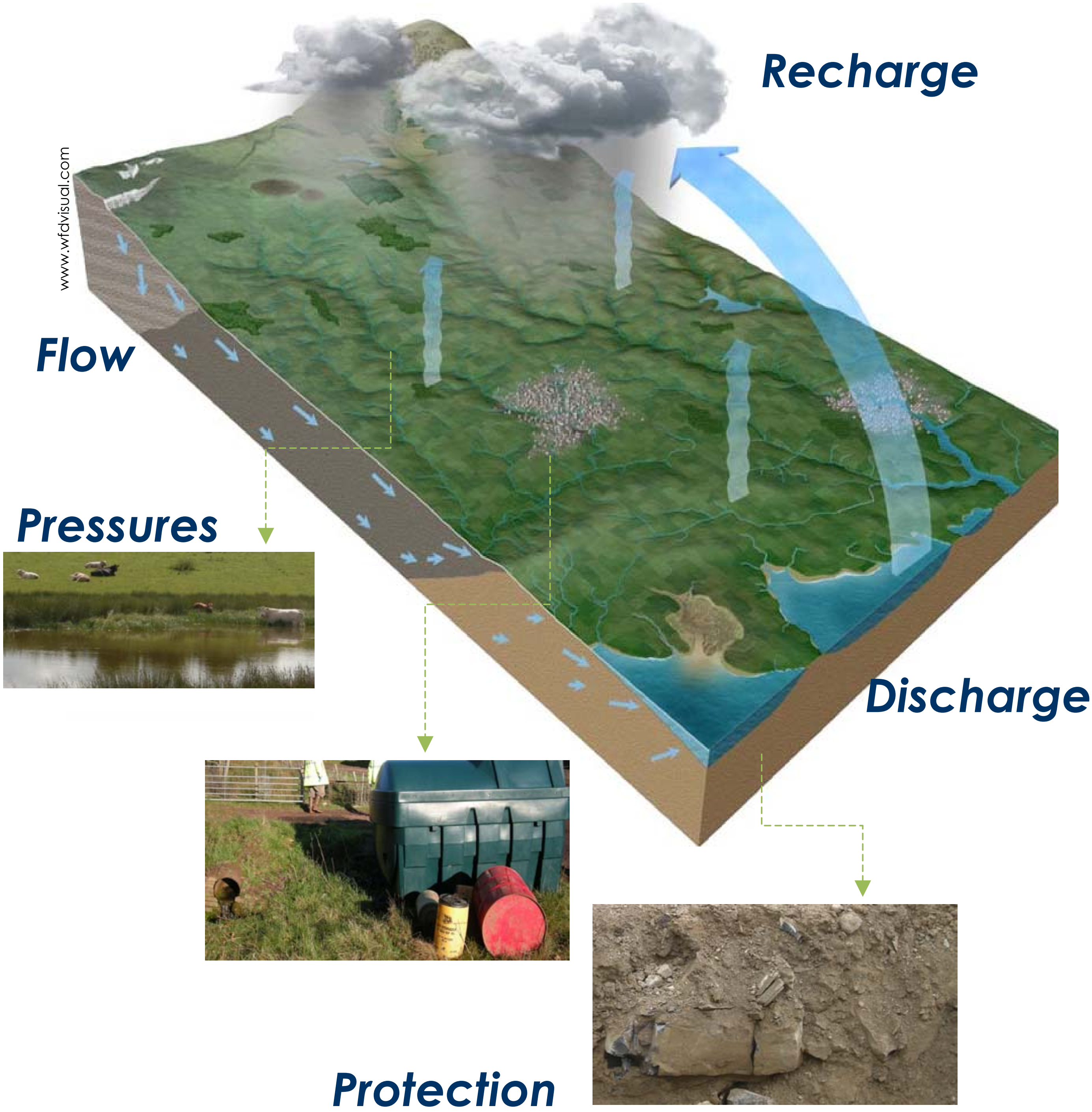
INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS – Irish Group

‘Groundwater in the Hydrological Cycle - Pressures and Protection’



Proceedings of the 30th Annual Groundwater Conference

Tullamore, Co. Offaly, Ireland

www.iah-ireland.org

INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS
(IRISH GROUP)

presents

‘GROUNDWATER IN THE HYDROLOGICAL CYCLE - PRESSURES AND PROTECTION’

Proceedings of the 30th Annual Groundwater Conference

Tullamore Court Hotel,
Tullamore,
Co. Offaly

20th & 21st April, 2010

The IAH would like to sincerely thank Tobin Consulting Engineers for their help in administrating the Conference registration.

For more information and contact details please refer to: *www.iah-ireland.org*

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Proceedings of the 30th Annual Groundwater Conference (International Association of Hydrogeologists, Irish Group will be made available digitally on the IAH-Irish Group website within approximately three months of the meeting.

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INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS (IRISH GROUP)

Founded in January 1976, the IAH-Irish Group membership has grown from 10 to over 130, and draws individuals from professional backgrounds ranging from academic to state agencies to private consultancies. The committee consists of a council of: President, Secretary & Burdon Secretary, Treasurer, Northern Region Secretary, Fieldtrip Secretary, Education & Publicity Secretary, Conference Secretary, plus a conference sub-committee.

Regular activities of the Irish Group consist of an annual two-day conference (currently held in Tullamore), an annual weekend fieldtrip, and a series of monthly lectures/technical meetings. Funding for the association is derived from membership fees and the annual conference. We welcome the participation of non-members in all our activities. Other activities of the IAH (Irish Group) include submissions to the Irish Government on groundwater, the environment and matters of concern to members, organising the cataloguing of the Burdon library and papers, which are now housed in the Geological Survey of Ireland Library, invitation of a guest speaker (often from outside Ireland) to give the David Burdon Memorial Lecture on a topic of current interest, and contributing to the Geological Survey of Ireland's Groundwater Newsletter.

The Irish Group provide small bursaries to students doing post graduate degrees in hydrogeology and pays the annual subscriptions of a few members in other countries as part of the IAH's Sponsored Membership Scheme. If you would like to apply for a student bursary, details can be found on the IAH (Irish Group) website shown below. IAH are encouraging members to highlight their local IAH Group to their colleagues/ students and to invite anyone they feel may be interested to join.

The IAH (Irish Group) is also a sponsoring body of the Institute of Geologists of Ireland (IGI).

For more information please refer to: *http://www.iah-ireland.org*
Future events: *http://www.iah-ireland.org/current/events.htm*
IAH Membership (new or renewal): *http://www.iah.org/join_iah.asp* *http://www.iah.org/payonline*

2010 Conference Objective

The 2010 Conference will be of great benefit to hydrogeologists, local authority engineers, consultants, planning officials, environmental scientists, public health officials, and many other professionals.

This year is the 30th Anniversary of the Annual IAH (Irish Group) Conference, with the theme 'Groundwater in the hydrological cycle - pressures & protection'. The two-day event will start by looking into how water recharges our aquifers through thick glacial overburden and then will move on to look at how groundwater flows through the many different rock types that Ireland possesses including karst limestone, poorly productive bedrock and sand and gravel systems. The second half of Day 1 will explore how groundwater discharges through surface water systems, groundwater dependent ecosystems and in extreme cases through flooding events. Finally, Day 1 will finish by looking at the diffuse agricultural and industrial point sources of contamination that can affect this precious resource and the implications of the Environmental Liability Directive for groundwater protection.

Day 2 will begin by looking at the new measures that are available to protect groundwater including legal, source protection zones and recent EPA guidance. The conference will close by looking at the impact that climate change is and may have on groundwater resources in Ireland and globally.

Evening entertainment will be provided on the first night of the conference at a local venue and is included in the registration fee.

‘Groundwater in the Hydrological Cycle – Pressures and Protection’
30th Annual Groundwater Conference

International Association of Hydrogeologists –Irish Group
Tullamore Court Hotel, Tullamore, Co. Offaly: Tues 20th & Wed 21st April, 2010

Programme Day 1, Tuesday 20th April

8.15 - 9.15 *Conference Registration; Tea, Coffee, & Exhibits*

INTRODUCTION

9:30	Welcome and Introduction Teri Hayes - President IAH Irish Group (WYG Environmental)

SESSION 1:	**RECHARGE & GROUNDWATER FLOW**
9.45	‘Groundwater Recharge: Influence of Glacial Tills on Resource Assessment and Groundwater Protection in Ireland’ - Bruce Misstear (Trinity College Dublin) and Dr. Les Brown (SLR Consulting)
10.10	‘Groundwater Flow in Karst Systems’ - Dr. David Drew (Trinity College Dublin)
10.35	Discussion, Q&A
10.55 - 11.25	*Tea & Coffee*
11.25	‘Groundwater Flow Regimes in Irish Poorly Productive Aquifers’ - Dr. Jean-Christophe Comte (Queens University, Belfast)
11.50	‘Groundwater Supply Development in Sand & Gravel Aquifers’ - Gerry Baker (WYG Ireland)
12.15	Discussion, Q&A
12.45 - 14.00	*Buffet lunch in Tullamore Court Hotel*

SESSION 2:	**DISCHARGE**
14.00	‘Surface Water - Groundwater Interactions’ - Dr. Geoff Parkin (University of Newcastle)
14.25	‘Groundwater Dependent Ecosystems’ - Dr. Geert van Wirdum (Deltares Subsurface and Groundwater Systems)
14.50	‘Multiple Flooding Mechanisms in Ennis Town and Environs’ - Mary Burke (Clare County Council)
15.10	Discussion, Q&A
15.30 - 15.50	*Tea & Coffee*

SESSION 3:	**PRESSURES**
15.50	‘Agriculture Landspreading & Set-back Distances’ - Donal Daly (Environmental Protection Agency)
16.10	‘The Assessment and Prioritisation of Point Sources of Groundwater Contamination in Ireland’ - Dr. Marcus Ford (Ford Consulting Group)
16.30	‘Groundwater Impacts from Mining: Irish Case Studies’ - Geoff Beale (Schlumberger Water Services UK Ltd)
16.50	‘Environmental Liability Directive Implications for Groundwater’ - Kevin Motherway (Environmental Protection Agency)
17.10 - 17.30	Discussion, Q&A

*The final panel discussion on Day 1 will be followed by a wine reception in the Tullamore Court Hotel sponsored by **City Analysts Limited**, followed by a BBQ meal at Hugh Lynch's Bar, Kilbride Street, Tullamore, sponsored by **In-Situ Inc**.*

‘Groundwater in the Hydrological Cycle – Pressures and Protection’
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Programme Day 2, Wednesday 21st April

SESSION 4:	**PROTECTION**
9.30	‘Legal Perspective on Groundwater Protection’ - Deborah Spence (Arthur Cox)
9.50	‘Delineating Source Protection Zones and Zones of Contribution for Monitoring Points’ - Coran Kelly (Tobin Consulting Engineers)
10.10	‘EPA Guidance on Protecting Groundwater Supplies’ - Margaret Keegan (Environmental Protection Agency)
10.30	Discussion, Q&A
10.50 - 11.20	*Tea & Coffee*

SESSION 5:	**CLIMATE CHANGE**
11.20	‘Impacts of Climate Change on Rainfall, River Discharge and Sea Level Rise’ - Dr. Tido Semmler (Met Éireann)
11.40	‘What will the Impacts of Climate Change be for Groundwater Systems in Ireland?’ - David Ball
12.05	‘Implications of Climate Change on Groundwater Resources’ - Dr. Kevin Hiscock (University of East Anglia)
12.30	Discussion, Q&A
12.50	Conference closing address: Shane Herlihy - Conference Secretary IAH Irish Group (RPS)
13.00	*Buffet lunch in Tullamore Court Hotel*

Table of Contents

SESSION I

GROUNDWATER RECHARGE: INFLUENCE OF GLACIAL TILLS ON RESOURCE ASSESSMENT AND GROUNDWATER PROTECTION IN IRELAND

Bruce Misstear, School of Engineering, Trinity College Dublin
Les Brown, SLR Consulting Ireland

ABSTRACT

An understanding of groundwater recharge is necessary for assessing and protecting groundwater resources. Recharge can be estimated by applying a recharge coefficient to the effective rainfall. The recharge coefficient is strongly influenced by the permeability of the glacial tills or other subsoils that cover most of the country's aquifers. Values of recharge were determined in four areas of contrasting subsoil permeability, using a variety of approaches. The results helped to validate previous estimates of recharge coefficient and were a key input in the preparation of a national recharge map for Ireland. The results of the case studies were also used to develop a quantified relationship between subsoil permeability, recharge, surface runoff and aquifer vulnerability. UK studies on the influence of tills on recharge are reviewed briefly and the issue of scale is highlighted. The paper concludes by emphasising the importance of tills in controlling recharge in Ireland. The recharge coefficient is useful for making preliminary estimates of groundwater resources at the river catchment or groundwater body scale. More detailed recharge investigations are desirable at the local scale, for example, in delineating source protection areas.

INTRODUCTION

Groundwater recharge is one of the key components of the hydrological cycle. An understanding of recharge is necessary for many applications, including: groundwater resources assessment, delineation of zones of contribution around water supply wells (source protection areas), contaminant transport investigations and groundwater vulnerability mapping.

In Ireland, recharge is strongly influenced by the glacial tills that cover large parts of the country. Where tills are thick and of low permeability, recharge to the underlying aquifer is impeded. Similarly, the vulnerability of the aquifer to pollution in such a situation is much less than in situations where the subsoil is highly permeable, thin or absent.

The case studies described in this paper are based on a research project on the linkages between recharge and vulnerability, which was undertaken for the EPA (Misstear and Brown, 2008). The paper will also review work carried out in the UK on the influence of tills on recharge.

CONCEPTS

There are two types of recharge: direct (diffuse) recharge, which results from percolation of rainfall where it falls, and indirect (point) recharge, which occurs after surface runoff, for example where streams enter swallow holes or other karst features. In Ireland, direct (diffuse) recharge is often estimated by first calculating the effective rainfall and then multiplying this effective rainfall by a recharge coefficient. The effective rainfall is the moisture surplus remaining after actual evapotranspiration is deducted from the total rainfall. This moisture surplus is sometimes referred to as potential recharge (although this term is also used in a more restricted way to describe the water that infiltrates below the root zone i.e. after surface runoff has been taken account of). The recharge coefficient is the proportion of effective rainfall that produces recharge.

The recharge coefficient is influenced by the thickness and permeability of the till (or other subsoil) overlying the aquifer, and also by the vertical hydraulic gradient. Saturated conditions are normally assumed for vertical flow within the tills. Using a soil moisture budget allied to a one-dimensional numerical model, Fitzsimons and Misstear (2006) demonstrated that the most important factor controlling recharge coefficient is the till permeability (more properly referred to as hydraulic conductivity). Furthermore, as can be seen in Figure 1, the recharge coefficient is particularly sensitive to till permeability in the range 0.001 to 0.01 m/d ($1x10^{-7}$ to $1x10^{-8}$ m/s), corresponding to the band of moderate permeability used by the Geological Survey of Ireland in its vulnerability mapping (Fitzsimons *et al*., 2003; Swartz *et al*., 2003).

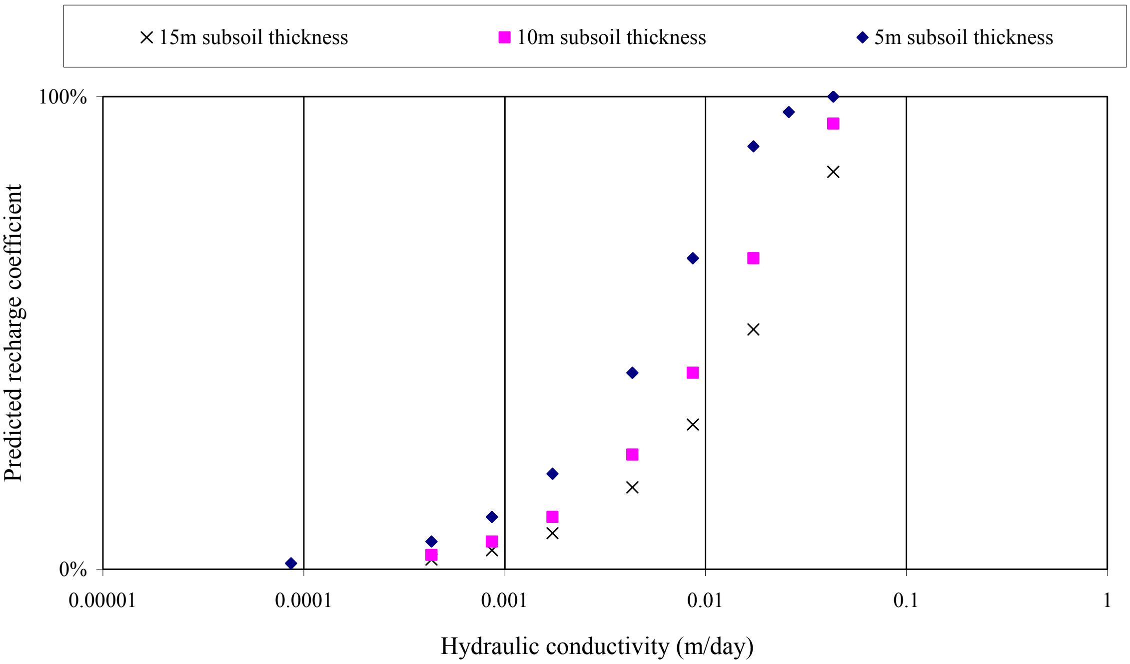

Figure 1: Relationship between recharge coefficient and the thickness and permeability (hydraulic conductivity) of till (Fitzsimons and Misstear, 2006)

CASE STUDIES

Recharge investigations were carried out in four areas of contrasting subsoil permeability. These included two areas with moderate permeability subsoils, in view of the high sensitivity to recharge within this permeability band. The study areas were: the Curragh gravel aquifer in County Kildare (high permeability subsoils), the Callan-Bennettsbridge lowlands in County Kilkenny (moderate permeability subsoils), the Galmoy area in County Kilkenny (moderate permeability subsoils) and the Knockatallon aquifer in County Monaghan (low permeability subsoils). The results are summarised in Table 1.

Table 1: Summary of the main results of the case studies (adapted from Misstear and Brown, 2008)

Study area	Setting	Methodology	Recharge coefficient
Curragh aquifer, County Kildare	Regionally important gravel aquifer. Thin (generally <3 m), moderate to low permeability till cover; high vulnerability. Lowland setting.	Soil moisture budget (SMB), hydrograph analysis, numerical modelling, natural tracers, catchment water balance	81-85%
Galmoy mine, County Kilkenny	Regionally important limestone aquifer. Till cover generally 5-10 m thick and of moderate permeability. Lowland setting.	SMB, natural tracers, water balance using dewatering discharges	55-65%
Callan-Bennettsbridge lowlands, County Kilkenny	Aquifer includes regionally important limestone and dolomite. Variable thickness of moderate permeability till and high permeability gravel cover. Mainly lowland topography.	SMB, river baseflow analysis	41-54% (for mod. perm. subsoils) [36-60% for entire subcatchments]
Knockatallon aquifer, County Monaghan	Locally important limestone aquifer. Thick (up to 50 m) low permeability till cover. Upland and lowland topography	SMB, dewatering discharges, baseflow analysis, natural tracers	<17% (and probably <5%)

THE CURRAGH AQUIFER

The Curragh sand and gravel aquifer (Figure 2) is glacio-fluvial in origin. The deposits generally range between 20 m and 40 m in thickness, with an estimated maximum thickness of 65 m (Daly, 1981; Hayes *et al.*, 2001). The variable composition of these deposits is illustrated by the photomontage of a road cutting in Figure 3. The till shown at the top of the cutting occurs over much of the Curragh aquifer (MacCarthy, 2008).

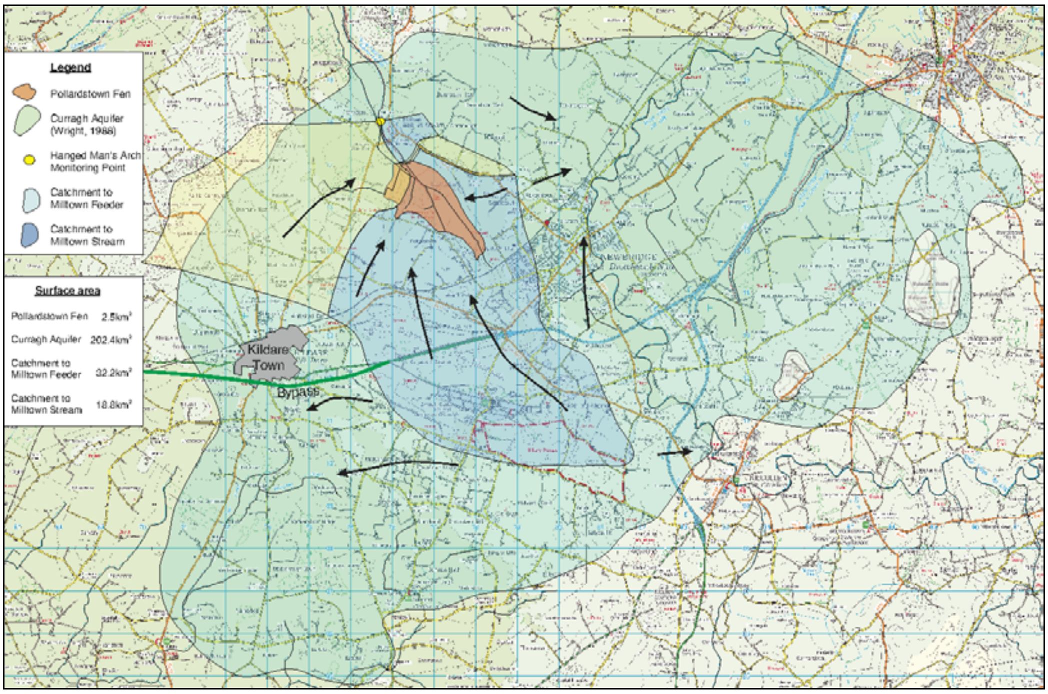

Figure 2: Curragh sand and gravel aquifer, including catchment area supplying Pollardstown fen (partly based on work by Kuczyńska, 2008)

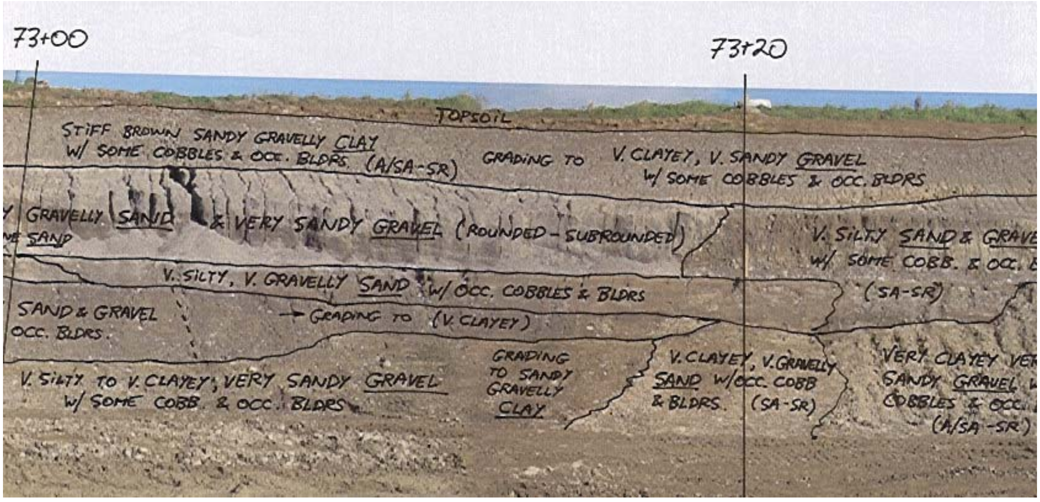

Figure 3: Geology of a 7 m high section of road cutting near Kildare town (prepared by AGL Consulting Engineers)

Recharge to the Curragh aquifer was estimated using several approaches, including soil moisture budgeting, well hydrograph analyses and a catchment water balance (Misstear *et al*., 2009b). The soil moisture budgeting was carried out using the Penman-Grindley and the FAO Penman-Monteith methods (see Allen *et al*., 1998) for a 30-year series (1971-2000) of daily rainfall and evapotranspiration data. The results, in terms of annual averages, were similar, with effective rainfall values of 334 mm/year and 321 mm/year for the P-G and FAO P-M methods, respectively.

The following simplified relationship between recharge (R), specific yield (S_y) and annual groundwater level fluctuation ($\Delta h/\Delta t$) was used in the analysis of well hydrographs:

$$R = S_y \frac{\Delta h}{\Delta t}$$

Applying a S_y value of 19% gave recharge coefficients in the range 72% to 100%.

A water balance was undertaken of the catchment area (estimated as 32 km^2) that feeds the springs and seepages at Pollardstown Fen (Figure 2). Based upon a total measured outflow for the fen of 9.14 x 10^6 m^3/yr (Kuczyńska, 2008), recharge was estimated to be 285 mm. With effective rainfall values of between 335 mm/yr and 351 mm/yr for the period under investigation (March 2002 to May 2005), the recharge coefficient was calculated at between 81% and 85%. This is considered to be a realistic range for a high permeability, high vulnerability sand and gravel aquifer.

THE GALMOY AREA

At Galmoy mine in County Kilkenny, the subsoil is largely sandy SILT, between 1 and 12 m thick, and has a permeability probably towards the upper end of the moderate permeability range (i.e. between 10^{-4} and 10^{-5} m/s). The subsoil overlies fractured dolomite and limestone bedrock.

Zinc and lead minerals are mined at Galmoy from depths of up to 150 m via a series of inclined roadways. Dewatering activities have created a large cone of depression focused around the mine workings. A water balance calculation based on the volume of groundwater abstracted from the estimated zone of contribution to the mine area (equivalent to 264 mm/year) versus the effective rainfall estimates for the area (407 to 428 mm/year) suggested a recharge coefficient of around 62-65%. However, the groundwater system was not in steady state with respect to dewatering, as groundwater levels in the immediate vicinity of the mine were continuing to drop at the time of investigation Therefore, the above calculation is likely to represent an overestimate of the recharge, with some of the abstraction volume being taken from aquifer storage. Chlorofluorocarbon (CFC) analyses of mine water samples also indicated that the dewatering involved a mixture of modern recharge and older groundwater from aquifer storage. Assuming about 10-20% of the dewatering represents depletion of aquifer storage, then the recharge coefficient was estimated (very approximately) at around 55%.

THE CALLAN-BENNETTSBRIDGE LOWLANDS

The subsoils in Callan-Bennettsbridge lowlands are typically silty clays, probably having a permeability towards the lower end of the moderate permeability range (i.e. between 10^{-7} and 10^{-8} m/s). The study area comprised two catchments: a subcatchment of the River Nore between the gauging stations at St John's Bridge in Kilkenny city and Mount Juliet downstream, and the catchment of the King's River, a tributary of the Nore which joins the right bank of the main river at Annamult, just upstream of Mount Juliet (Figure 4).

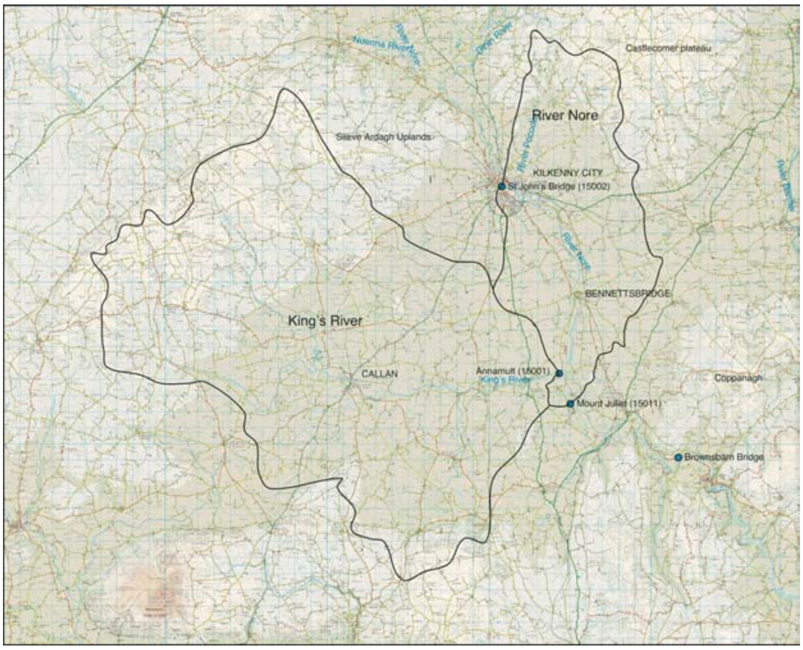

Figure 4: Location of study catchments, Callan-Bennettsbridge area

Recharge in these two subcatchments was estimated using soil moisture budgeting and river baseflow separation techniques. The Bennettsbridge subcatchment is part of the main River Nore catchment, and had been investigated previously (Misstear and Fitzsimons, 2007).

The stream flow data were analysed using the Boughton baseflow algorithm (Chapman, 1999). Examples of higher and lower estimates of baseflow are presented in Figure 5 for the King's River at Annamult. (Higher and lower estimates are given, in view of the inherent uncertainties involved in identifying the groundwater component of a stream hydrograph – see Misstear *et al*., 2009c).

With an average effective rainfall of 407 to 428 mm/yr - calculated using the FAO Penman-Montieth method (for a 30-year dataset from Kilkenny City) - the estimated recharge coefficients for the two subcatchments ranged between 36% and 60%. By calculating the areas of subcatchment covered by low or high permeability subsoils, and assigning likely recharge coefficient values to these subsoils, then the range of recharge coefficients for the moderate permeability subsoils within the subcatchments could be narrowed to 41-54% (Table 1).

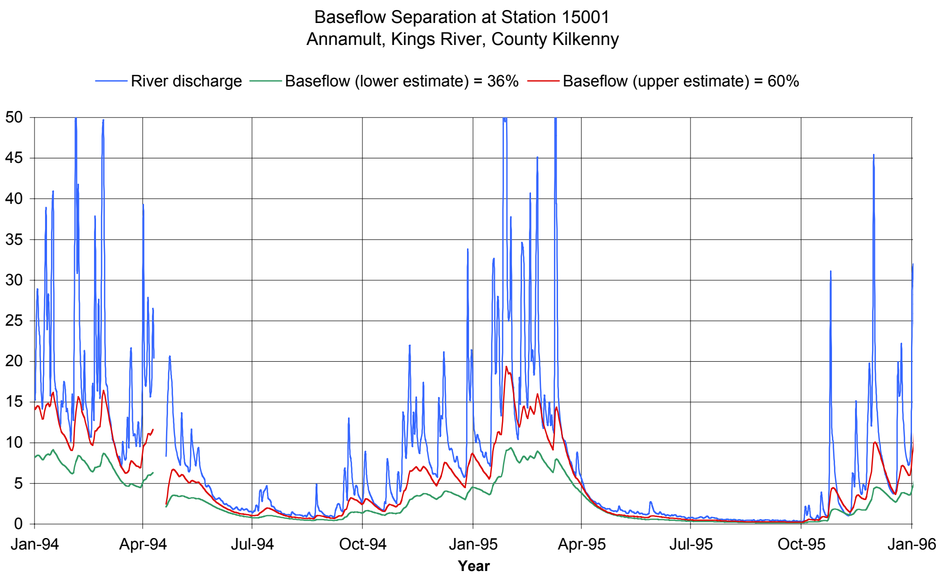

Figure 5: Baseflow analysis on the King's River, using the Boughton two-parameter algorithm

THE KNOCKATALLON AQUIFER

The Knockatallon aquifer is a fractured bedrock aquifer near the village of Tydavnet in north County Monaghan. The Knockatallon aquifer consists of two formations of limestone and other Carboniferous rocks: the Dartry Limestone Formation and the Meenymore Formation (Figure 6), with geophysical logging indicating that the Dartry Limestone is the main aquifer unit (Misstear *et al.*, 2008). The bedrock aquifer is covered by thick (up to 53 m), low permeability tills. Groundwater pumping by the Tydavnet Group Water Scheme (TGWS) from the 1980s onwards caused groundwater levels in the aquifer to fall substantially (up until the introduction of a treated surface water source in 2005, after which groundwater abstractions were cut back). In the period 2000 to 2005, an average of 1,000 m^3/day was abstracted from the wellfield (5 production wells).

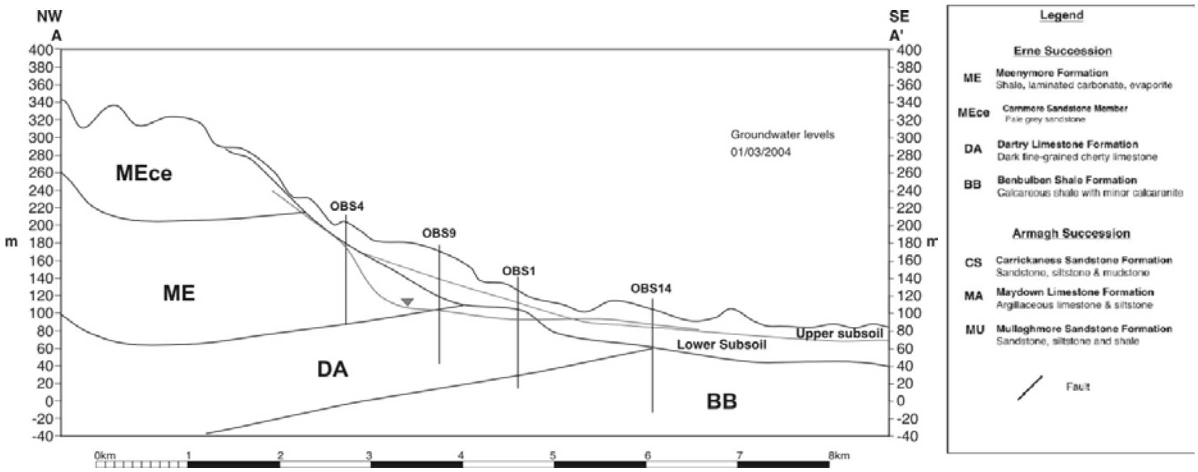

Figure 6: Cross section of the Tydavnet area showing subsoil geology, bedrock geology and potentiometric surface (Misstear *et al.*, 2008)

The lowering of groundwater levels in the vicinity of the wellfield is illustrated by the potentiometric surface contours in Figure 7. Following the steep declines in water levels during the early years of pumping, groundwater levels in several wells within the monitoring network were relatively stable, or showed only a slight lowering, in the period 2000-2005, suggesting that the abstraction rate of 1000 m^3/day by the TGWS was in near-equilibrium with recharge to the aquifer.

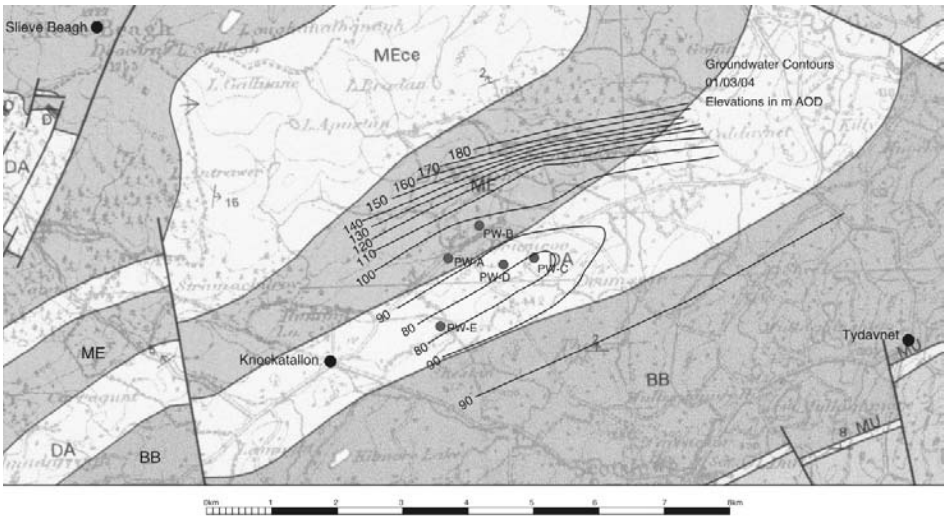

Figure 7: Potentiometric contours (m above datum) in the Knockatallon aquifer (01/03/2004). (Geology from Geraghty et al., 1997; refer to Figure 6 for the geological legend)

Owing to the presence of the thick, low permeability subsoil, direct recharge to the Knockatallon aquifer in the vicinity of the wellfield would be expected to be low. This was supported by a simple water balance calculation, which suggested that the annual average recharge through the till within the zone of contribution to the wellfield would be 26 mm, or 5 percent of the effective rainfall, if indeed this was the pathway for recharge. Dating of wellfield water (using chlorofluorocarbon isotopes), however, suggested that the main source of water at the wellfield was not from direct recharge through the overlying till, but rather from lateral flow from the Carnmore Sandstone bedrock upgradient of the wellfield (Figure 6).

River baseflow separation (again using the Boughton technique) on data from two gauging stations located several kilometres downstream of the main aquifer area suggested recharge coefficients of 12-17%. These are likely to be overestimates, in that the ‘slow flow’ component interpreted as groundwater baseflow is likely to include releases from peat deposits and other subsoils. Application of the Institute of Hydrology method (IOH, 1989) produced even higher baseflow estimates.

CASE STUDY RESULTS IN CONTEXT

The results of the four case studies were compared with previous estimates of recharge coefficient, as shown in Table 2. The table was prepared by the Working Group on Groundwater (2005) using the modelling work of Fitzsimons and Misstear (2006), allied with professional judgment. The findings of the case studies were consistent with the recharge coefficient bands within this table, thereby helping to validate those values.

