. GROUNDWATER SYSTEM CHARACTERISATION AND MONITORING NETWORK DESIGN

The design of the monitoring network is based on a conceptual understanding of the hydrogeological system and pressures, with monitoring data used to test or validate this understanding. Therefore, developing a good conceptual understanding of the hydrogeological system and pressures is of paramount importance when designing a representative monitoring network. Conceptual models of the hydrogeological system and the impact of pressures on the system were used to prepare the Article 5 Characterisation Report in 2005 and additional information, e.g. well design and water quality data, has been gathered subsequently. This information was used as a starting point for the design of the groundwater monitoring network (Figure 1).

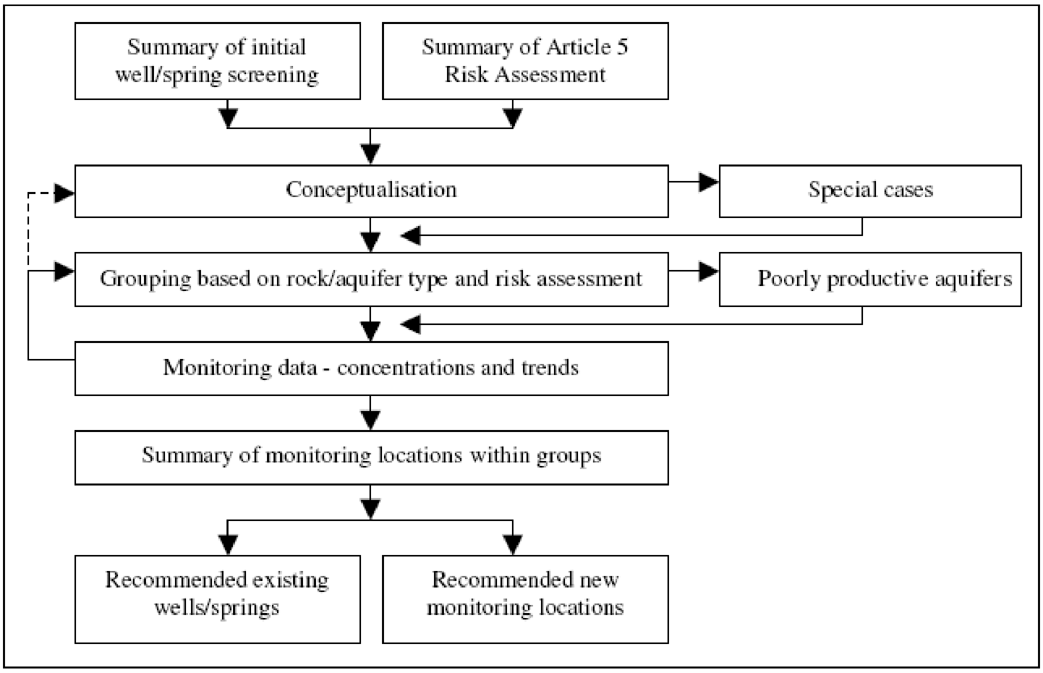

Figure 1 Monitoring Network Design Process

3.1 INITIAL SCREENING OF POTENTIAL MONITORING LOCATIONS

Monitoring locations with large zones of contribution[2] (ZOC) are preferred because the water quality is less affected by localised pressures. In this regard, monitoring locations with daily abstraction rates greater than 100 m^3 per day were prioritised for the groundwater quality network. Large springs have also been prioritised because the spring water quality effectively integrates the combined pressures impacting on the ZOC. Monitoring wells were further screened on well construction information – those incorrectly screened or with poor wellhead protection were not considered for potential inclusion the groundwater quality monitoring network.

3.2 GROUPING GROUNDWATER BODIES

The basic premise of grouping groundwater bodies is that it is not necessary to have a monitoring location in every groundwater body, as this would become prohibitively costly. In addition, reliance on water quality data from just one monitoring location is unadvisable because it is unlikely that the hydrogeological characteristics of, and pressures impacting on, the ZOC of a single monitoring point will account for spatial variation in hydrogeology and pressures across an entire groundwater body. Approximately 750 groundwater bodies were delineated nationally in Ireland for the Article 5 Characterisation Report, many of which have similar hydrogeological characteristics and pressures.

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[2] The monitoring location ZOC is primarily determined from the abstraction rate (to accommodate temporal variation, which was increased by 50%, as a safety factor), groundwater recharge within the ZOC and groundwater flow direction/topography. An allowance may have been made for uncertainties in groundwater flow direction, e.g. +/-20^{o} from the dominant flow gradient axis.

Groundwater bodies may be grouped for the purpose of monitoring if the aquifer characteristics, pathway susceptibility(ies), pressure(s) and confidence in the Article 5 risk assessment are sufficiently similar and the monitoring information still provides a reliable assessment of the WFD objectives. Therefore, by grouping groundwater bodies with similar hydrogeological characteristics and pressures, and through the establishment of a monitoring network that reflects the different hydrogeological characteristics and pressures in a groundwater body group, there is less reliance on data from a single monitoring location. Basic bedrock unit types that are common in Ireland (Table 1) have been used to group groundwater bodies at a national level. These groups have then been subdivided using the groundwater body classification and the Article 5 risk assessment, i.e. on aquifer type and risk category.

Table 1 Rock Unit Grouping in the Republic of Ireland

Rock Unit	Simplified Groups	Area (km^2)	Notes
Permo-Triassic Mudstones and Gypsum	Permo-Triassic	38	
Permo-Triassic Sandstones			
Namurian Sandstones	Silesian	5,908	
Namurian Shales			
Namurian Undifferentiated			
Westphalian Sandstones			
Westphalian Shales			
Dinantian Lower Impure Limestones	Dinantian Impure Limestones	13,900	
Dinantian Upper Impure Limestones			
Dinantian Shales and Limestones			
Dinantian (early) Sandstones, Shales and Limestones			
Dinantian Mixed Sandstones, Shales and Limestones			
Dinantian Dolomitised Limestones	Dinantian Pure Limestones & Precambrian Marbles	16,975	* may need to split out for consideration of Mg and Ca
Dinantian Pure Bedded Limestones			
Dinantian Pure Unbedded Limestones			
Precambrian Marbles			
Dinantian Sandstones	Devonian / Dinantian Sandstones	1,953	* note that Kiltorcan-type sst aquifer hydrochemistry varies N-S (see GW3 Characterisation Report)
Devonian Kiltorcan-type Sandstones			
Dinantian Mudstones and Sandstones (Cork Group)	ORS & Cork Beds	11,177	* note that Cork Beds can be variable, and that Kinsale Fmn will be more calcareous
Devonian Old Red Sandstones			
Ordovician Metasediments	Lower Palaeozoic	17,080	* note that Silurian and Ordovician Metasediment composition varies spatially and that, although it is classified as non-calcareous, it can have relatively high alkalinities particularly in the NE
Ordovician Volcanics			
Precambrian Quartzites, Gneisses & Schists			
Silurian Metasediments and Volcanics			
Cambrian Metasediments			
Basalts & other Volcanic rocks	Basalts	278	
Granites & other Igneous Intrusive rocks	Granites	3,781	

Although grouping groundwater bodies with similar hydrogeological settings and pressures resulted in the majority of groundwater bodies forming part of a groundwater body group, there were a few occasions when groundwater bodies could not be grouped. If there were no monitoring points situated within these groundwater bodies, it was necessary to propose the establishment of new monitoring sites.

3.3 MONITORING POINT REPRESENTATIVITY

Groundwater chemistry varies spatially across an aquifer because of natural variations between recharge and discharge areas and the impacts of anthropogenic pollutants. Therefore, the groundwater monitoring network is being designed so that the ZOCs of the selected monitoring locations adequately represent the variation in hydrogeology and pressures across a groundwater body or group of groundwater bodies. The ZOC of a single monitoring location will probably not be sufficient to confidently represent these variations. Consequently, a network of monitoring locations that can adequately represent the pressure and pathway variations across a groundwater body or groundwater body group is needed. A conceptual understanding of the factors that make a groundwater monitoring location representative is therefore required.

For a monitoring location to be representative of a groundwater body or groundwater body group it must be located in the same rock unit group, and the pressure and pathway characteristics (the impact potential, see Table 3) of the monitoring location ZOC must represent, in part at least, the impact potential of the overall groundwater body or groundwater body group.

The impact potential is determined by combining the pathway susceptibility (Table 2) with the pressure magnitude (Table 3). The diffuse pressure magnitude layers consider three data sources:

- the overall stocking density of foraging animals calculated using the Department of Agriculture and Food (DAF) data layer;
- the overall stocking density of pigs and poultry calculated using Central Statistics Office (CSO) data;
- the percentage of the groundwater body, groundwater body group or ZOC used for tillage as derived from the Corine land use maps.