Table 2: Comparison of case study results with recharge coefficients proposed by the Working Group on Groundwater (2005)

Vulnerability category		Hydrogeological setting	Recharge coefficient (rc)		
			Min (%)	Inner Range	Max (%)
Extreme	1.i	Areas where rock is at ground surface	60	80-90	100
	1.ii	Sand/gravel overlain by 'well drained' soil	60	80-90	100
		Sand/gravel overlain by 'poorly drained' (gley) soil			
	1.iii	Till overlain by 'well drained' soil	45	50-70	80
	1.iv	Till overlain by 'poorly drained' (gley) soil	15	25-40	50
	1.v	Sand/ gravel aquifer where the water table is ≤ 3 m below surface	70	80-90	100
	1.vi	Peat	15	25-40	50
High	2.i	Sand/gravel aquifer, overlain by 'well drained' soil [Curragh aquifer, Kildare]	60	80-90	100
	2.ii	High permeability subsoil (sand/gravel) overlain by 'well drained' soil	60	80-90	100
	2.iii	High permeability subsoil (sand/gravel) overlain by 'poorly drained' soil			
	2.iv	Moderate permeability subsoil overlain by 'well drained' soil	35	50-70	80
	2.v	Moderate permeability subsoil overlain by 'poorly drained' (gley) soil	15	25-40	50
	2.vi	Low permeability subsoil	10	23-30	40
	2.vii	Peat	0	5-15	20
Moderate	3.i	Moderate permeability subsoil and overlain by 'well drained' soil	25	30-40	60
	3.ii	Moderate permeability subsoil and overlain by 'poorly drained' (gley) soil	10	20-40	50
	3.iii	Low permeability subsoil	5	10-20	30
	3. iv	Basin peat	0	3-5	10
Low	4.i	Low permeability subsoil [Knockatallon aquifer, Monaghan]	2	5-15	20
	4.ii	Basin peat	0	3-5	10
High to Low	5.i	High permeability subsoil (sand and gravel)	60	85	100
	5.ii	Moderate permeability subsoil overlain by well drained soils [Callan-Bennettsbridge lowlands and Galmoy]	25	50	80
	5.iii	Moderate permeability subsoil overlain by poorly drained soils	10	30	50
	5.iv	Low permeability subsoil	2	20	40
	5.v	Peat	0	5	20

APPLICATIONS OF RECHARGE COEFFICIENTS IN IRELAND

The results of the recharge and vulnerability case studies assisted in the production of a national groundwater recharge map for Ireland. The map has been produced within a Geographical Information System (GIS), which includes layered information on rainfall, evapotranspiration, subsoil characteristics, soils information (soil wetness; peat), recharge coefficients and aquifers class - aquifer class is relevant, in that "recharge caps" are applied to poor and locally important aquifers. The map was prepared by CDM, Compass Informatics and the GSI, and the full methodology is described in Moe *et al*. (2007) and CDM (2008).

In addition to the utility of a recharge map for quantifying groundwater resources (at the river catchment or Groundwater Body scale), recharge values are a key input in the preparation of protection zones around individual wells or springs. Here, the application of the recharge coefficient methodology needs to take account of site specific data on subsoil variations, secondary (point) recharge, aquifer boundaries, groundwater level fluctuations, recharge acceptance and so forth.

The results of the case studies described in this paper were also used produce a quantified link between recharge and aquifer vulnerability, as shown in Table 3. For this analysis, the recharge coefficients are grouped into three categories: High (70-90%), Intermediate (30-70%) and Low (5-30%). For the 'High' category, the upper value of recharge coefficient was set at 90%, rather than 100%, since it is likely that some of the effective rainfall will almost always be 'lost' to runoff and/or interflow, even where subsoils are thin, absent or of high permeability. This 90% upper limit therefore makes some allowance for runoff due to factors such as topography and high intensity rainfall events. Again, a minimum value of 5% is proposed for the 'Low' category, since it is likely that some recharge will generally occur, even where the subsoils are thick and have a low permeability.

Table 3: Relationship between subsoil permeability, recharge, runoff and aquifer vulnerability (Misstear *et al*, 2009a)

Subsoil		Recharge	Runoff	Aquifer vulnerability
Permeability	Thickness			
High	1-3 m	High	Low	Extreme
	>3 m	High	Low	High
Moderate	1-3 m	High	Low	Extreme
	3-10 m	Intermediate	Intermediate	High
	>10 m	Intermediate	Intermediate	Moderate
Low	1-3 m	Intermediate	Intermediate	Extreme
	3-5 m	Low	High	High
	5-10 m	Low	High	Moderate
	>10 m	Low	High	Low

Table 3 also shows potential runoff (including interflow) categories, where these are related directly to the recharge class: High recharge = Low runoff; Intermediate recharge = Intermediate runoff; and Low recharge = High runoff. Hence, information on subsoil properties can be used to derive preliminary estimates of another main component of the hydrological cycle, namely surface runoff.

UK STUDIES

The effect of tills on groundwater recharge has also been considered in the UK, notably by Professor Ken Rushton (Rushton *et al*., 1988; Rushton, 2005). In his 2005 paper, Rushton summarised three approaches for estimating recharge through tills in UK, the selection of method depending on the heterogeneity and permeability of the till deposit (Figure 8).

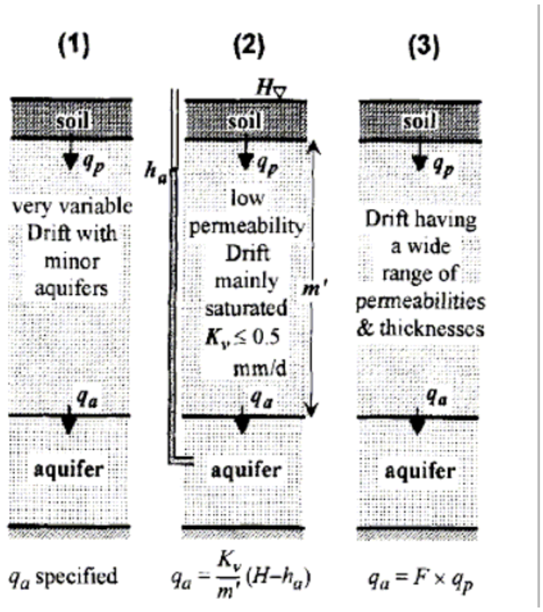

Figure 8: Alternative approaches for estimating actual recharge q_a when it is less than potential recharge q_p, owing to the presence of subsoil (drift) (Rushton, 2005)

In the first approach, a constant, low recharge rate through the till is assumed. In the second, the vertical infiltration rate through the till is calculated using Darcy's equation. For the third approach, the potential recharge (the effective rainfall) is multiplied by a 'recharge factor', the factor depending on the thickness and permeability of the till, and also the vertical hydraulic gradient. A more detailed account of the different approaches is given in the Environment Agency's guidance notes on regional groundwater modelling (2002). The recharge factor approach is the one that is most similar to the recharge coefficient methodology applied in Ireland.

In current groundwater modelling studies in the UK, recharge estimates are typically based on variations in the FAO Penman-Monteith soil moisture budgeting method. The presence of low permeability subsoils (referred to as "drift" in the UK) is sometimes accommodated directly as a separate layer within the numerical groundwater model, or by simulating interflow/rejected water within a separate runoff and recharge model. Recharge and runoff models include lumped (Catchmod), semi-distributed (Agency Recharge Code) and distributed codes (4R; Quinn *et al*., in preparation).

The local variability of superficial deposits and how this variability can lead to several different conceptual recharge models at a local scale is described by Cuthbert *et al*. (2009). Thus, for example, scenario C in Figure 9 illustrates the situation where water flowing laterally in a layer above a discontinuous low permeability till may recharge an aquifer indirectly. Although recharge for the different scenarios is not quantified by Cuthbert *et al*., their model is useful in highlighting the complexity of subsoils at a local scale.

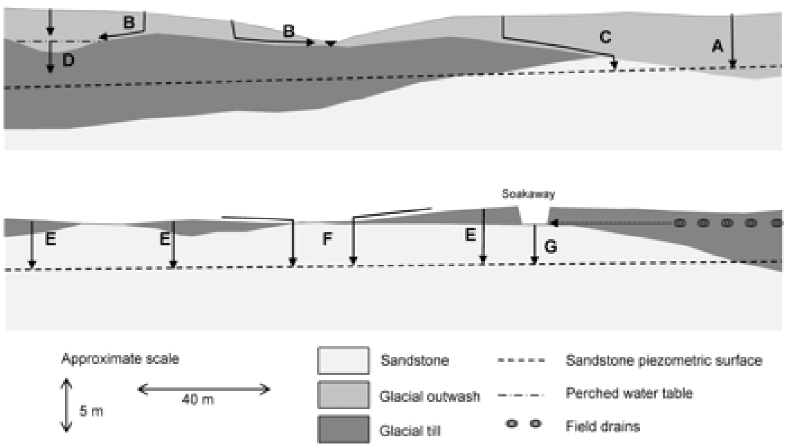

Figure 9: Local scale conceptual recharge models (Cuthbert *et al*., 2009)

DISCUSSION AND CONCLUSIONS

Glacial tills play a major role in controlling the recharge to Irish aquifers. The dominant till characteristic affecting recharge is permeability, although thickness is also relevant. Till permeability and thickness values can be used to derive recharge coefficients. When applied to effective rainfall values determined from a soil moisture budget, these recharge coefficients enable recharge to be quantified. The case studies described in this paper helped to validate previous recharge coefficient estimates. The results confirmed the strong linkage between recharge coefficient and subsoil permeability.

This approach to quantifying recharge is based on subsoils data, which are used to prepare vulnerability maps at a scale of 1:50,000. It does not take account of indirect (point) recharge or local variations in e.g. aquifer recharge acceptance. Therefore, the methodology is mainly useful for making preliminary assessments of groundwater resources at the river catchment or groundwater body scale. For local applications, including the delineation of source protection areas, it is desirable to take account of site specific data on features such as subsoil variations, secondary (point) recharge, aquifer boundaries, groundwater level fluctuations and aquifer recharge acceptance. The current EPA STRIVE Pathways research project should provide additional insights on groundwater recharge at the local scale, as the study areas are small sub-catchments (typically 5-30 km^2).

ACKNOWLEDGEMENTS

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KARSTIC GROUNDWATER SYSTEMS

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ABSTRACT

Karstified carbonate rock aquifers have many characteristics in common wherever they are located. However, there is also very commonly a case-specificity to any investigation in karst hydrogeology and this may cause difficulties in conceptualizing the groundwater system and involve the input of above-normal resources. Flow in karstic groundwater systems is via an integrated, hierarchical network of solutionally enlarged channels in which recharge modes strongly influence the throughflow systems and the throughflow system strongly influences the mode of discharge – primarily via springs. Recharge to karstic aquifers may be point or diffuse or a mix. Whatever the type of recharge, progressive sub-surface concentration of groundwater is also the norm. Flow velocities for groundwater are usually several orders of magnitude more rapid than under darcian conditions. Flow rates in Ireland derived from water tracings are similar to those reported from other karsts worldwide. Karst aquifers have been classified in a variety of ways including by flow type and by water chemistry.

DEFINING KARST GROUNDWATER SYSTEMS

Wherever in the world carbonate rocks are exposed to solutional weathering by acidified rain water, the results, in terms of surface landforms and the groundwater system are strikingly similar – a karstic terrain and a karstic groundwater flow system. Unfortunately, although the basic processes and results are universal, karst hydrogeology is notoriously site specific in some respects and understanding groundwater conditions in particular locales commonly requires the application of more resources than would be the case with more 'conventional' aquifer rocks. Thus, there is a limit as to the universal truths that can be confidently applied to any specific carbonate aquifer and even more so to any particular karstic water source. This paper briefly reviews present-day understanding of the functioning of karst groundwater systems but with particular reference to conditions prevailing in the Carboniferous limestone of Ireland. Useful perspectives on present-day thinking concerning karst hydrogeology are provided by White (1999), Worthington *et al* (2000) and capably summarized in Ford & Williams (2007).

As with other lithified rocks with secondary permeability, the basic control on the occurrence and behaviour of groundwater in limestone is the geological framework (including lithology, fracture density, bedding, faulting and folding) within which the water is stored and transmitted (Drew 2009). Uniquely however, as water moves through a carbonate aquifer it modifies the structure of the aquifer by differentially dissolving the rock and thus enlarging certain flow-paths for groundwater. These solutionally driven modifications increase exponentially through time and vary markedly in space. Within a short space of geological time (thousands or tens of thousands of years) the character of a carbonate aquifer may be utterly transformed by selective solutional erosion.
As the aquifer evolves through time:

- Flow systems increasingly resemble those of surface fluvial systems with a cover of rock;
- Storage decreases and transmission becomes more efficient;
- Groundwater flows become more concentrated, localized and integrated;
- Anisotropy and heterogeneity increase.

The term *channels* is used by Worthington and Ford (2009) to describe all solutionally enlarged openings ranging from fissures to conduits to accessible caves and to differentiate these openings from unmodified voids such as fractures or bedding partings. The relative proportions and locations of channels will vary greatly within and between karst areas but their existence is a fundamental characteristic of a karstified aquifer. The distinctive aspects of the karstic groundwater system are illustrated schematically in Figure 1.

Various definitions of what constitutes a karst groundwater system have been offered. For example:

- An aquifer dominated by solution conduits in which a turbulent flow regime occurs (Atkinson & Smart 1981);
- An aquifer with a permeability structure dominated by interconnected conduits dissolved from the host rock, which are organized to facilitate the circulation of water in a down-gradient direction wherein the permeability structure evolved as a consequence of dissolution (Huntoon 1995);
- An aquifer with self-organised, high-permeability channel networks formed by positive feedback between dissolution and flows (Worthington & Ford 2009).

Scale is an important consideration when investigating karst aquifers. The hierarchical, integrated nature of the groundwater flow system means that Representative Elemental Volume (REV) dimensions of several cubic kilometres are often necessary (catchment level) if the drainage system is to be conceptualized correctly. Extrapolation of groundwater conditions based on data from individual boreholes which sample only a tiny proportion of the aquifer are inherently risky.

In subsequent sections, recharge, throughflow and discharge from karst aquifers are considered separately. These distinctions are difficult to maintain however, as the three are inextricably connected genetically in a particular area. For example the mode of discharge is a function of the type of groundwater flow system whilst the flow system is in large part determined by the recharge mechanisms.

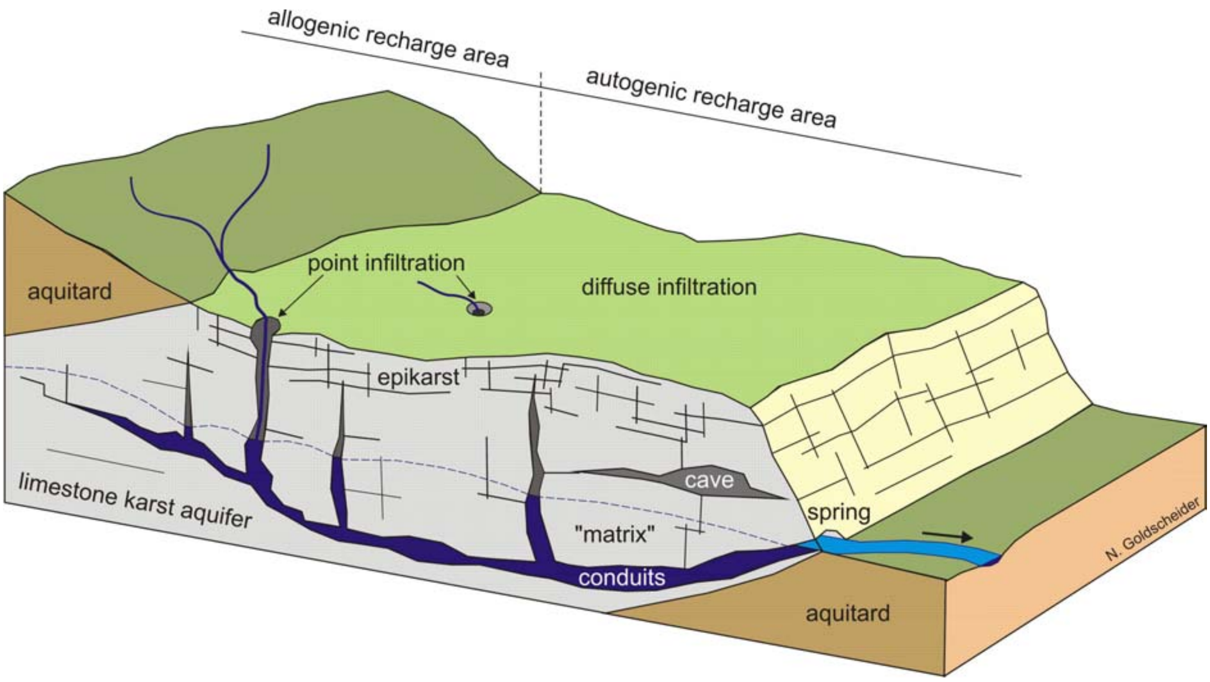

Figure 1: Schematic illustration of a heterogeneous karst aquifer system. After Goldscheider and Drew (2007)

ASPECTS OF RECHARGE

A striking characteristic of recharge in karst areas is the progressive down-gradient concentration of the recharging water into fewer but larger pathways. Initial concentration, at the rockhead will take place as the rainwater can only infiltrate where a secondary opening such as a joint, exists; but this is true for all fracture flow aquifers. Point recharge on a larger scale takes place where streams, generated on the limestone or flowing on to the limestone from adjacent carbonate rocks, sink underground at a point or as line-recharge. Such concentrated recharge is commonly distinguished from diffuse recharge as shown in Figure 1. If present, the epikarst may further concentrate (and store) water, focusing recharge on scattered points, 1-10m below ground level, which offer a preferred vertical pathway deeper into the unsaturated zone. Such recharge foci may evolve into the characteristic karstic landform of the doline or may remain without surface expression if mantled by a sufficient thickness of subsoil.

In Ireland, the great majority of the recharge is diffuse, though flow concentration at depth is probably the norm (Figure 2). Point recharge via sinking streams is typically within the range 5-10% of the total, rising to 40% or more on parts of plateaux karst such as the western Burren or the Cuilcagh massif or along the western flank of the Slieve Aughty Mountains in Co. Galway, where large allogenic streams flow on to the limestone and sink. Recharge concentration into dolines is widespread but difficult to quantify. For example, in Co. Roscommon doline densities reach 20 per km^2 in some areas (Hickey 2008).

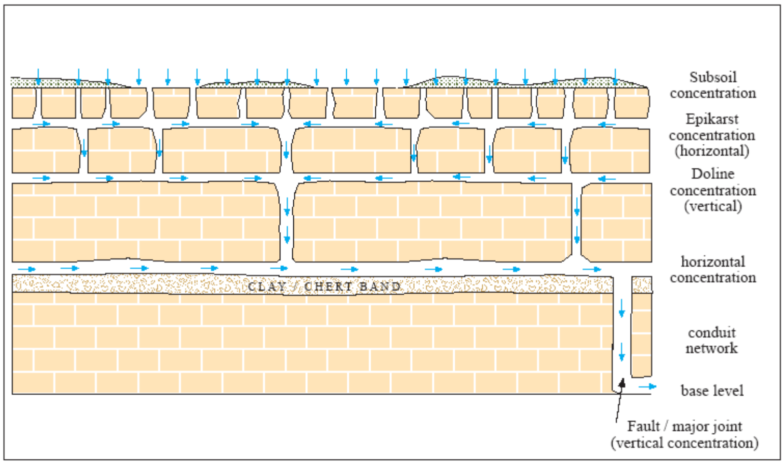

Figure 2: Schematic illustration of progressive concentration of recharge and groundwater flow in (for example) the Burren karst of Co. Clare.

Some surface streams especially in the lowlands of Clare, Galway, Mayo and Roscommon have stretches of channel which are inflowing to groundwater, at least under low stage conditions. Reaches of the River Clare and River Robe exhibit this behaviour whilst in extreme instances the rivers may dry completely for a part of the year over a part of their course; for example the Dunkellin and Lavally rivers in Co. Galway.

Finally, turloughs of which more than 135 are known, function as a unique mode of groundwater recharge in parts of the western lowlands (Coxon 1986, 1987). Storage in the larger turloughs may be considerable and they may function as significant sources of buffered point recharge over an extended period – usually in the late spring – early summer.

GROUNDWATER FLOW SYSTEMS IN IRELAND

A true karst groundwater flow system cannot exist until dissolution of the limestone has created a through route from recharge zone to an outlet point. This conduit then becomes a hydraulic low upon which flow converges and thus tributary conduit systems are formed and the network evolves progressively headwards. In immature karst systems this channel network will consist of a network of poorly integrated, small diameter channels whilst in a developed karst drainage system a hierarchy of channels, often ordered in a dendritic network, will convey recharge swiftly and efficiently to discharge points. Both end-members of this range are present in Ireland.

Flow systems in catchments with appreciable allogenic point recharge are dominated by cave (conduit) systems with very rapid flow through times (flow rates of several hundred metres per hour) and low storage, with consequent implications for potential contamination of the waters.

Hydraulic gradients in the upland limestones are often steep (>0.1) if the often deep unsaturated zone is taken into account. However, gradients within the saturated zone are similar in upland and lowland limestones, typically ranging between 0.001 and 0.01 (mean value 0.03). Representative conduit flow velocities in uplands are 20-300m/h and in the lowlands 5-250m/h. Figure 3 is a frequency graph of flow velocities in limestones in Ireland using data from 186 water tracing experiments. The mean apparent flow rate (assuming a straight line flow path) is 110m/h with the majority of values <125m/h and only a few exceeding 200m/h. This is comparable to the global value of 80m/h for more than 3000 tracings world-wide presented by Ford & Williams (2007). However, the values are not necessarily fully representative of flow rates in all limestones. For example, the sample is areally unbalanced with many data from Counties Clare, Galway and Roscommon but relatively few data from elsewhere. Also, the great majority are tracings from sinking streams and hence record conduit flows, which may not be representative of overall groundwater flow in some situations. For example, a tracing from a borehole to a spring probably via the epikarst, in Co. Offaly, gave a flow rate of 2-10m/h and this may be more typical of distributed flow in limestone. Also, it should be remembered that although almost all (>95%) transmission of water in a mature karst aquifer is via conduits, almost all the storage (>99%) is in small fissures and voids

Recorded groundwater flow rates are grouped by area or catchment in Table 1 and are further categorised into upland and lowland limestones. Flow rates vary five-fold between areas, with the highest velocities occurring in the lowland Gort-Kinvara aquifer where flow is dominantly in large (<20m diameter) conduits with a uniform gradient. Overall, no clear distinction between upland and lowland areas emerges.

Flow systems in the limestone aquifers may be dominated by conduit or by distributed flow or by a mixture of flow types in varying proportions. Conduit flow appears to dominate in upland areas but may be locally important in lowlands also – for example the Gort-Kinvara of south Co. Galway and in areas of east Galway, Mayo and Roscommon.

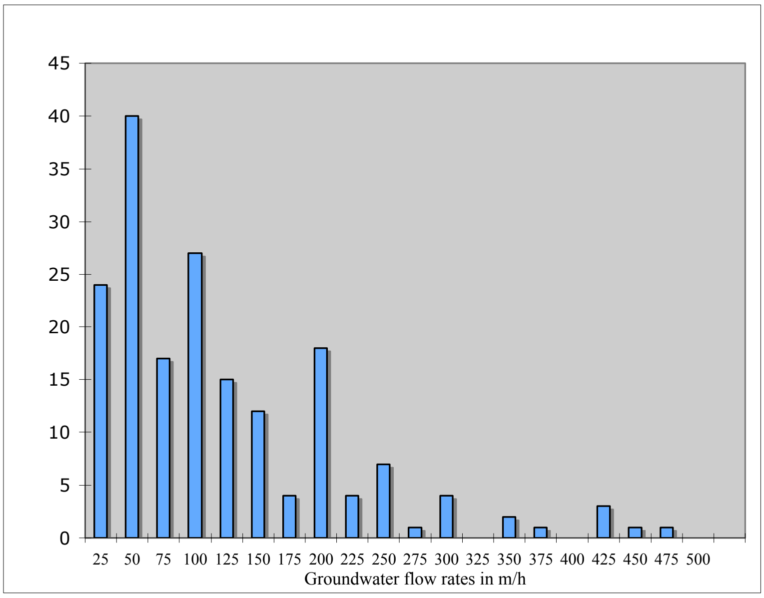

Figure 3: Frequency distribution of groundwater flow rates in Irish limestones (derived from water tracing data)

Table 1: Summary of Groundwater flow rates from water tracings in different areas of Ireland

CATCHMENT NAME	CATCHMENT TYPE	Number of water tracings	Range of Flow rates recorded (m/h)	Mean flow rate (m/h)
River Robe (Mayo)	Lowland karst	4	5-123	53
Lough Corrib (east Galway)	Lowland karst	7	6-440	117
Rivers Suck and Shannon (west) (Roscommon, Galway)	Lowland karst	43	2-279	140
Tipperary-Cork-Offaly	Lowland karst	14	9-135	37
Gort-Kinvara lowland (south Galway)	Lowland karst + concentrated allogenic recharge	70	7-1200	216
River Fergus lower (Clare)	Lowland karst	16	16-190	132
River Fergus upper (Clare)	Upland, Burren karst with concentrated allogenic recharge	19	21-330	83
Ballyvaughan springs (Clare)	Upland, Burren karst with limited concentrated allogenic recharge	10	---------	100
Upper River Shannon (Leitrim)	Plateau karst with concentrated allogenic recharge	13	3-206	74

GROUNDWATER DISCHARGE

Because Irish Carboniferous limestones are karstified to some degree groundwater flow in these aquifers is mainly focused on springs as point discharge points, though diffuse discharge into rivers does occur. More than 80% of the springs on limestone bedrock are small with a mean discharge of a few l/s and with correspondingly small catchments. Mean flow at c.200 springs exceeds 10 l/s and exceeds 25 l/s at c. 50 springs with catchments of several km^2. Two groups of springs are large in international terms: the intertidal zone springs at Kinvarra, Co. Galway, have an estimated mean discharge of 15 m3/s whilst the springs at Cong, Co. Mayo, which discharge both the sinking waters of Lough Mask and sinking streams generated on an area of non-limestone rocks to the west, have a mean outflow of c.20 m^3/sThe location of all springs on limestone with a mean discharge exceeding 10 l/s is shown in Figure 4. The great majority are located on the pure limestones or at the contact with impure limestones and are therefore most abundant in the west and the south. Only one spring on the impure limestone has a discharge >25 l/s.

The seasonal variation in discharge at springs is commonly <1:5 but a small number, mainly draining upland limestones have greater than tenfold variations in outflow. Killeany spring in the Burren, Co. Clare, is an extreme example, with a low:high discharge ratio of 1:60 (Drew and Chance 2007). Variations in discharge are accompanied by variations in water chemistry and both reflect the nature of the recharge and flow systems in the spring catchment. Table 2 summarises the discharge range, conductivity (a surrogate for water T.D.S.) range and variability and the frequency distribution of conductivity values for springs across a variety of limestone aquifers. Recorded conductivity values range from 230-765 but 70% of recorded values lie between 600 and 700 microsiemens/cm. Discharge and conductivity variations are greater for springs in the west (upland and lowland) and these springs exhibit a polymodal frequency distribution of conductivity values unlike the unimodal distribution in the midlands and south – a measure of the degree of karstification, and the recharge characteristics.

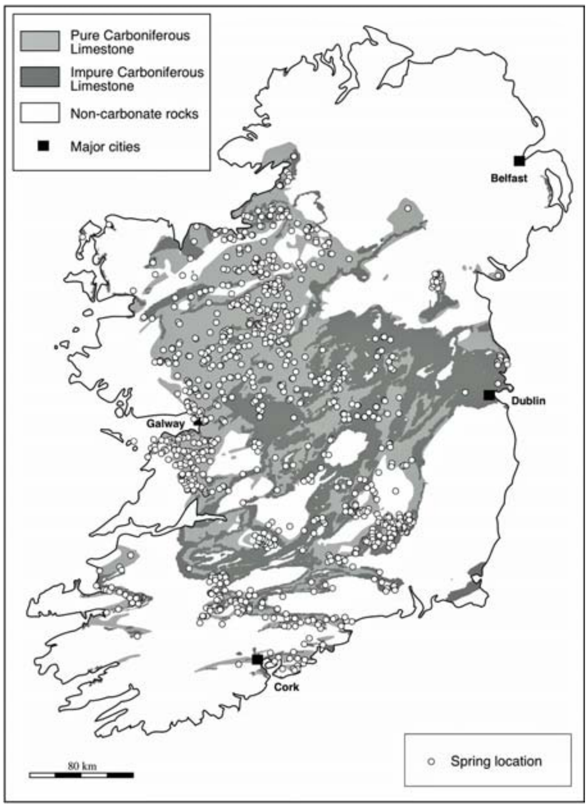

Figure 4: Location of all springs on limestone bedrock with mean discharge exceeding10 l/s (data from EPA Ireland)

Table 2: Discharge range and conductivity characteristics of some springs in regions of Ireland

LOCATION	Discharge range	Coefficient of variation of conductivity	Conductivity range (mean) microsiemens/cm	Frequency distribution type
Upland Burren Killeany spring	x60	5.3	230-500 (270)	Polymodal
Western lowland Bunadober spring	x8	6.2	630-755 (125)	Polymodal
Southern lowland Roaring Well spring	x2	0.06	660-670 (10)	Unimodal
Eastern Midlands Toberfin spring	0	0.5	750-765 (15)	unimodal

CONCEPTUALISING AND CLASSIFYING KARST GROUNDWATER SYSTEMS

Karst aquifers are evidently a special case of groundwater system. The three sets of hydrographs that comprise Figures 5, 6 and 7 illustrate this singularity. Figure 5 compares a borehole hydrograph for a 'conventional' sandstone aquifer with those for two karstified aquifers in the same locality over the same time interval. The 'flashy' nature of the groundwater regime in the karstic aquifers is apparent – a product of the peculiarities of their recharge, and groundwater flow mechanisms.

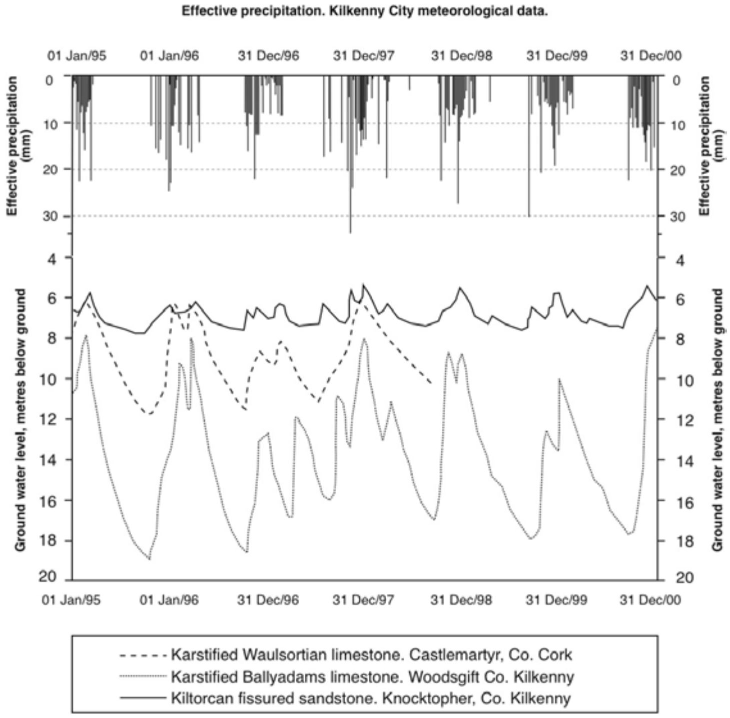

Figure 5: Well hydrographs 1995-2000 in karstified Waulsortian limestone, karstified Ballyadams limestone and Kiltorcan fissured sandstone (after Fitzsimmons and Misstear 2006)

However, internal differences within individual karst aquifers may also be considerable. Figure 6 shows the contrasting nature of borehole hydrographs over a four month period in a 6 km^2 area of highly karstified limestone near Ballinrobe, Co. Mayo whilst Figure 7 shows the varying responses of four boreholes, each only a few hundred metres apart near Rahasane turlough in Co. Galway, to rainfall. The marked differences in hydrological behaviour at the various locations in the above hydrographs are a reflection of the effects of scale – each borehole samples a part of the aquifer, which has its own distinct characteristics of void size, frequency and arrangement.

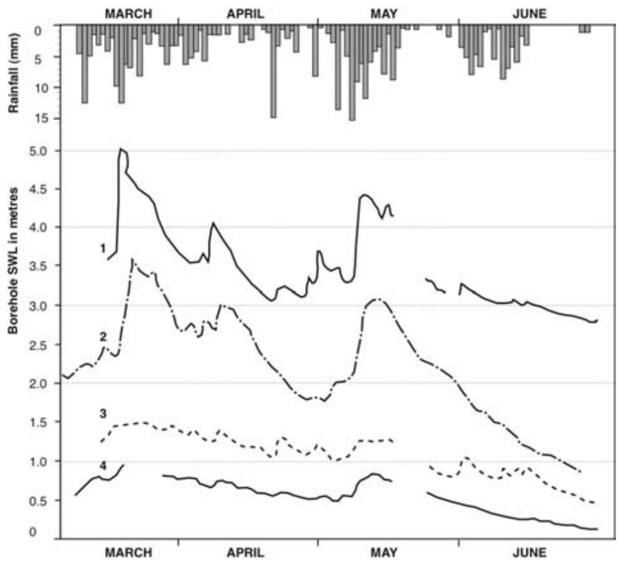

Figure 6: Well hydrographs March - June 1983, karstified limestone aquifer, Ballinrobe area, Co. Mayo, (after Coxon & Drew 1986)

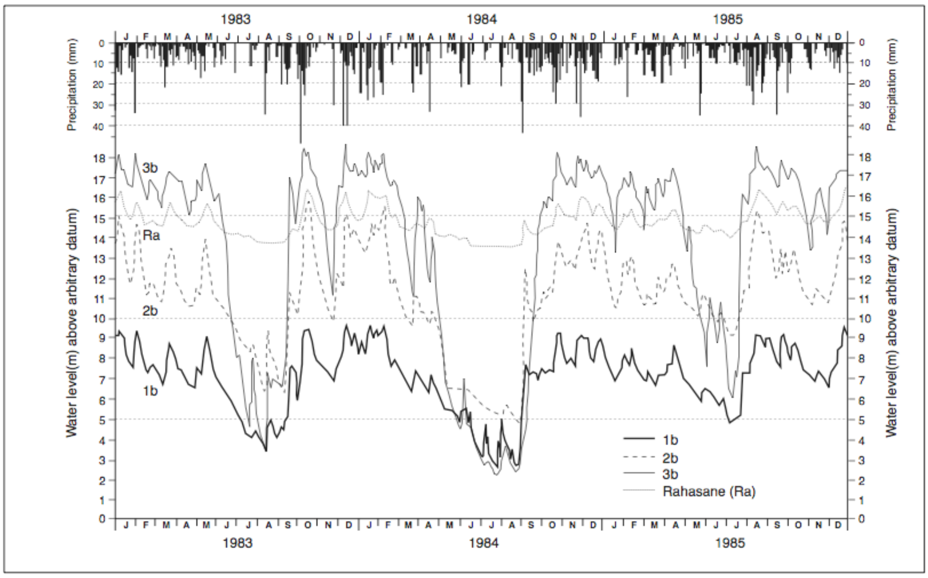

Figure 7: Water levels in boreholes around Rahasane Turlough, Co. Galway, 1983-85 (after Drew & Daly 1994)

A variety of ways of conceptualizing and hence classifying, karst aquifers have been proposed. Criteria have included strictly hydrogeological approaches such as flow regime, characterisation of the recharge, flow and discharge mechanism and also the use of indirect, surrogate, indicators of the nature of the aquifer such the chemical and physical profile of the spring waters. Figure 8 shows the classification suggested by Smart & Friederich (1987)

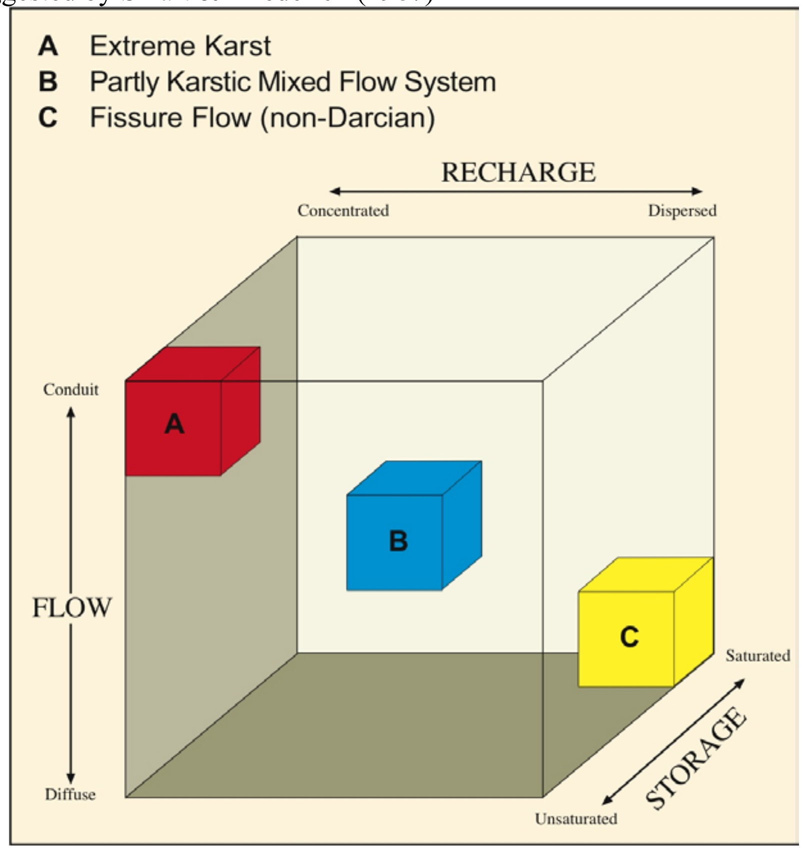

Figure 8: A classification of carbonate aquifers (after Smart & Friederich 1987)

This simple 3-way classification, qualitative rather than quantitative, allows a particular aquifer to be positioned within a cube based on the recharge, storage and flow types dominant. Type 'A' aquifers in Ireland would include many of the plateau limestones of the west and northwest. Type 'C' aquifers would presumably encompass the barely karstified areas of impure limestone whilst Type 'B' aquifers, with some degree of karstification, may well form the majority of groundwater systems within the Carboniferous limestone. Such attempts at classification, whilst inevitably somewhat arbitrary, are useful in assisting hydrologists to conceptualise what is happening in the particular aquifer under investigation.

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HYDROGEOLOGY OF IRISH POORLY PRODUCTIVE AQUIFERS: INITIAL FINDINGS

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ABSTRACT

Poorly productive aquifers (PPAs) underlie over 65% of Ireland. While these units are of limited hydrogeological significance from a water supply perspective, they can make a significant contribution to catchment water balances; they may strongly influence surface water quality, and thus affect aquatic ecosystem health, particularly during prolonged dry periods. Generic conceptual models view PPAs as consisting of fresh to heavily weathered fractured bedrock, often overlain by highly variable thicknesses of unconsolidated deposits. Hydrogeologically, flow in the superficial unit is commonly dominated by intergranular flow and has a relatively high storage capacity. On the other hand, the aperture and interconnectedness of low storage fractures strongly influence groundwater transmission through the bedrock.