Table 2 Pathway Susceptibility for a conservative contaminant (e.g. chloride)

PATHWAY SUSCEPTIBILITY			**Flow regime (horizontal pathway)**			
			Karst aquifers	*Fissured aquifers*	*Intergranular aquifers*	*Poorly productive aquifers*
Pathway susceptibility	Vulnerability	Extreme	E	E	H	E
		High	H	H	H	H
		Moderate	M	M	M	M
		Low	L	L	L	L
		High to Low *	H	H	H	H

* Indicates that the pathway susceptibility defaults to high where there is uncertainty surrounding the vulnerability map

Table 3 Impact Potential

IMPACT POTENTIAL		**Pathway Susceptibility**			
		Extreme	*High*	*Moderate*	*Low*
Pressure Magnitude	>2.0 LU ha^{-1} or >33% tillage	Extreme	High	High	Moderate
	1.5–2.0 LU ha^{-1} or 18-33% tillage	High	High	Moderate	Low
	1.0–1.5 LU ha^{-1} or 3-18% tillage	High	Moderate	Low	Low
	0.5–1.0 LU ha^{-1} or <3% tillage	Moderate	Low	Low	Low
	<0.5 LU ha^{-1}	Low	Low	Negligible	Negligible

Collectively, the impact potential combinations from the monitoring network should be representative of the mosaic of inputs from different pressures and hydrogeologies across the groundwater body or groundwater body group. When the impact potentials within the ZOCs of the monitoring network are proportionally similar to the impact potentials of the overall groundwater body or groundwater body group, then the monitoring network is deemed to be representative of the pressures and hydrogeology of that area. If the monitoring network is not representative of the groundwater body or groundwater body group, it may be necessary to adjust the groundwater body grouping or the monitoring network by dropping some of the monitoring locations or establishing other monitoring locations, either through the identification of existing non-monitored sources or by constructing new monitoring locations. The monitoring network will be routinely evaluated to determine if it continues to meet the requirements of the WFD.

3.4 MONITORING NETWORK DESIGN IN THE EASTERN RIVER BASIN DISTRICT

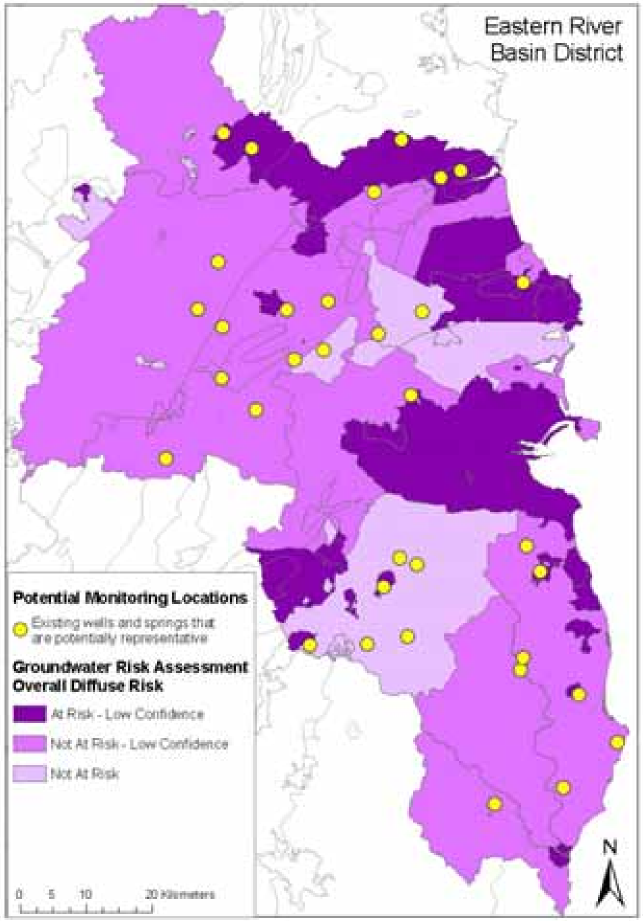
The principles of groundwater body grouping and representativity are exemplified below using an example from the Eastern River Basin District (ERBD). Seventy-five groundwater bodies were delineated in the Eastern River Basin District (ERBD) for the Article 5 Characterisation Report, of which two were classified as *at risk* from either point source or diffuse pressures, forty-eight were classified as *probably at risk* from either point source or diffuse pressures, fifteen were classified as *probably not at risk* from either point source or diffuse pressures and ten were classified as *not at risk* from either point source or diffuse pressures (Figure 2). Initial assessment of existing wells and springs in the ERBD indicated that 34 monitoring locations had abstractions greater than 100 m^3 per day and satisfied the other initial screening criteria.

Draft groundwater body groups were established using the rock units in Table 1 and these were subdivided using the groundwater body classification and the Article 5 risk assessment[3]. Existing groundwater quality information was used to check groupings e.g. if all the existing monitoring locations in a proposed group indicated that concentrations and trends for a particular determinand were similar, then the grouping was satisfactory for those determinands. However, if the water quality and trends were vastly different at the monitoring locations in a proposed group, then the grouping was revised. Additionally, if the water quality and trends at a small number of monitoring locations was vastly different to the majority of the other locations in a proposed group, and hydrogeological experience and judgement indicated that these differences should probably not occur, the monitoring locations were either excluded or treated with caution.

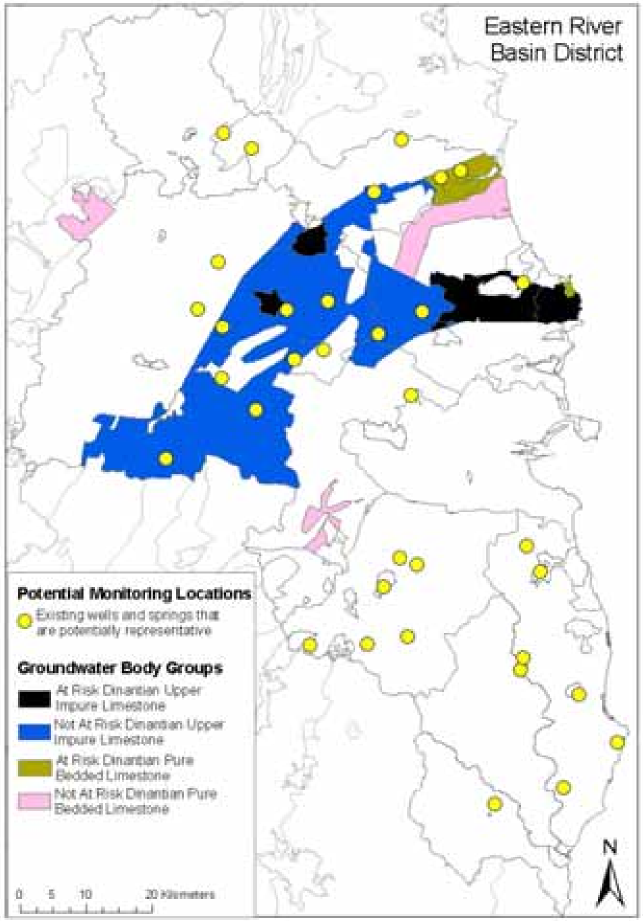
Major rock types in the ERBD include, the Dinantian Pure Bedded Limestones and Dinantian Upper Impure Limestones. These rock types were subdivided using the groundwater body classification and the Article 5 risk assessment to establish groundwater groups that were *at risk* or *not at risk* from diffuse and point source pressures (Figure 2). Of the twelve existing monitoring locations in the *not at risk* Dinantian Upper Impure Limestone grouping, the combined impact potential from four monitoring locations was deemed to be representative of the impact potential across the groundwater group (Figure 2). There were no existing monitoring locations in the *not at risk* Dinantian Pure Bedded Limestone grouping and hydrogeological experience and judgement was used to determine that three new monitoring locations would be required to represent the variations in impact potential across this groundwater body grouping. There was a single existing monitoring location in the *at risk* Dinantian Upper Impure Limestone grouping, and hydrogeological experience and judgement was used to determine that two additional monitoring locations would be required to represent the variations in impact potential across this groundwater body grouping, whilst an additional monitoring location may be required in the *at risk* Dinantian Pure Bedded Limestone grouping because the combined impact potential from both existing monitoring locations is not deemed to be representative of the variations in impact potential across the groundwater body grouping.

Currently work is ongoing to determine the monitoring network requirements for point sources, urban areas and groundwater dependent terrestrial ecosystems in the ERBD, although monitoring will largely focus on individual groundwater bodies associated with these pressures and receptors.

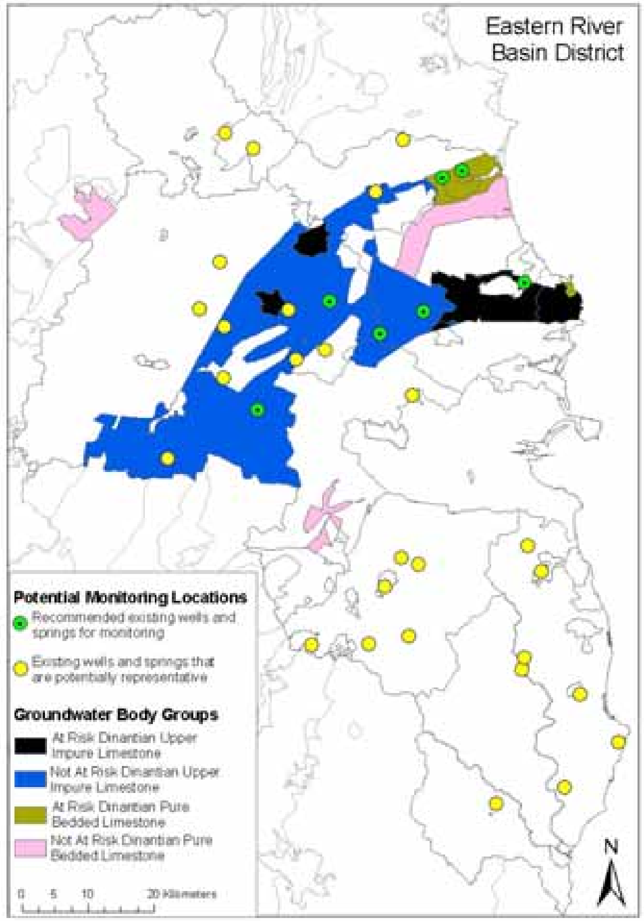
[3] Currently the risk assessments are being revised to incorporate new groundwater vulnerability and soil maps and efforts are being made to try and improve confidence in the Article 5 risk assessment, so the groundwater bodies will be classified as either *at risk* or *not at risk* only. Therefore, until this work has been completed, the groundwater body grouping is a draft, with *probably at risk* considered to be *at risk* and *probably not at risk* considered to be *not at risk*.