Investigations into the hydrogeology of poorly productive aquifers completed through the Griffith Geoscience Research Programme aim to establish the validity of the existing generic model and how it needs to be better adapted to Irish conditions. Refining the conceptual model(s) permits the model to be employed in the development of catchment management tools to evaluate future climatic and human impacts on the wider hydrological cycle, with the aim of improving the effectiveness of proposed programmes of measures for the WFD. Achieving this goal involves developing a multidisciplinary investigation methodology combining geological, geophysical and geochemical techniques to characterise the structure and connectivity of groundwater pathways in PPAs at the catchment scale. Application of this approach is illustrated using preliminary results collected from the EPA monitoring site at Gortinlieve, Co. Donegal.

INTRODUCTION

The EU Water Framework Directive (2000/60/EC) requires a comprehensive understanding of the entire water cycle. This has resulted in a revised approach to monitoring and protecting groundwater resources that aim to illustrate interactions between groundwater and surface water, and the role played by groundwater in influencing the health of aquatic ecosystems.

In the past, hydrogeological regimes/conditions in poorly productive bedrock aquifers (PPAs) (including hard rock aquifers) have been largely overlooked in an Irish context (Robins and Misstear, 2000). These units now form the focus of a research effort, sponsored under the Griffith Geoscience Research (Griffith) Programme, to better understand groundwater flow and transport regimes, and the extent to which groundwater derived from PPAs contributes to the overall water balance at a catchment scale. The programme faces significant challenges given that studies completed elsewhere in similar geological settings suggest that the highly variable structural heterogeneity in PPAs substantially influences flow regimes and determines contaminant transport pathways and attenuation rates. Indeed, understanding groundwater flow and mass transport in fractured bedrock is recognised as one of the most challenging areas of hydrogeological research due to both the discontinuity and the heterogeneity of fractured systems (Faybishenko and Benson 2000, Neuman 2005), where the absence of a characteristic length scale precludes definition of a representative elemental volume (Black, 1994; Bonnet *et al*, 2001). Characterisation may be further complicated by the bedrock geochemical properties which can give rise to contrasting weathering histories that influence both physical and hydrochemical interactions between rock mass and groundwater. Despite these challenges, understanding the physical and chemical characteristics of PPAs remains critical for illustrating not only groundwater contributions to the catchment flow balance but also for better characterising the key processes influencing contaminant transport and attenuation.

The objectives of this research programme include testing the validity of commonly applied conceptual models (e.g. Marechal *et al*. 2004, Dewandel *et al*. 2006) of flow and transport in hard rock aquifers, and refining a methodology which can be confidently applied to characterise groundwater flow and mass transport in Irish catchments underlain by weathered/fractured poorly productive bedrock. Tackling this issue requires a multi-scale and multi-disciplinary approach employing geological, geophysical, hydraulic and geochemical techniques to provide a reliable basis for the development of coherent conceptual hydrogeological models. Ultimately these models will provide the foundation for mathematical simulations that will utilise techniques commonly applied to heterogeneous aquifers and in hydrocarbon reservoir characterisation, but have not yet been widely applied in Irish hard rock hydrogeology.

FIELD SITES

Five field sites have been selected for detailed investigation of hydrogeological conditions in PPAs by Griffith researchers. These cover a range of metamorphic (including low grade metasedimentary) and igneous rock types in northern and western Ireland. They consist of four sites instrumented by the EPA as part of its groundwater monitoring programme in counties Donegal, Galway, Mayo and Louth, along with an additional NIEA/GSNI site in Co. Down (Table 1 provides summary data for each site).

Table 1: Field site characteristics and work to date

Field Site	Geology	Catchment Area	Features	Works done and in progress…				
				Hydraulic tests (pump/slug/ packer)	Borehole Geophysical Logging (T, C, Calip., HiRAT, Res., nat.γ)	Surface geophysics (ERT, EM, Seismics, MRS)	Hydrochemistery (major and traces elem., envir. isotopes)	Groundwater Numerical Modelling
Gortinlieve, Co. Donegal	Single Unit: Southern Highland Group (Pelitic & psammitic schist, phyllite & marble).	~2km²	NE/SW trending fault.	2009	2009	first ERT and Seismics survey 2008 (APEX) ERT, EM, MRS planned 2010	2009	2011-…
Mount Stewart, Ards Peninsula, Co. Down	Silurian Gala Group Greywacke; Permian/Triassic Sherwood Sandstones	~5km²	Sandstone/ Greywacke contact	in progress 2010	2010	ERT, EM, MRS, Seismics in progress 2010	in progress 2010	2011-…
Mattock, Collon, Co Louth	Silurian metasediments; Ordovician volcanics and metasediments	~30km²	Thick alluvial fill at lowest part of transect.	planned 2010	planned 2010	first ERT and Seismics survey 2008 (APEX) ERT, EM, planned 2010	planned 2010	2011-…
New Village, Oughterard, Co Galway	Granites, Precambrian quartzites, gneisses and schists, dinantian pure bedded limestone, marbles	~ 5km²	Forested/Unforested comparison	2008	2008	ERT, EM, Seismics planned 2010	2008	2011-…
Glencastle, Belmullet, Co. Mayo	Precambrian quartzites, gneisses and schists	~10km²	Complex geology, faulting, densely fractured transition zone	2008	2008	ERT, EM, MRS planned 2010	2008, 2009	2011-…

The monitoring infrastructure at the EPA sites includes a groundwater monitoring network consisting of three borehole clusters installed along transects, approximately 1 km long, in each catchment. Each cluster contains up to four separate wells screened in the subsoil, transition, shallow and deep bedrock units. To better characterise the hydrological cycle at each study site, the groundwater network is complemented by an EPA surface water gauging point and additional points installed by QUB as part of related research projects. EPA monitoring of water quality and water levels in both groundwater and surface water, supplemented by additional targeted monitoring efforts completed by Griffith researchers, aim to provide a more integrated insight into the interaction of the different components of the hydrological cycle.

The EPA sites contrast in layout with the NIEA/GSNI site in Co. Down, where 16 boreholes (9 open holes and 7 screened piezometers) are distributed across ~3 ha. Groundwater is assumed to discharge into the adjacent Glen Burn, where flow rates will be continuously monitored following on-going installation of two weirs, located immediately upstream and downstream of the site. The more detailed monitoring infrastructure installed at this site will permit investigations to be carried out to evaluate the degree of hydrogeological variation that may be anticipated at the scale of 10s of metres in this hitherto poorly characterised unit.

Land use in the Donegal, Louth and Down catchments consists of pasture and tillage. The influence of the these activities on groundwater and surface water quality and processes leading to their attenuation along the various pathways connecting sources to aquatic receptors form the focus of associated EPA STRIVE pathways research, which is being carried out concurrently with the Griffith programme. Characterisation of these attenuation processes will rely strongly on confident characterisation of the hydrological and hydrogeological setting of each catchment. The results of both Griffith and EPA STRIVE pathways research will feed into a catchment management tool to be generated at the end of the latter project for use by river basin district managers.

METHODOLOGY

The initial approach adopted for characterising each PPA site employs a suite of routinely-employed geological, hydraulic, geophysical and geochemical characterisation techniques. These include:

1. Desk and field studies to determine/verify catchment boundaries, bedrock geology, overburden thickness and composition leading to a conceptual model landscape development;

2. Outcrop mapping of bedrock fracture characteristics to identify/constrain dominant and hydraulically active fracture sets;

3. Mapping and sampling of the overburden to determine subsoil type, thickness and geomorphological origins;

4. Geophysical characterisation working down from large scale surveys covering the entire area of each catchment (EM, ERT) to more localised investigations (2D/3D seismics, cross-hole/3D ERT, MRS, GPR, magnetometry, when applicable) at specific sub-scales;

5. Borehole drilling and well installation (completed by EPA contractors/GSNI personnel);

6. Geochemical analysis of bedrock and overburden;

7. Borehole geophysical logging (temperature, conductivity, resistivity, natural gamma, high resolution acoustic televiewer (HiRAT), heat-pulse flow meter) is undertaken to map both bedrock structure and groundwater characteristics with depth and identify hydraulically active fractures. They also provide valuable data for ground truthing surface geophysical investigations;

8. Hydraulic testing: pumping tests, slug tests, and packer tests;

9. Geochemical sampling of surface water and groundwater and with analysis being carried out at all sites for field hydrochemical parameters and major anions/cations, and for trace elements, and environmental isotopes in selected catchments.

CASE STUDY – METASEDIMENTARY BEDROCK AQUIFER – GORTINLIEVE, CO. DONEGAL

The following case study presents preliminary results from research completed in the latter half of 2009, focussing on a 2 km^2 catchment site at Gortinlieve, Co. Donegal (Figure 1), as part of the initial site characterisation programme. Results have been generated by applying the above suite of approaches to develop and a revised conceptual model of flow and transport in the catchment, with emphasis on characterisation of the PPA and how it corresponds to the generic model employed to date.

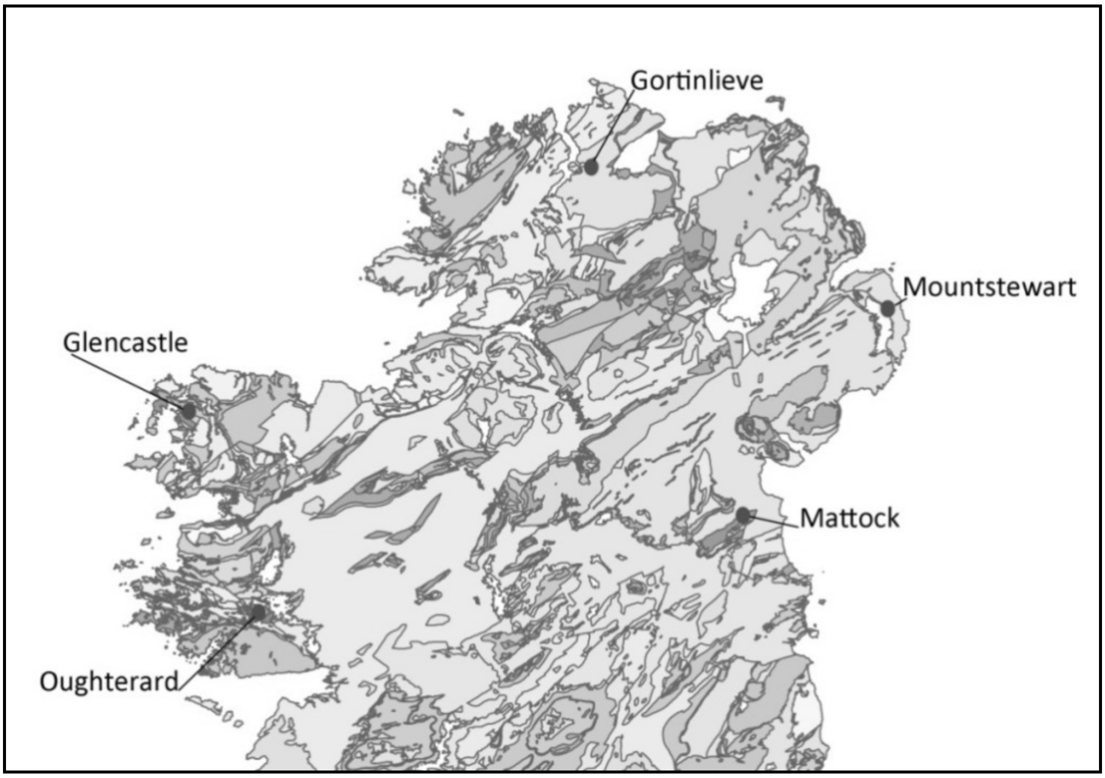

Figure 1: Field site locations overlain on the 1:1000000 scale simplified bedrock map of Ireland (GSI, 2003)

Desk studies of topographic maps and aerial photographs and field-based (hydro)geological mapping helped define the catchment boundary of the stream flowing through the study area, while helping identify hydrologically significant geomorphological features. Field mapping of exposures of the Dalradian psammite bedrock and overlying subsoils (consisting of thin layers of peat and glacial till, with thicker alluvium deposits in the vicinity of the stream) provided useful supplemental information to that available from geological maps. Similarly, surface geophysical measurements carried out by Apex Geoservices Ltd. permitted subsurface heterogeneity to be assessed and notably detected a steeply dipping electrically conductive feature at the same location as a spring line (occurring at a break in slope).

Mapping bedrock fracture properties (strike, dip, length) at outcrops in the catchment and in a nearby quarry suggested that a steeply dipping joint set covered in iron staining acted as a significant pathway for groundwater flow in bedrock. Elsewhere comparison of degrees of weathering based on staining and rock hardness suggested high spatial variability in the degree of alteration experienced by rock. Overall, the degree of weathering decreases, albeit irregularly, with depth. Drilling completed under supervision of EPA consultants (O'Callaghan Moran Ltd.) using downhole hammer showed that conditions in buried bedrock were broadly consistent with those observed in outcrop, while significant water strikes demonstrated the rock to be capable of yielding significant quantities of groundwater.

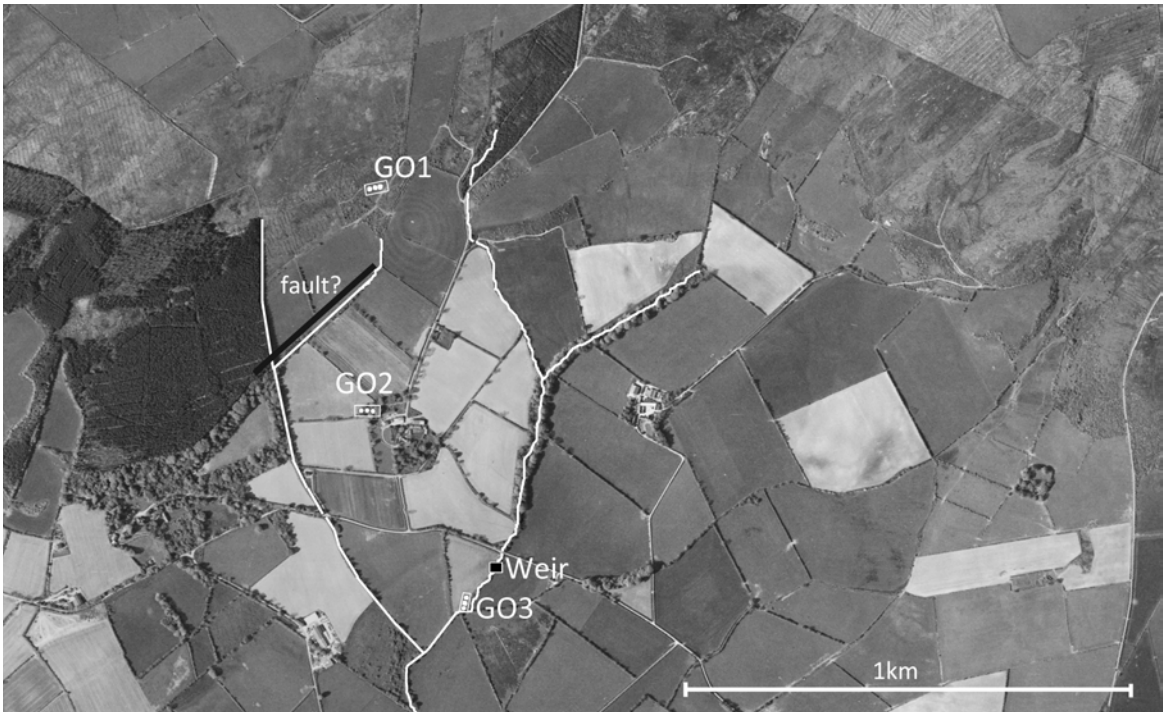

Figure 2: Aerial photograph (Ordnance Survey of Ireland, 2010) of the field site at Gortinlieve, near Newton Cunningham, Co. Donegal showing the borehole clusters GO1, GO2 and GO3 and weir installed by the EPA. Land use is predominantly pasture and tillage with coniferous plantation in the upper catchment and mixed deciduous and coniferous plantations in the estate lands at the west. Indicated between GO1 and GO2 is the location of a possible fault line which is considered to impact on the groundwater flow regime and will be the subject of further work.

Completion of the borehole clusters as monitoring wells (Figure 2) permitted pumping tests to be carried out to further investigate the hydrogeological properties of the bedrock and the overlying transition zone occupying the base of the overburden and the uppermost part of the bedrock. Figure 3 summarises test results and suggests a significant decline in hydraulic conductivity (determined by dividing transmissivity by saturated thickness) and storativity from the transition zone to the underlying bedrock. The higher hydraulic conductivity observed at depth at GO-3 contrasts with the decline in values observed at the other monitoring well clusters (Nitsche, 2009). Nonetheless, these overall characteristics observed are consistent with common hard rock models that attribute a storage function to the transition and shallow bedrock and a more transmissive function to deeper parts of bedrock.

Water quality monitoring completed during pumping tests permitted an evaluation spatial variations in hydrochemistry across the catchment to be completed. Results generated to date reveal notable spatial variability across the Gortinlieve catchment as well as in the vertical profiles of the monitoring well clusters (Figure 4). The trend towards an increase in total dissolved solids, pH and alkalinity with depth in the upper reaches of the transect, contrasts with that observed in groundwater samples collected from the cluster located at GO-3.

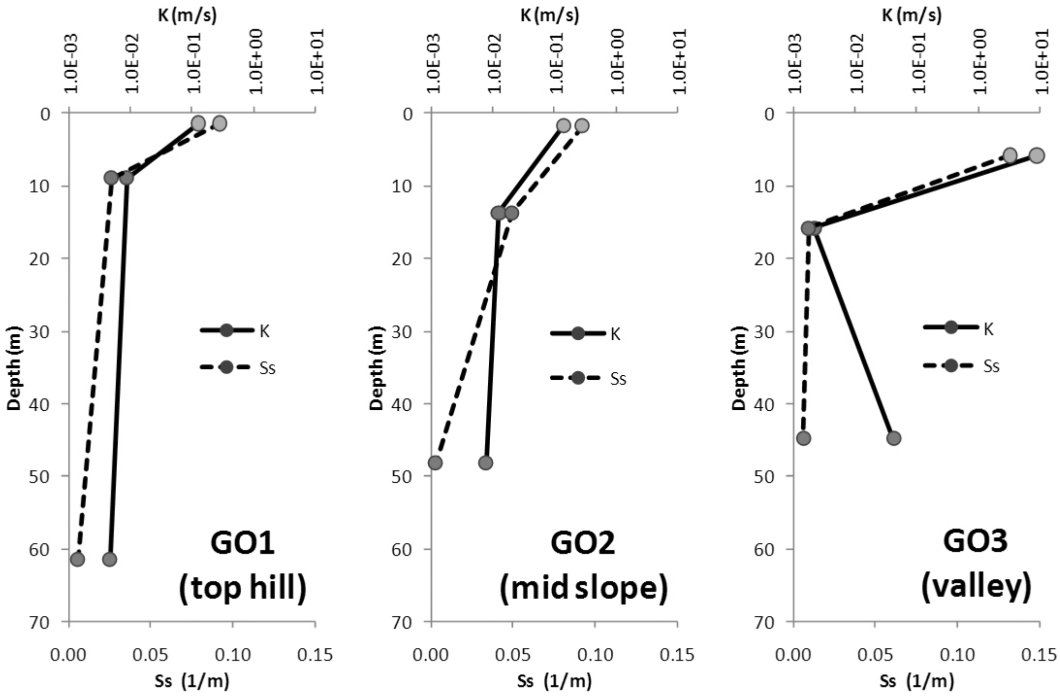

Figure 3: Variation in hydraulic conductivity (K) and storativity (Ss) with depth at each of the well clusters from GO1 (upper catchment) to GO3 (lower catchment)

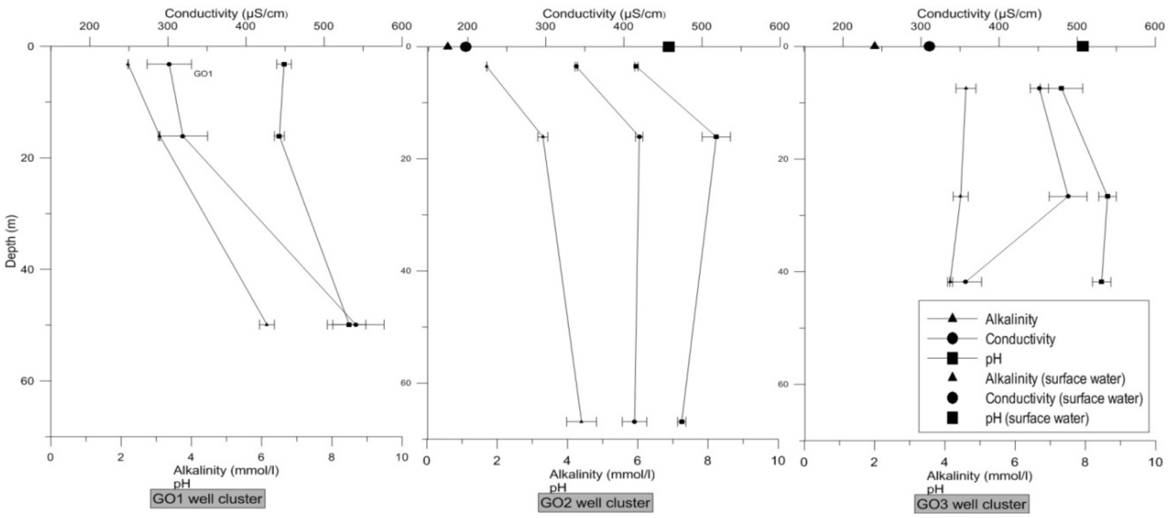

Figure 4: Variation of conductivity, pH and alkalinity with depth in the three well clusters, Gortinlieve. The values are averaged from a single well pumping tests of 0.5 - 6 hours duration

Although drilling provided valuable information on the nature of bedrock in the subsurface and the location of water strikes, downhole hammer data was unable to provide detailed information on the nature of the features supply water to the completed monitoring wells. Borehole geophysical logging provided a means of further investigating condition in accessible open holes. Calliper, HiRAT, natural gamma and differential temperature/water conductivity logs provided useful additional information on fracture density, geometry and hydraulic activity. Figure 5 provides an example of the output generated for GO-2 (Deep BH).

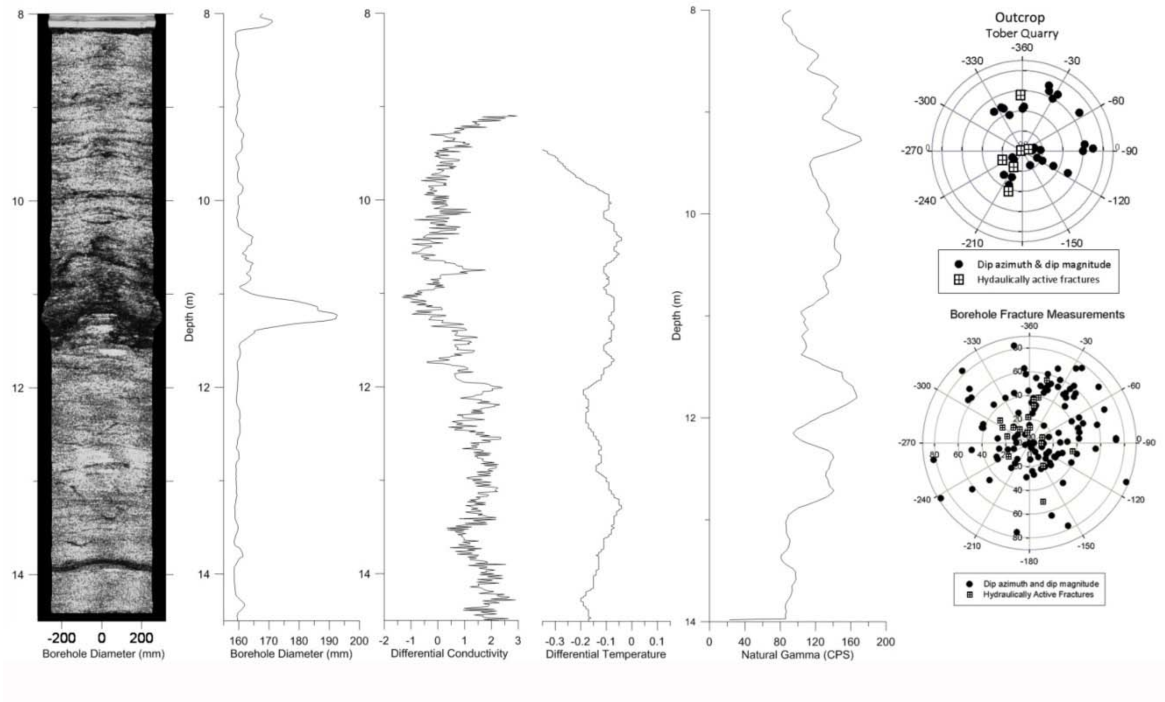

Figure 5: Geophysical borehole logs for a section of GO2 deep, Gortinlieve, with (from left) HiRAT image of the borehole walls, caliper log showing variations in borehole diameter, differential conductivity and differential temperature reflect groundwater discharge to the borehole using contrasts in hydrochemistry; natural gamma reflects variations in minerals containing ^{40}K, assumed in this case to be clays

Geophysical logging results indicated that not all fractures detected in open monitoring holes were hydraulically active. Comparison of fracture patterns generated by geophysical logging revealed a northerly dipping fracture set to be hydraulically active in those holes tested. This fracture set had also been identified in outcrop (Figure 5) and suggests that that the set permits water to flow to depth in a hydrogeologically anisotropic unit. Curiously, fracture analyses completed at GO-3 in the valley, suggested greater fracture density than elsewhere, despite the lower apparent transmissivity determined from pumping tests. Preliminary natural gamma log data suggest that the bedrock in this interval may be more weathered and that the occurrence of weathering residues in fracture sets may restrict hydraulic activity. In contrast, fractures encountered at depth at this location are suspected to have experienced less weathering and are capable of transmitting groundwater more efficiently. At higher ground elevations, the weathered horizon is suspected to be thinner and relatively unweathered fractures thus occur closer to the top of bedrock.

The contrasts in hydrochemistry observed reflect processes typical of silicate-dominated systems. In general less conductive, more acidic waters are believe to reflect shorter residence times, where water has yet to reach equilibrium with the minerals in the host rock, while the more alkaline, higher conductivity waters encountered reflect the increasing age of groundwater with depth. However, this pattern has not been observed consistently. For example in the vicinity of the stream, an inverted hydrochemical pattern is suspected to arise as a consequence of potentially rapid travel times along deeper more transmissive fractures. Further tracer-based approaches are required to clarify the processes giving rise to this phenomenon.

The inconsistency in hydraulic and hydrochemical conditions observed between monitoring well clusters is believed to reflect the influence of geological heterogeneity on groundwater flow and mass transport. The presence of hydraulically active steeply dipping fracture sets suggests that bedrock hydraulic conductivity anisotropy may play an important role in the hydrogeological regime across the catchment. Hydrogeological heterogeneity further complicates flow. The occurrence of a spring line along the monitoring well transect, and the corresponding geoelectric anomaly, point to larger scale features influencing groundwater flow. The spring line may arise because of a hydraulic barrier, such as that generated by fault gouge, which causes groundwater to stratify, with younger shallow water discharging to the ground surface (Hypothesis 1, Figure 6). Alternatively, the feature may be more conductive than its surroundings and permit deeper hydrochemically more distinct water to discharge. Further hydrochemical and geophysical investigations promise to shed more light on this issue (Hypothesis 2, Figure 6).

Integrating the data collected in this study during the final the six months of 2009 has permitted a refinement of the generic hydrogeological conceptual model (Figure 6). This model, as well as the dipping conductive feature revealed by resistivity profiles will be the focus of the next phase of geophysical investigations in the catchment, while natural tracer testing will be attempted to determine if this particular feature is acting as either a hydraulic barrier to flow from the upper catchment or as a highly conductive zone.

CONCLUSIONS & PERSPECTIVES

The results presented at this stage of the research programme are based on an initial six month investigation, yet already suggest that modifications to the generic conceptual model of poorly productive aquifers may be required. Similarly, new findings indicate that on-going refinements to the conceptual model(s) may have potentially significant implications not only for characterisation of their contribution to catchment water balances, but also potentially for programmes of measures, depending on groundwater body status.

In spite of the above modifications, investigations completed to date suggest that the physical aspects of conceptual model adopted at the start of this study remains broadly consistent. On the other hand, as results presented in the Gortinlieve case study indicate, geochemistry, and more notably weathering history, may play a more significant role in affecting hydrogeological conditions than previously suspected. More specifically the presence of minerals constituents that may alter to clays may play a potentially important role in reducing the hydraulic conductivity of hydraulically active fractures. As a corollary to this point the occurrence of clay may also significantly alter water quality and the ability of bedrock to attenuate contaminants. These issues remain to be investigated in further detail.

More generally, the preliminary investigations completed at all EPA test sites using conventional hydraulic and geological methods and suggest that groundwater can play an important role in maintaining stream flow/quality and thus affecting aquatic ecosystem health. Forthcoming research, to be completed from the summer of 2010 onwards, will focus on a number of issues in further detail that will permit conceptual models to be more confidently developed. Further research will include characterising the role played by bedrock geochemistry/weathering in influencing groundwater flow regimes, using both chemical and physical methods. This will further incorporate geochemical data with techniques that have been widely applied elsewhere but to a limited degree in Ireland, e.g. packer sampling/testing and stable isotope sampling/analyses. In addition, proposed multi-scale and multi-methods geophysical investigations should provide useful spatial information (2D/3D) on the influence of geological heterogeneity on relevant hydrogeological features, as well as for the interpolation of borehole data. These data promise to provide important information needed for the construction and parameterisation of groundwater numerical models.

Apart from widely used methods, researchers are also currently investigating further application of innovative tracer research techniques using comparative particle and solute tracer testing in fissured media, which have been pioneered with support from the Griffith Research Programme. Findings to date have demonstrated the potential importance of matrix diffusion as a means of generating persistent release of solute contaminants following the end of their application (Flynn & Sinreich, 2010). Combined use of solute and particle tracers thus provides a means of assessing the importance of matrix diffusion for solutes.

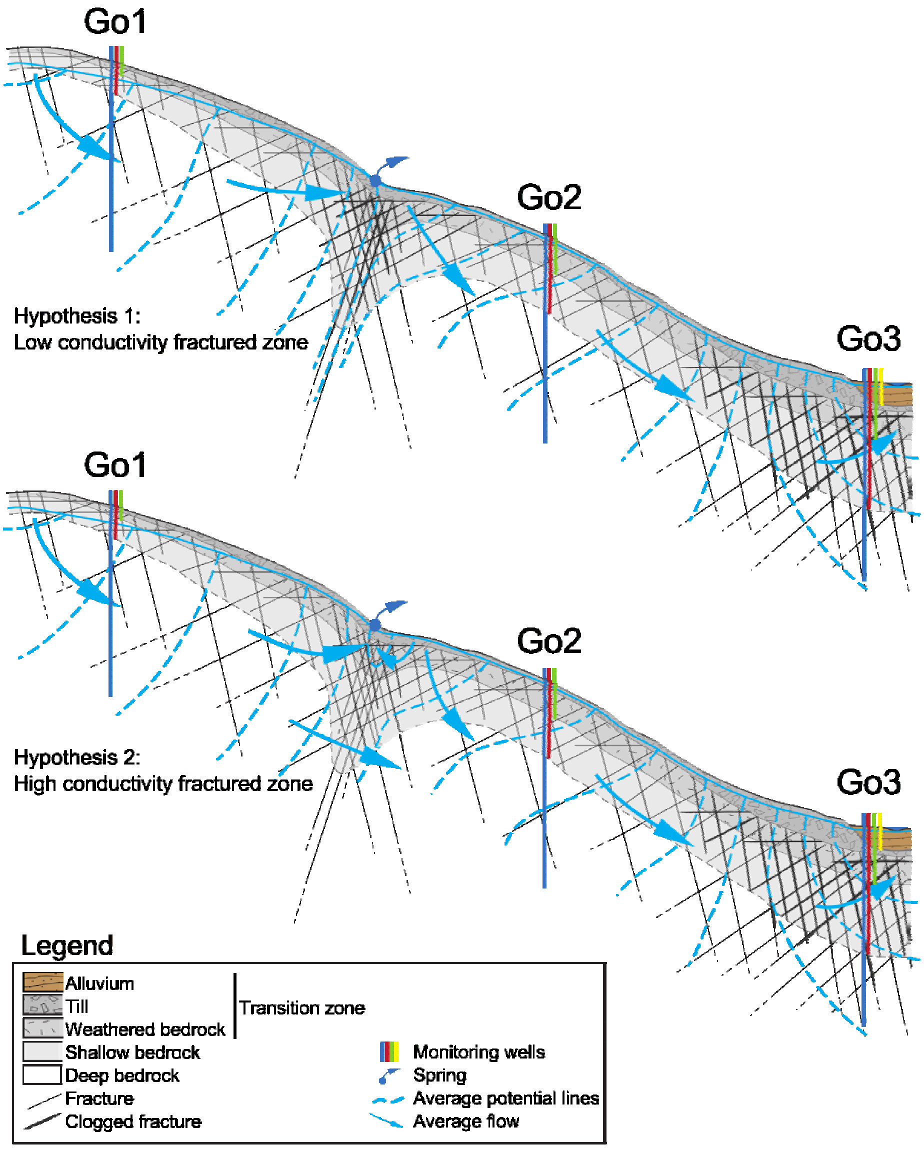

Figure 6: Conceptual models of flow along the well cluster transect, Gortinlieve, integrated and synthesizing all hydrogeological investigations carried out to date (not to scale)

The information gathered in this programme promises to have potentially significant implications not only for characterising groundwater flow regimes and mass transport processes in poorly productive rock types, but also for evaluating the effects of programmes of measures, and how/when these may result in improved surface water quality. Proposed investigation programmes will involve co-ordinated data acquisition and analysis with researchers involved in affiliated programmes such as the EPA Strive Pathways Research Programme and should serve to increase the confidence with which RBD managers can make appropriate decisions concerning the role of groundwater in surface water quality in areas underlain by poorly productive bedrock aquifer systems.

ACKNOWLEDGEMENTS

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GROUNDWATER SUPPLY DEVELOPMENT IN SAND & GRAVEL AQUIFERS

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ABSTRACT

Gravel Aquifers are an important aquifer in Irish Hydrogeology. These aquifers support larger groundwater abstractions, baseflow to rivers and support GWDTEs. It is crucial that we understand the mechanisms that influence groundwater flow and quality in the aquifers. There are significant differences between the hydrogeology of gravel aquifers and bedrock aquifers. These differences are largely related to the intergranular nature of groundwater flow and the higher storage provided by the aquifers. Developing gravel aquifer supplies requires informed drilling programmes with detailed design and suitable drilling equipment being the key to success. An example is provided of a gravel aquifer development in Co. Wicklow where detailed staged investigations were completed.

INTRODUCTION

Gravel aquifers can be found throughout Ireland. Their importance and productivity as an aquifer is derived from their high permeability and storage. Gravel aquifers have been successfully developed for groundwater supplies throughout the country and are also now being developed for geothermal projects for example in the Lee Valley in Cork City. Gravel aquifers also host important Groundwater Dependant Terrestrial Ecosystems (GWDTE) e.g. Pollardstown Fen and provide baseflow to rivers. The hydrogeology of gravel aquifers differs significantly from our bedrock aquifers. A good understanding and appreciation of the hydrogeology of these aquifers is therefore very important.

GRAVEL AQUIFER PROVENANCE

Gravel deposits in Ireland are mostly derived from the Quaternary Period (<1.6 Million Years Ago). Ireland underwent a series of glaciations and warmer spells (interglacials). Most Quaternary Deposits were laid down during the last glaciations that took place between 70,000 and 10,000 years ago. The deposits found in Ireland are broadly classed as either Glacial (deposited directly from the ice) or Glacio-fluvial (deposited by running water beneath or downstream of the ice). The Glacio-fluvial aquifers tend to be well sorted due to being deposited in running water.

Gravel aquifers can form characteristic geomorphological features such as Eskers, Outwash Fans, and Moraines. Or the aquifers can develop as features that are less obvious from the land surface including Buried Valleys, lacustrine and alluvial deposits. The largest gravel deposits in Ireland include the Mid-Kildare Gravel Aquifer (the Curragh), the Nore Valley Gravels and the Barrow Valley Gravels. Counties Kildare, Laois, Carlow, Kilkenny, Offaly and to a lesser extent Westmeath have the largest number and density of gravel aquifers. This may in part be due to the improved mapping in these counties as a result of Groundwater Protection Schemes having been completed there.

GRAVEL AQUIFER CHARACTERISTICS

The grain size of sands and gravels are sufficiently large that when packed together water can pass through them in sufficient magnitude to constitute an aquifer. The permeability of unconsolidated aquifers can be directly related to the grain sizes present. Indeed numerous methods are provided to derive permeability based on an effective grain size of the material with one of the earliest dating to 1911 (Hazen 1911). With reducing grain sizes the amount of water that can flow through is not enough for a deposit to be considered an aquifer. Deposits consisting of sediments smaller than sand (silt, clay) do not constitute aquifers.

While the presence of sand and/or gravel in a deposit is important, it is the smaller particles present in deposit that have the greatest control on the ultimate permeability. Therefore the absence of clays and silts from the interstices of the aquifer is crucial. Where the deposit is poorly sorted the finer particles plug the spaces between the sand and gravel grains and exclude groundwater.

Sand & Gravel Aquifers have a high effective porosity and specific yield as a result of the space present between the grains. The higher specific yield means there will be significantly less drawdown in the aquifer as a result of pumping in comparison to a bedrock aquifer. Bedrock Aquifers in Ireland are typically >400Ma and have very little primary porosity. As a results the permeability and storage in these aquifer is provided only by secondary features such as enlarged fissures. While large fissures can transmit considerable amounts of water (we often hear of underground rivers in karst aquifers) they do not act as a good store of water the size/volume of the fissures is small. The effective porosity of bedrock aquifers in Ireland is generally below 1.5% where as in gravel aquifers this figure can be up to 30%. This means that, for a unit drop in head, 20 times more water will drain from a unit volume of gravel aquifer than a bedrock aquifer.