Overall Diffuse Groundwater Risk Assessment



Groundwater Body Grouping



Recommended Monitoring Locations

Figure 2 An example of monitoring network development in the ERBD

3.5 MONITORING IN POORLY PRODUCTIVE GROUNDWATER BODIES

Approximately 65 per cent of the bedrock in the Republic of Ireland is classified as poorly productive aquifers. The groundwater flow paths in poorly productive aquifers can be considered in four categories: (i) an upper weathered zone, (ii) an interconnected fissured zone, (iii) larger isolated fissures in massive rock, and (iv) permeable fault zones. Within these rock types, flow path lengths are typically only 10's to 100's of metres, and groundwater contribution to surface waters[4] is estimated to be less than 20 per cent.

Monitoring of poorly productive rocks is particularly challenging given the large number of small, localised flow systems. Whilst larger abstractions are usually associated with fault zones, small private abstractions from the poorly productive rocks are probably associated with isolated fissures or more transmissive horizons at shallower bedrock depths, and have ZOCs of only a few 10's m^2. Consequently, water quality may be influenced by localised anthropogenic pressures, which do not necessarily reflect the pressures on the groundwater body as a whole.

Since groundwater contributions from poorly productive rocks to surface water receptors are limited, and the impacts on groundwater relate to local pressures, the development of a regional monitoring network of sufficient size to record all the variations in impact potential in the poorly productive aquifers would be very expensive and would not represent good value for money. Instead, monitoring of the poorly transmissive rocks will be carried out at a small number of locations nationally to confirm assumptions made regarding the conceptual model of poorly productive rocks. A consensus has been developed in the groundwater working group that monitoring in poorly productive areas should be subject to separate detailed pilot studies with hydrogeological information gathered during the installation of nested piezometers. The pilot studies will mainly focus on a few areas with existing surface water quality problems, thereby enabling an estimation of the contribution from groundwater to associated surface water bodies. Monitoring will also be carried out at larger abstractions in poorly productive areas because these are generally associated with locally important fault and fracture zones (e.g. pegmatites in granite) with larger ZOCs.

3.6 RECEPTOR MONITORING

Under the WFD, if pollutant concentrations in groundwater arise as a result of anthropogenic alterations and these significantly compromise the environmental objectives of an associated surface water receptor, POMs must be introduced to reduce the impact from groundwater on the associated surface water receptor.

Therefore, the groundwater contribution to associated surface water receptors and the concentration of pollutants in groundwater must be known to determine if groundwater is detrimentally impacting on the quality of water in the associated surface water receptor. Monitoring locations will be situated in groundwater discharge areas where the associated surface water receptors are known to be *at risk* from pollution, and groundwater is potentially a significant contributor to the problem. Terrestrial ecosystems that are dependent on groundwater will also be monitored to determine the flow and level of dependence on groundwater, the impacts of groundwater on the ecosystem, and whether the ecosystem is significantly damaged. If groundwater is known to be a significant contributor to a damaged terrestrial ecosystem, then monitoring data will be used to confirm that groundwater contributions are the cause of the problem, and that applied POMs are effective.

Article 7 of the WFD also requires monitoring of groundwater abstractions from Drinking Water Protected Areas that are greater than 10 m^3 per day or supply on average more than 50 persons. Unlike monitoring for the Drinking Water Directive, samples of raw water (prior to treatment) will have to be taken. If the water quality deterioration is significant enough to potentially result in

[4] A project managed by the South West River Basin District is currently investigating groundwater contributions to surface water for seven different hydrogeological scenarios in Ireland, and their work should help to provide a greater understanding of groundwater contributions to associated surface water receptors.

additional purification/treatment, the associated groundwater body will be considered to be at poor status. In such cases POMs will be required to address the deterioration in groundwater quality.

3.7 PREVENT OR LIMIT MONITORING

Surveillance and operational monitoring will be used to provide a coherent overview of groundwater status, although there will be a need for additional operational groundwater monitoring that will be specifically aimed at point source pressures. However, if there are unexpected water quality problems that are associated with an unknown risk or if there are persistent trends indicating water quality deterioration that have not been reversed by POMs, then investigative monitoring should be established to assess the problem and monitor the effectiveness of any future remedial action.

Groundwater monitoring sites are already sampled for certain IPPC and waste licensed activities, and also where the conditions of planning regulations stipulate groundwater monitoring. Some of these monitoring sites will be incorporated into the operational monitoring network, although additional monitoring upgradient and downgradient of the point source may be required, if only to demonstrate the effectiveness of POMs. Water quality data from these assessments will be integrated into operational monitoring programmes. There may also be a need to monitor the impacts of multiple point sources on groundwater quality, for example, to study urban impacts on groundwater quality. Monitoring the myriad of potential point source pressures across all urban areas in Ireland would require substantial resources. The Eastern River Basin District (ERBD) is currently leading a separate POM project to assess the likely impacts of urban areas on groundwater quality. Results of this project will be integrated into final monitoring system designs, prior to publication of river basin management plans in 2008.

4. REPORTING

Historically, reporting on groundwater quality has been used to study the impacts of point source pollution, or determine the level of treatment at drinking water sources, or to provide information on general water quality trends. Under the WFD, monitoring data should be reported to Europe at the end of the River Basin Management Plan i.e. every six years. However, internal reporting will probably take place annually to accommodate modifications of the monitoring network and ensure the effectiveness of POMs.

Reporting will include documentation on the monitoring location, monitoring frequency, water quality parameters investigated, assumptions made and status assessments based on data from the monitoring programme. Confidence limits associated with the quality of monitoring data gathered and the interpretation used for the status assessments must also be reported.

Improvements in data collection technologies, such as data loggers and the introduction of remote downloading technologies that enable the transfer of data over a mobile phone connection will reduce the cost of data collection. Although data loggers have been developed to include basic field chemistry parameters such as conductivity, temperature and pH, for the majority of chemical determinands, samples will have to be gathered in the field. However, the introduction of palm pilot technologies and the establishment of the Water Information System for Europe (WISE) data storage repository will enable faster reporting, with access granted to those relevant parties who supply data and those who will report on the data.

5. CONCLUSIONS

The implementation of the WFD in Ireland has brought about a need for expanded and/or improved groundwater monitoring programmes. Whereas historically groundwater quality data was gathered to allay fears surrounding drinking water abstractions and to monitor the impacts of point source contamination, the WFD has introduced a framework that requires monitoring networks that have the potential to improve the understanding of processes associated with groundwater and it’s interaction with surface water and other receptors, potentially resulting in improved water quality in all water bodies.

The hierarchical approach described in this paper enables an overall assessment of groundwater quality to be made, with more focused monitoring applied in those areas that are deemed to be *at risk* from pollution. Limited monitoring in poorly productive aquifers conserves resources for monitoring in the productive aquifers, which can be focussed on groundwater bodies that are *at risk* from pollution and *at risk* surface water receptors that are dependent on groundwater contributions. The creation of a representative monitoring network maximises resources further because this network provides data that aids conceptual understanding of the hydrogeology and pressures, provides information that verifies the Article 5 risk assessment, and provides data that may be used to assess the effectiveness of programmes of measures.

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7. ACKNOWLEDGMENTS

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ESTABLISHING NATURAL BACKGROUND LEVELS FOR GROUNDWATER IN IRELAND

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ABSTRACT

The Water Framework Directive (WFD) requires that the status of all water bodies be established. As part of the establishment of the chemical status of groundwater bodies in Ireland this study was undertaken to assess the natural background concentrations of water quality parameters. It is envisaged that the findings of this study will be used as part of the process of defining Environmental Quality Standards (EQS) and ultimately groundwater body status. A methodology developed for the analysis of natural background levels in the South Eastern River Basin District (SERBD) was modified and applied using data from each River Basin District (RBD) to assess Natural Backgrounds Levels (NBL) for a range of parameters on a national scale. A tool was developed as part of the study to identify monitoring points least likely to be affected by anthropogenic activity. This resulted in the identification of 42 monitoring points, yielding 892 water quality records distributed nationally across all seven RBDs. Variation is expected in natural background levels based on the composition of the aquifer units from which the groundwater samples are collected. A simplified grouping process was undertaken to combine monitoring points located in aquifers of similar lithological composition. Ranges of NBLs have therefore been proposed based on the groups derived. It was possible to establish a single NBL for some parameters which were considered not to be influenced by lithology. The study produced proposed NBLs for 15 out of 28 parameters identified for analysis by the Environmental Protection Agency (EPA). NBLs could not be derived for 13 parameters. These were primarily heavy metals which are not typically tested for, and because of the high number of limits of detection (LOD) for these parameters in the database.

INTRODUCTION

As part of the Further Characterisation programme required under Article 5 and Annex II of the Water Framework Directive (WFD) 2000, the South Eastern River Basin District (SERBD) project team were assigned the task of establishing Natural Background Levels (NBL) for Irish groundwaters. The task was part of a larger process involving the establishment of Environmental Quality Standards (EQS) for all waters. EQS are required to determine the effectiveness of Programmes of Measures to improve or preserve water quality status required under Article 11 of the WFD. O'Callaghan Moran & Associates (OCM), the SERBD consultant appointed by the project team to address groundwater aspects of the SERBD plan, were requested to undertake the analysis of NBL for Irish groundwaters. OCM completed the task in consultation with the Irish Groundwater Working Group, the Geological Survey of Ireland (GSI) and the Environmental Protection Agency (EPA).