This higher storage in the aquifer has a number of interesting and practical implications. For instance well hydrographs from monitoring wells in Sand & Gravel aquifers will have a much more muted and sinusoidal shape than a bedrock aquifer hydrograph. The change in water level in response to recharge is defined by the equation $R=\Delta h*Sy$ where R is the recharge, Δh is the change in water table elevation and Sy is specific yield. It follows that there is an inverse relationship between Δh and Sy – therefore aquifers with a large Sy (e.g. gravel aquifers) will have a small change in water table elevation (Δh) for a given recharge when compared to a bedrock aquifer with a large Δh. This difference can be clearly seen when comparing bedrock and gravel hydrographs with the gravel hydrographs fluctuating over a much smaller scale (a few metres) and at a much slower, smoother rate. Another interesting difference in the hydrographs is that peaks and troughs occur later in the gravel aquifer due to the slower response.

The equation to calculate average linear groundwater velocity shows that v (average linear velocity) is equal to Ki/n_e. Where K is permeability, i is hydraulic gradient and n_e is effective porosity. So in comparing a bedrock and gravel aquifer with similar permeability we see that the groundwater velocity in the gravel aquifer will be much slower because velocity is inversely proportional to the effective porosity which is much higher in the gravel aquifer. So while gravel aquifers can provide large abstractions this does not mean the groundwater is moving fast through the deposit; rather, there is more water in the aquifer so it doesn't need to move as fast.

This slower velocity has direct implications for aquifer protection. The GSI defines the inner source protection area (SI) as being the 100 day Time of Travel (ToT). This is the distance a drop of groundwater will travel to the well in 100 days. As the velocity in the gravel aquifer is much slower the resulting ToTs tend to be much smaller than in bedrock aquifers.

The equations for groundwater flow to a well under confined (Theis, Jacob) or unconfined (Neuman) show that the extent and rate of expansion of the cone of depression is inversely proportional to the storage coefficient. This also implies it will take longer for the pumping test to stabilise in a high storage aquifer. It will take longer for the cone of depression to reach any aquifer boundaries that may be of importance to the operational productivity of the well. Longer pumping tests should therefore be scheduled in Sand & Gravel Aquifers in comparison to bedrock aquifers.

GSI GRAVEL AQUIFER CLASSIFICATION

The GSI have developed a classification scheme for gravel aquifers (DoELG/EPAGSI 1999). The classification system of a gravel deposit as an aquifer is based on the following criteria:

- The deposit has sufficient permeability to be considered an aquifer ($>10^{-4}$m/s);
- Silt/clay fraction is below 7% and typically less than 5% (OSuillebhain 2000);

- The deposits need to have an area of at least 1 square kilometre (some exceptions may apply);
- The deposit normally needs either a saturated thickness of at least five metres or a total thickness of at least ten metres (where information on saturated thickness in unavailable);
- The continuity of the gravel deposit must also be taken into account i.e. can groundwater move throughout the entire aquifer or is it compartmentalised due to sections of lower permeability.

The GSI classifies Sand & Gravel Aquifers into two classifications: **Rg**: Regionally Important Gravel Aquifer – Area must be > $10km^2$ and **Lg**: Locally Important Aquifer – Area 1 – $10km^2$.

GSI VULNERABILITY CLASSIFICATION OF GRAVEL AQUIFERS

The GSI also provides a specific classification of unconfined gravel aquifers. The aquifer classification table where the water table is within 3m the aquifer vulnerability is classified as Extreme. In all other cases the aquifer vulnerability is classified as High. It is notable that the vulnerability of the gravel aquifer does not reduce to below High irrespective of the depth to the water table. Gravel aquifers are typically high vulnerability when examined in completed vulnerability maps. However there are some exceptions where a gravel aquifer is mapped but the aquifer is confined or buried under a protective layer of lower permeability deposits (e.g. Tills). In these cases the vulnerability of the aquifer may be less than High.

EPA/ TEAGASC GRAVEL MAPPING

The EPA Soil & Subsoil Mapping Project (formerly Teagasc FIPS-IFS) identifies a number of gravel deposits throughout the country. The gravel deposits follow the nomenclature G*xxx,* where the G signifies a gravel deposit and this is followed by a code that indicates the parent material of the gravels. For example GDSs is Gravel deposits derived from Devonian Sandstone Parent Material. This implies the gravel clasts are made of eroded Devonian Sandstone rather than the gravel having being deposited in the Devonian (as previously mentioned Irish unconsolidated aquifers are Quaternary in age). These types of deposits have the potential to be classified as aquifers provided there is sufficient extent, thickness and permeability.

Other deposits which may be high permeability are identified on these maps however they may not be suitable as aquifers due to limited extent e.g. Eskers (Esk) or proximity to the coast Marine Sands & Gravel (MGs), Raised Beaches (MBs & MBg). Other deposits that may be worth investigating are Alluvial Sands and/or Gravels (As / Ag) and Lacustrine Sands and/or Gravels (Ls/Lg).

It is also important to note that the EPA Soil/Subsoil maps are developed from remote sensing data, landform mapping with field checking. The mapped subsoil deposits are representative of the top most strata. Therefore it is possible that gravel deposits may exist buried under unproductive till deposits and these gravels may not be identified on the map.

GRAVEL AQUIFER EXPLORATION AND DEVELOPMENT

Gravel aquifers can be exploited using a number of means including Boreholes, Dug Wells, Infiltration Galleries, Collector Wells and pumping from springs. The most common means of gravel aquifer exploitation in Ireland is by vertical boreholes. Dug Wells are the traditional means of exploiting gravel aquifers and are still in use for both private and public water supplies. They require that the water table is relatively close to surface and that the annual variability in water level is not very high. Infiltration galleries, which consist of horizontal wells draining towards a sump, also require a shallow water table with little fluctuation in levels. These are more suited to alluvial or terrace gravels and can sometimes be drilled directly underneath rivers.

Drilling boreholes in gravel aquifers requires different approach to that in a bedrock aquifer. Typically a trial well will be required to establish the depth and nature of the deposits. There can be significant vertical variability in gravel deposits. Narrow diameter trial wells can be drilled at much reduced cost and their final productivity is not an issue. The production well design should be based on the findings of the trial well. A number of drilling methods can be applied to gravel aquifers. The success of the drilling program will be greatly improved by knowing the materials that will be encountered.

Where drilling is to be completed using Down the Hole Hammer (DTH) drilling the flushing media can be air or mud. Where mud drilling is chosen the density of the mud in the hole will keep the borehole open until the casing is installed. It is however crucial that adequate time is allowed for proper development of the well to remove mud's to ensure the optimal yield is derived from the well. Where the flushing media is air the quickest and most successful method for drilling is the Symmetrix type system. Under this system the casing is advanced in tandem with the drill, by means of a casing-shoe, to ensure the walls stay open. The drill is retracted and the screen can be installed inside the casing. The casing can subsequently be retracted allowing the deposits to collapse around the screen.

It is possible to develop a natural gravel pack in gravel wells. The development involves flushing or pumping the well after completion. This can be completed using the air injection system in the drill rig or dual-lift systems that have the ability to suck or blow sediments. The purpose of the development is to remove the finer particles from the aquifer material in direct contact with well screen. These finer particles can damage pumps and also inhibit the flow of water to the well. Coarse grained (d_{10} > 0.25mm) poorly sorted aquifers are suitable for natural gravel pack development. The slot size used under these circumstances is typically smaller (e.g. 1mm) but not so small as to limit groundwater inflow.

Artificial gravel packs are more suited to fine, well-sorted aquifers e.g. sand deposits (Misstear *et al*. 2006). The main advantage of artificial gravel packs are that they allow screens with larger slot sizes to be used, they reduce the risk of sand pumping and screen blockage, they increase the effective diameter of the well. Artificial gravel packs require a wider drilling diameters, which has significant cost implications and suitable pack material must be purchased.

In any gravel well it is advisable that machine slotted screen is used (e.g. Boode). For the type of diameters and depths of gravel wells PVC screen material is sufficient; more expensive slotted steel casing is not necessary. The slot size of the casing material can be chosen and typically ranges between 1-3mm. Where a natural gravel pack has been developed, a 1mm slot size may be more appropriate. The hydrogeologist should define the section of the wells that require screen and those which require plain casing. There is little to be gained from screening layers of low permeability, fine sediments that provide no water to the well and can lead to issues with turbidity and pump damage.

PUMPING TESTS IN GRAVEL AQUIFERS

Gravel aquifer extent and geometry is highly variable and the hydraulic connectivity of the aquifer cannot be fully understood by drilling and geophysical surveys alone. A pumping test is crucial to understand the hydrogeology of the aquifer and the long term sustainable yield of a well. The pumping test duration should be sufficient to allow the water table to stabilise in the aquifer. Gravel aquifers will take longer to stabilise because of their higher storage.

Initial driller's yield estimates can be misleading is smaller gravel aquifers. Where the aquifer is of limited extent e.g. buried valleys and there is no direct recharge from any surface water features the yield of the well can vary throughout the year as storage in the aquifer becomes depleted towards the end of the summer.

CASE STUDY – WOODENBRIDGE WELL FIELD, ARKLOW, CO. WICKLOW

The Woodenbridge well field consists of five production wells drilled in the glacio-fluvial sand & gravels in the R. Avoca valley at Woodenbridge. The sand & gravel deposits in this area are mapped as Undifferentiated Alluvium (A) under the EPA Subsoil Mapping and as Gravelly Alluvium under the GSI Quaternary map. The gravels are underlain by the Kilmacrea Formation (KA), which is a Dark Grey Slate with minor pale sandstone. The bedrock is part of the Ordovician Metasediment hydrostratigraphic rock unit group and is classified by the GSI as a Locally Important Bedrock Aquifer (Ll).

Extensive trial well drilling in the bedrock aquifer by Wicklow County Council (WCC) has found the bedrock aquifer to be largely unproductive except in local zones, which agrees well with the GSI aquifer classification (WYG 2009). Trial wells were drilled in the gravel aquifer in Woodenbridge between 2004 and 2005. Following the successful testing of these wells production wells were drilled including PW12 – PW13. PW12 was located adjacent the R. Aughrim in the Woodenbridge Golf Club. PW13 – PW16 were drilled in Coillte land southeast of the Golf Club along the banks of the R. Avoca.

The gravel deposits were 12.5 – 19.5m thick and they extended up to the steeply sloping hills of the valley. The gravels were found to be highly variable in vertical and horizontal horizons with layers of clays and cobbles interbedded with the sands & gravels. The gravel wells were drilled at 300mm diameter using the Symmetrix drilling system. PW12 – PW15 were fitted with 250mm diameter screen with 3mm wide slots. PW16 has 168.5mm screen installed. Following the installation of the screen the outer casing was retracted to 3m below ground and the gravel material was allowed to collapse around the screen to form a natural gravel pack.

The top 1.5m of the well annulus was filled with a bentonite grout to prevent the ingress of surface water to the well. The shallower sections of the well were not screened in order to try to prevent shallow groundwater from entering the wells. Following the installation of the well materials a well development programme was completed at each well for a period of 24 hours using a Dual-Lift system whereby the compressed air was injected into the well for two hours, then the system was reversed and the water was vacuumed out of the well for a further two hours. There was a significant reduction in the quantity of sand present in the groundwater in each well by the end of the twenty four hour period. A 72 hour stepped pumping test was undertaken on each production well followed by a 10 day constant rate combined yield test. The steps were analysed using the Hantush-Biershank (Misstear *et al*. 2006) method to assess the well efficiency at the maximum output of the well. The wells are very efficient (>90%) at all but the highest pumping rates where there is a reduction in efficiency but not such to cause concern.

The results of the step test showed there was minor interaction between the wells. All wells were capable of reproducing the maximum abstraction rate during the combined constant rate test. The results of the 10 day constant rate test show the drawdown in the wells stabilised after 1 day of pumping. Analysis of the pumping test data suggested this stabilisation was due to the interception of a recharge boundary, which was most likely to be the R. Avoca.

The total abstraction from the wells PW13 – PW16 during the 10 day test was 6,264m^3/d. The recharge to the aquifer is limited due to the relatively small aerial extent of the gravels. The up-gradient catchment of the bedrock aquifer is also quite small. The total annual average potential recharge in this area represents c. 1,500m^3/d. This suggested the aquifer must receive recharge from another source, which was likely to be the R. Avoca. Subsequent water table contouring confirmed the flow of groundwater from the river to the wells.

The wells can therefore to be considered as a River Bank Filtration system (RBF). RBF systems are well documented throughout Europe and the USA. The yield available from such wells fields is regularly >100,000m^3/d. The sustainability of these supplies is considered primarily in the context of the potential impact on the river they receive recharge from. In the case of the Woodenbridge wells initial estimates suggested 75% of the abstractions came from the river. The 95 percentile flow (Q95) of the R. Avoca at the PW13 – PW16 well field is estimated to be 2.25m^3/s (194,400m^3/d). This comprises 1.75m^3/s estimated for the Woodenbridge Station No.10008 and 0.5m^3/s at the Knocknamohill Station (No. 10028) on the R. Aughrim. The portion of the abstraction from PW13-

PW16 that is sourced from the river is estimated to be c. 4,700m^3/d, which represents 2.4% of the Q95 flow of the R. Avoca. The potential impact on the river flow and ecology is therefore considered to be negligible.

River Bank Filtration systems work as a natural filtration system for river water and are developed in favour of direct surface water abstractions for that reason. The resulting water is typically free of microbial contamination and heavy metals, which are filtered out in the riverbed.

An issue with RBF systems can be the reduction in well yield with time due to the clogging of the riverbed due to continued filtration. Any filtration system will clog over time due to the constant entrapment of suspended matter in the percolating water. In manufactured filtration systems backwashing is used to clear the out the filtrate. This is not possible in RBF systems without ceasing production and allowing groundwater flow to revert back towards the river. The clogging is in some cases balanced by episodes of erosion of the riverbed during flood events. The ultimate impact of clogging on the yield will be determined by the balance between sedimentation and erosion of the individual system.

The R. Avoca suffers from poor water quality as a result of acid mine drainage received from the now decommissioned Avoca Mines. Discharges from mine adits flow directly into the R. Avoca loading the river with elevated metal concentrations (Al, Fe, Mn, Pb, Cu, and Zn).

During the 10 day pumping tests sampling was completed on six occasions from the wells and the R. Avoca. An extensive list of parameters was analysed including total and dissolved metals. The results showed the metals in the R. Avoca, which exceeded drinking water limits (Al, Fe, and Mn), were not found in elevated concentrations in the production wells. The Total Aluminium results for the R. Avoca (average 311ug/l) were considerable higher than the Dissolved Aluminium (av. 79ug/l). Similarly the Total Iron results (av. 337ug/l) were higher than the Dissolved Iron (68ug/l). Total Manganese (av. 58ug/l) and Zinc (av. 105ug/l) were more similar to the dissolved portions (52ug/l and 100ug/l respectively).

This difference between the total and dissolved portions of the metal is a result of the prevailing river water conditions (pH). The total metal portion will be filtered out in the riverbed as this will be bound up in colloidal/particulate matter. The dissolved portion is more likely to flow through the aquifer towards the wells. The results from the production wells showed values were mostly very low or below detection limit for the metals. Some higher results for Mn and Zn were observed in PW14.

Following the completion of the pumping test a 2D groundwater model was developed for the PW13-PW16 site to assess the groundwater flow times from the river to the wells. The model indicated that the travel time along the most direct flow paths from the river to the wells would take a couple of days but there would also be longer flow paths where river water could take up to 100 days to reach the wells. The model indicated stabilisation of the river contribution to the wells would not be reached for at least 40 days. It was decided a longer pumping test was required at the site to assess the water quality at the wells over a longer duration of pumping. A 90 day pumping test was completed on PW13-PW16 from 19/08/09 to 11/11/09.

In order to gain a fuller understanding of the aquifer additional monitoring wells were drilled between each of the production wells and the river. The monitoring wells were drilled using a Shell & Auger rig, which allowed visual examination of the relatively undisturbed samples. The water bearing strata were found to be well sorted sub-rounded gravels. A number of samples from MW15 (located between PW15 and the river) were analysed for particle size distribution. The results show that the % fines in the gravel layers ranges from 0 – 5%. The samples were 65% Gravel and 35% Sand. There were also discreet layers of clay/silt.

The production wells, rivers and a bedrock spring at an adjacent road cutting were sampled on a weekly basis during the pumping test. Automatic water level and flow loggers were installed in the production wells with daily manual dipping of these and the monitoring wells. Physio-chemical parameters were also monitored on site on a daily basis. It was not possible to install a flow/stage gauge in the R. Avoca at the location due to the size of the river and the absence of any suitable structures to attach the gauge. River water level was surveyed on a number of occasions during the pumping test and data was available from an automatic gauge on the R. Aughrim in the vicinity.

The results of the pumping tests show the wells were able to maintain their yield during the course of the test with no reduction is specific capacity of the wells. This is an indication that clogging of the riverbed did not occur, or did not impact the river water contribution during the test. Water level monitoring showed the groundwater level was directly linked the river water level with aquifer levels rising and falling in tandem with the river.

Water quality results show microbial contaminants and metals were filtered out during the entire course of the test. Results for Aluminium and Iron were well below expected levels based on the percentage contribution from the river and were mostly below detection or very low. The results for manganese also showed very low levels in all wells, however there were some higher results recorded at PW13 towards the end of the pumping test, but these were below the drinking water limit.

The results of the pumping test were used to develop a more detailed 3D model of the aquifer. This model assessed a much larger area than previously assessed (3km x 4km extent). The model was used to assess the contribution of the R. Avoca to the wells, the potential impact of different clogging scenarios and whether there could be groundwater flow induced from the Shelton Abbey Tailings Pond. The Shelton Abbey Tailings Pond was the main dump for mine waste from the Avoca Mines. The site is located on the opposite bank of the R. Avoca to the production wells and is located downstream (and down hydraulic gradient) of the wells. Groundwater monitoring at the down-gradient of the tailings heap by the GSI suggests show elevate metal concentrations in the groundwater however this is also due to ponding of runoff from the heap in the vicinity of the well. There is no reduction in the river water quality of the R. Avoca downstream of the Tailings Heap.

The groundwater model was successfully calibrated for water level and quality at the site and was capable of accurately representing the trend conductivity observed at the wells. The model showed the system had almost completely stabilised during the course of the 90 day test. Therefore the water quality results from the second half of the test represent the water quality with the full contribution from the river.

The model indicated the capture zone to the wells did extent to the opposite bank of the R. Avoca up gradient of the wells field but did not cross the river down gradient of the well field. This implies the capture zone to the wells does not include the Tailings Heap and this will therefore have no influence on the water quality at the site. The model was run for a number of scenarios where clogging of the riverbed resulting in a reduction in the riverbed permeability. The literature suggests a reduction of two orders of magnitude, of the riverbed permeability, will occur c. 100m up and down stream of the production wells.

Under the clogging scenarios the wells did draw water from under the river at PW16 (the well closest to Shelton Abbey) but the capture zone did not extend very far down gradient to result in any significant contribution from Shelton Abbey. Under the clogging scenario there was some limited additional drawdown (0.5m in some wells), which is not expected to result in any reduction in well yield.

CONCLUSION

Gravel aquifers are an important aquifer in Irish Hydrogeology. These aquifers support large groundwater abstractions, baseflow to rivers and support GWDTEs. It is crucial that we understand the mechanisms that influence groundwater flow and quality in the aquifers. There are significant differences between the hydrogeology of gravel aquifers and bedrock aquifers. These differences are largely related to the intergranular nature of groundwater flow and the higher storage provided by the aquifers. Developing gravel aquifer supplies requires informed drilling programmes with detailed design and suitable drilling equipment being the key to success. Gravel aquifers are a relatively untapped resource in Irish Hydrogeology. Potentially massive yielding supplies could be developed from riverbank filtration systems e.g. consider a gravel aquifer along the R. Shannon. The sustainability of such supplies must be considered in the context of the direction contribution of the river water systems they are supported.

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SESSION II

QUANTIFYING GROUNDWATER-SURFACE WATER INTERACTIONS FROM THE SUBSTRATE TO THE CATCHMENT SCALE – HISTORICAL PERSPECTIVE AND FUTURE PROSPECTS

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ABSTRACT

Integrated management of water resources at the catchment scale is now generally recognised as being necessary for holistic environmental protection, and is required by legislation such as the EU Water Framework Directive. Good management requires sufficient scientific understanding of physical and biochemical processes with appropriate supporting datasets and modelling tools to support decision-making. Historically, groundwater and surface water have been considered separately, and this is now reflected in the modelling tools available, which retain a division between hydrogeology and surface water hydrology. Recently, there has been a significant international research focus on integrated water management, with specific focus on interfaces, which are recognised to be more important than previously considered. In particular there is considerable interest in the hyporheic zone, the region beneath or close to a river where there is mixing between groundwater and surface water with associated hydrochemical changes and impacts on ecological systems. The ecosystem services provided by this interface are only just starting to be understood, and the scales (spatial and temporal) over which processes act at this interface are smaller than have previously been considered in aquifer/catchment management. This paper presents a view of the historical development of understanding and modelling capabilities for integrated groundwater/catchment management, tracing some key aspects through to recent developments, and provides some thoughts on where modelling capabilities and associated studies may develop in the future.

INTRODUCTION

The need for a more integrated approach to the management of groundwater and surface water that focuses on the protection and enhancement of ecosystems is being driven by legislative frameworks such as the EU Water Framework Directive (European-Commission, 2000). These frameworks require consistent approaches to water resources, water quality and ecological objectives, recognising that these are closely inter-dependent. Integrated catchment management requires better understanding of these interactions, and appropriate modelling tools and methods to simulate physical, chemical and biological processes from the source to the sink following a source-pathway-receptor approach.

It is becoming increasingly recognised that the behaviour of interfaces between traditional environmental compartments is critical both to understanding, and to the efficacy of management actions. In particular, river - aquifer interactions (RAIs) exert significant control on the movement of water and migration of pollutants through catchments, and on the ecology of river corridors (Jones and Mulholland, 2000). The interface between aquifers and rivers is a dynamic environment in which chemical, biological and physical processes occur at a range of spatial and temporal scales, and which result in various gradients between surface and groundwater (Smith, 2005). It provides a range of ecological services, including energy, carbon and nutrient cycling, pollutant degradation and retardation, habitat opportunities for interstitial fauna, spawning beds for fish, and a rooting zone for aquatic macrophytes (Brunke and Gonser, 1997). Managers require appropriate tools to represent the relevant processes at the river – aquifer interface and these should be at the appropriate scales (Smith *et al*., 2008).

The broad environmental management themes that provide a context for RAI functioning are summarised in a multi-author report produced from the Hyporheic Network (a NERC-funded knowledge transfer network on groundwater – surface water interactions and hyporheic zone processes, supported by the Environment Agency of England and Wales) – "The Hyporheic Handbook" (Environment Agency, 2009):

1. "Sustainable management of water resources
2. Protection and improvement of water quality
3. Protection and improvement of lotic (e.g. river, streams or spring) ecology",

with cross-cutting issues identified as:

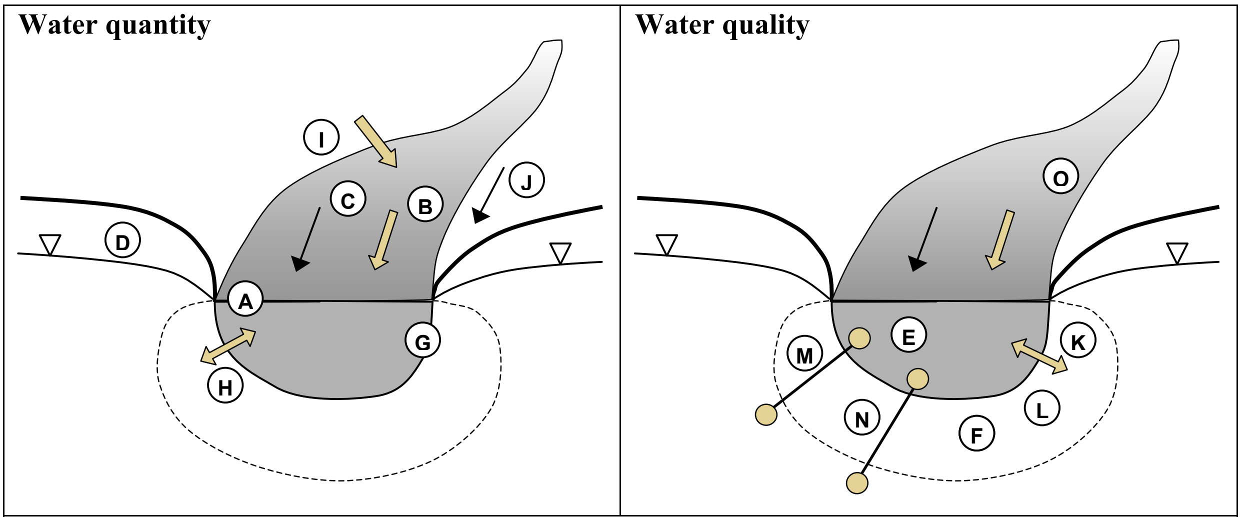
4. "Environmental monitoring and investigation
5. Risk assessment, modelling and forecasting
6. Restoration and remediation".

This paper outlines some of the current modelling approaches that are being used to help answer key environmental management questions that require modelling of RAIs.

CONCEPTUAL FRAMEWORKS FOR RIVER-AQUIFER INTERACTIONS

There have been a number of excellent reviews of the complex interactions between groundwater and surface water. Some of these focus explicitly on the inter-relationships between different disciplinary perspectives; for example Sophocleous (2002) reviews the interactions in relation to climate, landform, geology, and biotic factors. Many others, while recognising the multi-disciplinary nature of the subject, take a disciplinary perspective: Winter et al. (1998) give a summary of the hydrologic framework of surface water–ground water interactions; Jones and Mulholland (2000) summarize studies of solute transport, retention and transformations associated with hyporheic zones; Hancock *et al.* (2005) review hyporheic zone and river-aquifer interactions from an ecological perspective; Pachepsky *et al.* (2006) review the transport and fate of manure-borne pathogens in the hyporheic zone from a modelling perspective; and Gandy *et al.* (2007) review the attenuation of mining-derived pollutants in the hyporheic zone.

A central issue in modelling river-aquifer interactions is the scales at which a) predictions are required, and b) models (supported by available data) are capable of providing predictions. Dahl *et al.* (2007) reviewed classification systems and proposed a multi-scale typology based on geomorphological, geological and hydrological concepts, identifying a critical Riparian Hydrogeological Type which has interactions with a Landscape Type at scales of greater than 5 km, and with rivers at an intermediate or reach scale of 1 – 5 km and at a local scale of 10 – 1000 m. This division into three spatial scales is a convenient basis for analysis. Requirements for modelling outputs can then be assessed at these scales. Figure 1 shows an illustrative conceptual model of a river-aquifer system, indicating the main variables that are of interest for management. Some variables are of direct relevance, for example river levels, velocities, wetted perimeter, water and bed temperatures, and sedimentation/erosion rates are all related to ecological habitats, and knowledge of hyporheic zone residence time distributions may be critical for assessments of water quality compliance. Other variables are indirectly important, for example quantification of bulk river-aquifer exchange flows, locations of discharge 'hot-spots', and changes in Eh, pH and DO across the hyporheic zone are required for characterising hydrochemical fluxes and water quality.



A – River levels
B – River flows
C – Average river water velocity
D – Groundwater hydraulic heads in the riparian zone
E – River water temperature
F – River bed / hyporheic zone temperature
G – River wetted perimeter
H – Bulk river-aquifer exchange flow
I – Groundwater discharge ‘hot-spots’ (locations, areas, fluxes)
J – Downstream water flows through river bed substrate
K – Bulk river-aquifer hydrochemical fluxes
L – Hyporheic zone residence times (and residence time distribution)
M – Differences in hydrochemical parameters (Eh, pH, DO) across the hyporheic zone
N – Concentration differences of environmental pollutants across the hyporheic zone
O – Sediment erosion/deposition locations and rates.

Figure 1: Schematic cross-sections of hyporheic zones beneath rivers, with generic model output variables across and along the hyporheic zone, which are potentially of use in environmental assessment and management

HISTORICAL MODELLING PERSPECTIVE

MODELLING APPROACHES

Many computer modelling approaches that are in common current use have their origins in the 1980’s when personal computers came into widespread use, based on emerging ideas from the 1960’s and 1970’s. Computational models can be classified into different types, with often-used classifications being into either empirical (models based on relationships between variables) or physics-based (those based on fundamental physical laws), and lumped (with no spatial detail being represented) or spatially-distributed. Increasingly, the boundaries between these classifications have become blurred, but an important and relevant aspect here is the domain (i.e. the physical region) which models represent. Based on the historically recognised disciplines of hydrogeology and hydrology, many models have focussed on either groundwater or surface water systems. Increasingly, single-discipline models have been being coupled together (so-called ‘external coupling’) to provide a more integrated modelling approach, although some integrated modelling systems (with ‘internal coupling’) were developed from an early stage. Most models can represent systems at different scales, but each application of a model usually represents spatially distributed systems at essentially a single scale, and most models address scale issues through an increasingly detailed but essentially uniform subdivision of space (the ‘reductionist’ approach).

GROUNDWATER MODELS

The industry standard numerical groundwater flow model is MODFLOW, which was developed by the US Geological Survey. The history of MODFLOW is given in McDonald and Harbaugh (2003). MODFLOW considers two or three dimensional groundwater flow using a finite-difference representation of the equations governing flow in confined or unconfined multi-aquifer systems. In common with many groundwater models, rivers are represented in the original MODFLOW model as a head boundary condition represented by river levels, with the model calculating exchange flow rates based on a conductance parameter which represents the (inverse of the) resistance between the aquifer and the river. With this approach, only baseflow to the river is calculated on a cell by cell basis, and there is no feedback from changes in river levels. Various additional modules have been added to MODFLOW to calculate river flows with various levels of complexity (e.g. the BRANCH module with a diffusive wave approximation of the Saint-Venant open channel flow equations, the similar MODFLOW/DAFLOW model, the STREAM module, and the SFR2 module which includes unsaturated flow beneath river channels). Barlow and Harbaugh (2006) describe future developments of MODFLOW including the further development of coupled models.

MODFLOW and other models with a similar basis have been extensively used to support management of groundwater systems at a regional scale, and can provide spatially-varying information on bulk river-aquifer interactions at this scale (Rushton, 2007); Figure 2 shows some typical output on river-aquifer flows from the Environment Agency's National Groundwater Modelling System (NGMS), based on a finite-difference regional groundwater model.

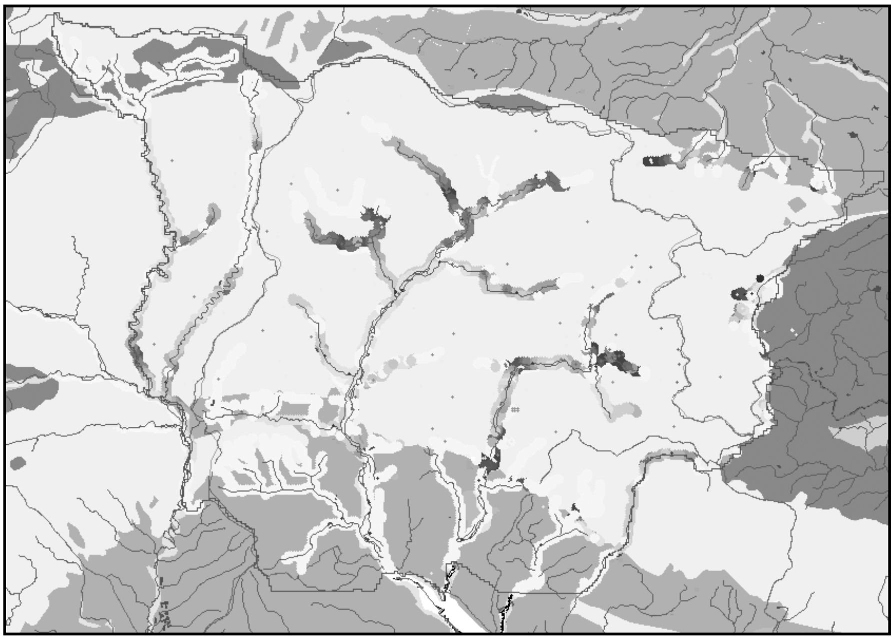


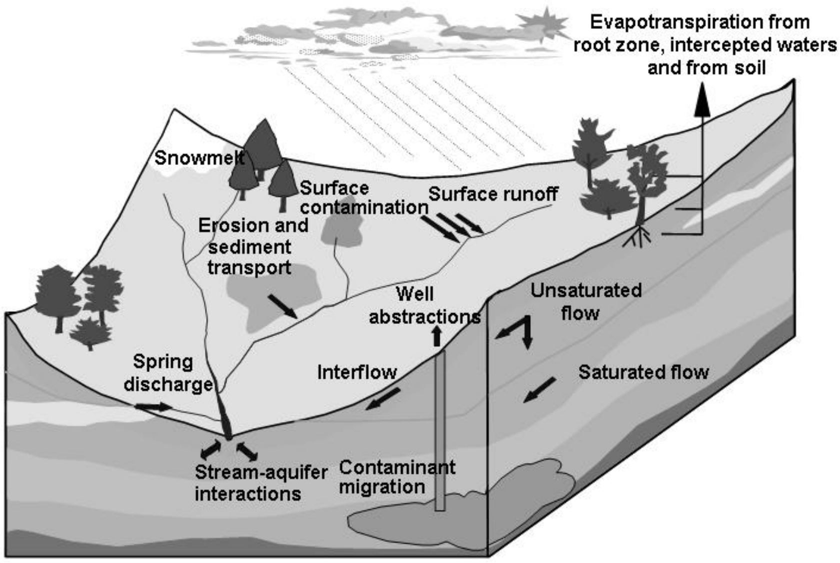
Figure 2: Example of river-aquifer flow rates in the Test-Itchen Chalk aquifers, from the Environment Agency's National Groundwater Modelling System (NGMS); darker shaded spots along the rivers represent higher rates of exchange flows at each cell in the model grid

SURFACE WATER MODELS

There are a large number of catchment modelling systems that approach the river-aquifer interaction problem from more of a surface water perspective, a well-known example being the SWAT suite of models (Gassman *et al,* 2007). Catchment models calculate total hydrological flows in rivers, and may make simple assumptions about baseflow components, using these as inflow boundary conditions. If more detailed instream information is required, for example cross-sections of water velocities for habitat assessments, localised river reach hydraulic models may be appropriate, taking inflows from catchment rainfall-runoff models as boundary conditions.

INTEGRATED CATCHMENT MODELS

One of the leading original integrated catchment modelling systems was the SHE model, based on a blueprint paper which recognised that there existed a basic understanding of each of the processes in the hydrological cycle. The SHE model has been further developed into two widely-used models – Mike-SHE, developed by the Danish Hydraulic Institute (DHI), and the SHETRAN model, developed by Newcastle University (Ewen *et al*., 2000; see http://www.ceg.ncl.ac.uk/shetran for further information). These models represent all key processes in each part of the hydrological cycle, with interaction between components for groundwater and surface water being transferred through internal exchange of information during a model simulation, and water flow variables such as groundwater – river exchange flows being used as the basis for solute transport calculations (Figure 3). These models represent spatially distributed parameters and outputs on similar finite-difference grids to groundwater flow models (Figure 3).



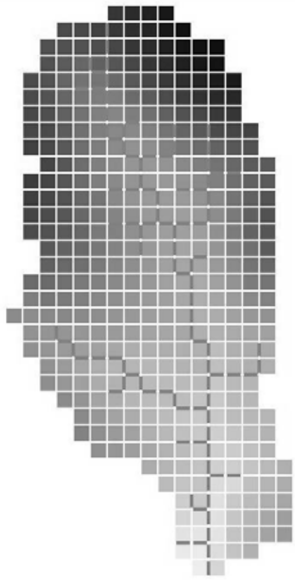


Figure 3: SHETRAN schematic processes and typical distributed modelling grid

MODELS OF REACH-SCALE CONTAMINANT TRANSPORT

There exist a considerable number of models representing small-scale instream cycling of contaminants based on the Transient Storage Model (TSM) approach, which represents solute exchange between river water and transient storage zones (for example the hyporheic zone) using first order mass exchange coefficients or diffusion models to describe lateral solute diffusion in the hyporheic zone (e.g. De Smedt, 2007). Some studies (e.g. Zaramella *et al.,* 2003) have found that TSM models can represent advective exchange with shallow beds but do not do a good job of representing exchange with a relatively deep sediment bed, although other examples (e.g. Harvey *et al.,* 2006) have used similar models to represent decadal timescale interactions between surface water and ground water.

HETEROGENEITY AT THE BEDFORM SCALE

Although regional modelling can provide broad scale understanding of flow pathways, aquifer heterogeneity, in particular fissure flow, and 'hot-spots' of river-aquifer exchange flows may provide the critical controls on contaminant transport, and these are not well understood. In particular, the use of spatially homogeneous regional scale parameters such as river-aquifer conductances may not represent these key controls, especially in relation to localised upwelling and downwelling, affecting the length of time for which the water is in contact with mineral surfaces in the hyporheic zone where there are rapid variations in biochemical processes (Sophocleous, 2002).

The importance of high permeability deposits in controlling 'hot-spots' of high flow rates was investigated in a theoretical study at Newcastle University (McAleer, 2006). An artificial section of a hypothetical river bed was constructed in the laboratory using combinations of medium and high permeability sediments (Figure 4a). A spatially uniform water input from below the sediments was generated to represent a constant baseflow, and a salt solution introduced instantaneously into the flow. The breakthrough of the tracer in the surface water (representing the river water) was measured using an electrical conductivity (EC) meter, and the resulting breakthrough curves analysed using both an analytical transfer function model and a numerical model (Figure 4b). A series of experiments were carried out with different ratios of areas of high to low permeability sediments, the high permeability sediments representing 'hot-spots' of preferential exchange flow between groundwater and surface water. A key finding was that the percentage of tracer mass that flowed through the 'hot-spots' exceeded 90% when only less than 10% of the river bed area comprised of high permeability sediments. This result demonstrates the danger of making assumptions of heterogeneity when considering the potential of bed sediments for contaminant attenuation, and the need to consider small-scale heterogeneities even in large scale models.