METHODOLOGY

ERTDI METHODOLOGY

The Environmental Protection Agency (EPA) commissioned a study in 2003 as part of the Environmental Research Technology Development and Innovation Programme 2000 – 2006 (ERTDI) to develop a methodology for the characterisation of unpolluted groundwater. The study was undertaken by Environmental Simulations International Ltd (ESI) and TMS Environment Limited (TMS). The primary purpose of the project was establishing natural background levels for a range of parameters to use as a standard against which the impact of anthropogenic activity could be measured.

The project involved a review of international groundwater characterisation methodologies with a focus on methodologies proposed by other EU member states. The methodology was developed using available monitoring data in the SERBD. Any data gaps in the existing monitoring system were to be identified.

The ERDTI methodology recommended a hierarchy of preferred analytical techniques to establish NBLs. The most preferred technique was historical groundwater data analysis followed by recent groundwater data analysis. The lower ranked options were the analysis of rainfall inputs and the analysis of surface water chemistry. The approaches are outlined briefly below.

Historical Data Analysis

The highest ranking technique is the analysis of historical data from monitoring points that were considered to be unaffected by anthropogenic activity. Data from monitoring points collected prior to 1973 were considered to be representative of groundwater that was not significantly affected by anthropogenic activity. The year 1973 was used as it marked the introduction of the Common Agricultural Policy in Europe and Ireland's entry into the European Economic Community (EEC).

For this study a limited amount of pre 1973 data was provided to OCM by Geoff Wright from the GSI. The data was of limited use because of uncertainty and limitations of the analytical techniques used at the time, precise location of some monitoring points and also some concern about the reliability of construction of some of the monitoring points. It was concluded that the use of the limited pre-1973 data could not be used to establish NBL for the study.

Recent Data Analyses

While it may appear that all areas are affected by anthropogenic pressures some areas are less affected due to natural protection (e.g. low vulnerability) or lower intensity of pressures (e.g. upland areas). Where recent analytical data are to be assessed at least 50 data records from several representative monitoring points are required. Of the suggested methodologies this approach was most applicable in the Irish context.

Atmospheric Inputs

This technique involved the use of rainfall quality data for the following range of parameters: SO_4, NO_3, NH_4, Cl, H, Na, K, Ca, Mg, alkalinity and electrical conductivity. This technique assumed recharge and actual evapotranspiration rates are well defined and established and also that the fate of species in the soil and subsoil are known. Because of uncertainties in the use of rainfall data and limited information on the fate of species migrating through the soil zone it was decided not to apply this technique in the study following consultation with the Irish Groundwater Working Group (IGWG).

Surface Waters

Finally, the use of surface water quality during recession periods to determine NBLs in groundwater was considered. While it is clear that river flow during extended recession periods is dominated by groundwater discharges to the river, there is considerable uncertainty about how much of the total flow is derived from groundwater at other times. Even if this is known it is more problematic to discern the proportions of the species measured in the river coming from groundwater versus that representing surface runoff. In addition, only historical i.e. pre-1973 data, could be used and such data is very limited. It was decided not to apply this technique in the study following consultation with the IGWG because of these uncertainties.

INITIAL RESULTS

The proposed methodology was trialled by ESL & TMS in four lithological groupings distributed across the SERBD to reflect a range of hydrogeological settings. The methodology and results were published in 2004 and reviewed by the IGWG. It was decided to apply the methodology in more detail to the SERBD and to increase the range of parameters originally included in the analysis.

Initial results from the SERBD indicated that a more stringent method of selecting representative monitoring points was required and that this would exclude many monitoring points used in the original assessment. The IGWG considered that the methodology needed to be applied to a larger data set. The methodology was re-trialled drawing on monitoring data from each River Basin District to ensure a national distribution of derived values.

DATA VALIDATION

It was clear from the initial trial that relatively few monitoring points were suitable for inclusion in the analysis. A methodology for selecting monitoring points which were representative of natural background conditions and which should not be significantly affected by anthropogenic activity was developed. In addition to methods suggested in the ERTDI report the IGWG developed a screening methodology which incorporated key concepts and methods used in Article 5 Risk Assessment of Groundwater Bodies to exclude monitoring points and data which were not suitable for inclusion in the analysis. The methods suggested in the ERTDI report are outlined below.

Analysis of Time Series

Data were plotted for individual points in time series to highlight, and where appropriate, remove anomalous points. These anomalous points were sometimes identified as errors in data entry (e.g. a decimal point in wrong place, or the use of incorrect units). While it was not always possible to explain anomalous values, where there was a clear discrepancy such values were eliminated from the dataset.

Ionic Balance Calculations

Where there were sufficient data ionic balance calculations were performed. Where the balance is in error by +/- 10% it can suggest an error in the reported results. The method requires data on the major ions (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, SO_4^{2-}, Cl^{-}, CO_3^{2-} + HCO_3^{-}) which in many instances were not all available. For this reason not all data with a poor ionic balance were excluded, rather it was used as an indicator for a greater level of inspection of the data and for caution when using the data in the study.

Monitoring Point Screening

In 2003 OCM completed a screening programme to assess the suitability for inclusion of groundwater wells in a monitoring programme. The screening process was developed further by the IGWG. The screening methodology was used in 2005 nationally to ensure a consistent approach to screening of groundwater monitoring wells for inclusion in a groundwater monitoring programme. The IGWG methodology was used to determine if a monitoring well was representative of the aquifer it was located in. This involved a site visit and the documentation of all relevant aspects of the monitoring points. Details on borehole construction, well head protection, abstraction rates and local point sources of pollution were important in determining if a point was suitable. Any monitoring point which was not deemed suitable by this methodology was not included in the analysis.

Monitoring Point Representativity

While screening ensured that the monitoring point was representative of the aquifer it was situated in, it was also necessary to determine if that point of the aquifer is representative of natural background conditions.

The IGWG developed a series of groundwater risk assessment tools for Article 5 Characterisation. One key methodology was the development of Impact Potential maps, which are generated in a Geographic Information System (GIS) by combining information on aquifer type and vulnerability, soil types, and pressure magnitude. In the risk assessment GWBs with a high percentage of high or moderate impact potential were considered to be potentially at risk of being affected by anthropogenic activity. This method was further developed to determine if monitoring points were representative of natural background conditions. Where less than 10% of a Zone of Contribution (ZOC) of a monitoring point was at moderate or high impact potential it was considered to be suitable for consideration in the NBL study. ZOCs were delineated for all monitoring points to be included in the analysis. Where the GSI had already delineated a ZOC that data was used to delineate the ZOC.

SELECTED MONITORING POINTS

Monitoring Points were identified in each River Basin District using the above methodology. The distribution of points nationally and the number of available monitoring records was as follows:

Table 1 – Monitoring Points included in Natural Background Analysis

RBD	Monitoring Points	Records
SERBD	12	221
SWRBD	3	33
ShRBD	3	225
WRBD	9	66
ERBD	10	284
NSRBD	5	63
Total	42	892

DATA ANALYSIS

ANALYSIS METHODOLOGIES

The ERTDI methodology can be used to define an upper limit (and lower limit if necessary) of Natural Background Levels. Beyond this limit the quality of a given sample can no longer be said to be representative of natural background conditions. Two techniques were used to analyse the data. Within the context of this report these are called the Statistical Approach and Cumulative Frequency Plots (CFP).

Analysis of Limits of Detection

Limit of Detection (LOD) values are frequently recorded in water quality data sets. These can cause problems when trying to determine the significance of such a value and in using it statistically. For this analysis LOD records were substituted with a value which was half of the LOD i.e. <0.02 was replaced with 0.01. Where over 50% of the data were LODs the dataset was not suitable for analysis and it was therefore considered that the NBL established would be an LOD limit.

Cumulative Frequency Plots

Upper and lower natural background limit values can be obtained from cumulative frequency plots. The shape of the curve on a cumulative frequency plot can be used to assist in understanding the geochemistry of the groundwater body. Some of the qualitative information that can be obtained is shown in Figure 1. Log normal distributions are to be expected for many solutes in naturally occurring systems. Hence the concentrations are plotted on a log scale (except pH as it is already a log variable). Upper and lower limits are obtained by extrapolating the distribution identified as the natural distribution (i.e. the straight line section of the graph) up or down to the 95^{th} or 5^{th} percentiles. The approach is interpretative and there is a certain degree of subjectivity in deriving limits using this technique.

The Statistical Approach

Sinclair (1974) recommends a minimum number of 100 data points to produce a cumulative frequency plot. However because data was likely to be limited, where at least 50 records were available the plots were still considered to be statistically viable in the ERTDI methodology. Where there were not enough data to create a suitable CFP the following statistical approach was used.

1. The base 10 logarithm (y_i) of all concentrations (C_i) is calculated, substituting for LOD values with 50% LOD where suitable. Compute the mean of the log data ($\bar{y}$) and the unbiased standard deviation of the log data (s_y).

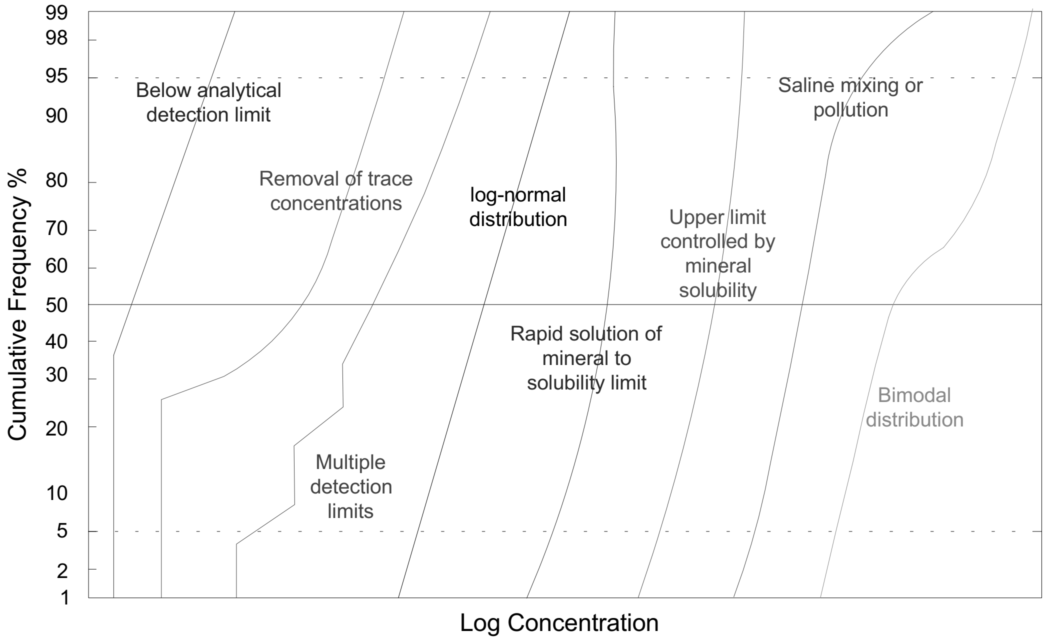
$$y_i = \log_{10}(C_i) \qquad \bar{y} = \frac{1}{n}\sum_i^n y_i \qquad s_y = \sqrt{\frac{\sum_i^n (y_i - \bar{y})^2}{n-1}}$$

2. The upper limit for natural background concentration (C_{upper}) is set at ten to the power of the mean log plus two log standard deviations. The lower limit (C_{lower}) is to be set at ten to the power of the mean log of the data set minus two standard deviations.

$$C_{Upper} = 10^{\bar{y}+2\cdot s_y} \qquad C_{Lower} = 10^{\bar{y}-2\cdot s_y}$$

3. Where the limit is less than the LOD for the species the limit is set to the lowest LOD in the data series.

Figure 1. Examples of concentration profiles on cumulative frequency plots (EPA 2004).



NATURAL VARIATIONS IN HYDROCHEMISTRY

Aquifer lithology influences the hydrochemistry of the groundwater moving through it. Data was grouped according to the lithological influence on various parameters. A grouping strategy was developed in conjunction with the ERTDI author (Steve Buss) and the GSI. It was concluded that certain parameters were not significantly influenced by lithology. These are described herein as "global parameters" and include Nitrate, Chloride, Sulphate, Zinc, TOC, DO and Orthophosphate.

The following four lithological groups were proposed: Karst Limestones, Devonian Sandstones, Lower Palaeozoics and a mixed group including other lithologies not represented by the first three. The mixed group included Dinantian impure limestones, sandstones and shales, Namurian and Westphalian sandstones and Precambrian marbles. It is acknowledged that a more detailed breakdown to reflect lithological variation could have been developed. However, given the limitation of the data sets and numbers and distribution of monitoring points it was considered by the IGWG that a simplified grouping strategy was reasonably representative.

RESULTS

Results are presented for the "global" parameters in Table 2 and for grouped parameters in Table 3. The cumulative frequency plots for the "global" parameters are shown in Figure 2. (Other cumulative frequency plots are included in the SERBD Report on the study which will be available in June on the WFD website). Due to the high number of LODs (>50%), analysis was not possible for the following parameters: Aluminium, Ammonium, Arsenic, Boron, Cadmium, Chromium, Copper, Lead, Mercury, Nickel and Nitrite.

Table 2 – Natural Background Levels for ungrouped parameters

Parameter	IGV	95th percentile	5th percentile	Mean
Nitrate (mg/l NO_3)	25	8.9	0.1	3.0
Chloride (mg/l)	30	21	8	14
Orthophosphate (mg/l $P0_4$)*	0.03	0.07	0.0015	0.01
Zinc (ug/l)	100	61.28	1.1	21
Sulphate (mg/l SO_4)	200	36	11.9	21
T.O.C. (mg/l)		4.8	0.2	1.5

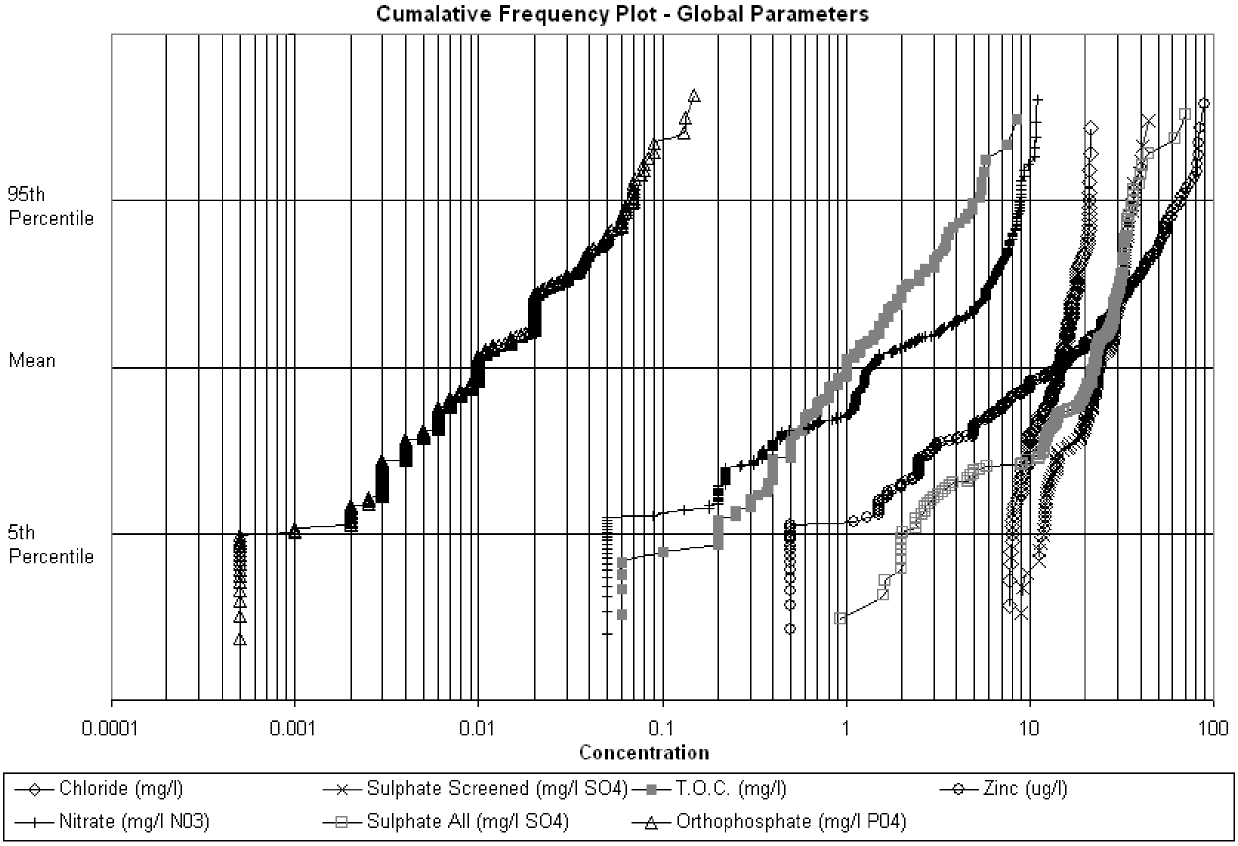
* Values were not included in NBL results and are included only for information.

Table 3 – Natural Background Levels for parameters grouped by lithology

Parameter	IGV	95th percentile				5^{th} percentile				Mean			
		Lower Palaeozoic	Devonian Sst	Karst Lst	Mixed	Lower Palaeozoic	Devonian Sst	Karst Lst	Mixed	Lower Palaeozoic	Devonian Sst	Karst Lst	Mixed
EC (uS/cm)	1000	706	228	768	781	260	134	414	372	497	181	638	547
Alkalinity (mg/l $CaCO_3$)		379*	92	382	399	176*	11	170	139	263	58	282	251
pH	6.5-9.5	7.8	7.4	7.7	7.8	6.9	5.3	7.0	6.7	7.3	6.3	7.4	7.4
Hardness (mg/l $CaCO_3$)		388	96	437	381	59	19	172	146	229	64	301	246
Potassium (mg/l)	5	5.5*	2.5	4.7	5.4	0.6*	0.4	0.5	0.5	2.0	1.3	2.0	2.8
Sodium (mg/l)	150	40.8*	19	29	95	8.6*	7.8	6.8	9.4	19.9	11	16	25
Calcium (mg/l)	200	116*	34	143	32	40*	4	70	2	70	18	104	16
Magnesium (mg/l)	50	54.1*	7.8	28	32	7.8*	2.1	4.6	1.8	23.0	4.3	15	16
Iron (ug/l)	200	-	-	324	-	-	-	8.9	-	-	-	106	16
Manganese (ug/l)	50	420*	-	483	-	7*	-	1	-	100	-	79	-

*Results Defined using the Statistical Approach.
"-" Analysis not possible because there were >50% LOD present in the data or insufficient data.