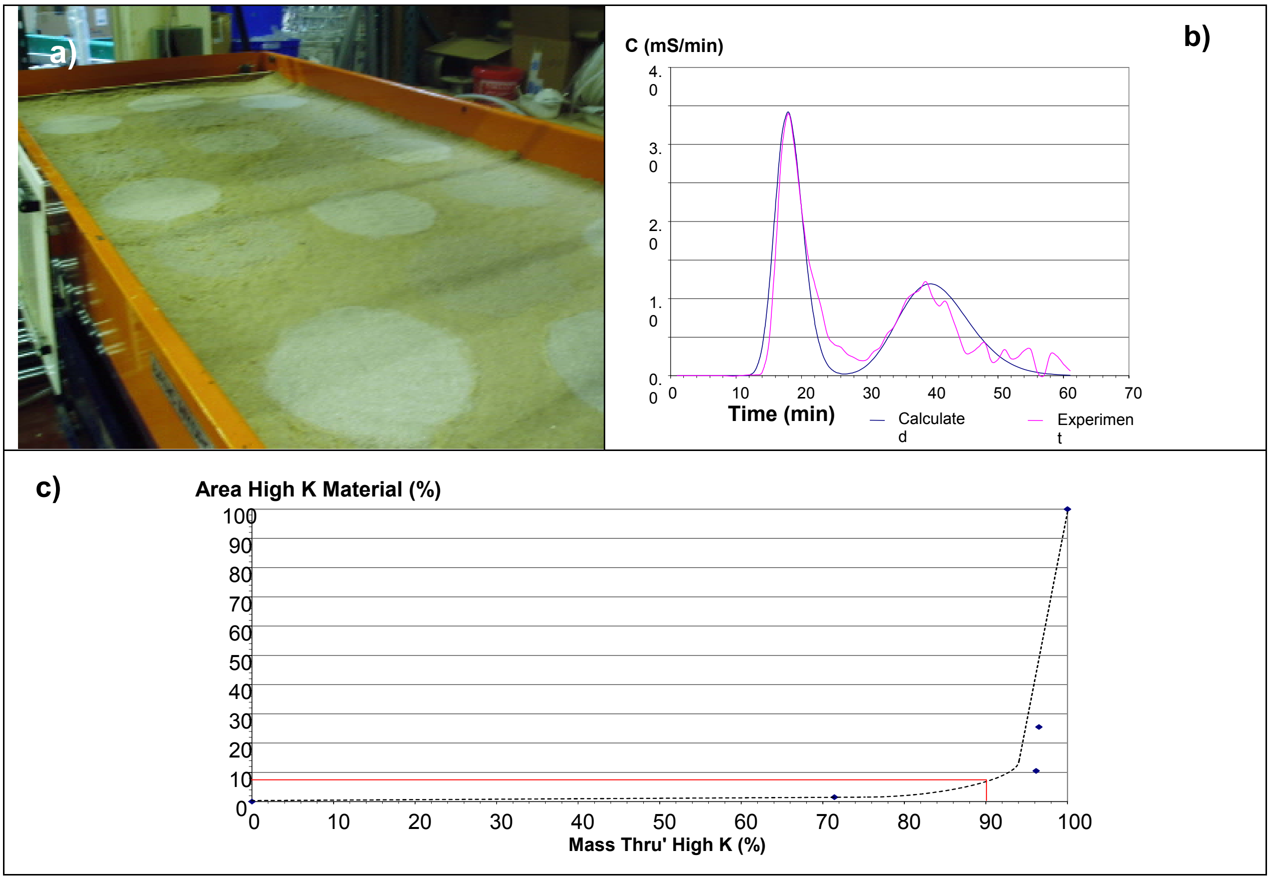


Figure 4: Laboratory and modelling experiments on breakthrough of conservative substances through hypothetical stream bed structures: a) experimental apparatus, showing arrangement of high permeability 'patches'; b) graph of transport flux against time, showing bimodal breakthrough rates; c) relationship between fraction of area of high permeability sediments against percentage of transport flux passing through these sediments

CONCLUSIONS – TOWARDS A FUNCTIONAL INTEGRATED MODELLING TOOLKIT

The Hyporheic Handbook (Environment Agency, 2009) identifies a number of recommendations for further research needed to develop the understanding and capabilities to support management needs, covering areas including (amongst others): the significance of groundwater – surface water interactions and feedbacks (e.g. cumulative effects of aquatic plant growth on sedimentation and nutrient attenuation) at the catchment scale; short time-scale system responses, including the role of flow variability and extremes; long time-scale responses, for example morphological changes; ecosystem health and the factors influencing it, particularly related to microbial and invertebrate benthic ecosystems; and human impacts in both rural and urban environments. More specifically in relation to the theme of this paper, the report highlights the need for better baseline data, rapid monitoring tools that can characterise hyporheic zones at detailed spatial and temporal resolution over a range of spatial scales, and dynamic models that incorporate changing boundary conditions and temporal property changes, as a result of internal system feedbacks.

Over the last decade, there have been significant changes in perceptions of the water environment, with establishment of the view that surface and groundwater resources must be managed in an integrated way. Over a similar period, there has been an increasing recognition that the interfaces between environmental compartments, which were previously treated separately, are critical parts of the overall hydrological system. However, despite massive advances in computer power over the last decade and the development of increasingly sophisticated user interfaces which make model use far easier than before, models which can represent these multi-disciplinary river-aquifer interaction processes at appropriate scales have been relatively slow to develop, and most models which are used in support of environmental management are based on process representations at this critical interface which are essentially unchanged from those of 10 or 20 years ago.

A new functional modelling tool (or toolkit) could be envisaged to move beyond some of these limitations of existing models, providing capabilities driven by needs of environmental managers, and based on an integrative systems approach following some key principles:

- A consistent multi-disciplinary conceptual basis;
- Usable with minimal data in ‘conceptual’ mode, but models to accept multi-scale spatial data as it becomes available or is required for management decisions;
- Output variables which incorporate the effects of natural variability in the environment at an appropriate scale;
- A flexible model structure that allows different levels of detail to be represented which are appropriate to a particular management issue;
- Consistency between (approximate) representations of local-scale processes in large scale models, and corresponding small-scale models;
- Feedbacks to model structure, internal boundary variables, and parameters, from long-term morphological changes;
- Outputs including water balances and hydrological responses, and substance storage and residence times, at any scale.

Development of such a modelling system would be a major undertaking, but even during development would be likely to provide new challenges and insights into the multi-scale process representations required for effective environmental management.

ACKNOWLEDGEMENTS

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GROUNDWATER DEPENDENT ECOSYSTEMS – A SMALL EXHIBITION

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ABSTRACT

This paper provides some elements of the ecohydrology of groundwater dependent ecosystems. First, attention is paid to a functional approach of the ecosystem concept. According to this, the ecosystem derives its character from a combination of local factors and its location in the regional hydrological system. The appreciation of this is important for the management needed for conservation, as through the Habitats Directive and Water Framework Directive. It is shown that a colloquial, correlative linkage of cause and effects may not always hold true for specific cases. Examples given include the effect of a lowering of groundwater tables through increased nutrient availability to the vegetation, and a description of the influence of groundwater on chemical site characteristics. A last example concerns the importance of long time series in order to assess the normal occurrence of extremes in the local groundwater situation and species responses.

INTRODUCTION

Rather than providing an in-depth treatment of one theme in the field of groundwater dependent ecosystems, this paper is intended as a small and incomplete, yet appetising exhibition lining up several elements of an ecohydrological approach of the matter. Groundwater dependent ecosystems rely on groundwater for several reasons. Most directly, the groundwater supports site wetness and supplies water to plants and animals. Groundwater also often has a pronounced influence on the chemistry of water and soil, governing the availability of nutrients and potentially toxic substances. Where groundwater discharges, flow rates are an ecological factor in themselves and they control site temperature and several other things that matter for organisms. Although certain floral and faunal assemblages are almost exclusively found in groundwater dependent ecosystems, they may frequently find their requirements satisfied by alternative environmental fabrics. So how do ecologists know they are groundwater dependent in the first place?

Present understanding of ecosystems is largely based on heuristic studies of the correlation between observed flora and fauna with intuitively described or measured site characteristics, such as ‘wetness’, average, minimum or maximum groundwater levels, soil or water pH, hardness, or, increasingly nowadays, “full” chemical analysis of water and soil. Little attention is paid in many such studies to the processes at work, the hierarchy between factors, differences between variables and parameters, and between state and rate factors. Moreover, in the absence of relevant timeseries, the ecosystem is often implicitly considered ‘stable’ within a small range of variation, biologically as well as physically and chemically.

The mere classification of ecosystems or habitat types as groundwater dependent on the basis of such studies, while a useful hint of the possible importance of the groundwater system for them, does not warrant the understanding needed for the effective management of the relevant groundwater systems in our naturally changing and man-changed reality. Hence it is paramount to answer the question whether and how in any real, local ecosystem of concern the groundwater part ‘works’ (or should ‘work’) in favour of the conservation of habitat types and species.

In this paper I will shortly present a conceptual model of an applied ecosystem, the ecodevice model. The merits of this model are (1) its functional, rather than just descriptive definition, and (2) making the functional approach consistent with physical science in the fundamental distinction of driving forces, conditional parameters and operational fluxes.

Based on this conceptual model, I will then argue that any reliable prediction of the response of vegetation to groundwater levels requires an account of the hydraulic parameters of the relevant ecodevice. The actual responses provided in some important cases suggest that changed nutrient rather than water fluxes between soil and vegetation determined the main impact on the favourable state of conservation.

My next point is to demonstrate what makes a groundwater dependent ecosystem stand out hydrochemically and how there are several, hydrologically very different pathways to make that sensible to species at ground level. A practical model is provided to evaluate the hydrochemical attribute in the context of the water cycle.

The last point I want to stress is the decisive importance of time series of local observations, yielding a really different perspective. I will use the Pollardstown Fen case for illustration of the length researchers went there to locally derive biological requirements as well as to provide time dependent references for hydrological observations. It will be obvious that a long journey still remains before us, but I hope you have got the appetite to take the challenge!

THE ECODEVICE AS A CONCEPTUAL MODEL

Even though ecosystems may be attributed intrinsic value, the concept of intrinsic value uses no standards by which a groundwater dependent ecosystem can be attached *more* intrinsic value than, say, a groundwater independent dryland. However, in today's context, the label groundwater dependent ecosystems apparently flags machines legally protected in order to contribute to sustainable biodiversity. Their state of conservation is *extrinsically* valued, against standards provided in legal documents. Hence they can perform better or worse, functionally succeed or fail, while ecosystems just vary. Van Leeuwen (1982) and Van Wirdum (1982, 1986) therefore call them *ecodevices*: ecodevices are ecosystems that can fail in regard of what we wish to maintain, manage and protect them for. Any debate about success or failure of an individual ecodevice touches on definitions, standards, measurements and their functional interpretation. The ecodevice model was conceived such as to equate the functional interpretation to the generalised physical approach of transport processes by defining driving forces, device coefficients or conditions, and operational fluxes: Flux ≡ Driving force times coefficient.

In view of an explicit spatial representation of ecodevices, the driving forces are said to be located in the environment around the device, the so-called ecological field (Figure 1), where they are measured in gradients of concentrations. The driving forces perceived can vary with time and with any device's position in the field. As long as an ecodevice can maintain fluxes of any essentials into and out of plants and animals within the range between the minimum required and the maximum tolerated, the device is successful in protecting them (Figure 2). There is great freedom to adapt the definitions of fields and devices to specific cases investigated.

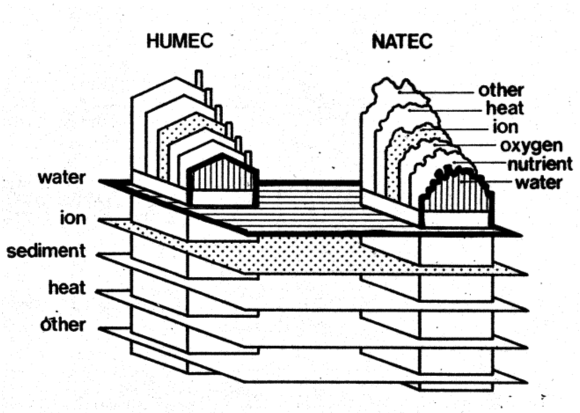


Figure 1:
Two ecodevices, the *humec* serving humans and the *natec* serving biodiversity of nature, operated in the same ecological field, shown here with gradients in water, ion, sediment, heat and other concentrations. The fluxes through the *humec* and the *natec* differ according to their different positions in the field and their different device coefficients.

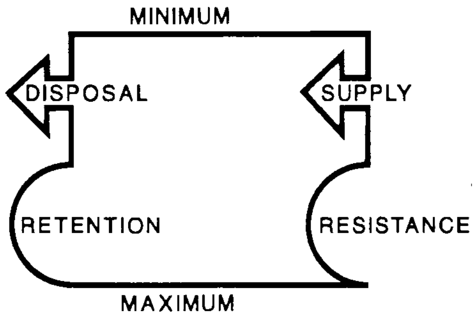


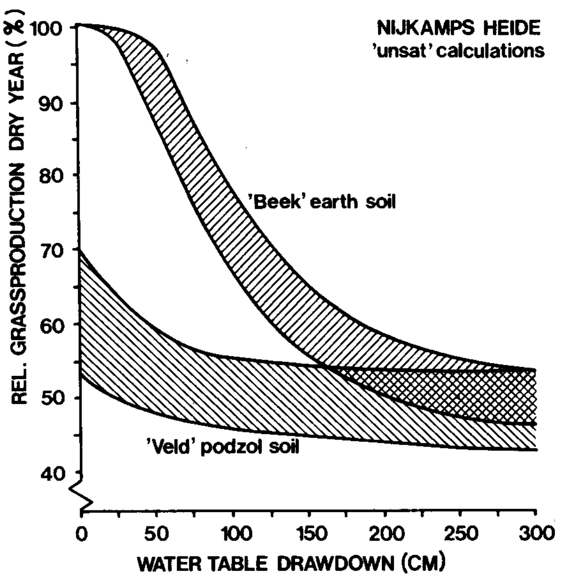
Figure 2:
Ecodevices serve living organisms by controlling input and output fluxes between the ecological field and their living sites between the minimum (required) and maximum (tolerated) levels: their *juste milieu*.

For managers, it is important to be aware that failure may result from device defects, as well as from unexpected variance of the driving forces. Indeed, extreme variance of these may destroy the ecodevice. Note, that a home (Greek: oikos, providing the root for all eco-words) is just about the best known example of an ecodevice. Ecodevices can be very small or very large, and they may vary much in complexity. Their service capacity for biodiversity increases as their complexity naturally increases with time, e.g. when a canal becomes silted up and overgrown. Once a problem is diagnosed according to the conceptual model of ecodevices in an ecological field, further investigations can take advantage of that as will be shown in the following chapters.

FLUXES OF WATER AND NUTRIENTS TO WILD FLORA IN ECODEVICES

Even today the calibration of the occurrence of species of wild flora against characteristics of the groundwater regime, such as 8-year mean groundwater levels, 15- or 85-percentile levels, or average spring maxima is practiced in attempts to replace more intuitively formulated wetness indicator scales, such as Ellenberg's. Such practice is prone to confuse driving forces for wetness with plant received wetness. It led to debates in The Netherlands about the question whether or not a, say, 5 or 10 cm lowering of average groundwater level would lead to water supply shortage for semi-natural grassland vegetation, as many conservation ecologists said.

Agricultural specialists and hydrologists in the 1970s disagreed with this on the basis of soil moisture and water flux modelling. From such models it was immediately apparent that plants do not directly observe groundwater level unless they or their roots are drowned or run dry. In fact, water supply is a function of groundwater levels and soil hydraulic properties, and actual water use also depends on the capacity of roots to grow further down as soil dries. A detailed study of this in a small nature reserve in the eastern part of The Netherlands showed, as a first result, that the same groundwater lowering in one soil type, a 'veld' podsol soil, would have a clear effect on water use by the same vegetation, whereas it would not in another, 'beek' earth soil at 50 m distance (Figure 3, Bannink & Pape (internal report 1979), Van Wirdum 1981).



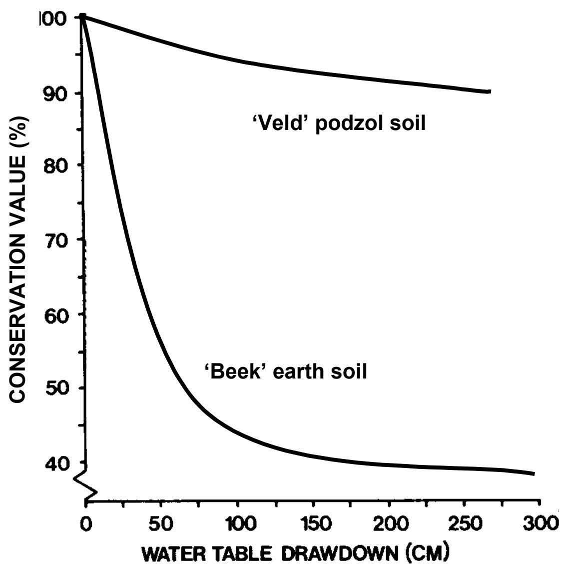


Figure 3:
Effect of water table drawdown on the supply of soil water through the unsaturated root zone, expressed as production of a standard grass vegetation in a dry year (10 % probability of occurrence). Bandwidth of calculations with parameters sampled within 10x10 m^2 squares. The overall effect in the relatively wet earth soil is bigger, but it only starts when drawdown exceeds ca. 25 cm.

Figure 4:
Expected effect of water table drawdown on the conservation value of the local vegetation, judged from similar, actually dewatered sites elsewhere. Indicator analysis and changed productivity of the sites suggest that the initial effect in the earth soil is caused by increased availability of nutrients, followed by drying out, leaching and acidification of the soil at further dewatering.

This was counterintuitive in view of the general experience in conservation ecology that semi-natural grassland vegetation on 'beek' earth soils was much more sensitive to even very little dewatering. While not accurately measured, the species changes and performances during the initial lowering suggested a response to increased available nitrogen, rather than to drying, probably due to exposing more organic material in the topsoil to oxidation. The second result from the same study then was that the lack of any water supply effect actually was a prerequisite for the nitrogen boost to become sensible to the vegetation. Little more than a hypothesis, needing more firm analytical and observational proof, this rationale became a corner stone of ecohydrology in The Netherlands. In the legal context of the European directives, it is important for sustainable land-use development to provide further validation and quantification to this for any individual protected area possibly influenced.

THE WATER CYCLE AS A FRAMEWORK FOR CHEMICAL GROUNDWATER DEPENDENCE

Site chemistry is a measurable attribute of groundwater dependent ecosystems. Gibbs (1970) provided evidence that river waters worldwide reflect the groundwater systems their waters are derived from. A comparison of surface water and groundwater samples in The Netherlands (Van Wirdum, 1991, Van Wirdum, Den Held & Schmitz, 1992), further tested with other European and world-wide data, suggests that much of the ecologically relevant variance in wetland ecosystems can be explained by changes in the water composition as the water travels from the atmosphere through the ground (lithosphere) and through rivers towards the oceans. I based the LAT (litho-atmo-thalasso; thalassa being Greek for ocean) framework on this. My first approach to a graphical ordering of water analyses, using electrical conductivity, calcium and chloride concentrations, after all appears very closely related to Gibbs' earlier work. In addition to these data-poor methods, I also defined a similarity coefficient to assess the similarity between natural waters on the basis of a more complete analysis of major ions in solution. Extreme members of the three main types in the water cycle were chosen to arrange individual water samples in regard of these main types (Figure 5). This method has proven valuable in diagnostic studies as well as in trend analyses and in visualising the interplay between different sources of water to a wetland ecosystem. A detailed description is given in Van Wirdum (1991).

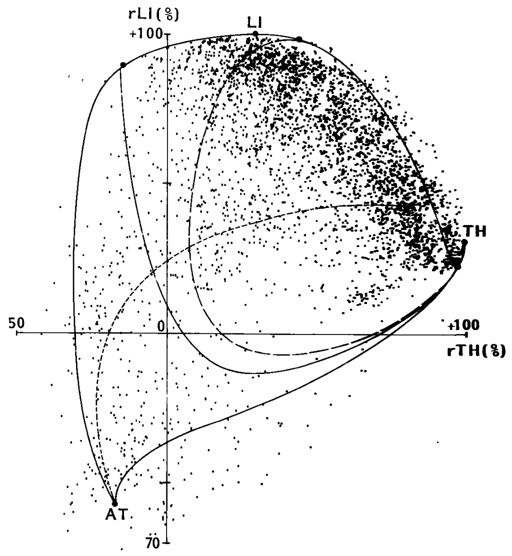


Figure 5:
Arrangement of 5000 mainly surface water analyses from nature reserves in The Netherlands according to their chemical similarities (electrical conductivity, pH, 7 major ions) to seawater (rTH, x-axis, from –50% to 100%) and calcareous groundwater (rLI, y-axis from –70% to 100%). AT, LI and TH: benchmark analyses representing rainwater, calcareous groundwater and seawater, respectively. The broken line around the dense cluster of points in the right half of both graphs marks a 20% admixture of polluted Rhine water. The full line left of that marks a 90% admixture of rainwater.

Unpolluted, natural water analyses would be mainly expected in the area between AT (rainwater) and LI (calcareous groundwater) in the graph. From Figure 5 it is apparent how much the supply of Rhine water as a water management strategy in The Netherlands has suppressed that natural, chemical character of groundwater and surface water dependent ecosystems in The Netherlands, in favour of a dominance of polluted Rhine water. This process has mainly taken place since 1950, and only recently this situation is somewhat improving again.

It should be well understood that different mechanisms can lead to a similar chemical signature of the water. Hence, a dominance of lithotrophic water could indicate upwelling of groundwater from deeper layers, as well as local, recent groundwater in calcareous soils, or a strong inflow of clean river water. Indeed, the standard freshwater composition in Northwest Europe bears this character. Several sites known by botanists as characterised by upwelling groundwater in fact appeared to owe their special character to a downward flow of water, reaching the site from rivers, canals, or upslope springs. In fact, in many cases such a mechanism would slow down a succession towards rainwater dependent ecosystems more than would a local upwelling. In all these cases, however, the present method provides information about the presence of a chemical mark somehow left by groundwater influence, either active and local, or rather by superficial ingress or even only due to a calcareous local deposit. The actual mechanism can often be found by gradient measurements in the ecological field around the relevant ecodevice. Temperature and electrical conductivity are often suitable quantities, which can be more efficiently measured with soil probes than actual chemical concentrations can (Van Wirdum 1991). Of course, many other chemical differences exist, which are not directly addressed by this method and should be considered a next step in the data analysis.

LONG-TERM VARIATION OF WATER REGIMES IN A GROUNDWATER DEPENDENT ECOSYSTEM AT POLLARDSTOWN FEN (IRELAND)

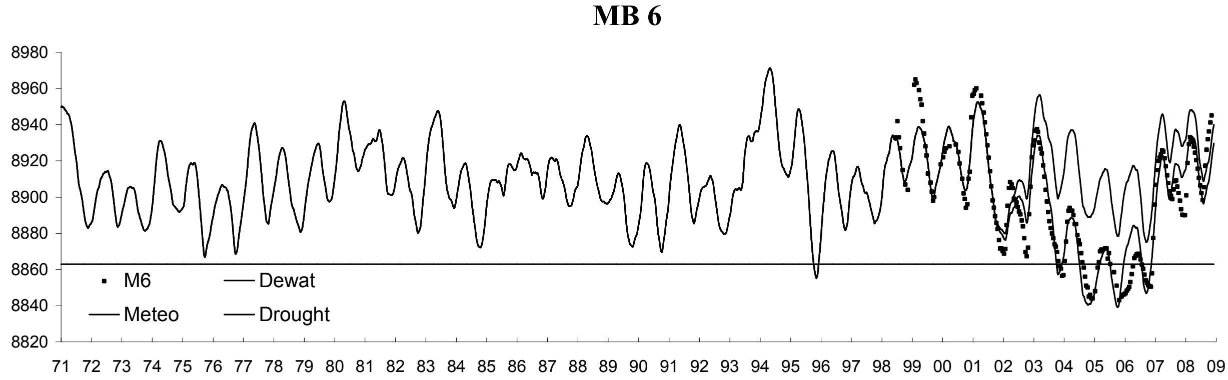
Detailed studies were made in Pollardstown Fen (Kildare, Ireland) in view of the possible effects of temporary construction dewatering for the Kildare Bypass. The investigations, final reports of which will become available in the course of 2010, were made by a large group of hydrogeologists and ecologists (several reports, WYG 2010). Pollardstown Fen is a peatland receiving discharge from the Curragh aquifer, largely via springs and diffuse seepage areas. The study design roughly followed the ecodevice scheme. The groundwater regime in the Curragh aquifer was studied as the main aspect of the ecological field. A fen interface study was made to find out more about the transfer of water between the aquifer and the sloping southern margin of the fen, where the most sensitive flora and fauna occurred. From this study, it appeared that the local geology and surface slope determined where springs and seepage slopes were located. With changing groundwater levels, the fluxes through them would change, but their locations would remain within the same 'predetermined' zone.

The most important result from the study of local, phreatic regimes and their influence on flora and fauna was the strong association between certain plant species in a short, open vegetation structure, with the rare and sensitive Geyer's whorl snail and with an extremely small variation of phreatic groundwater level in the 1997-2002 period. Apparently, if the potential groundwater level in the aquifer sufficed, phreatic level would never sink more than a few centimeters below ground level. Since these sites were on a slope, groundwater levels would never substantially exceed ground level either.

Observations of hydraulic potential in the aquifer since 1997 have shown a relative drought from late 2003 to late 2006. Due to the coincidence of a transition from relatively wet to relatively dry weather with the possible influence of construction dewatering, it was impossible to prove and quantify the latter on the basis of the groundwater level observations. The uncertainty in the results obtained with the regional Kildare Aquifer Model, which was initially developed to predict possible impacts, and therewith serve as a reference to compare observations to, also remained too big to accurately assess these. However, with statistical time series analysis of groundwater level observations, in conjunction with weather and dewatering data, it was possible to estimate a contribution of construction dewatering.

When the drought, probably together with a changed grazing regime, appeared to have contributed to a local vegetation shift and withdrawal of Geyer's whorl snail on the most upslope part of the southern margin of the fen, the question arose whether or not this drought was related to dewatering and whether it was historically unique. Only a relatively short series of historical observations of groundwater levels, plant species and snails was available as a reference. These series, however, are interesting enough. Especially a comparison of botanical data since 1979 showed variation in wetness indication well before construction dewatering started and provided a new view of the dynamics of the fen.

The statistical analysis of observed groundwater levels with meteorological and dewatering data made it possible to estimate a no-impact reference for comparison. It was thus possible to make a rough estimate of the long-term fluctuations of groundwater levels. However, due to the special nature of the fluctuations of the phreatic regime at the fen margin the estimations for this regime are less reliable. The statistical analysis is in fact forced to attribute the low water levels to dewatering influences, but continuation of the monitoring would be needed to see whether the model continues to perform well.



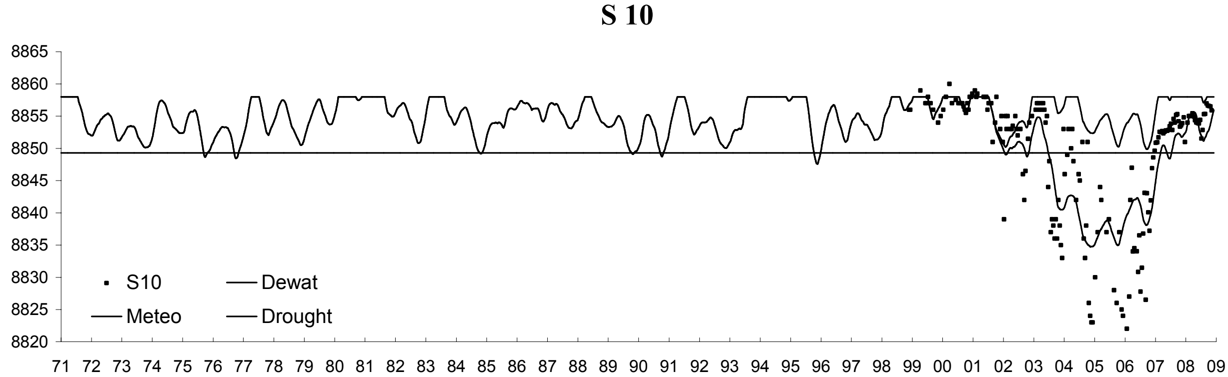


Figure 6:
Long-term groundwater regimes in the aquifer South of Pollardstown Fen (MB 6) and near the surface of the fen margin on site A (S 10; run-off level added as a boundary condition). Regimes explained in time series models (v9r0) with weather variation only (upper curves) and taking account of cutting dewatering (lower curves) are shown together with the observed groundwater levels (dots). The numerically determined drought threshold, shown as a straight line in each graph, was used in risk evaluation.

Although fluctuations are clear from the graphs in Figure 6, it is also apparent that the groundwater level in the aquifer during 2003-2006 was not sufficient to maintain the usual phreatic regime on the uppermost part of the fen. According to the statistical analysis, this would not have been the case if there had not been dewatering. This only happened in this particular, most elevated location and thus provides an explanation for the ecological responses observed. Groundwater levels have since recovered. The botanical investigations do indicate that there is some recovery of the vegetation, but the optimal conditions for Geyer's whorl snail had not yet returned when the programme finished late 2008. It appears that the shift in vegetation is more persistent, and perhaps even in part caused, due to a changed grazing regime, whence experimental cutting has been applied, of which the results are presently unknown to me. All in all, the temporary hydrological effect from dewatering has been underestimated in 1999, but no measurable permanent hydrological effect is expected. It is therefore expected that ecological recovery will follow, providing it is not impeded by other changes, such as the diminished grazing.

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MULTIPLE FLOODING MECHANISMS, ENNIS, CO. CLARE, NOVEMBER 2009

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Mary Burke, Senior Executive Chemist, Clare County Council

ABSTRACT

On the days around 19th November 2009 Ennis town witnessed severe flooding due to a prolonged period of heavy rain followed immediately by three days of intense rain. In addition to surface water flooding from the River Fergus, the town and its environs experienced flooding from multiple sources including karst areas, combined sewers, flood plains, tidal affected and flash floods. This paper describes the why, where and how of the multiple flooding mechanisms that were observed during the November 2009 flood event and in particular, describes the flooding from the town's karst features.

INTRODUCTION

The River Fergus flows through Ennis, Co. Clare and outfalls to the Fergus Estuary at Clarecastle through a tidal barrage. En route from Ballyallia Lake (located 2.5 km north of the town centre), the River Fergus intercepts flows from numerous sources including Lough Girroga, Drumcliff Springs, the Claureen River, Cahercalla Stream, the town's surface water network, the River Gaurus, Flannan's Stream, Ballybeg Lough and the backdrains at Clare Abbey, Cappahard, Bunnow and Skehanagh flood plains. During November 2009, following an extremely prolonged period of heavy rain, each of these sources were in flood and both separately and combined, directly and indirectly, flooded large areas of the town and its environs.

The Ennis Main Drainage and Flood Study Report, which was prepared by J B Barry and Partners and WYG Ireland Ltd. Consulting Engineers on behalf of Clare County Council, identified that large areas of Ennis town were at a high risk of flooding from the River Fergus and its tributaries. Studies were undertaken, in conjunction with University College Galway Hydrology department and K T Cullen & Co. Ltd., to estimate the flood risk and to propose flood protection works for the town and its environs. Following this report, a flood protection scheme for the town centre was designed by JBB and WYG on behalf of the OPW; Phase 1 of which has been constructed and Phase 2 is due to commence on site later this year. Feasibility Studies are currently being prepared by Hydro-Environmental Ltd. for the remaining flood risk areas outside the town centre.

While the Flood Study Report had identified the potential flooding mechanisms, however the magnitude and the prolonged duration of the flooding which was observed during the November 2009 flood was, in places, unexpected. This paper discusses the November 2009 event and briefly explains the flooding mechanisms with particular emphasis on the karst features flooding.

RIVER CATCHMENTS

In order to understand the flooding in Ennis during November 2009, it is first necessary to define the catchments that drain into the Ennis area. The estimated drainage catchment areas and the typical subsoil conditions are shown in Table 1.

Table 1: Drainage Catchment Areas

River / Tributary	Estimate Catchment Areas – km^2	Comment
Fergus (u/s of Ballycorey)	564.3	Drains numerous large lakes and generally karst (limestone) areas (much of the Burren) but also some Namurian (shale) areas
Claureen	55.2	Drains Namurian (shale) Areas
Gaurus	27.7	Drains Limestone Till Area with lakes
Ballybeg Lough	4.8	Karst Area with lakes
Flannan's Stream	4.7	Karst Area with on lake
Lough Girroga	4.6	Karst and Limestone Till Area, with lakes and flood plains
Bunnow	2.4	Flood plain (karst)
Cahercalla Stream	2.2	Urbanised and karst area
Skehanagh	1.0	Flood plain (karst)
Town Centre and Other Areas	9.7	Mostly Urban and Suburban with areas of flood plain.
TOTAL	**676.6**	**Catchment Area Upstream of Clarecastle Barrage**

AVAILABLE DATA

RAINFALL DATA

The drainage areas' rainfall data for the period leading up to the November 2009 flood was gathered from the weather stations at Kilmaley (Claureen River), Carran (River Fergus) and Crusheen (River Fergus, Lough Girroga and Gaurus). The total monthly rainfall for the three stations was 492mm, 439mm and 365mm respectively with the average calculated at 432mm. Table 2 presents the rainfall data for the immediate period before the peak flood event demonstrating the intensity of the rainfall that led to the severe flooding.

Table 2: Rainfall Data

Location	Month total, mm	12th to 19th November 09		17th to 19th November 09	
		Total, mm	Average, mm	Total, mm	Average, mm
Kilmaley	492	209.4	26.2	126.4	42.1
Carron	439	170	21.2	104.9	35
Crusheen	365	152.6	19.1	92.5	30.8
Average	432	177.3	22.2	107.9	36.0

Figure 1 presents the average total daily rainfall for Fergus and Claureen River Catchments based on the records from Kilmaley, Carran and Crusheen rain gauges. The peak rainfall event occurred on the 18th November when an average of 40.2mm of rain fell (note: the peak rainfall for Carran of 49.3mm occurred on the 17th November).

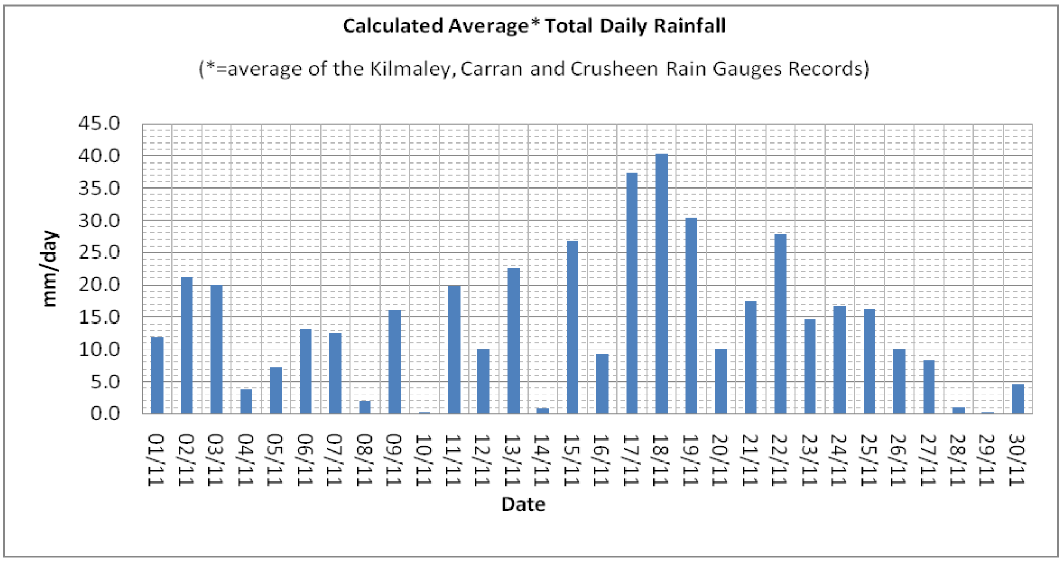


Figure 1: Average Total Daily Rainfall for the Fergus and Claureen Catchments (Nov 2009)

Drumcliff Rain Gauge (Ennis Town) rainfall data from 13th Nov to 16th Dec is presented in Figure 2 below. For the period between October 31 and November 27 468mm total rainfall was recorded at Drumcliff. 42mm and 41mm of total rainfall were recorded on the 18th and 19th November respectively which resulted in significant surface water flooding in the town centre. The 20th November, however, proved to be a sunny day with only 3mm total rainfall recorded.

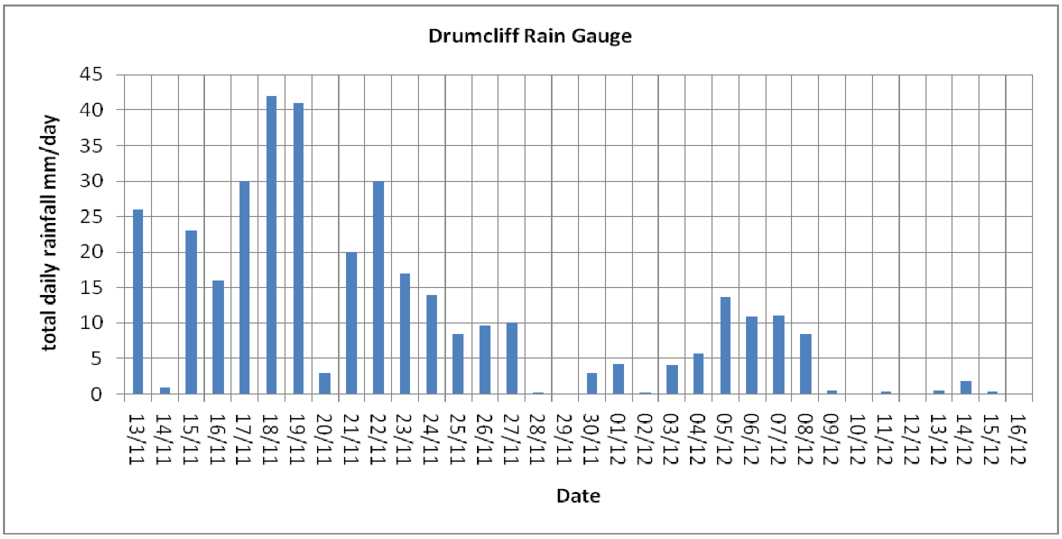
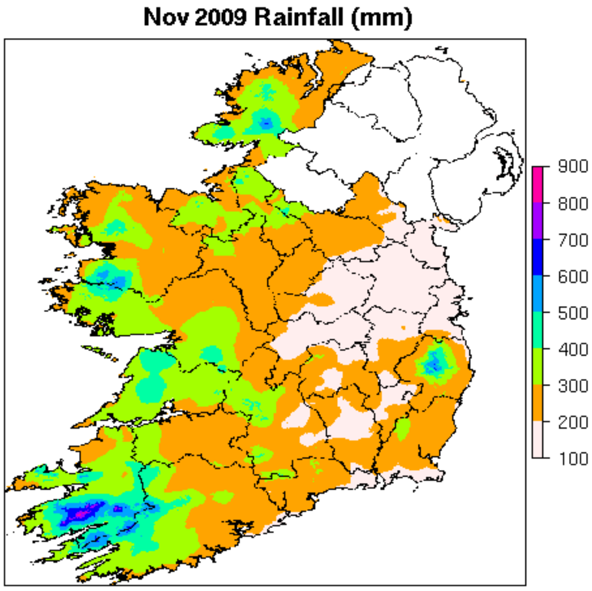
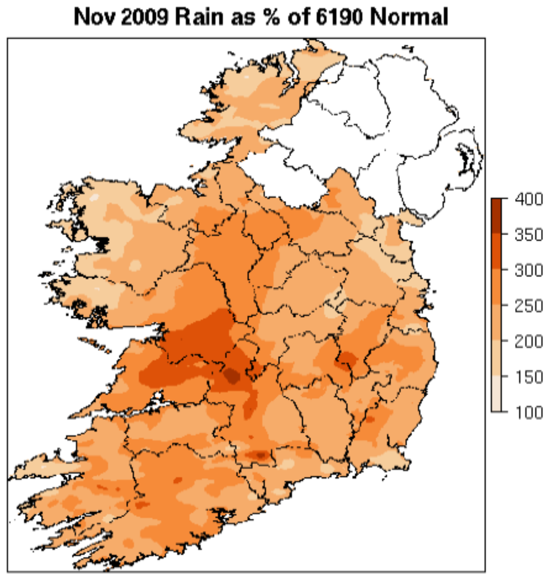


Figure 2: Total Daily Rainfall for the Ennis Town (13th Nov to 16th Dec)

The following maps (taken the Met Éireann report on the November 2009 rainfall) of the November 2009 rainfall show that the catchments draining to the Fergus experienced 300mm to 500mm total rainfall or between 200% to 300% normal November rainfall amounts.





TIDAL DATA

In 1954, under the District of Fergus Act 1943, a tidal barrage was constructed at Clarecastle, the Lower Fergus was embanked along much of its length and a flood plain at Doora and Gaurus, with an area of c. 100 ha, was designated as an area reserved for flooding. During high spring tides the barrage gates close preventing direct tidal flooding of the low lying areas in Ennis while forcing river flows to be stored in the Lower Fergus embanked channel and associated flood plains. When tide levels drop sufficiently the gates re-open allowing the Lower Fergus to drain down. The duration of the gates closure can be prolonged by southerly gales, storm surge and low pressure in the Fergus estuary. Conversely, northerly gales can reduce the tide level and shorten the duration of gate closure.

The mid November highest spring tide occurred on the 18th November with a calculated maximum level (excluding storm surge and atmospheric conditions) of circa 3.0mOD. It is interesting to note that the peak rainfall days coincided with the spring tides. Figure 3 presents the calculated high tide levels at Clarecastle Bridge based on the Foynes Harbour Tide Tables (including an approximation that Clarecastle Bridge high tide levels are 0.9m higher than Foynes).

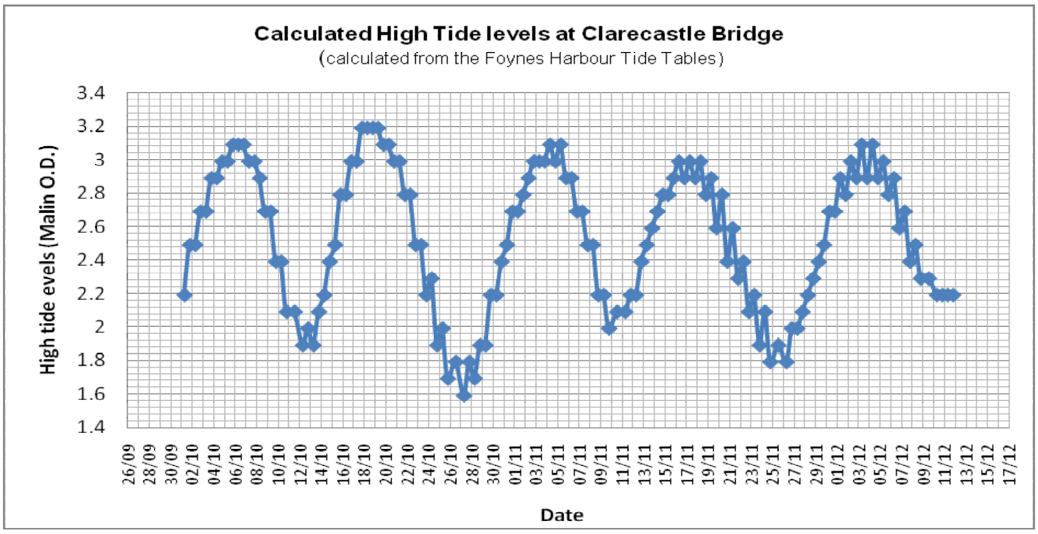


Figure 3: Calculated High Tide Levels at Clarecastle Bridge (late Sept 09 to mid Dec 09)

RIVER GAUGE DATA

There are a number of river gauges in the Ennis Area with the principal ones located at:

- Ballycorey Weir (River Fergus Upper)(AR);
- Inch Bridge (Claureen River) (AR);
- Victoria Bridge (River Fergus Major) (SG);
- Club Bridge (River Fergus Major) (AR);
- Knox's Bridge (River Fergus Major) (SG);
- Doora Bridge (River Fergus Lower) (SG);
- Staff gauge at Clarecastle Bridge (Fergus Estuary) (SG).

(AR=automatic recorder, SG= staff gauge)

The OPW hydrometric section has a telemetry link to the Ballycorey, Club Bridge and Inch Bridge river gauges. Figures 4 - 6 show the water level (based on staff gauge datum) for the three sites during November.

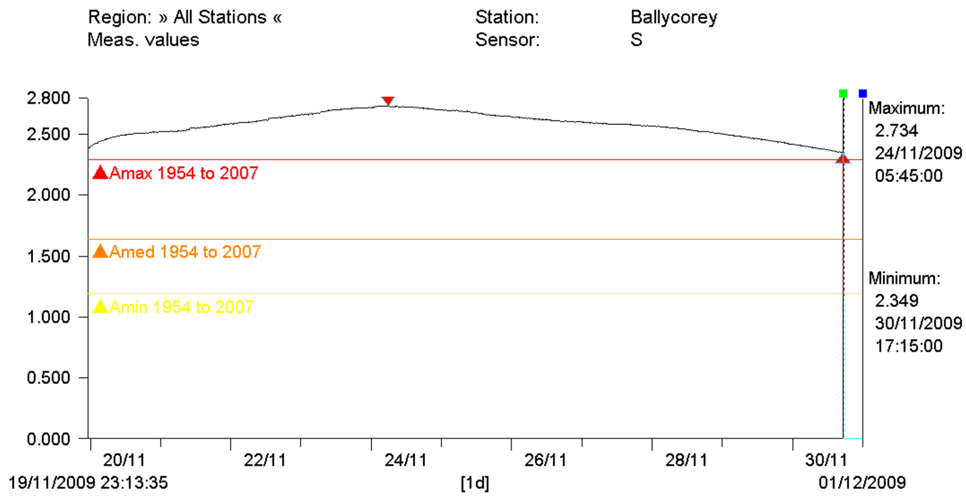


Figure 4: Ballycorey Gauge (20th to 30th Nov) (peak level on the 24th November)

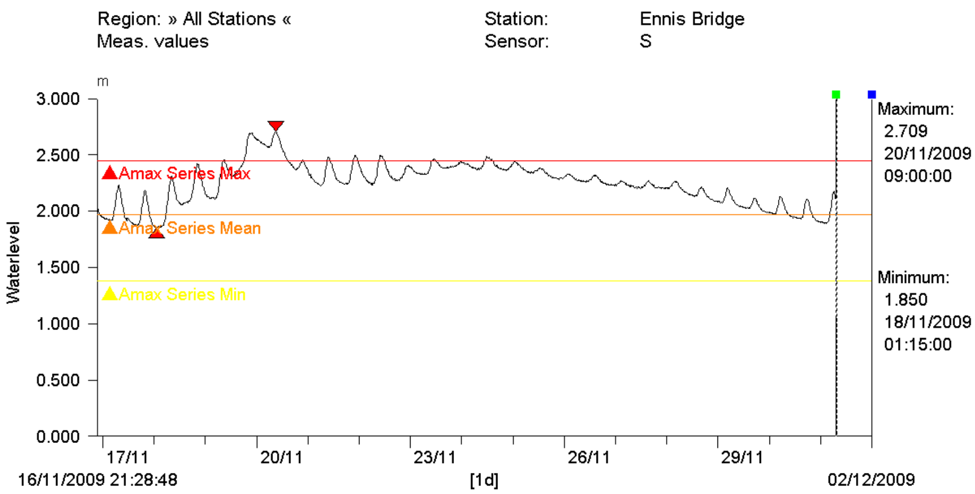


Figure 5: Club Bridge (Garda Station) Gauge (17th Nov to 2nd Dec) (peak level on the 19th and 20th November)

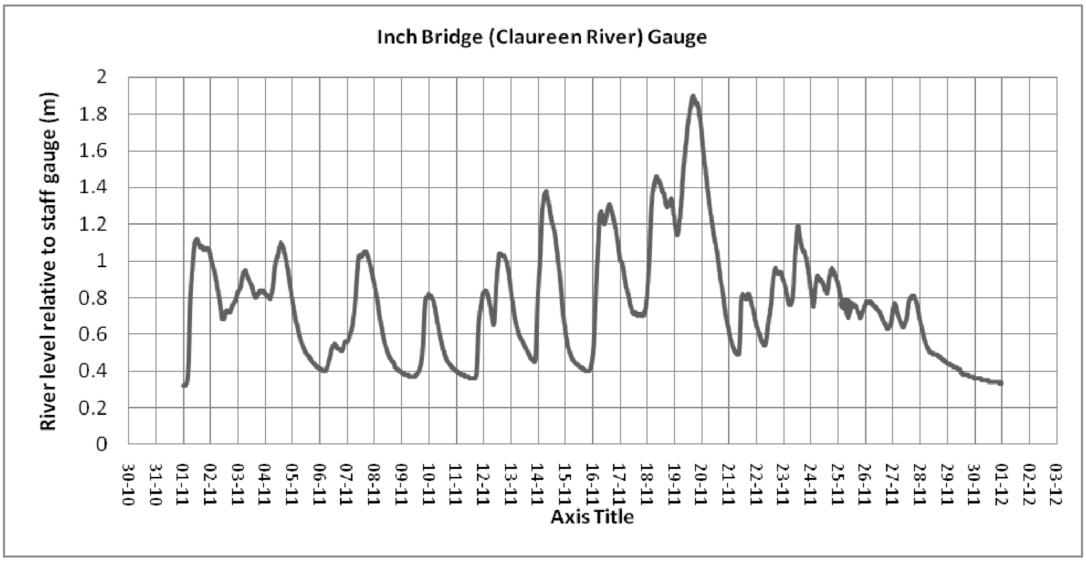


Figure 6: Inch Bridge Gauge (30th Nov to 1st Dec) (peak level on the 19th November)

Following the 18th November Ennis Town Council kept twice daily records of the levels at the various river gauges. A summary of peak flood gauge levels recorded during the flood event is given in Table 3.

Table 3: Peak flood Gauge Levels Recorded

Location	Flood Level, mOD	Date
Ballycorey Gauge	7.08	24th November
Claureen Gauge	11.39	19th November
Club Bridge Gauge	3.68	20th November
Knox's Bridge Staff Gauge	3.0	19th November
Doora Bridge Staff Gauge	2.8	19th November

During the flood event the effect of the tide / river levels in the Lower Fergus was most apparent at Doora Bridge, Knox's Bridge and Club Bridge while upstream at Bank Place Bridge the effect was negligible. It was reported that the level of Lough Girroga was observed to be fluctuating by as much as 40mm between high and low tide which may suggest a subterranean connection exists between the lake and the Lower Fergus.

FLOOD PROFILE FORECASTING

During the flood emergency response the gathered data was relayed to WYG and Hydro-environmental Ltd. who in turn used this data, rainfall data and gauge rating curves to forecast the prevailing flood flows. Then, by using the calibrated hydraulic river model (prepared for the Ennis Flood Study), this flood flow data was used to accurately forecast the flood profile for the town. This profile was then compared with the town's topographical data which allowed high flood risk areas to be identified which greatly assisted the co-ordination of the local authority's emergency flood alleviation works.

OBSERVED FLOOD LEVELS

Table 4 lists some of the flood levels in the town and environs based on observations and anecdotal evidence.

Table 4: Observed flood levels

Location	Level	Flood Source
Gort Road (Industrial Estate)	Circa 7.25mOD	Lough Girroga
Fíor Uisce (Gort Road)	>4.1mOD	Fergus Minor
Kevin Barry Avenue Bridge	>3.6mOD	Fergus Minor
Drumcliff Road	>7.0mOD	Drumcliff Springs and environs
Gort Road (Elm Park)	>5.7mOD	Fergus / Ivy Hill Turlough
Claureen Bridge, Lahinch Road	>6.2mOD	River Claureen
Watery Road	>5.7mOD	Fergus / Claureen
Cusack Lawn, Lahinch Road	>5.6mOD	Fergus / Claureen / Sewers
Mill Bridge / Circular Road Lower	>5.0mOD	Fergus / Sewers
Springfield Orchard (Harmony Row)	>4.1mOD	Fergus / Sewers
Bindon Lane	>3.9mOD	Fergus / Sewers
Abbey Street Lower	>3.75mOD	Fergus
Newbridge Road	>3.5mOD	Fergus / Sewers
Castlelawn	>2.8mOD	Fergus / Sewers
Saint Flannan's College	9.0mOD	Flannan's Stream
Saint Flannan's Drive	>9.7mOD	Flannan's Stream
Ard Aoibhinn housing estate, Limerick Road	>6.5mOD	Flannan's Stream
Killadysert Cross / Ballybeg Road	>4.5mOD	Ballybeg Lough
Abbey Ville housing estate	>2.7mOD	ClareAbbey Floodplain / Lower Fergus

ESTIMATED FLOOD FLOWS

The Ennis Flood Study Report estimated that the 1 in 100 year (design) flood flow for the River Fergus (main channel) through the centre of Ennis at circa 92 cumec. On 29th November the OPW hydrometric section measured and calculated the flow at Ballycorey Gauge at 68.0 cumec when the gauge level was 2.43m. This data was compared to the existing gauge rating curve and it was found that the curve would underestimate flows. A provisional revised curve has been used for this study.

Using the river gauge data and the rational method for the ungauged catchments (considering the catchments to be fully saturated due to the prolonged wet spell) to calculate the average flow for the karst areas, the average flood flow through Ennis on the night of the 19th and morning of the 20th November 2009 has been estimated at 90.0 cumec with circa 10 cumec flowing in the Fergus Minor and the remainder, 80.0 cumec, flowing in the Fergus Major through the centre of town. The average flow in the Lower Fergus was estimated at 112.8 cumec. Table 5 below presents the estimated average flood flows.

Table 5: Estimated Average Flood Flow on night 19th / morning 20th November 2009

Source	Estimated Average Flood Flow (cumec)
Fergus (u/s of Ballycorey)	*63.2*
Lough Girroga	*2.2*
Others (including Drumcliff Springs Area)	*0.8*
Claureen River	*22.8*
Cahercalla Stream	*1.0*
(Fergus Minor) Bifurcation of Fergus u/s of town centre	*-10.0*
Average Flow Through Town	***80.0***
Flow Through Town	*80.0*
(Fergus Minor) Confluence with Fergus d/s of town centre	*10.0*
Town Catchment	*3.9*
River Gaurus	*12.9*
Flannan's Stream	*2.2*
Ballybeg Lough	*2.2*
Bunnow	*1.1*
Skehanagh	*0.5*
Average Flow to Clarecastle Barrage	***112.8***

The flood flow through the centre of town, which occurred at approximately 6:30pm on 19th November, is estimated at 83.9 cumec as summarised in Table 6.

Table 6: Estimated Peak Flood Flow at 6:30pm on 19th November 2009

Source	Estimated Peak Flood Flow (cumec)
Fergus (u/s of Ballycorey)	*60.5*
Lough Girroga	*2.2*
Others (including Drumcliff Springs Area)	*0.8*
Claureen	*31.4*
Cahercalla Stream	*1.0*
(Fergus Minor)	*-12.0*
Peak Flow Through Town	***83.9***

Whereas the majority of the flood sources peaked around the 19th and 20th November the River Fergus at Ballycorey did not peak until the 24th November with an estimated flow of 83 cumec. On comparison of the rain gauge (totally daily rainfall) and river gauge (average daily level) data it was identified that, following a period of prolonged heavy rainfall:

- The Claureen River at Inch bridge responds to short duration rainfall events (1-day or shorter) with a lag of approximately a day; and
- The River Fergus at Ballycorey Bridge responds to longer duration rainfall events (approximately 6-days) with a lag of approximately 5 to 6 days.

The fact that the Claureen River was estimated to have contributed 20% of the average flood flow to the total flow (to Clarecastle Barrage) during the November flood, while only been 8% of the overall catchment area, demonstrates the significance of this flashy river in relation to flooding in Ennis.

If the Claureen River peak had coincided with the River Fergus peak the flood flow through Ennis town centre would have been in the order of 100 cumec.

The peak flow, relative to the Ballycorey and Inch Bridge gauges only, is estimated to have occurred on 23rd November and at 93.4 cumec while the peak flow on the 19th was estimated at 91.9 cumec. Fortunately for Ennis the peak flow coincided with the neap tides. Figure 5 (Club Bridge Gauge) clearly shows the river flow peak on the 23rd and 24th Nov. Figure 7 shows the estimated flows and total flow between 19th November and 1st December.

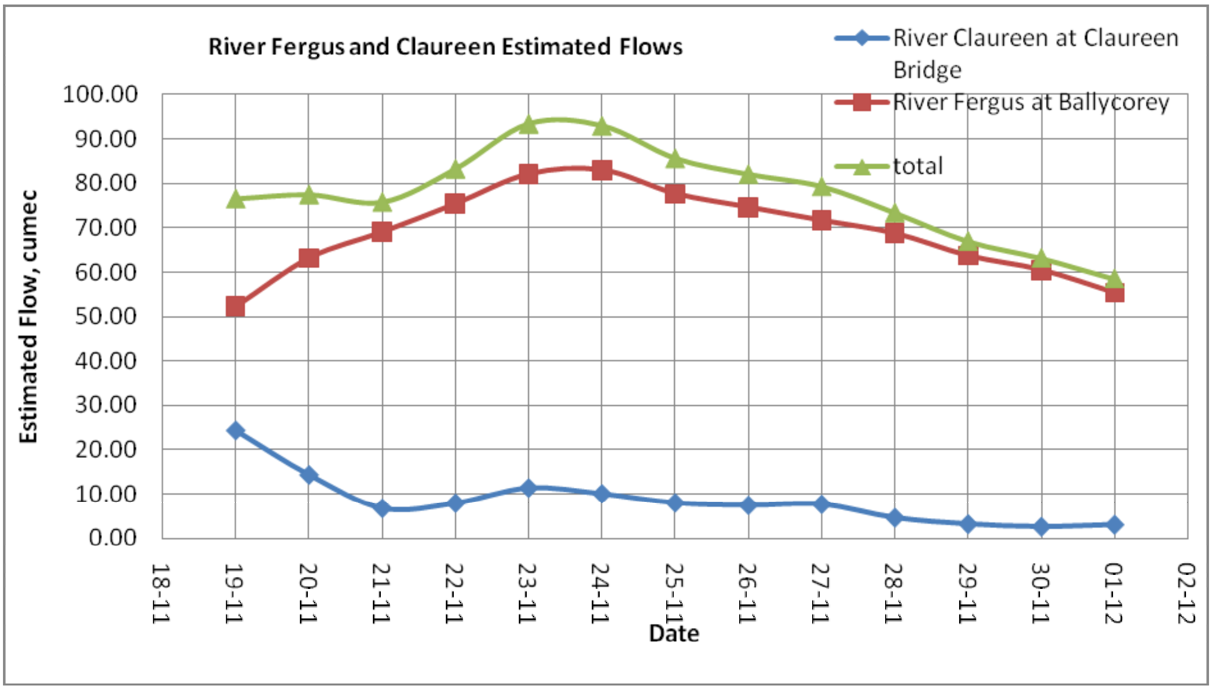


Figure 7: Estimated River Fergus and Claureen flows from 19th Nov to 1st Dec 09.

FLOODING

The pattern of flooding that occurred in Ennis during November 2009 was complex due to the multiple flooding mechanisms, the varying flood durations and the wide spread nature of the flooding. The flooding mechanisms identified include:

- High river levels overtopping river banks;
- High river levels breaching or leaking through river walls;
- Backflow through unchecked surface water outfalls;
- Flooding of low lying areas from the combined sewer network;

- Springs / groundwater flooding;
- Turlough flooding;
- Swallow hole flooding;
- Lower Fergus / Tidal Affected Section: overtopping and breaching embankments and flooding through faulty sluice gates.

WHERE AND HOW DID THE FLOODING OCCUR?

Starting at Ballycorey Weir and continuing downstream flooding occurred at the following locations (that exhaustive) for the following reasons.

Location	Reason for flooding
Auburn Lodge	High river levels in the Fergus
Gort Road Industrial Estate	High river levels in the Fergus and Lough Girroga overtopping the Gort Road
Aughanteeroe and Fíor Uisce	High river levels in Fergus Minor and back flow through surface water sewers
Watery Road and Elm Park / Gort Road	High river levels in the Fergus, Ivy Hill turlough overtopping the Gort Road, back flow through surface water sewer network and flooding from the combined sewer
Drumcliff Road	Ground water flooding (Drumcliff Springs)
Brookville and Cloughleigh	High river levels in the Claureen
Cusack Lawn and Lahinch Road	High river levels in the Fergus and back flow through surface water sewer network
Mill Wheel	Leakage through old river walls
Mill Road	Back flow through surface water sewer network and flooding from the combined sewer
Bindon Lane	High river levels in the Fergus and flooding from the combined sewer
Abbey Street Car park	Leakage through old river walls and back flow through surface water sewer network
Abbey Street Lower and Francis Street	High river levels in the Fergus overtopping the river bank and flowing to low lying streets and flooding from the combined sewer
Newbridge Road, College Road, and Coláiste	Leakage through old river walls and back flow through surface water sewer network, turlough flooding, combined sewer flooding and ground water flooding
Garda Station and Cusack Park (GAA)	High river levels in the Fergus
Fergus Park	Overtopping and leakage through embankments and back flow through surface water sewer outfalls
Francis Street Pumping station	Backflow from storm overflow due to high river levels which in turn prevent the effective operation of storm pumps, flooding from combined system and surface water flows from the town centre
Castlelawn	Overtopping and leakage through embankments, back flow through surface water sewer network and flooding from combined system
Cappahard	Back flow through surface water sewer network, ground water flooding and Overtopping and leakage through embankments. Cappahard Lodge was flooded from the combined system.
Clonroadmore	Faulty sluice gates and leakage through embankments
St. Flannan's Drive, College	High river levels and swallow hole flooding resulting to surface water flows through college
Ard Aoibhinn and Toberteascain	Surface water flows from swallow hole flooding, surface water sewer flooding and ground water flooding
Abbeyville	High flood levels in the floodplain due to faulty sluice gates and high river flows
Killadysert Cross	Swallow hole flooding and combined sewer flooding

WHY THE FLOODING OCCURRED?

Following a wet summer and a prolonged unsettled and heavy rainfall period starting in mid October and continuing through November 2009, the river catchments draining to the River Fergus were saturated resulting in high water levels, high water tables and full flood plains. From the 11th November the weather became increasingly wet with average total daily rainfall amounts for the Fergus Catchment exceeding 20mm/day and peaking at 42mm/day on the 18th November.

The River Fergus, which responded at a slower rate to the exceptional wet spell due to a number of factors including attenuation of flows through upstream lake-land and turlough area and slow discharge from the Burren karst area, was already by the 18th November close to matching the maximum flood levels recorded at Ballycorey.

The Lower Fergus river levels and flood plain water levels were remaining high due to a combination of the high river flows and the high spring tides, which peaked on the 18th November. The exceptional wet spell on the 18th and 19th led to flash flooding from the both the shale and smaller karst river catchments (i.e. run-off was high due to the saturated catchment). The combination of these flood events led to the extreme flood levels in Ennis town on the evening of the 19th November.

The Fergus flows continued to rise until the 24th November when the maximum level of 2.734m was recorded at Ballycorey Weir. Fortunately this peak flood flow coincided with the month's neap tides and less wet days (note: still greater than 10mm/day) and therefore did not cause a repeat of the extreme flood levels experienced in the town centre on the 19th and 20th November.

FLOODING FROM KARST FEATURES

Whereas the most extensive flooding occurred directly as a result of high river levels either overtopping banks, leaking through old river walls and embankments or backing up sewers, some of the most prominent flooding that occurred in Ennis during November 2009 was due to relatively low flows from flooded karst features such as turloughs and swallow holes. The flooding 'why, where and how' of some of the karst features is discussed below.

KARST FEATURE FLOODING: LOUGH GIRROGA

Lough Girroga, which is located to the north of Ennis, drains an estimated catchment area of 4.6 km2 that includes large areas of low lying flood prone lands at Ballymaley and Ballyduff which are interconnected by small streams and ultimately discharge to Lough Girroga. The lough is situated in an enclosed depression (bounded by the Gort Road (Old N18), Limerick to Galway Railway line, Our Lady's Hospital complex and drumlins) with no definite surface water outlet and is reported to discharge slowly through swallow holes to the River Fergus.

During periods of prolonged rainfall, which results in flows to the lough that exceed the flow capacity of the swallow holes, the lake floods the depression to a level typically less than 6.5mOD. Following the exceptional rainfall events of November 2009 the lake levels continued to rise to levels in excess of 7.0mOD before overflowing the Gort Road and flooding through the Gort Road Industrial Estate en route to the River Fergus at the rear of the estate. The average flood flow was estimated at 2.2 cumec.

This flooding resulted in the closure of the Gort Road (formerly the national primary route (N18) between Limerick and Galway), temporary closure of the industrial estate businesses and nearly forced the ESB to shut down the substation that supplies much of North Clare. Figures 8 & 9 show the normal lake level and November 2009 flood; whilst Figure 10 shows the flooding across the Gort Road.

Figure 8: Lough Girroga Depression with lake in background (24th March 2010)

Figure 9: Lough Girroga Depression in flood (21st November 2009)

Figure 10: Gort Road and Industrial Estate flooded (21st November 2009)

KARST FEATURE FLOODING: FLANNAN'S SWALLOW HOLE

The Flannan's Stream, which is located to the south of Ennis, has a catchment area of 4.7 km2 and drains to a swallow-hole located in sport grounds to the rear of St. Flannan's College. The swallow-hole has been proven (by KT Cullen as part of the Ennis Main Drainage Report) to be connected to springs located 880m east, at Toberteascain, via an underground karst system, which in turn discharge to Clare Abbey floodplain.

Following relatively short durations of heavy rainfall the capacity of the swallow capacity is quickly exceeded resulting in the Flannan's Stream backing up and flooding the lands in the vicinity of the swallow-hole. During times of exceptional rainfall the flood waters leak through and overtop the sport grounds boundary wall and flows through the college grounds to a low lying area in front the college at the Limerick Road Boundary wall where it slowly discharges to groundwater.

During the period of 18th to 21st November 2009, when the average flood flow in the Flannan's Stream was estimated at 2.2 cumec, the flood water levels rose behind the Limerick Road wall and eventually dramatically cascaded over it before crossing the Limerick Road and flowing downhill through the Ard Aoibhinn housing estate. From there, following cleaning of gullies and opening of manhole covers on the storm sewer line, the flood waters drained towards Toberteascain where they flooded up out of manholes and in places erupted through the road surface, before flowing through properties en route to the Clare Abbey flood plain.

This flooding resulted in the closure of the St. Flannan's Drive and the Limerick Road (formerly the national primary route (N18) between Limerick and Galway), closure of the college for several days, flooding of houses at Honeywell and Ard Aoibhinn, shutting down of an ESB substation, flooding properties at Toberteascain and damage to roads. Figures 11 & 12 show the swallow hole and Limerick Road wall under normal conditions and Figure 13 shows the Limerick Road wall being overtopped during November.

Figure 11: Saint Flannan's College Swallow Hole (24th March 2010)

Figure 12: Saint Flannan's College Limerick Road Wall (24th March 2010)

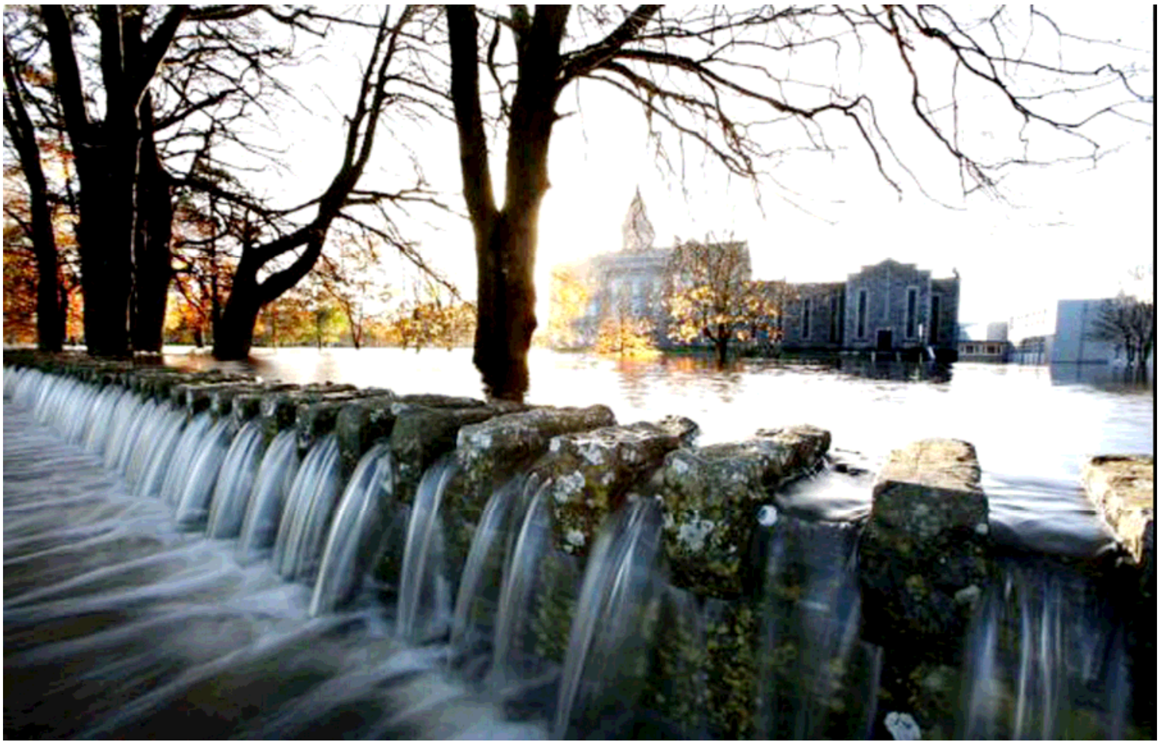


Figure 13: Saint Flannan's College Limerick Road Wall being overtopped (20th November 2009)

KARST FEATURE FLOODING: BALLYBEG LOUGH AND IVY HILL TURLOUGH

Flooding was also recorded at Killadysert Cross, located to the south of Ennis, when the flood flow from Ballybeg Lough exceeded the capacity of its associated swallow hole at Limerick Road. Flooding in this area is a regular occurrence following prolonged wet spells resulting in flooding of a small number of houses, flooding of the foul sewerage network in the area and the closure of the Killadysert Road.

Exceptional flooding was recorded during November 2009 at Elm Park and Ivy Hill on the Gort Road, located to the north of the town centre, due to a combination of flooding mechanisms including high river levels backing up a trunk surface water sewer, combined sewer flooding and flooding from

Ivy Hill turlough. Normally the flood level of the turlough is controlled by an overflow to the Elm Park trunk surface water sewer, however due to the high river levels in the Fergus this overflow was ineffective and resulted in the turlough level continuing to rise until it overflowed the Gort Road and flooded through Elm Park. This flooding resulted in the flooding of approximately 40 houses, the closure of the Gort Road and Drehidnagower Road and the flooding of a main lift combined sewer pumping station. Figure 14 shows the Ivy Hill turlough under normal conditions.

Figure 14: Ivy Hill turlough under normal conditions (25^{th} March 2010)

CONCLUSION

The flooding that was experienced in the Ennis area during November 2009, following a prolonged period of heavy rain followed immediately by three days of intense rain, was due to a multiple flooding mechanisms including high river levels, high tide levels, sewer flooding and karst feature flooding. While flooding from the main river channels was predictable, the extents of flooding from the karst features was unexpected and resulted in flooding of the Gort Road industrial estate, housing estates, St. Flannan’s College, the closure of main roads and disruption to electricity supplies.

Phase 1 of the Ennis Flood Alleviation scheme has recently been completed and successfully protected much of Ennis town centre during the November floods. Phase 2 of the scheme is programme to commence on site later this year. Further to recommendations given in the Ennis Main Drainage Report and recently prepared feasibilities Studies the local authority propose to undertake emergency flood alleviation works for Lough Girroga, Elm Park, St. Flannan’s and Killadysert Cross.
It is recommended that a flood early warning system be put in place for Ennis to allow the local authority to organise the necessary emergency flood alleviation measures and resources promptly with a view to mitigating the impact of future flooding in the town.

REFERENCES

Ennis Main Drainage and Flood Study Report: JBB and WYG

A Flood Study of a Tidally affected Town (2001): C. Cunnane (UCG) and A. Cawley (HEL)

Report on Rainfall of November 2009 (Feb 2010): S. Walsh (Met Éireann)

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SESSION III

LANDSPREADING OF ORGANIC FERTILIZERS AND SETBACK DISTANCES FROM GROUNDWATER SOURCES

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Environmental Protection Agency

ABSTRACT

The landspreading of organic fertilizers poses a threat to groundwater in certain physical/hydrogeological settings, particularly with regard to microbial pathogens and nitrate. Protection of drinking water sources from these pollutants is necessary. Arbitrary fixed radii are commonly used internationally in delineating planning and protection zones to aid decision-making on the location of potentially polluting activities. They are a good 'first step' in delineating a protection area, but if a large distance is chosen, they are not always scientifically defensible. A risk-based approach to delineation of landspreading exclusion zones using readily available or obtainable hydrogeological information is recommended in this paper. The most important hydrogeological factor is the groundwater vulnerability, as represented by the permeability and thickness of the subsoil. In addition, the design and construction of wells are important factors that need to be taken into account.

INTRODUCTION

The EU Nitrates Directive (91/676/EEC) requires Member States to take specific measures to protect surface water and groundwater from nitrate contamination arising from agricultural activities. The *European Communities (Good Agricultural Practice for Protection of Waters) Regulations,* implementing the Directive, were first introduced in 2006 with amendments in 2007. The Regulations give statutory effect to certain elements of the Nitrates Action Plan. A consolidated regulation came into effect on the 31st March 2009 – *European Communities (Good Agricultural Practice for Protection of Waters) Regulations* 2009 (S.I. No. 101 of 2009) (called the GAP Regulations in this paper). These revised Regulations provide for strengthened enforcement provisions and better farmyard management in order to comply with the European Court of Justice Judgment in relation to the *Dangerous Substances Directive* in the context of phosphorous discharges from farm installations and to provide the legal basis for the derogation under the Nitrates Directive granted to Ireland, as well as including minor technical amendments.

The aim of Article 17 of the GAP Regulations is to prevent pollution of water from fertilizers and soiled water. Article 17 (2) specifies setback distances for application of organic fertiliser and soiled water on land in the vicinity of water abstraction points, such as wells, springs, watercourses and lakes. These distances vary from 25 to 200 m, depending on the daily abstraction amount or number of people served by the source. Article 17 (5) (a) enables a local authority to specify alternative distances following prior investigations and consultation with the Environmental Protection Agency (EPA).

This paper presents recommendations, based on a draft EPA Advice Note, on the implementation of Article 17 (5) as it is applied to groundwater sources. It recommends delineation of landspreading exclusion zones in the vicinity of groundwater abstraction points, rather than the arbitrary radii given in Article 17 (2). A recommended risk-based approach is outlined.

BACKGROUND

POLLUTION THREAT TO WELLS AND SPRINGS FROM LANDSPREADING OF ORGANIC FERTILIZERS

Microbial pathogens and nitrate are the two most widespread pollutants, from a public health perspective, in groundwater. In addition, phosphate in groundwater, while not generally a public health issue, can pose a threat to surface water ecosystems. These pollutants are present in organic fertilizers (e.g. slurry) and soiled waters. However, the threat posed by landspreading, undertaken in compliance with the EC (Good Agricultural Practice for Protection of Waters) Regulations 2009, is generally less than from on-site wastewater treatment systems (OSWTSs) (such as septic tank systems for single houses), farmyards, grazing animals and inorganic fertilizers, for the following reasons:

- The nutrient loadings are substantially less than that arising from grazing animals (15-35% of the total organic loading arises as slurry depending on the duration animals are housed).
- The pathogen loadings are also lower than from grazing animals, not only because their relative input is less but also because some pathogen die-off occurs in slurry pits and dungsteads.
- Organic fertilizer is spread evenly in a thin layer (approximately 3-5 mm) over the land surface, thereby facilitating uptake of nitrate and die-off and attenuation of pathogens. This contrasts with the focused loading in urine patches, dung pats and areas where localised spreading of soiled water occurs; or release of effluent 0.5-1.5 m below the surface in the case of OSWTSs.