Figure 2 – Cumulative Frequency Plot for "Global" Parameters



DISCUSSION

Results of the natural background analysis are shown in Table 1 and 2. It was concluded by the IGWG that for parameters such as nitrate, the mean value is more representative of NBL than the 95th percentile, for example 3mg/l for nitrate as NO_3. The 5th and 95th percentile are considered to be levels beyond which a given concentration cannot be regarded as natural background for a parameter e.g. the 95th percentile value of 9mg/l for nitrate as $N0_3$. The IGWG considered that the NBL mean value for nitrate was representative and is consistent with previous research and the experience of the group members.

Dissolved Oxygen NBLs could not be established in the study. D.O. was generally reported in the data set as a percentage of saturation which must be converted to a concentration for use in the analysis. The conversion to a concentration is temperature dependant. Temperature was not recorded in many instances and may not have been measured in the field. The distribution of the D.O. values may be more affected by the degree of confinement and redox potential in the aquifer at that point than the lithology of the aquifer. Further research is required in order to establish the natural background limits for D.O.

Other parameters affected by the degree of aquifer confinement are sulphate, iron, manganese, and in some cases, nitrate. The methodology for selecting representative monitoring points results in a bias towards the selection of monitoring points in confined locations as these will be the best protected from anthropogenic activity and have the lowest vulnerability rating. This is best illustrated in the CFP for sulphate. When plotted on the CFP the lower values of the distributions did not conform to the trend in the upper portion of the line ("Sulphate All" in Figure 2). Data from monitoring points in confined aquifer conditions produce a more consistent trend ("Sulphate Screened" in Figure 2).

Orthophosphate results do plot a straight line on the CFP. The upper limit of 0.07mg/l given by the method is not considered to be realistic. While only 18% of the data are LODs, there are a variety of LOD values from different laboratories ranging from 0.001 to 0.04. The median and mean values are less than some of the LOD within the data set. There is also some uncertainty relating to the nature of the laboratory technique as not all samples may have been filtered to remove particulate matter before analysis. For these reasons an NBL was not derived in the study for orthophosphate. Further research is required in order to establish the natural background limits for orthophosphate.

It was not possible during the study to derive NBLs for 11 parameters because of low confidence in using CFPs, or in the statistical treatment due to either the identified data sets comprising high proportion of LODs or parameters were not consistently monitored. The lack of large parameter ranges for individual monitoring events also affected the data validation as it was not possible to complete an ionic balance for some analytical suites.

CONCLUSIONS

The study undertaken by OCM demonstrated that the ERTDI Methodology can be used to derive NBLs for some groundwater parameters. The EPA identified 28 parameters for which NBLs should be established. The study established NBLs for 15 parameters. NBLs could not be derived for 13 parameters. These were primarily heavy metals which are not frequently tested for and there is a high number of limits of detection (LOD) for these parameters in the database. Limited ranges of data also inhibited data validation, particularly for ionic balance calculations. The NBL will be established for these parameters by the EPA based on current analytical methods and limits of detection.

A significant effort was made to validate and screen the data. This was done to identify monitoring points that were representative of natural background conditions by eliminating those influenced by anthropogenic activity. The study established NBLs for a limited range of parameters on a national basis and for a range of lithological settings. Forty two monitoring points, including 892 water quality records, were identified nationally as suitable for use in the assessment.

Lithology was seen to have an important influence for some parameters e.g. hardness, E.C., alkalinity, pH, calcium and magnesium. Hydrochemistry of monitoring points in confined aquifers was found to vary from those in unconfined aquifers as would be expected. This influence had to be considered when selecting NBLs for parameters such as sulphate, D.O., iron and manganese.

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DETAILED AND INTEGRATED MONITORING OF THE POSITIVE AND NEGATIVE IMPACTS OF LARGE SCALE WATER ABSTRACTIONS AND TREATED WASTE WATER DISCHARGES ON HABITATS AND ECOSYSTEMS

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ABSTRACT

A water and wastewater utility company in southern England is carrying out investigations to assess the impact of its discharges of treated wastewater on sites protected under the Birds and Habitats Directives and on Ramsar sites. While these sites are surface water features, many are complex interdependent hydrogeological, hydrological and ecological systems. The water company has commissioned a detailed study of 21 wastewater discharges into 12 catchments with the aim of understanding whether the discharges have the potential to impact the sites, whether there is evidence that effluent is reaching the site, the proportion of nutrient loading to the site contributed by the discharge, and whether the discharge is having an actual impact on the site. This paper describes the design and implementation of an integrated monitoring programme targeted at answering specific regulatory questions. A description of a similar investigation and monitoring programme designed to assess the impacts on protected sites of groundwater abstractions by another water utility company with be given in the presentation for comparative purposes.

INTRODUCTION

The public water supply and wastewater industry in England and Wales, although privately-owned, is regulated under a five-year Asset Management Plan (AMP) cycle, whereby prices are set in parallel with an agreed programme of service and environmental improvements. The current AMP4 period (2005-2010) includes a regulatory Environmental Driver H8 "Investigations agreed by the conservation agencies and the [Environment] Agency to assess the impact of water company assets on the requirements of the Directive", the Directive being the Habitats Directive (92/43/EEC) and Birds Directive (79/409/EEC). UK government policy is that Ramsar sites are to be afforded the same level of protection as they would under the Habitats and Birds Directives.

Company A is a large integrated water supply and wastewater treatment utility company providing services across the south of England between Kent and Hampshire. Under its regulatory agreement, it is assessing the impacts of 21 of its wastewater treatment works (WTW) on a number of protected sites in 12 catchments.

This paper describes the design and implementation of an integrated investigation and monitoring programme intended to answer these questions at one of the sites. The lessons drawn from the practical experience of implementing the whole programme of works are summarised.

In the presentation, comparison will also be made with the AMP4 programme of works being implemented by another water supply company to assess the impacts of its groundwater abstractions on similar protected areas in eastern England. In this case the focus is largely on water levels and flow rates rather than on water quality.

DESCRIPTION OF THE SITE

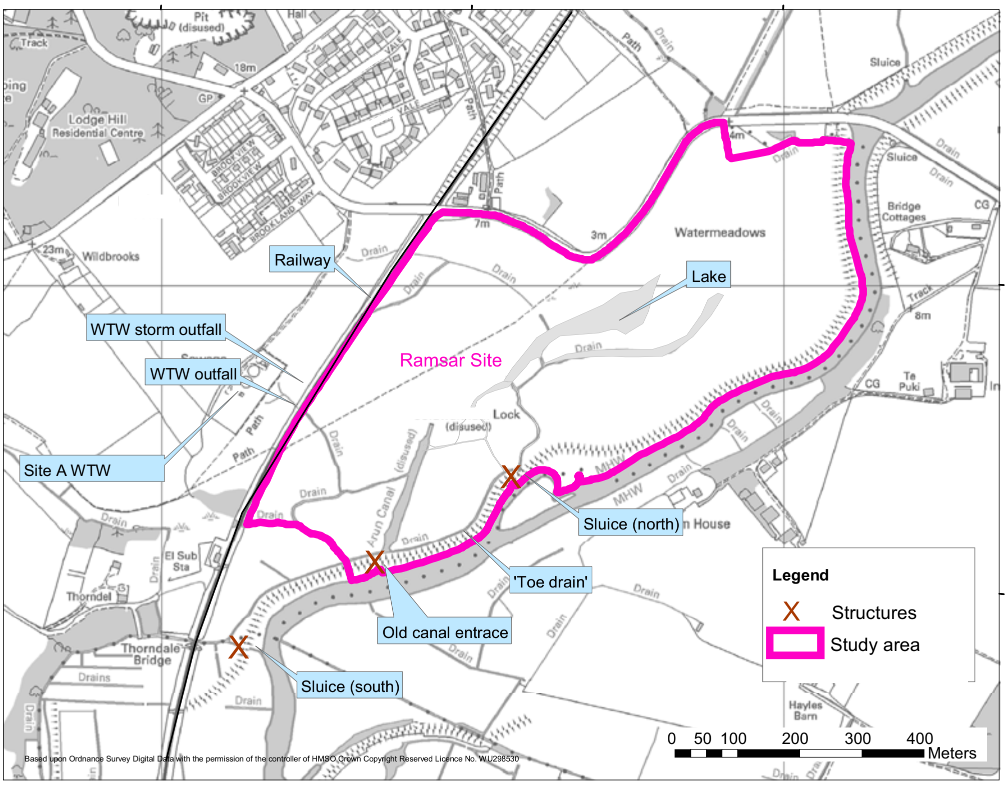
PROTECTION STATUS

The WTW discharges into Site A through a Site of Special Scientific Interest (SSSI, designated under the Wildlife & Countryside Act 1981), which is itself part of a Special Protection Area (SPA, protected under the Birds Directive) and Ramsar site (a wetland of international importance). The features of interest specified in the SPA citation are:

Bewick's (Tundra) Swan
Over 27,000 wintering waterfowl, including shoveler, teal and wigeon

The Ramsar designation refers also to seven wetland invertebrates, eight scarce plant species and the diverse and rich flora in the site's ditches. A map of the site is shown on Figure 1.