Nevertheless, organic fertilizers contain nutrients and microbial pathogens and pose a threat to the water quality of wells and springs. The purpose of the requirements of Article 17 is to minimise the risk to these wells and springs. The risk depends largely on well head protection, borehole construction and the hydrogeological settings in the vicinity of the drinking water sources.

ASSESSING RISKS TO GROUNDWATER SOURCES – GENERAL PRINCIPLES

For any particular potential source of pollutants, the following factors influence the risk to a well or spring:

- Zone of contribution (ZOC) (or catchment area) of well/spring, as activities outside of this area cannot pose a threat.
- Proximity to the well/spring, in particular position relative to the Inner Source Protection Area (SI)[1] as microbial pathogens arising outside this area are unlikely to reach the well/spring.
- Groundwater vulnerability (a combination of subsoil permeability and thickness, and the presence of karst features that enable bypassing of the subsoil), as the risk is greatest in extreme vulnerability areas. There are four vulnerability categories: extreme (E), which is sub-divided into outcrop and shallow rock (X) and 1-3 m soil/subsoil (E); high (H); moderate (M); and low (L).
- Hydrogeological properties of the aquifer, as they influence pollutant attenuation and pollutant travel times. For example, groundwater flow velocities are rapid (>5-100s m/d) in karstified limestone aquifers.
- Existing groundwater quality; where the quality is historically poor, it may indicate a well/spring that is susceptible to pollution, and further deterioration must be prevented.
- Well head completion and borehole construction, as these influence whether surface water and shallow groundwater can enter directly into the drinking water source.

THE ROLE OF BUFFER ZONES IN PROTECTING GROUNDWATER SOURCES

Buffer zones or setback distances are a commonly used as a means of reducing the likelihood of impacts on water from human activities. For instance, in circumstances where the soil in the vicinity

[1] Further details are given in DELG/EPA/GSI, 1999. Groundwater Protection Schemes. Published by the Geological Survey of Ireland. Available on www.gsi.ie

of a stream or lake is a gley, the presence of a buffer zone would reduce the impact of landspreading, and the bigger the buffer zone, the lower the risk. Also, in the case of wells, where the soil/subsoil has a low permeability there could be lateral movement of pathogens at the surface or at a shallow depth and then down the outside of casing where construction/grouting is inadequate. Therefore a small buffer zone is justified as a means of reducing the likelihood of pollution in this circumstance. The use and benefits of larger buffer zones will depend on site specific considerations. Where there is a pressure, such as landspreading, in addition to a horizontal setback distance, a critical factor in protecting wells from pathogens is the overlying soil and subsoil, i.e. the vertical dimension or perspective, expressed as vulnerability. Other important factors in reducing the risk to human health are good well design and construction, and water treatment. Consequently, while they have a role, it is recommended that there should not be sole reliance on setback distances in the vicinity of wells and springs.

SETBACK DISTANCES – USE OF ARBITRARY FIXED RADII

The use of "arbitrary fixed radii" around wells/springs is a common approach taken internationally in delineating planning and protection zones to aid decision-making on the location of potentially polluting activities. They have the advantage that they are easy to administer and enforce, require little technical expertise and give some protection. However, they tend to over-protect the land area down-gradient of a well/spring, may under-protect the land area up-gradient of the water supply, they take no account of vulnerability or pollutant attenuation and are difficult to justify scientifically. Yet, in certain circumstances, their use is unavoidable, as the information required for a more scientifically-based alternative may not be readily available. For instance, if the ZOC and Inner Protection Area (SI) have not been delineated, alternative distances are not an option in most circumstances.

If alternative distances are not being considered, the arbitrary fixed radii in Article 17 (2), which vary from 25-200 m, must be applied. Where the abstraction is >100 m^3/d, the required setback distance is 200 m; this represents an area of 12.6 ha. In many circumstances, this area is larger than is required to protect the well/spring from landspreading of organic fertilizers and soiled water. Consequently, it is recommended that local authorities avoid the use of arbitrary fixed radii, except perhaps as a 'first step' or where short distances are being proposed for practical reasons to deal with particular issues.

ROLE OF WATER TREATMENT

Protection of drinking water supplies is an important facet of the Nitrates Directive and the Water Framework Directive and delivery of clean potable water is a requirement of the Drinking Water Directive. These two aspects combine together in the Water Safety Plan approach whereby its objective is to provide 'safe and secure' drinking water. The water safety plan requires that the hazards associated with the catchment, treatment process and distribution system be identified and managed.

A multi barrier approach is required to ensure a safe and secure supply particular with reference to the risk of *Cryptosporidium*. Disinfection as a minimum is required to be put in place for all public groundwater supplies. In cases where there is an insufficient natural barrier (in-situ subsoil thickness that provide protection from *Cryptospiridium*) additional treatment barriers are necessary. The extent of the natural barrier is directly linked to the groundwater vulnerability and the management measures within the catchment. The groundwater vulnerability and the degree of water treatment should be determining factors in the assessment of proposed 'landspreading exclusion zones'.

USE OF ALTERNATIVE SETBACK DISTANCES

Article 17 (5) enables local authorities to specify alternative distances, following prior investigations and consultation with the EPA. The purpose of this Article is to enable prohibition of landspreading in areas where it is, as far as possible, scientifically justifiable. In certain circumstances, the area where landspreading should be prohibited may need to be at a distance from the well/spring in addition to the area in the immediate vicinity of the well/spring; therefore, the term 'landspreading exclusion zone' is considered to be preferable as a concept than 'alternative distance' or 'setback distance', although they are used interchangeably in this paper.

LANDSPREADING EXCLUSION ZONES – INFORMATION REQUIREMENTS

RECOMMENDED APPROACH

In considering applications under Article 17 (5), the EPA has to assess the **risk** to drinking water sources from all potential pollutants arising from landspreading of organic fertilizer and soiled water in the ZOC of abstraction points. Therefore, a focussed **risk-based approach** to considering this issue, using the Source – Pathway – Receptor (S-P-R) framework, is required. The EPA requires assessment of factors related to the pressure or hazard (in this case organic fertilizers including soiled water), the receptor (well or spring) and the pathway(s) for potential pollutants to the abstraction point. Information should be provided in a concise form.

PRESSURES

- A brief description of land use.
- Details in relation to the organic materials to be landspread, i.e. the origin of the material (cow/cattle farmyard slurry/manure, soiled water, intensive activities). *[Landspreading of organic fertilizers from intensive agricultural activities (e.g. piggeries) and industrial and municipal sludges may pose a greater threat than farming practices where the organic fertilisers generated on the farm are recycled, i.e. dairy, cattle, sheep, etc.]*
- The likely pollutants present in the materials, e.g. nitrogen, phosphorus, metals, pathogens, including a conclusion on the pollutant/s posing the greatest threat to the abstraction source (usually these will be microbial pathogens and nitrate).
- Information on lands, if present nearby, that have a derogation under the Nitrates Directive.
- Location of areas, if present, used for landspreading of soiled water using irrigation systems.
- An evaluation of the nutrient loading rate from organic materials and other sources of nutrients, such as inorganic fertiliser.

RECEPTOR AT RISK

- Type of receptor – well or spring groundwater source.
- Type of drinking water supply – public water supply, public group water scheme, private group water scheme, small private supply or exempted supply. Where relevant provide details on the water supply scheme code.
- Population served by the water scheme and volume of drinking water supplied per day.
- Summary details on abstraction source – location, well depth and diameter, abstraction rate, depth to bedrock, etc.
- Details on well construction, position of pump and pumping regime.
- Summary of existing relevant water quality data, including parameters that are indicative of contamination, such as nitrate, ammonia, chloride, potassium, *E. Coli*. Where possible, graphs of pollutant concentrations showing temporal variations should be included.
- Details on existing or proposed water treatment.
- Details on the *Cryptosporidium* Risk Assessment Score and Risk Classification, including the individual Catchment Risk Score and the Treatment, Operation and Management Risk Score.

PATHWAYS

The required pathway information will vary depending on the type of receptor being considered, the hydrogeological setting, groundwater quality and likely risk to the source. A list of possible pathway factors for groundwater sources is given below.

- Inner and Outer protection area boundaries, including basis for these boundaries.
- Vulnerability within the zone of contribution (ZOC), including basis for category, i.e. type, permeability and thickness of subsoil.
- Aquifer category.
- Karst features, if bedrock is limestone.
- Soil types.
- Where source is in a sand/gravel aquifer, thickness of unsaturated zone in vicinity of source.
- Summary information on the hydraulic properties of the aquifer.

- Estimates of groundwater velocities in the bedrock aquifer based on estimated permeabilities and hydraulic gradients.
- Assessment of likelihood of denitrification in soil, subsoil and bedrock.
- Conceptual model of zone of contribution.

Information on several of the above factors is available on the GSI website: www.gsi.ie

DELINEATION OF LANDSPREADING EXCLUSION AREA – RECOMMENDED APPROACH

BASIS FOR DECISION-MAKING

The EPA view on the delineation of exclusion areas is influenced by the following factors:

- A risk-based, 'weight of evidence' approach is preferred, as there is usually no definitive scientific solution to delineating exclusion zones.
- Adequate information must be available and provided to enable a good conceptual understanding of the hydrogeological setting (this information should, in any case, be already available to enable proper development and protection of the source).
- Two pollutants – microbial pathogens and nitrate – pose the greatest threat from landspreading.
- Maintenance of the safety and security of drinking water sources, by ensuring the absence of microbial pathogens, is based on taking a dual approach – prevention and water treatment, particularly in relation to *E. coli* and *Cryptospiridium*.
- In areas where nitrate concentrations are relatively high, landspreading of organic fertilizers is not likely to be the main source of nitrate where the requirements of the GAP Regulations and good nutrient planning are followed. In addition, where average nitrate concentrations are >50 mg/l, it is unlikely that prohibiting landspreading, except in the autumn, will reduce concentrations significantly.
- Disposal of soiled water by landspreading (via soiled water irrigation systems) has been shown to cause excessive leaching and localised plumes of high nitrate (Bartley, and Johnston, 2005.) Therefore, it is essential that there is compliance with Article 18 (5) and Article 18 (6) of the GAP Regulations. Also, it is preferable to dispose of soiled water outside the ZOCs of sources.
- Where the hydrogeological setting indicates that there is a low risk from landspreading, minimum setback distances should be based on the standard of the well head protection and the degree of water treatment.

The alternative setback distance in the vicinity of a well/spring and, where relevant, exclusion areas at a distance from a well/spring are based on consideration of three issues, which can be considered independently:

1. Protection from microbial pathogens;
2. Ensuring that nitrate concentrations do not exceed 50 mg/l;
3. The adequacy of the well head construction and protection.

While Article 17 (5) of the GAP Regulations (S.I. 101) allows a local authority to specify alternative distances to those outlined in Article 17 (2), the Regulations do not cater for a local authority introducing other measures that reduce the risk to drinking water supplies. However, other measures (e.g. prohibiting application of soiled water in the ZOC during the prohibited application periods for landspreading of organic fertilizers in Schedule 4 of the GAP Regulations) could be considered, where agreement for such measures can be reached with the landowner concerned.

PROVIDING PROTECTION FROM MICROBIAL PATHOGENS – EPA RECOMMENDATIONS

In circumstances whereby an application for an alternative distance is being considered, the water supplier must provide an assurance to the EPA that the treatment is appropriate and has relevant monitors and alarms in place to ensure that the drinking water supplied is wholesome and clean[2]. The exclusion area for landspreading shall be influenced by the existing groundwater quality:

- Where samples of untreated water contain zero *E. Coli* or are in the range 1-10/100 ml the exclusion area is based on consideration of nitrates and well head construction.
- Where *E. Coli* numbers in untreated water are >10/100 ml in one or more samples, the reasons for the 'gross' contamination should be identified and appropriate exclusion areas delineated. For instance, options include extending setback distances along sinking streams in karstified aquifers or excluding landspreading in outcrop and shallow rock areas (X vulnerability) in the Inner Source Protection (SI) area.
- *Cryptospiridium* may enter groundwater from the land surface in areas of extreme (E) vulnerability. If areas such as this are present in the ZOC, an adequate treatment barrier to *Cryptosporidium* must be installed.

PROVIDING PROTECTION FROM NITRATE – EPA RECOMMENDATIONS

- Where the mean nitrate concentration is <25 mg/l, the exclusion zone should be based on well head construction and providing protection from microbial pathogens.
- Where the mean nitrate concentration is in the range 25-37.5 mg/l with peaks above 37.5 and below 50 mg/l, disposal of soiled water should generally be prohibited in the ZOC during the prohibited application periods for landspreading of organic fertilizers in Schedule 4 of the GAP Regulations. Some exceptions may be acceptable in moderate (M) and low (L) vulnerability areas. The aim of this is to reduce the likelihood of the 50 mg/l maximum acceptable concentration being exceeded.
- Where the mean nitrate concentration is <50 mg/l with peaks above 50 mg/l, the exclusion area should consist of the SI/X area. In addition, no landspreading of organic fertilizers or soiled water should take place in the ZOC after 1st September. The aim of these requirements is to reduce rapid leaching and therefore the likelihood of peak concentrations exceeding 50 mg/l. If agreement on this cannot be obtained from the relevant land owner, consideration could then be given to extending the landspreading exclusion area. However, these measures alone, i.e. use of landspreading exclusion zones, may not be adequate to prevent the drinking water limit from being exceeded and consequently the public health implications must be considered.
- Where the mean concentration is >50 mg/l, other measures are needed to reduce the nitrate concentrations and the public health implications must be considered. As a means of reducing peak concentrations, the exclusion area should consist of the SI/X area. In addition, it is recommended that no landspreading of organic fertilizers or soiled water should take place in the ZOC after 1st September.

SETBACK DISTANCES BASED ON ADEQUACY OF WELL HEAD PROTECTION

The following arbitrary fixed radii are recommended around wells and springs:

- Where well construction can be demonstrated that it was undertaken in accordance with the IGI Water Well Guidelines (2007) and the subsoil thickness is >3 m (i.e. either H, M or L vulnerability), the <u>minimum</u> recommended distance is **10 m**. Where grouting has not been undertaken to the standards of the IGI Water Well Guidelines (2007), but is regarded as 'good', the <u>minimum</u> distance should be **20 m**.
- Where there is >10 m subsoil ('low' or 'moderate' vulnerability depending on subsoil permeability), the <u>minimum</u> buffer would be dictated by the quality of the well construction, i.e. **10-30 m**.
- Where well head construction and protection is inadequate, a 'fallback' arbitrary minimum distance of **30 m** is used; in circumstances where there is a steep slope to a well/spring and the soil has a low permeability, a greater distance is recommended, to reduce the likelihood of polluted surface water and shallow groundwater collecting around the well. A similar distance

[2] The definition of wholesome and clean is provided by Regulation 5 of the European Communities (Drinking Water) (No. 2) Regulations 2007 (S.I. No. 287 of 2007)

is recommended up-gradient of springs, with a reduced distance down-gradient, which should depend on the slope away from the spring.
- The relevance of well head protection is illustrated in Figure 1.

CONCLUSIONS

1. Landspreading of organic fertilizers undoubtedly poses a threat to groundwater in certain physical/hydrogeological settings. Microbial pathogens and nitrate are the two main pollutants arising from landspreading.
2. However, the threat is generally less than that posed by on-site wastewater treatment systems, farmyards, grazing animals or inorganic fertilizers.
3. Arbitrary fixed radii are a good 'first step' in delineating a protection area, but if a large distance is chosen, they are often not scientifically defensible.
4. A risk-based approach to delineation of landspreading exclusion zones using readily available or obtainable hydrogeological information is recommended in this paper. The most important hydrogeological factor is the groundwater vulnerability, as represented by the permeability and thickness of the subsoil.
5. In addition, well design and construction are important factors.
6. In many, though not all, circumstances the resulting landspreading exclusion zones will cover less area than that delineated using the arbitrary radii.
7. The EPA recommends the delineation of scientifically-based landspreading exclusion zones in the vicinity of groundwater abstraction points.

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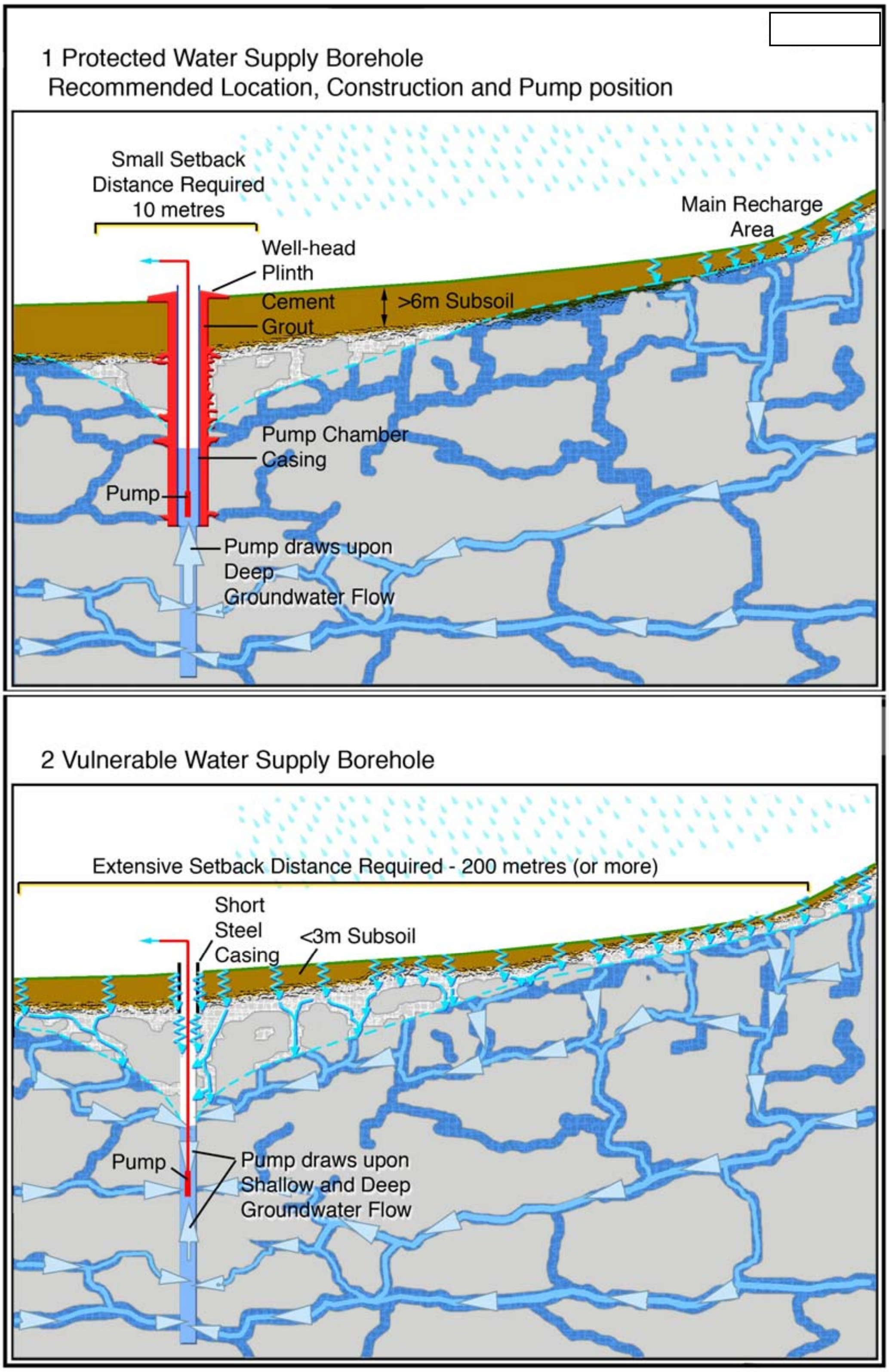

Figure 1: Schematic sections illustrating of the importance of the borehole location and construction in producing wholesome and clean water and in determining the setback distances (Acknowledgement: Both the original concept and the illustrations are by David Ball).

THE ASSESSMENT AND PRIORITISATION OF POINT SOURCES OF GROUNDWATER CONTAMINATION IN IRELAND

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ABSTRACT

Point sources of groundwater contamination are probably more important in the Irish context than previously considered. Anywhere where there has been an intensity of anthrogenic activity there is potential for such a point source. Obviously industrial and waste management sites stand out as particularly good potential examples of this, but it is important to realise that small potentially unregulated sites can cause the same type of problems. A lot of work is needed to address identified issues linked to the groundwater so that the WFD "good status" objectives can be achieved as early as 2015, and to prevent future deterioration of groundwater already at good status.

This paper is designed to provide a concise summary of how best to assess point sources of groundwater contamination. It is something that needs to be done in a consistent, methodical way, building a conceptual understanding of site issues as the work progresses to allow all important risks to be considered (including non-groundwater related ones) and appropriate corrective action to be designed and implemented, as needed, with well defined and achievable remedial objectives and criteria.

One of the most important contributors to point sources of pollution in Ireland at industrial sites is the former use of chlorinated solvents. Their presence in groundwater, sometimes at only trace to ultra-trace concentration, is relatively widespread. There are extreme cases where the scale of the CHC release appears to have been very large (tens of tonnes over many years) and the impact on the underlying groundwater very considerable, sufficient to potentially cause long term deterioration in groundwater quality over many years to decades unless there is intervention, which will be in breach of the Water Framework Directive requirements. Even with corrective action, the time scale for significant improvement can be very long.

INTRODUCTION

The EU Water Framework Directive (2000/60/EC) (WFD) sets three main objectives (i) to achieve good chemical quality status and ensure no deterioration of status, (ii) prevent or limit the input of pollutants and (iii) put in place measures to reverse any significant and sustained upward trends in pollution. The Daughter Directive 2006/118/EC, on the protection of groundwater against pollution and deterioration, established specific measures (criteria) in order to prevent and control groundwater pollution. In Ireland the European Communities Environmental Objectives (Groundwater) Regulations (2010) have just been published and are designed to deliver the requirements of the above at a National level, with measures:

- To prevent or limit the input of pollutants into groundwater and the deterioration of the status of all bodies of groundwater, with further measures to protect, enhance and restore them as needed;
- Establishing criteria and procedures for assessing groundwater chemical (and quantitative) status and procedures for the identification of significant and sustained upward (pollution) trends and how to define (and demonstrate) trend reversal;
- To establish rules for the presentation and reporting of groundwater monitoring results, trend assessment and the classification of chemical status of Groundwater Bodies.

Groundwater status assessments should be undertaken for all Groundwater Bodies (GWBs) identified as being At Risk from contaminated land site pollution. This assessment should be based on a good conceptual understanding of source and contaminant transport within the GWB which should be subdivided into areas reflecting the distribution of chemical pressures and pathway susceptibility.

A GWB can have Good or Poor Status. A number of GWBs (14% of the country's land area) were classed as Poor Status (interim status assessment in 2008). In the case of Good Status this classification is assigned a High or Low level of confidence. The GWB is also assigned an At Risk (AR) status for which there can be High or Low Confidence. In the initial WFD Risk Assessment (2005) only very few site specific point sources of pollution registered as being At Risk (High Confidence) however 295 GWBs were considered at risk in some way due to point source pollution (Daly & Craig, IAH 2009). With improving understanding of the magnitude and extent of site specific groundwater issues, via the IPPC and Waste Management licensing regimes, the level of confidence will increase so that the actions required will become much easier to define.

In principle point sources of contamination may be considered relatively low contributors of the degradation of GWBs, given Ireland does not have a protracted industrial heritage, and because of their localised and finite nature by definition. However the work done in the past 6-12 months has highlighted a high percentage of larger, longer established and/or major chemical use sites as having groundwater contamination, some of which is very significant. The mass of contamination associated with an individual point source can be tonnes if not 1-2 orders of magnitude greater than this. Concentrations can be so elevated, that where the contamination is resistant to degradation etc, then the groundwater impacts can be observed for many decades. Contaminants can be carcinogenic, mutagenic, ecotoxic, etc, and some are volatile so can impact people who live above a plume, particular if close to the source.

OVERVIEW OF APPROACH TO THE ASSESSMENT OF POINT SOURCES OF GROUNDWATER CONTAMINATION

The approach to assessment of point sources of contamination should be consistent and phased. There is little point in rushing out and undertaking detailed investigation unless there are exceptional circumstances (e.g. catastrophic release). The following tasks may usually be needed:

1. Chemical source audit & desk study;
2. Preliminary investigation;
3. Main Phase II intrusive investigation;
4. Quantitative Risk Assessment;
5. Corrective Action (as needed).

Chemical Source Audit & Desk Study: This has to be the place to start. Understanding the operational history of a site, whether it is associated with industrial manufacture, waste management or another anthropogenic activity, is a fundamental building block. There may be exceptional circumstances where no significant site-specific information exists, but this will be very rare. Some of the things you want to find out detailed information on are as follows:

- Site manufacturing operations and chemical use in the process;
- A full inventory of hazardous chemicals (where stored, how transferred, where used);
- A full inventory of hazardous wastes (as above, including possible historic disposal);
- Information on support operations and associated chemical use (e.g., boiler house; power supply; maintenance);
- Site drainage systems and on-site effluent treatment;
- Chemical spill/loss records.

Of critical importance is an assessment of how the above changed with time during the evolution of the site, from original Greenfield status to the current day. With environmental regulation and licensing the environmental security of many sites in Ireland has greatly improved in the past decade or more. Sites that date back to the 1980s and beyond are usually worthy of most scrutiny.

It is important to remember that the type and properties of the chemical as well as the volume stored is important. A small (and perhaps seemingly innocuous) volume of one chemical may be much more significant to understand than the bulk storage of another. You can sense this from the range in target concentrations for groundwater criteria that exist (e.g. Groundwater Threshold Values (GTVs) for boron (750 μg/l) versus mercury (0.75 μg/l). The same applied to organic contaminants, which can be relatively mobile in soil-groundwater systems. Other useful pointers can be:

- Underground storage can be key, not necessarily linked to tank failure;
- Underground transfer pipes, notably linked to fuel supply, can be an Achilles Heel;
- Effluent drains and sometimes storm water drainage inappropriately used can be a pathway leading to secondary (line) sources of release. Floor drains and sumps in and around production buildings can represent a particular (high strength) source;
- The drum storage you see today is often far from that that existed 20 or 30 years ago;
- Peripheral (support) activities can have a significant chemical footprint;
- Chemical needs change with time (production changes; hazardous chemicals get phased out);
- Beware of former “outback” areas of a site or small and old chemical storage/use sites.

A good source audit will lay the foundation stone for a focussed site investigation. It can involve one or more days on-site with identification and interviewing of long-serving and even retired management and employees if needed. Gaining access to detailed old site plans and aerial photographs is considered key.

The Desk Study element is designed to establish the environmental setting and sensitivities of the site and together with the source audit allow a robust initial Conceptual Site Model (CSM) to be developed. It requires the collation and assessment of the standard publically available information on geology, hydrogeology, hydrology, land-use and groundwater (and potentially surface water) users. If possible this should be supplemented with site specific information which may be obtained from geotechnical reports, other site borehole, monitoring well and/or abstraction well data and local site features (streams, topography). At this stage preparation of a good initial CSM is important to complete. It will evolve as more data becomes available which reduces uncertainty and focuses efforts. The undertaking of a preliminary risk assessment (CLR11 type) can have limited value being too broad to deliver a useful level of focus if applied inappropriately. Through this process it should be possible to see the “wood from the trees” and generate real focus to the follow-on investigations.

Preliminary Investigation & Risk Assessment: A site may only need a small level of intrusive investigation to characterise a potential groundwater contamination issue. However for larger and/or more complex sites, a preliminary relatively small investigation serves to gain initial information to provide a step change in understanding of the site condition, focusing the more major follow-on intrusive investigations. Such an investigation would mainly target priority areas of the site (as access allows) and might include:

- Near-surface soil sampling (areas of staining; close to potential sources; background data);
- Soil vapour surveying as a screening tool (PID; FID; compound specific);
- Shallow (perched) groundwater sampling, including some deeper soil sampling;
- Gas monitoring (landfill gas; other);
- Sample from on-site abstraction well(s) if in existence;
- Possible surface water sampling;
- Sample analysis for the identified Chemicals of Potential Concern (COPC).

The above work involves typically a few days on-site up to about one week. It purposely does not investigate to depth for risk of creating preferential pathways for shallow contamination to migrate down. It may target up to 5-10 identified potential sources areas where some level of contamination may be expected, based on source audit findings. The number of samples collected (from all media) is perhaps unlikely to be >15-20, although a range of selective laboratory analyses may be required to characterise the COPCs. Broad suites of chemical analysis can be useful at this stage (e.g. VOCs, sVOCs including PAHs, the so-called Maxi suite). It must always be remembered that potential site specific COPC may not appear on standard suites (this varies between laboratories) and need to be specifically requested, to be reported as TICs or as specific analysis results. Examples of such compounds include THF, alcohols and Chromium 6+.

Monitoring well design must be planned and implemented with care. Too often wells are poorly designed and/or installed and either "fail" after a relatively short period (e.g. silt up), allow cross contamination by having long screened sections, and for similar reasons provide mixed data on water levels and therefore groundwater flow. Subtle vertical head gradients can be crucial to understand. Shallow wells (as proposed here) are usually easier to get right. Deeper wells and particularly ones installed into important GWB need very careful consideration by a qualified and experience practitioner (hydrogeologist).

Monitoring wells installed at this stage can be combined to allow gas and groundwater monitoring. If on-site abstraction wells exist that the pumping regime must be considered before preparing to sample. Pumped samples towards the end of an operational period are ideal, with samples collected from the rising main and not from the header tank or distribution system if possible. This is particularly important for sensitive (unstable) parameters. Surface water sampling can have limited value given inherent variability of the water body, and gas results as a one off may be misleading (e.g. uncharacteristically low due to the ambient conditions (high pressure)).

Now there should be sufficient data to up-date the CSM and complete a preliminary risk assessment in which all the main potential source – pathway – receptor relationships are considered and focussed down to the ones that appear to really matter at the subject site (potential pollutant linkages). In this particular case we are focussing on pollutant linkages to groundwater, whether the groundwater is considered a receptor in its own right or as a potential pathway to a user (well), surface water body or land users overlying a plume (e.g. if the contamination is volatile). This risk assessment will allow decisions to be made as to whether more work is needed, and if this includes detailed site investigation then the scope and cost of this.

One of the big debates is what assessment criteria should be used to assess reported groundwater or soils contamination levels. In the case of groundwater the decision is reasonably straightforward, with IGVs and now some GTVs published for Ireland. Beyond these Drinking Water Standards (or possibly US or Dutch criteria) may be used for other compounds not listed elsewhere. The use of drinking water standards should not be seen as unduly conservative in most cases in that it is where you apply them that matters (e.g. typically at an agreed compliance point not directly below the site but down gradient thereof). For soils the decision is more difficult. In Ireland background soil values exists for some chemicals. Beyond this Dutch Multi-functionality based Criteria (Soil Remediation Circular 2006, as amended on 1-10-2008) or US EPA criteria may represent a useful starting point. The UK SGVs or SSTLs are not appropriate in this case because they are designed to assess human health not groundwater related issues.

Main Phase II Intrusive Investigations: There appear to be many 10s, and possibly 100s of site specific point sources in Ireland that may have potential to breach the requirements of WFD in terms of groundwater quality. The magnitude and extent of most of these is yet to be fully characterised. They are often multifaceted (a number of point sources within the "site specific" point source) and they do not respect site boundaries so "off-site" issues are not uncommon. Characterising the chemical character, magnitude and extent of contamination, at a working facility, particularly if it extends beyond the site boundary can be difficult. Access particularly in and around buildings and structures is often limited. Underground services can be a key restriction. Drilling off-site is inevitably difficult and often seen as a last resort by the problem holder; other potential sources of the same contamination may exist which compounds their concern. Modelling combined with site data may be the optimum approach in some cases. The Phase II Investigation may include:

- Targeted soil sampling using back-hoe excavator or drilling techniques (more perched and shallow groundwater data);
- Deeper soil and groundwater sampling, into the main groundwater flow zone, which may be a bedrock aquifer (drill into top section only);
- All installations and significant other features (surface water bodies; deep drains; topographic breaks in slope) need to be levelled in to Ordnance Datum;
- Collection of soil property (e.g., PSDs; Foc) and soil partitioning (Leach Test) data as well as detailed chemical (COPC) analysis information for soils and groundwater horizons;
- Assessment of aquifer properties (hydraulic conductivity etc) using simple or longer term hydraulic testing methods (Well development needed first);
- Site drainage inspection, testing and/or sampling if this is a potential source or pathway.

If such investigations are large scale and complex due to the magnitude of the contamination issue and/or the complexity of the geology/hydrogeology then it may be best to conduct them in more than one phase and involve a higher degree of technical innovation to collect the data (e.g. temporary or permanent purpose designed multilevel sampling devices; flow through cells for key field parameters if able to purge and sample using a pump). Too often deep boreholes are drilled and monitoring wells installed without due consideration of the value of the data and the risk posed. If long screened sections exist geophysical logging, flow logging and depth sampling may be necessary to demonstrate whether such a contaminant migration pathway now exists. This is particularly crucial in areas where chlorinated and fluorinated organic compounds exist and there is potential for DNAPL in the system. Wells should be sealed up after sampling and testing if this is determined to be a genuine risk.

This investigation is about an appropriate investment to collect appropriate and robust data and characterise a site to the point that full and appropriate risk assessment can be performed and remedial options considered should they be needed. Compromising on data collection at this or any stage of the process risks missing something important. It may cause the project to fail in some way in its delivery of the optimum understanding of the contamination issues and portrayal of the true risk to groundwater and/or a wider environment. This of course can also lead to the wrong or non-optimal remedial solution to be selected with potentially more cost to the client than is necessary and limited net improvement to the environment.

Quantitative Risk Assessment: Risk Assessment fundamentally underpins the whole process and should be considered throughout it (initially qualitatively). Risk assessment informs the investigation process and vice versa. Too often the assessment is picked up as the next discrete task in the process, not uncommonly by a separate expert risk assessor or risk assessment team, with insufficient consideration during the investigation phase. The alternative can be that a member of the site investigation completes a risk assessment; however they may not be expert in the process and just use black box type methods.

Both approaches are considered flawed and in themselves may well be a high risk strategy. A consistent team approach is the ideal one. Undertaking risk assessment without a strong dose of professional judgement can be like building a wall without mortar; it does not necessarily stand up to the rigors of time. Simply launching into a process of deriving site specific remedial goals using a standard methodology or software package may well not serve to conceptually deliver what is needed (seen as too black box by some and much too conservative by others). Considering the risk posed by the identified contamination, to the receptor(s) that matter and refining this based on improved data and professional judgement is the route to success. Once this iterative process is completed then derivation or so called remedial goals or clean up targets becomes the natural and transparent finale to the process, if needed.

How far the risk assessment is taken and the precise tools that are used will be dependent on the point source in question. There is no doubt that simple quantitative risk assessment for groundwater, initially in the process at least, adds a lot of value and is much more transparent. This involves using simple empirical calculations and estimates to assess:

- Contaminant specific assessment of likely fate and transport (how far are they likely to go);
- Groundwater concentration trends and gradients; potential mass in the system (ball park);
- Groundwater mass flux down hydraulic gradient and potential time for depletion;
- Recharge estimates and likely dilution of the plume.

The above are often essential tools in the understanding and ready communication of the magnitude, extent and predicted fate of the groundwater contamination emanating from the point source in question. In many cases they provide a better description of what is happening than models that generate simple risk quotients or clean-up criteria. You need both to generate and communicate the "right" answer and put it in context.

When difficult sites (point sources) are encountered use of relatively simple groundwater risk assessment tools such as the UK EA Remedial Targets Spreadsheet, RAM and CONSIM may be insufficient. Others are better for broader risk assessment including that linked to human health generally. It may be they are unable to deal with the complexity of the migration pathway or the fact the point source is multifaceted. In such cases more sophisticated modelling may be expected, using 2D or 3D flow and contaminant transport models to generate transient flow outputs that simulate plume evolution within the GWB down gradient of the point source with time. The contaminant fate and transport side of these models incorporate the wide range of attenuation mechanisms that can control migration and limit the extent of the plume off-site. When contamination is already off-site and operational history is poorly understood, reliance on such modelling can be paramount.

More complex models can take many weeks to build and calibrate and of course rely on robust site investigation data. Ultimately they can be used in the same way as simpler models with back calculation to estimate an acceptable contaminant concentration beneath the source area or at some compliance point down gradient thereof. Subject to communication and discussions with the Regulatory Authorities there can be a need to validate the findings of the risk assessment modelling by drilling off-site data and confirming groundwater concentrations and impacts.

Corrective Action (as needed): Remediation is the most active form of corrective action involving process or physical processes to pro-actively reduce the contamination. Monitored Natural Attenuation (MNA) represents another form of corrective action that is focussed on confirming natural processes are effective at reducing contamination with time with no interim or residual risk to a wider environment to be expected. CLR11 defines remediation objectives as site-specific objectives that relate solely to the reduction or control of risks associated with one or more pollutant linkages that have been demonstrated through risk assessment to represent unacceptable risks.

Quite a lot of point sources have been investigated in Ireland now. However, few have been fully understood and fewer remediated. The limited remediation that is taking place is often linked to groundwater pump and treat systems which should be considered as containment or pathway control mechanisms rather than active remediation. This can be the preferred option because hydraulic control is often easier to implement than understanding and remediating a range or point sources, particularly on an active site. Complexity of the geology and hydrogeology in Ireland linked to the preponderance of fractured/karst bedrock also lends itself to hydraulic control, provided you intercept the main flow zones. However, it must be appreciated that pump and treat can be a very inefficient way of addressing a point source with mass removal rates as low as a few 10's kg/year. If the source area mass is believed to be tonnes+ then decades or more may be needed, even with the assistance of natural attenuation and specifically *insitu* degradation. Source area soil and groundwater remediation may be needed in such cases.

Remediation must target the critical pollutant linkage and either “break” it (as in the case of a containment) or address the actual source and/or residual source zone within a timeframe required under the remedial objective. Under the WFD which requires the prevention of deterioration of the status of all bodies of groundwater, it is clear that this timeframe may be short for point sources that are causing significant deterioration of groundwater (and specifically a GWB) with 2015 the first main marker, or possibly up to 2027 if an extended deadline is allowed. In most cases it is residual (old) contamination we are dealing with although this must be proven. If an ongoing (active) chemical loss is occurring this will typically need to be addressed immediately.