Figure 1. Site map



GEOLOGY, HYDROGEOLOGY AND HYDROLOGY

Site A forms part of a lowland river floodplain and is characterised by low lying and flat grazing marsh between 0 and 5 maOD. The regional bedrock geology is formed of the Lower Cretaceous Folkestone Beds (sandstone, and a major aquifer) and Sandgate Beds (silts and clays, a minor aquifer). Superficial deposits include River Terrace Deposits (sands and gravels), Head Deposits (clay, silt and sand) and recent alluvium. A geological cross section through the site is shown on Figure 2.

Site A has a complex of drainage ditches and small water courses linked to a tidal river. It is also crossed by a disused canal linked to the river. At 190 m from the river there is an old lock, beyond which the canal is dry.

A conceptual water balance model of the site is shown on Figure 3.

Figure 2. Geological cross section through Site A

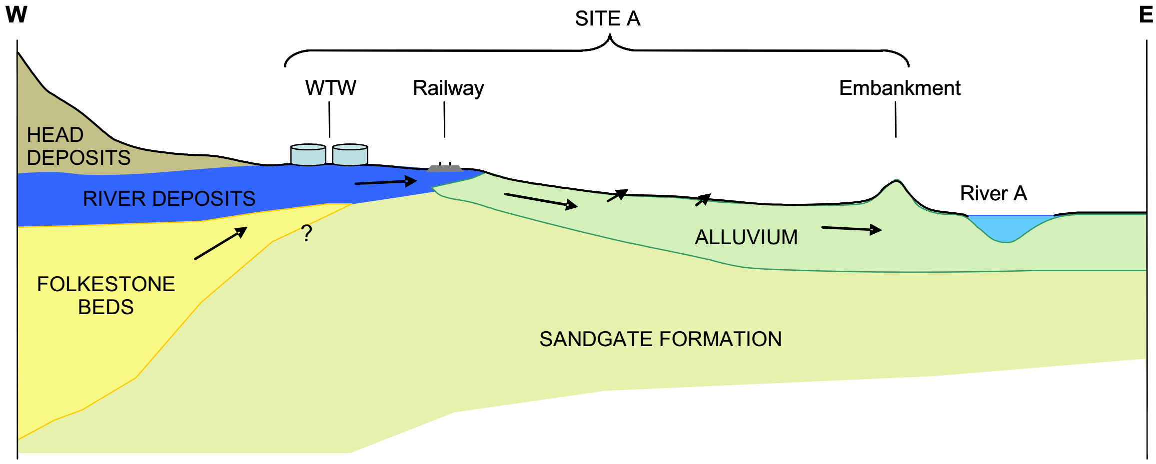
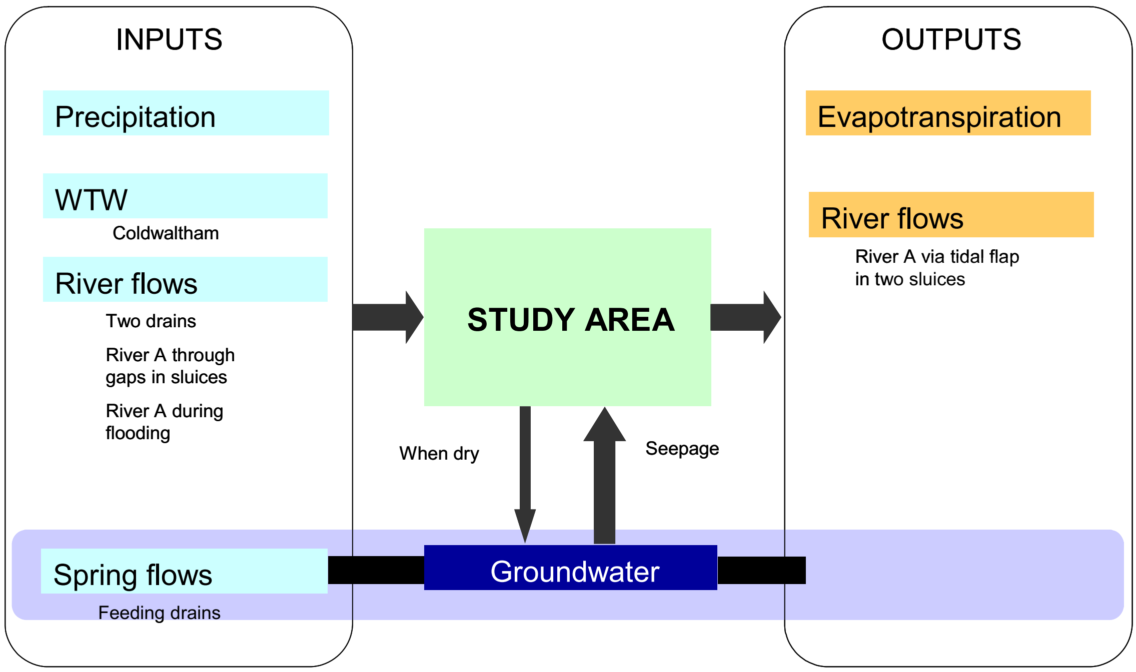


Figure 3. Conceptual water balance for Site A



The study focussed only on the impacts on the component of the site represented by the SSSI. The majority of the SSSI lies on poorly drained river alluvium juxtaposed with better drained grasslands in the western part where river gravels and the Folkestone Beds are present. Due to the naturally poor drainage together with interventions by conservation organisations to improve the site's attractiveness to wildlife, a semi-permanent lake has formed in part of it.

CONSERVATION AND MANAGEMENT

The site is owned by the county Trust for Nature Conservation, whose management included ditch maintenance, clearing part of the old canal of reeds, reducing the number of drainage ditches, and maintaining water levels at optimum levels by means of sluices to allow water both to enter and leave the site.

Site A is a registered Common, allowing sheep and cattle grazing at certain times of the year, although grazing has recently been sporadic.

THE WASTEWATER TREATMENT WORKS

The WTW has both treated effluent and stormwater outfalls. The discharges enter a ditch that flows under a railway line and then into the site proper, eventually reaching the river. The WTW consists of a primary settlement tank, secondary treatment (two biological filters and two double humus tanks) and tertiary treatment (a drum filter to remove suspended solids and BOD). There are no stormwater retention tanks. The WTW accepts the effluent from a population of about 800. Average effluent flows in recent years have been 168 m^3/d, although there are no records of stormwater flows.

Average concentrations of main effluent parameters over the past five years are given below:

Suspended solids	8.1 mg/l
BOD ATU	3.7 mg/l
Orthophosphate	8 mg-P/l
Nitrate	24 mg-N/l
Ammoniacal nitrate	0.56 mg-N/l

DESIGN OF STUDIES AND MONITORING PROGRAMME

REGULATORY AGREEMENT

The programme of studies was agreed between the water company (and Mott MacDonald as its consultant), the Environment Agency as the designated regulator for Environmental Driver H8, and English Nature, the organisation responsible for SPAs under the Birds Directive. The agreement identified:

The precise spatial area of concern and its key interest features
Data already available from previous studies and monitoring
The specific questions the studies and monitoring programmes should answer
The scope of work required to answer each question
The deliverables
The scope of consultation with stakeholders
Facilities and data to be provided by the different parties

The agreed spatial **area of concern** and **key interest features** are as in the site description above.

The specific questions to be answered are:

Does the WTW have the potential to impact on the protected site of concern?
Is there evidence of eutrophication and organic enrichment?
What is the proportion of the total phosphate and organic load contributed by the WTW?
Is the phosphate and organic load discharged by the WTW detrimental to the interest features of the site?
Can an acceptable level of phosphate and organic load in the receiving waters be obtained?

Considerable debate was held on which features and issues were sufficiently uncertain to require additional fieldwork, which had been studied in detail but required reanalysis of existing data, and which were well enough understood for no further work to be needed. For example, it was agreed that no new ecological surveys needed to be carried out, but that a critical review and analysis of existing data was needed.

STUDIES AND SURVEYS

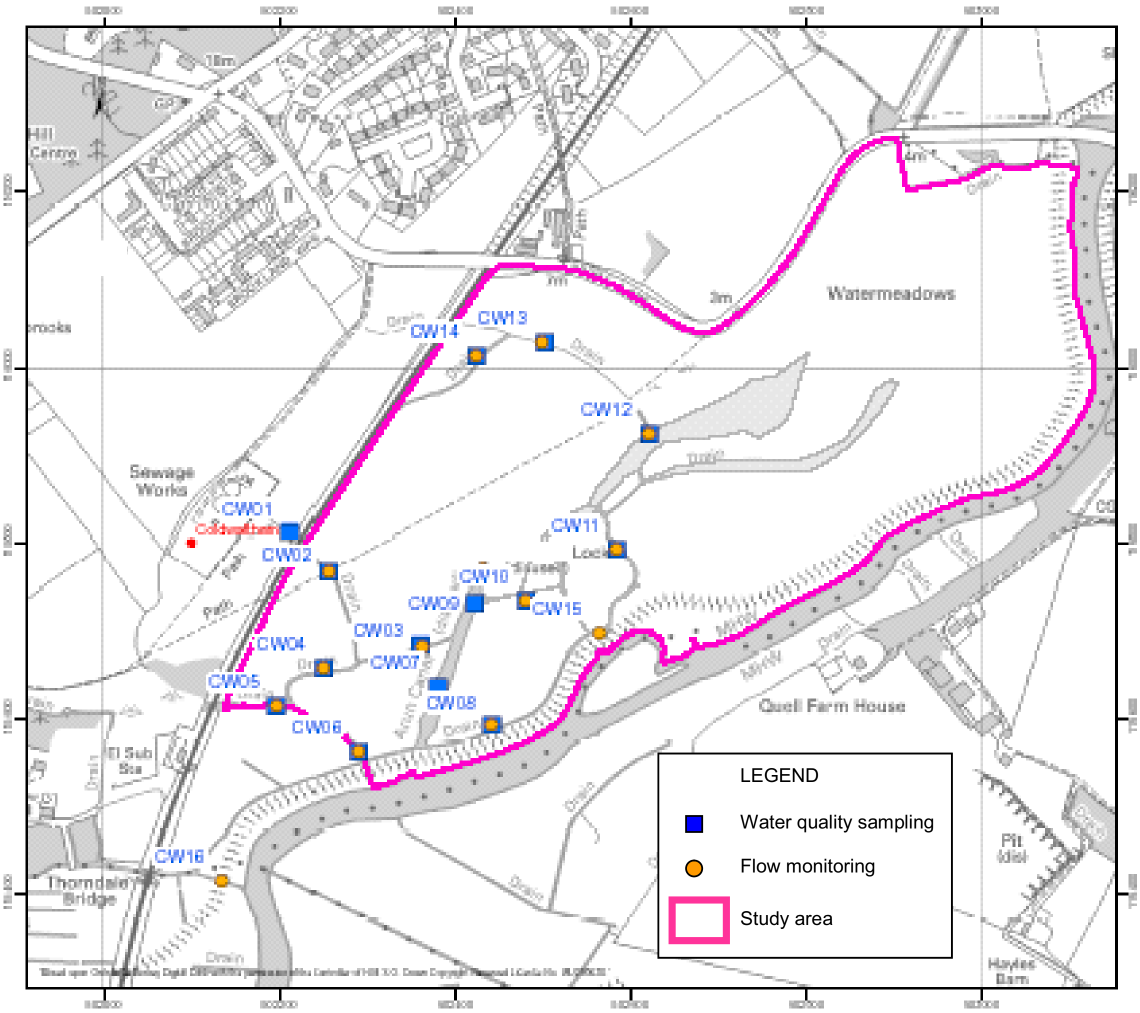
The final programme of work that was agreed and implemented is summarised below:

Does the WTW have the potential to impact on the protected site of concern?	▪ Identification of flow path directions and flow rates across the site using electromagnetic flow meters and biodegradable floats. ▪ Observation of flood events. ▪ Sampling and analysis of water along flow paths for sewage signatures, including metals, chloride and caffeine. ▪ Provisional allowance for a tracer test from the WTW.
Is there evidence of eutrophication and organic enrichment?	▪ Reanalysis of existing data, including: mean trophic rank (MTR), trophic diatom index (DTI) and macroinvertebrate surveys; analysis data for chlorophyll, nitrate, ammonia, orthophosphate and BOD; channel surveys for width, substrate, shade and turbidity.
What is the proportion of the total phosphate and organic load contributed by the WTW?	▪ Flow and quality (sewage signature parameters as above) monitoring at 10 locations at 20 day intervals over a 6 month period, including one ditch where effluent is unlikely to reach. ▪ Calculation of mass balances of phosphate and organic load, taking account of WTW discharges, surface water and groundwater flows, agricultural runoff, wildlife contributions.
Is the phosphate and organic load discharged by the WTW detrimental to the interest features of the site?	▪ Desk study to map trends in ecological structure and function over time against channels in zones influenced by flows from the WTW.
Can an acceptable level of phosphate and organic load in the receiving waters be obtained?	▪ Desk study to assess the impacts of changes in effluent quality on water quality in different zones of the site. The potential impact of these changes on interest features is to be estimated by reference firstly to data obtained during the surveys (comparison of actual water quality and actual ecological status) and secondly to published species tolerance levels.
Stakeholder consultation	Meetings with stakeholders to obtain local knowledge about the site and changes in it over time (e.g. management, hydraulic regime, fish stocks, bird numbers, agricultural practices) and any other issues relevant to the questions to be answered. Stakeholders include: the Environment Agency, English Nature, Internal Drainage Board, the county wildlife trust, the site warden, angling societies, local government, local farmers and landowners and others.

IMPLEMENTATION OF STUDIES AND MONITORING PROGRAMMES

Figure 4 shows the selected main sampling and monitoring locations at Site A.

Figure 4. Sampling and monitoring locations



The programme of works described above covered a six month period, with field works completed by the end of December 2005. This was one of 12 such programmes, many of which will extend through 2006, and all of which comprised combinations of ecological surveys and monitoring, groundwater and surface water monitoring and other environmental studies. Such a large programme had implications for project management, logistics, health and safety, quality control and information management. Some particular issues are commented on below.

Project management: each of the 12 studies for individual catchments were established internally as independent projects with their own project managers, under one overall project manager.

Staff: two dedicated field teams were established covering different geographical areas. Most members were recruited specifically for the work, including some on short-term contracts, although each team included at least one long-term permanent Mott MacDonald staff member to assist with implementation of common standards and liaison with the project managers. An office-based member of staff was designated as field teams manager who facilitated scheduling of field work. A data manager was recruited under a short term contract for information and data management and quality control. In all cases, temporary field and office staff had environmental qualifications to ensure a certain level of quality assurance.

Health and safety: intensive field work adjacent to, in or on water over a long period by a significant number of staff, some of whom were new to the company raised health and safety concerns. An overall project health and safety plan was drawn up and was the subject of briefings to all project staff. For particular activities where existing company guidelines were not specific enough, new "Hazard Information Sheets" were prepared. Examples included using an inflatable dinghy to sample from open water, and the use of ethanol and formaldehyde in biological sample preservation. Particular effort was spent to ensure that where independent subconsultants were used they not only signed up to the health and safety plan and guidelines, but fully "bought-in" to their implementation.

Equipment: significant quantities of field equipment were purchased, ranging from simple devices for sampling rivers and streams from the bankside to flow meters, field water quality instrumentation and an inflatable dinghy. Validation and calibration was a significant activity, and a designated field equipment manager was appointed.

Methodologies: a field and office (data processing) methods manual was prepared, often based on pre-existing method statements, but in all cases revised specifically for this programme.

Information management: a database and GIS was established for storage, management and reporting of information as it was received from the field and laboratory.

Consultations: early consultations were held with stakeholders, the Environment Agency and English Nature in particular, but also with key land owners, conservation agencies and others, to ensure that the field programme could be implemented as planned from the outset without having to make significant changes part way through.

Quality management: although last in this list, it was recognised that the successful implementation of the field surveys would be judged ultimately by the completeness and quality of the data that was obtained. The data was received from a diversity of sources, including field staff, data logger downloads, laboratory reports and results of analysis by Mott MacDonald staff (the latter particularly for invertebrate microscopic identification). As a result, much data were received on paper. A dedicated data manager was responsible for collating and entering all field data, and he was required to ensure that all his entries were checked by a member of staff not otherwise responsible for that particular monitoring area.

LESSONS LEARNED

The programme of field surveys and monitoring will run through to 2007, although the intensity of work has subsided from its peak in late-2005. Reporting of the outcomes of the studies is also ongoing. However, certain lessons have been learned from the experience of designing and implementing the field studies programme:

Design of field studies: early intense consultations with stakeholders were vital for ensuring that the field programme was not subject to significant change after it started. A series of technical discussion papers was issued on such matters as invertebrate monitoring (e.g. diatoms) and how to detect residual effluent at a distance from a WTW (caffeine was settled on as a key indicator). Discussions with stakeholders often focussed on the precise interest features of concern at the protected sites and their reasons for citation as an SPA, SAC, SSSI or Ramsar site, and how the sensitivity of such features to water quality and flow should be assessed.

Changes in monitoring points: Some sampling points had to be changed after the start of the field programme as a result of information obtained in the first weeks, such as: improved understanding of flow patterns in drainage networks meant that better monitoring locations could be found; discovery of unmapped discharges to protected areas; dredging of watercourses by landowners and managers that rendered further monitoring pointless.

Field team management: the management of the field teams was an intensive activity and repaid the designation of an office-based field team manager and equipment manager. The early establishment of standard methodologies was also successful.

Health and safety: this issue required constant monitoring to ensure that field staff maintained their awareness of risks and that new risks were recognised. Many of the monitoring points required walking with equipment and samples in winter to some distance from road tracks. To date, no accidents have occurred during actual fieldwork, although there have been two minor road accidents. The rise of the bird flu risk has meant that new guidelines have been prepared.

Equipment: the dedicated equipment manager has been particularly busy in maintaining and recalibrating equipment. Electromagnetic flow meters and water quality instruments (especially dissolved oxygen probes) have been particularly sensitive to damage (especially to cables) while being carried across country.

Methodologies: in the early stages of the field programme some areas in particular required particular attention. The *measurement of flow in very slow flowing ditches* was identified as an issue, and the methodology was revised to ensure reproducible and realistic measurements were obtained. A related issue was the sampling of water in very shallow ditches where it was difficult to ensure *silt-free sampling*. The *safe use of ethanol and formaldehyde* in biological sampling was of particular concern for its safety implications during carriage of the liquids in the field, the storage of samples prior to examination, and the safe examination of samples. *Consistency of macrophyte surveys* between individual ecologists was identified as a possible source of confusion, and addressed by discussions between the ecologists about appropriate approaches.

ACKNOWLEDGEMENTS

The description of this work has been given with the permission of the water company. It is acknowledged that staff from Company A were an integral part of the discussions that led to the designs of the studies and investigations. The studies at site A were managed by Stine Jensen and the overall programme of studies for Company A is managed by Jon Pavey, both of Mott MacDonald.