PRIORITISATION OF POINT SOURCES OF GROUNDWATER CONTAMINATION

There is a need to achieve good status for all GWBs within a defined time frame, and the first and main deadline of 2015 is not far away. Prevention of the deterioration of those waters that have been classified as good already is of course also a requirement. The magnitude and potential extent of some point sources of contamination in Ireland can be expected to threaten the very success of this groundwater protection programme.

Groundwater status assessments need to be undertaken for all GWBs that have already been identified as being At Risk (whether High or Low Confidence level). The next groundwater status assessment for Ireland is due in 2014, ahead of the 2015 deadline. If classified as Poor Status measures will be required to address these, unless this is technically infeasible or disproportionally expensive. There is potential for more so-called orphan sites to materialise in Ireland in the coming years, due to business insolvency or similar and with the possibility of the unfolding environmental liability being the straw that broke the camel’s back in some cases. This of course may shift the burden of addressing some of the identified issues to the State. In the 2008 Status Assessment, 3 individual sites were identified as being contributors to GWBs being at Poor Status, as these sites are still subject to investigation or monitoring rather than remediation. Some 44 sites were classified as Good Status but At Risk, with a need to collect more data.

Given the timescales involved, there now appears to be a reasonably urgent need to progressively draw together greater knowledge (investigation and monitoring data) and/or expert opinion of the chemical character, magnitude and extent of such point sources of contamination and superimpose this on the existing expert knowledge of the overall status of the relevant GWBs. It will be key to understand whether there is sufficient overall attenuation capacity within them to effectively ameliorate the deleterious effects of many of these point sources of groundwater contamination and associated plume(s). In some cases this may be difficult and perhaps impossible in practice because of the number of point sources that could potentially affect an individual GWB (only major sites or ones that have had known pollution incidents are in the centre of the radar screen at the moment).

Therefore investigations and risk assessments for sites need to be completed and the appropriate corrective action (remediation) options assessed and agreed with the appropriate regulatory authority. This perhaps needs to happen in the next 1-2 years if possible so appropriate actions can be implemented soon thereafter. Such enforcement should not just be applied to site operations with so-called deep pockets. Sites with limited or no financial resources need to be subject to same level of scrutiny, if possible.

CHLORINATED SOLVENT CONTAMINATION

Between 25-50% of industrial sites with groundwater issues that I have reviewed in the past 9 months as part of the EPA Framework Contract have had chlorinated aliphatic hydrocarbon compounds (CHCs) as key COPCs and/or risk drivers. They have occasionally been used as a production carrier solvent however by far their greatest application has been for cleaning and degreasing metal parts and equipment, whether as a interim production preparation step or as part of general maintenance. Such applications can be expected to extend into SMEs and therefore into the unregulated as well as the licensed business sector. The good news is that their use was typically phased out by the 1990s. The bad news is they have properties that make them a particular problem for groundwater, as follows:

- They are complex with 3 main parent and a number of so-called daughter compounds;
- Parent compounds are not susceptible to biodegradation if the groundwater is oxygenated. Such degradation requires specific microbiota to catalyse the process. Highly reducing conditions are preferred;
- All the main CHCs have just the wrong sort of aqueous solubility – high enough to be mobile in groundwater at very significant concentration levels but low enough (typically <1%) to allow free product to accumulate if a release occurs (so called DNAPL) which acts as an ongoing secondary residual source. This DNAPL is dense (specific gravity of up to about 1.5) which means it can descend under their own weight through the unsaturated and saturated zone and deep into GWBs if sufficient volumes are released;
- If daughter compounds are formed they are typically more susceptible to biodegradation but not necessarily under the same conditions in which they may be expected to have formed;
- Vinyl chloride, one such breakdown product is extremely volatile so tends to migrate up into the unsaturated zone (good from a groundwater standpoint) but is particularly toxic (proven human carcinogen) so there are particular sensitivities if it is found and it has accumulated;
- Finding DNAPL in a fractured aquifer is like finding a needle in the proverbial haystack as Ireland's specific hydrogeological conditions to not lend themselves to straightforward remediation, particular given this is typically old contamination.

Of the groundwater CHC sites that have been reviewed as part of the EPA Framework Contract, a number appear to have a magnitude and extent of contamination sufficient to cause local impact to the underlying GWB for decades. They continue to be investigated by the EPA.

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GROUNDWATER IMPACTS FROM MINING: IRISH CASE STUDIES

Geoff Beale (Schlumberger Water Services UK Ltd)

NOTES

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ENVIRONMENT LIABILITY DIRECTIVE IMPLICATIONS FOR GROUNDWATER

Kevin Motherway, Irish Environmental Protection Agency[3]

ASBTRACT

In the wake of a number of high profile environmental disasters after which the European taxpayer had to pay to remdiate major environmental damage and a recognition that the polluter pays principle was not being enforced, the Environmental Liability Directive (2004/35/CE) introduced a regime across all Member States whereby polluters responsible for environmental damage (damage to water, land and habitats &species) will be held accountable for all costs associated with the remediation of such damage. The implications for groundwater are twofold in that the directive focuses on the status of water bodies under the Water Framework Directive and the risk posed to human health by contaminated soil. The Environmental Protection Agency (EPA) is the sole competent authority for the implementation the directive in Ireland and in effect is now responsible for assessing cases of environmental damage and the overseeing of remediation and the recovery of costs associated with such damage. The effect of the transposition of the directive is that Ireland now has a de facto contaminated land regime, which focuses on risk posed to human health. While there are clear, established criteria for the assessment of the status of water bodies under the water framework directive; there are no such criteria established in Ireland for the assessment of the risk to human health posed by contaminated land. The EPA is currently reviewing such assessment criteria for use in the implementation of the environmental directive in Ireland.

THE ENVIRONMENTAL LIABILITY DIRECTIVE

ORIGINS OF THE ENVIRONMENTAL LIABILITY DIRECTIVE

The polluter pays principal has been a cornerstone of EU policy since 1973[4], formally mentioned in council recommendations[5], treaties[6] and in our own national legislation[7]. It is also outlined as Principle 16 of the Rio Declaration on Environment and Development. However in spite of the widespread adoption of this principal as a tenet of environmental justice it has not to date been applied in a uniform and consistent manner across the EU.

In the last 20 years there have been several high profile environmental disasters where operators were clearly responsible, identifiable and were as such clearly "liable"; however they did not pay for the entire cost of the pollution that they were responsible for. One such incident was the 1998 Los Frailes, Aznalcollar tailings dam collapse near Donada National Park in Spain where 1.5 M m^3 of sulphide tailings and 5.5 M m^3 of low pH water were released into the environment[8]. The cleanup in the immediate area of the Los Frailes site was undertaken by the operator, however the Spanish (and in effect the EU) taxpayer also had to pay out in the order of €240 M in cleanup costs. Given the apparent wide discretion exercised by Member States (MS) in whether or not to pursue operators who

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

[4] Member States meeting with the Council of 22 November 1973 (3)

5 Council Recommendation 75/436/EURATOM, ECSC, EEC of 3 March, 1975 recital 1;

6 Article 130r (2) 1992 Treaty of Maastricht

7 Section 52(2)(d) of the EPA Act 1992

8 Eptisa, Servicios de Ingenieria S.A. 1998. Investigation of the Failure of the Aznalcóllar Tailings Dam: Summary. Report for Boliden Apirsa, dated November 1998.

pollute and the inherent motivation for operators not to invest in preventive measures or cleanup costs in light of the unwillingness of MS to pursue them, the EU took the view that MS should be required to pursue those operators who cause damage to the environment. The resulting Environmental Liability Directive (2004/35/CE) (hereafter referred to as the ELD) requires MS to recover the costs of remedying damage to the environment from operators responsible for causing the damage. The ELD also has provisions to oblige operators to take action in cases of an imminent threat of damage to the environment, so avoiding/minimising consequences and in the long run reducing costs for both industry and MS.

TRANSPOSING LEGISLATION AND COMPETENT AUTHORITY

While the ELD has been partially transposed into Irish law by the *European Communities (Environmental Liability) Regulations 2008 (SI 547 of 2008)* and is in full force in Ireland since 1/4/2009, an Environmental Liability Bill (due before the Dáil in 2010) is required to transpose some aspects of the directive (regarding some legal defences, genetically modified organisms and a number of other provisions) and so this paper refers to the provisions as set out in the directive. In the Irish transposing legislation the Environmental Protection Agency (EPA) has been designated as the sole competent authority. This means that the EPA have sole responsibility for enforcing the provisions of the ELD.

TEMPORAL LIMITATIONS OF THE ELD

The ELD came into force on the 30th of April 2007, with the Irish regime coming into force on the 1st of April 2009. There is statute of limitations of 30 years on ELD cases, whereby the ELD does not apply if a period of 30 years elapses between when the emission or incident occurred and the environmental damage results.

ENVIRONMENTAL DAMAGE

The ELD outlines the scope of the directive by introducing a special case of major impact on the environment known as *environmental damage.* There are three types of environmental damage: land damage; water damage; and damage to habitats & species; with the criteria for determining if environmental damage has occurred in each case, clearly set out in the directive. Diffuse sources of pollution are not covered under the ELD.

Land damage is any land contamination that creates a significant risk of human health being adversely impacted as a result of direct or indirect contact introduction, in, on or under land of substances preparations organisms or micro-organisms.

Water damage is any damage that significantly adversely affects the ecological, chemical and/or quantitative status and/or ecological potential as defined in 2000/60/EC (the Water Framework Directive- hereafter WFD).

Damage to habitats and species is any damage that has significant adverse effects on reaching or maintaining favourable conservation status for species and habitats as set out in the habitats and the Birds Directive.

The three types of damage can be interrelated depending on the type of event; e.g. a spill of hydrocarbon on an SAC bounding residential properties (into which vapours could migrate) and overlying a highly vulnerable aquifer close to a designated WFD monitoring point. The implications of land damage and water damage are discussed in more detail below; however damage to habitats and species is not discussed in detail as it is not the main focus of this paper, but is outlined well in other works.[9]

[9] Derham J, 2008: "Remediation Environmental Damage under the Environmental Liability Directive: a competent authority strategy", Proceedings of IAH (Irish Group) 2008 Annual Conference

OPERATORS AND LIABILITY REGIMES

Under the ELD an “operator” can be any individual as well as companies, organisations, private and public bodies (e.g. Local Authorities) in control of an economic activity. If environmental damage has occurred then the key aim of the ELD, recovering costs from the party responsible, can only be achieved if there is a clearly identifiable operator, who can be held accountable. In the absence of a clearly identifiable operator the competent authority may take action to clean up Environmental Damage as a last resort; however such a decision would be very serious one given the potential costs involved for the public purse and that it is contrary to the aims of the ELD that the polluter should pay. In all likelihood, in the absence of a clearly identifiable operator responsible for environmental damage an ELD case cannot proceed. The judgment as to who is the responsible operator is one that would be made on the grounds of probability as in civil matters and not beyond reasonable doubt as in criminal matters. This judgment would be taken by the EPA based on the available data and/or any investigations undertaken and would be subject to challenge in the civil courts by the operator.

There are two regimes for operators under ELD; with *strict liability* for ELD Annex III operators, which covers the vast majority of permitted and licensed activities (such as IPPC and waste licences, water discharge licences, etc.); and *fault based liability* for non Annex III activities (i.e. all other operators). Strict liability means that liability can be imposed on an operator regardless of whether or not fault on their part has been proved. Annex III operators are treated by the ELD as a higher risk category that can cause any type of environmental damage, whereas non-Annex III operators are only capable, through fault or intent, of causing damage to habitats and species. The reason non-Annex III operators are included in the ELD is that even a small “low risk” operator such as a construction firm could cause damage of national significance to a protected species; for example destroying a roost of protected bats during demolition works. Therefore only Annex III operators are capable of causing land damage or water damage.

REMEDIATION

The ELD formally outlines the manner in which damage to the environment should be remedied, including the concepts of primary, compensatory and complementary remediation for remedying water damage and damage to habitats and species. This means that the potential costs of environmental clean up now are no longer restricted just to trying putting things back to how they were, but may extend to making up any shortfalls in the cleanup and any interim losses of amenity while the damage is being put right. For land damage the imperative outlined in the ELD is that the contaminated land be remediated such that it no longer poses any significant risk of adversely affecting human health, with no complementary or compensatory remediation required.

LIABILITY: CRIMINAL AND CIVIL

The aim of the ELD is to focus on the cost recovery and the remediation of environmental damage and it does not speak to civil or criminal liability. This means that any criminal prosecution that may be related to an incident of pollution by an operator can proceed independently of any ELD action, as could any third party action against the operator seeking compensation. For example; if there was a release of a pollutant which resulted in water damage from an IPPC licensed installation, which also failed to inform the EPA of the incident, the EPA may pursue the remediation of the water damage using the ELD while simultaneously prosecuting the operator for failure to comply the obligation to inform the EPA of the incident in accordance with the conditions of their licence. Notwithstanding this ability to tackle an incident on several legal fronts, one of the intentions of the ELD is to afford operators the opportunity to put things right with no requirement for any court action; unless the operator chooses to appeal any directions issued by the EPA. Remediation of damage is achieved by the operator submitting proposals for remediation and the EPA, having considered these, then serving a direction on the operator to put in place measures, as the EPA deems necessary to achieve the remediation. There is a cost recovery element to the ELD whereby not only is the operator responsible for the remediation of any environmental damage but is also responsible for any costs incurred by the EPA (or any of its agents), such that the public purse is not impacted. In this manner the polluter pays not just for the making good of the environmental damage, but also for all costs associated with the management of that damage event by the state.

IMPLICATIONS FOR GROUNDWATER

WATER FRAMEWORK DIRECTIVE
Water damage as defined in the ELD focuses solely on the status of a water body under the WFD. If a pollution incident occurs which significantly adversely affects the status of a water body it is classed as water damage under the ELD and the directive could be used as one of the legislative tools to achieve restoration of the status of the water body.

In order for water damage to occur the water body and its baseline status have to be well understood so that the impact of any potential incident can be assessed. If an operator were responsible for a significant pollution event but this event did not actually result in a change in the status of the water body then it would not qualify as water damage. A short-term event such as a minor fish kill, due to an acute rise in BOD while serious in terms of the local environment may not qualify as water damage but could of course be pursued under other legislation such as the Water Pollution Act or fisheries protection legislation.

Diffuse pollution, which is not covered under the ELD, is frequently the cause of poor status of water bodies. Phosphorous from diffuse agricultural sources, is commonly responsible for poor status of surface water bodies but most probably the ELD could not be used to resolve such an issue. While there is provision for multi-party causation in the ELD; if no clear operator(s) responsible for the pollution can be identified, as is frequently the case in diffuse pollution, an ELD case could not proceed. In terms of groundwater a similar analogy would be that of elevated Nitrate levels in intensive agricultural areas.

Cases of water damage that could be pursued under the ELD are more likely to originate from point sources. In England the authorities are dealing with their first case of water damage; due to a malfunction at a pumping station associated with a sewer network which resulted in a significant fish kill, with the impact on fish numbers sufficient to lead to a less favourable status for the water body. In this case the operator[10] is fully cooperating with the ELD regime and has agreed to remediate the river and is negotiating a package of compensatory and complementary remediation.

In terms of groundwater to date there have been no recorded cases in the EU of groundwater related water damage. One of the probable areas where groundwater water damage could occur in MS is over-abstraction leading to poor quantitative status or saline intrusion.

IMPLICATIONS FOR CONTAMINATED LAND

There is a legislative vacuum in Ireland with regard to soil contamination, with no clear regime or policy in place[11]. Apart from EPA licensed sites, the majority of contaminated land sites fall under the remit of local authorities. To date the assessment of contaminated land sites in terms of their impact, need for remediation and any cleanup level to be achieved has been done on an *ad hoc* basis using a number of different soil standards from other MS.

It is significant to note that land damage results from any significant risk of human health being affected. The ELD is silent on any implications for the wider environment. It is also of note that there is no mention of the level or seriousness of the impact on human health. This is in contrast to the contaminated land regime in the UK, which requires that there be Significant Possibility of Significant Harm (SPoSH), i.e. the impact on health must be non-trivial. So in effect the UK now have two

[10] The details of this case are as yet not in the public domain.

[11] "Critical Analysis of the Land Damage Provisions of the Environmental Liability Directive", Shields, A.; Irish Planning and Environmental Law Journal - Vol.16, No. 2 Summer 2009.

contaminated land regimes: a SPoSH regime (known as Part 2a)[12] and the ELD land damage regime (which has no health impact threshold).

Whether human health will be impacted by contaminated land is a function of the land use in the area where the land contamination has occurred. A major contamination event in a rural area where there is no pathway or receptor (e.g. drinking water abstractions or dwellings) that could affect human health may not qualify as land damage, whereas a minor fuel spill in an area surrounded by residential properties into which vapours could migrate would most likely qualify as land damage. Any land which has a designated future use which could result in human health being impacted could also qualify as land damage (e.g. there could be planning permission for houses); so the decision as to the whether land damage has occurred and whether the ELD applies could depend on the planning status of the land in the vicinity of a pollution event.

Annex II of the ELD outlines the approach to be taken in remediation of environmental damage including a sub heading on remediation of land damage. Annex II outline that a risk assessment approach should be used, but does not specify the methodology. Just as there is no contaminated land regime in Ireland, similarly there is no established risk assessment tool, with assessment tools such as CLR11 (UK Environment Agency) or the Target and Intervention Values (VROM *Dutch Ministry of Housing*) being used on an *ad hoc* basis by competent authorities. It is of note that in England and Wales the competent authority for the ELD and contaminated land are the numerous local authorities and that while the CLR11 system is recommended in the UK ELD guidance Document[13], it is not obligatory to use it.

It is of note, that while List I substances[14] are prohibited to be allowed to be discharged into soil or water; Annex II of the ELD, while advocating a risk based approach and outlining how natural recovery should be considered, does not except List I substances or indicate that they are exempt from such consideration. This could have an important implication which means that while strictly speaking List I substances are not to be allowed to remain in soil or groundwater, in the case of land damage they are not exempt from risk assessment or that natural recovery should be considered. This could represent a softening of the position of the Commission in recognising that while it is not permissible that a contaminant should be allowed to pose a threat to human health, that risk assessment may now be used to assess the impact of List I substances, rather than a prescriptive position that they be removed from the water body, which is many cases is unfeasible. This is a matter that requires further legal analysis.

FUTURE DEVELOPMENTS

The EPA is drafting an ELD guidance document and is also currently reviewing contaminated land assessment tools. Both the guidance document and a recommendation as to what land damage assessment methodology(ies) should be used will be issued in late 2010, after a consultation process. The effect of the EPA being the sole competent authority for the ELD means that the Agency are in effect the only body that will be responsible for contaminated land which poses a significant risk to human health. For all other aspects of contaminated land relating to general impact to environment (i.e. non human health related), the EPA is only responsible for it's licensed and permitted sites, with responsibility (via the Water Pollution Act) for all other contaminated land sites resting with the Local Authorities. While the intention of the ELD transposing legislation is not to set up a contaminated land regime in Ireland, the effect will be that the EPA will be responsible for contaminated land in Ireland, but the remit will be restricted to the effects on human health. While this restriction and the

[12] The UK 1995 Environment Act inserted Part IIA into the Environmental Protection Act (EPA) 1990

[13] DEFRA November 2009 ELD Guidance Document

[14] Council Directive 76/464/EEC of 4 May 1976 on pollution caused by certain dangerous substances discharged into the aquatic environment of the Community

design of this regime is a side effect of the ELD transposition, the restriction of the contaminated land regime to risk to human health is consistent with the contaminated land regime in the UK and some other MS. The effects of contaminated land on groundwater will still fall under the remit of other legislation such as the Water Pollution Act, with no prescribed methodology for assessment of impact or risk based approach advocated. This incremental development of a contaminated land regime in Ireland via the transposition of the ELD, while welcomed in some ways, also serves to highlight the need for legislative reform in this area, as has been noted by other commentators in the field.[11]

SESSION IV

LEGAL PERSPECTIVE ON GROUNDWATER PROTECTION

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ABSTRACT

Ireland has benefited from having had available to it a robust suite of remedies at law to protect water (including groundwater) since at least the Fisheries Consolidation Act of 1959 and more particularly since the local government (Water Pollution) Act of 1977. That said, there is no question but that the protection afforded to waters was not consistent and did not represent joined- up thinking from the regulatory management perspective. Furthermore, the focus was not so much on prevention, monitoring and improvement but on responding to polluting incidents or the threat of polluting incidents. Emerging from implementation of the Water Framework Directive of 2000 and the Groundwater Directive of 2006, is a new and welcome approach, attempting to approach groundwater protection from the perspective of lifecycle and sustainability, introducing clear environmental objectives as well as groundwater quality standards and threshold values for the classification of groundwater and protection from pollution and deterioration. This is a positive obligation placed on public authorities which is a widely defined term in the legislation, including bodies such as the NRA, the DDDA, Coillte and Bord na Móna as well as all relevant Local Authorities and the EPA. Accordingly, there is a growing amount of legislation which crosses over the divide between the protection of groundwater and other environmental matters and it is not always easy to locate the remedies or the relevant protections in any given case. It will be necessary to review a number of different pieces of legislation in order to do so.

THE WATER FRAMEWORK DIRECTIVE

The Water Framework Directive (2000/60/EC, the "WFD") consolidates and codifies EU measures on water uses and pollutant discharges. The Directive establishes medium-term environmental objectives to protect and enhance the status of aquatic ecosystems, including surface and coastal waters and groundwater.

The WFD requires Member States to establish river basin districts[1], for which management plans must be adopted to achieve specified environmental objectives. These objectives include:

- To prevent or limit the input of pollutants into groundwater and prevent deterioration of the status of all bodies of surface water and groundwater, immediately; and,
- To protect, enhance and restore all bodies of surface water and groundwater, with the aim of achieving "good status" by the end of 2015.

Exceptions are made for heavily modified water bodies (whose character is changed substantially due to physical alterations caused by human activity) and where the necessary improvements cannot be achieved for technical reasons, disproportionate expense or natural conditions. Less stringent objectives then apply. This is the same where human activity makes improvement of the status impossible or disproportionately expensive.

In Ireland, the European Communities (Water Policy) Regulations, 2003 to 2008[2] (the "Water Policy Regulations") provide the structure within which the relevant objectives are to be achieved. The Water

[1] Art. 3(1) Water Framework Directive
[2] There are three European Communities (Water Policy) Regulations: S.I. No. 722 of 2003, S.I No. 413 of 2005 and S.I. 219 of 2008.

Policy Regulations provide for the characterisation of river basin districts by the end of 2004, with the establishment of environmental objectives and management plans by October 2009. Together with the development of a programme of measures to achieve the environmental objectives, these plans are to be reviewed not later than 16 October 2015 and every subsequent six years.

River basin management projects have been established for each of the eight designated river basins throughout the island of Ireland. These projects involve all the local authorities concerned with the river basin. The local authorities involved in these projects are assisted by the Department of Environment Heritage and Local Government (DoEHLG) and the projects are funded through the National Development Plan and EU Interregional community initiative in the North. In order to give a picture of the complexity of such a project, the Shannon River Basin District project involves eighteen local authorities. To date the final River Basin Management plans have not been published on the websites of the individual projects but the Department of the Environment, Heritage and Local Government has indicated on its website that the adoption of the final River Basin Management Plans would take place in December 2009.

The Water Policy Regulations impose an express duty on relevant public authorities such as the ESB and the Minister for Communications Marine and Natural Resources to exercise their functions in a manner that is consistent with the provisions of the WFD and which achieves or promotes compliance with the requirements of the WFD.

EU Member States are required to establish by the end of 2009 a programme of measures for achieving WFD environmental objectives (e.g. abstraction control, pollution prevention or control of pollution measures) that will be operational by the end of 2012. Basic measures include, in particular, controls of groundwater extraction, controls (with prior authorisation) of artificial recharge or augmentation of groundwater bodies (providing that it does not compromise the achievement of environmental objectives). Point source discharges and diffuse sources liable to cause pollution are also regulated under the basic measures. Direct discharges of pollutants into groundwater are prohibited subject to a range of provisions listed in Article 11. For example, the programme of measures has to be reviewed and if necessary updated by 2015 and every six years thereafter.

River basin management plans have been prepared in draft form for each of the eight river basin districts on the island of Ireland. Four of these river basin districts are entirely in Ireland, one is entirely in Northern Ireland, and three are cross-border international river basin districts. These should have been with the DoEHLG since December 2009.

THE GROUNDWATER DIRECTIVE AND THE EUROPEAN COMMUNITIES ENVIRONMENTAL OBJECTIVES (GROUNDWATER) REGULATIONS 2010 [S.I. NO. 9 of 2010]

The Groundwater Directive 2006/118/EC has been developed in response to the requirements of Article 4(1)(b) of the Water Framework Directive. The European Communities Environmental Objectives (Groundwater) Regulations [S.I. No. 9 of 2010] (the "2010 Regulations") transpose the Groundwater Directive into Irish law. The 2010 Regulations came into operation on 27 January 2010.

In Ireland the original Groundwater Directive [80/68/EEC] (the "Groundwater Directive") set a baseline standard of quality for groundwater. The Groundwater Directive was primarily transposed into National legislation through: the Local Government (Water Pollution) Acts 1977 to 2007, the Local Government (Water Pollution) Regulations, 1978 [SI No 108 of 1978], the Protection of Groundwater Regulations, 1999 [SI No 41 of 1999] and the Local Government (Water Pollution) (Amendment) Regulations, 1999 [SI No 42 of 1999].

The 2010 Regulations are much more forward looking than the Groundwater Directive, because the 2010 Regulations introduce clear environmental objectives as well as groundwater quality standards and threshold values for the classification of groundwater and the protection against pollution and deterioration. They require clear environmental objectives to be achieved in groundwater bodies within specified timeframes and introduce a risk-based approach to implementing the legal obligation to prevent or limit inputs of pollutants into groundwater. This risk-based approach is evidenced by Article 14, which allows the EPA to create exemptions for certain pollutants provided that they are adequately monitored. The 2010 Regulations provide a more flexible, proportionate approach to implementing the legal obligation to prevent or limit inputs of pollutants into groundwater by granting a considerable degree of discretion to the EPA in determining the standards to be applied. This is evidenced by Article 10, which gives the EPA discretion to engage with public authorities to help limit pollution to groundwater.

The main obligations of the 2010 Regulations are set out in Article 4 of the 2010 Regulations, which require that:

- Public authorities must prevent or limit the input of pollutants into groundwater and prevent the deterioration of the status of all bodies of groundwater[3];
- All bodies of groundwater must be protected, enhanced and restored to ensure a balance between abstraction and recharge of groundwater, with the aim of achieving good groundwater by 22 December 2015[4];
- There must be an overall reversal of any significant and sustained upward trend in the concentration of any pollutant resulting from the impact of human activity[5]; and
- Standards for groundwater dependent protected areas must be achieved by 22 December 2015[6].

Parts IV and V of the 2010 Regulations require the EPA to take measures to determine groundwater quantitative and chemical status and monitor groundwater, while part VI requires the EPA to establish procedures for the identification of significant and sustained upward trends and the definition of the starting point for trend reversal.

The 2010 Regulations place obligations on the EPA, local authorities and other government bodies listed in Schedule 1 in respect of groundwater quality. These bodies are required to prevent and reduce pollution of groundwater in order to achieve good groundwater quantitative status by 2015. The regulations prohibit discharge of pollutants into groundwater in the absence of a permit, and require that permitted pollution should not cause deterioration in groundwater status. Although Part III makes it an offence not to comply with the regulations, exemptions set out in articles 16 to19 serve to ease this burden on the relevant public authorities. These exemptions are as follows:

Article 16 allows for the good groundwater status deadline to be extended provided that no deterioration occurs in the status of the affected body of water and all of the following conditions are met:

(a) It must be shown that the necessary improvements cannot reasonably be achieved within the timescales set out in Regulation 4(b) for at least one of the following reasons:

   i. The scale of improvements can only be achieved in phases exceeding the timescale for reasons of technical feasibility,

   ii. Completing the improvements within the timescale would be disproportionately expensive, or

[3] Art. 4(a) Groundwater Regulations 2010
[4] Art. 4(b) Groundwater Regulations 2010
[5] Art. 4(c) Groundwater Regulations 2010
[6] Art. 4(d) Groundwater Regulations 2010

iii. Natural conditions do not allow timely improvements in the status of the body of water;

(b) Extension of the deadline, and the reasons for it, are set out and explained in the relevant river basin management plan;

(c) Extensions are limited to a maximum of two further updates of the river basin management plan except in cases where the natural conditions are such that the objectives cannot be achieved within this period; and

(d) A summary of the measures envisaged as necessary to bring the bodies of water progressively to the required status by the extended deadline, the reasons for any significant delay in making these measures operational, and the expected timetable for their implementation are set out in the river basin management plan. A review of the implementation of these measures and a summary of any additional measures must be included in updates of the river basin management plan.

Article 17 of the 2010 Regulations allows for less stringent requirements to be applied to groundwater that is heavily affected by human activity, provided that the following conditions are met:

(a) The environmental and socio-economic needs served by such human activity cannot be achieved by other means, which are a significantly better environmental option not entailing disproportionate costs;

(b) The highest chemical and/or quantitative status possible is achieved in relation to any individual groundwater body, given impacts that could not reasonably have been avoided due to the nature of the human activity or pollution;

(c) No further deterioration occurs in the status of the affected body of water; and

(d) The establishment of less stringent environmental objectives, and the reasons for it, are specifically mentioned in the river basin management plan and those objectives are reviewed every six years.

Article 18 allows for a temporary deterioration in groundwater status, provided that the following conditions are met:

(a) All practicable steps are taken to prevent further deterioration in status and to protect other water bodies not affected directly by the said circumstances;

(b) The conditions under which circumstances that are exceptional or could not reasonably have been foreseen are documented in the river basin management plan;

(c) The measures to be taken under such exceptional circumstances are included in the programme of measures and will not compromise the recovery of the quality of the body of water once the circumstances have ceased;

(d) The effects of the circumstances are reviewed annually and subject to consideration of scale, technical feasibility, cost and natural conditions, all practicable measures are taken to restore the body of water to the status that obtained prior to the effects of those circumstances as soon as reasonably practicable;

(e) A summary of the effects of the circumstances and of such measures taken or to be taken to restore the body of water to the status that obtained prior to the effects of those circumstances is included in the next update of the river basin management plan.

Article 19 creates an exemption for the failure to achieve good groundwater status due to alterations to groundwater levels, provided that:

(a) All practicable steps are taken to mitigate the adverse impact on the status of the body of groundwater;

(b) The reasons for these alterations are specifically set out and explained in the river basin management plan and the objectives are reviewed every six years;

(c) The reasons for these alterations are of overriding public interest and/or the benefits to the environment and to society of achieving the objectives established by Regulation 4 are outweighed by the benefits of the new alterations to human health, to the maintenance of human safety or to sustainable development; and

(d) The beneficial objectives served by these alterations of the water body cannot for reasons of technical feasibility or disproportionate cost be achieved by other means, which are a significantly better environmental option.

It should be noted that these exemptions echo the exemptions set out in Article 4 of the Water Framework Directive.

THE ENVIRONMENTAL LIABILITY DIRECTIVE

The Environmental Liability Directive ("ELD") introduced a harmonised liability regime for environmental damage across the Member States of the European Union. It requires those whose activities cause or have caused an imminent threat of environmental damage to take preventative actions, and where such damage has occurred, to remediate it.

Environmental damage is defined in Article 2 of the ELD as including damage to protected species and natural habitats, water damage and land contamination that creates a significant risk of human health.

The ELD provides for two distinct but complementary liability regimes. Strict liability is incurred in respect of environmental damage or the imminent threat of such damage associated with an operational activity listed in Annex 3 of the ELD. It is not necessary to demonstrate that there has been fault or negligence attributable to the operator of an activity in order to incur strict liability. A fault based liability attaches to all forms of occupational liability other than those listed in Annex 3. However, fault will only arise in respect of damage to protected species and natural habitats and only where there has been some fault or negligence on the part of the operator.

The operational activities listed in Annex 3[7] (the "Annex 3 Activities") incorporate industrial and agricultural activities of particular risk to the environment. This heightened risk has been addressed by a series of European environmental initiatives and regimes listed previously. In principle these areas attract strict liability because of the particular threat that they pose to the environment and because activities in these areas are already heavily regulated and managed such that environmental damage should rarely occur.

The key feature of the fault-based liability regime is that it only arises in respect of damage to protected species and natural habitats. This can be contrasted with the broader definition of environmental damage in respect of which liability is incurred by operators engaged in Annex 3 Activities. 'Environmental Damage' is defined to comprise damage to protected species and natural habitats, water damage and land damage (being land contamination which creates a significant risk of

[7] As amended by Directive 2006/21/EC of the European Parliament and of the Council of 15 March 2006 on the management of waste from extractive industries and amending Directive 2005/35/EC.

human health being adversely affected as a result of the indirect introduction, in, on or under land, of substances, preparations, organisms or micro-organisms).

The ELD will be transposed in Ireland by means of primary and secondary legislation. We initially decided to transpose the ELD through the promulgation of the European Communities (Environmental Liability) Regulations 2008 [S.I. 547 of 2008] (the "ELR"). The remaining provisions, which include the areas in which the Member States have discretion as to the transposition of various options and penalty provisions, will be transposed by an Environmental Liability Act, which is currently in draft (Bill) form. Although a general scheme has been developed and published for this Bill, its detail is not publicly available and it has not yet been laid before the Oireachtas.

LOCAL GOVERNMENT WATER LEGISLATION

We in Ireland already have robust protection measures available to us in the form of existing local government legislation. This legislation gives a range of functions to the EPA and local authorities and deals with the licensing of discharge of trade and sewage effluent; construction, operation and maintenance of sewers; prevention and control of any polluting matter entering waters; and the regulation of all waste water discharges. The relevant pieces of legislation are:

- The Local Government (Water Pollution) Acts 1977 to 2007,
- The Waste Management Acts 1996 to 2010,
- The Environmental Protection Agency Acts 1992 to 2007,
- The Water Quality (Dangerous Substances) Regulations 2001,
- The Waste Water Discharge (Authorisation) Regulations 2007 and
- The Water Supply Act 2007.

Section 1 of the Water Pollution Act 1977 (as amended) defines 'waters' to include-

(a) Any or any part of any river, stream, lake, canal, reservoir, aquifer, pond, watercourse or other inland waters, whether natural or artificial,

(b) Any tidal waters, and

(c) Where the context permits, any beach, river bank and salt marsh or other area which is contiguous to anything mentioned in paragraph (a) or (b) and the channel or bed of anything mentioned in paragraph (a) which is for the time being dry, but does not include a sewer.

Court Orders may be sought requiring the person to terminate the entry or discharge, to mitigate or remedy the effects discharged, caused or permitted or to pay the costs incurred in investigating, mitigating or remedying the effects[8].

Under the Water Pollution Act 1977 (as amended) it is an offence to cause or permit polluting matter to enter waters; to discharge trade or sewage effluent without a licence, or not to comply with a notice or order[9]. Convictions under the Acts attract fines and/ or imprisonment.

The Acts provide for the following defences:

> *That the activity concerned is authorised by a licence; that all reasonable care has been taken to prevent the entry to waters; or that the activity is in accordance with an approved nutrient management plan.*

[8] S. 10 Water Pollution Act 1977, as amended
[9] S. 10(3) Water Pollution Act 1977, as amended

A person may also recover damages in respect of any injury, loss or damage caused by the trade effluent, sewage effluent or other polluting matters entering waters. Section 23 of the 1990 Act also provides that prosecutions for offences committed by a body corporate may also be brought against any director, manager, secretary or other officer, or any person acting in that capacity, who consented or connived in the commission of the offence or to whom the offence is attributable by reason of his or her neglect.

Under the existing Local Government water pollution legislation, a polluter can be required to mitigate or remedy any effects of the entry or discharge into water. This would include the replacement of fish stocks, the restoration of spawning grounds, the taking of measures to prevent the continuance of the entry or discharge, the removal of polluting matter from waters, the treatment of affected waters so as to mitigate or remedy the effects of the entry or discharge, the making of alternative arrangements for the supply of water for domestic, commercial, industrial, fishery (including fish-farming), agricultural or recreational purposes, the making good of any damage to plant or equipment or to any water abstraction or treatment work and any consequential losses incurred by reason of the entry of polluting matter into waters.

The Water Services Act 2007 defines groundwater as "all water below the land surface that is not in a pipe or similarly contained"[10]. However, the 2010 Regulations employ a different definition, describing groundwater as: "all water which is below the surface of the ground in the saturation zone and in direct contact with the ground or subsoil"[11]. It is not clear whether these definitions would have a different effect in practice as it appears from the legal perspective to mean largely if not wholly the same thing. I would welcome your views on this.

Where an authorised person under the Water Services Act 2007 has reasonable grounds for suspecting a risk to human health or to the environment or an offence is about to be committed in the carrying out of water services which includes the storage of groundwater, he or she can enter onto the relevant premises to investigate whether or not an offence is being committed[12].

It can be seen that there are various means of protection available where groundwater is at risk.

WATER LEGISLATION

- Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy (the Water Framework Directive)
- Environmental Liability Directive (2004/35/EC)
- Local Government (Water Pollution) Act 1977
- Local Government (Water Pollution) Amendment Act 1990
- Waste Management Acts 1996
- Environmental Protection Agency Act 1992 (as amended)
- Water Quality (Dangerous Substances) Regulations 2001
- Waste Water Discharge (Authorisation) Regulations 2007
- Water Services Act 2007
- European Communities Environmental Objectives (Groundwater) Regulations (S.I. 9/2010
- Local Government (Water Pollution) Regulations, 1978 (SI No 108 of 1978)
- The Protection of Groundwater Regulations, 1999 (SI No 41 of 1999)
- Local Government (Water Pollution) (Amendment) Regulations, 1999 (SI No 42 of 1999).

[10] S. 2 Water Services Act 2007
[11] Art. 3 Groundwater Regulations 2010
[12] S. 22(2)(a) Water Services Act 2007

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DELINEATING SOURCE PROTECTION ZONES AND ZONES OF CONTRIBUTION FOR MONITORING POINTS

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ABSTRACT

The national EPA groundwater monitoring network comprises 280 boreholes, wells and springs, the majority of which are public or group scheme supplies. The status of groundwater bodies depends largely on an assessment of groundwater quality data that are obtained from this network. Such assessment requires knowledge of where the water is coming from, i.e., the catchment to the well or spring, the vertical pathway between the ground surface and the aquifer and the horizontal pathways. These factors must be considered when defining Zones of Contribution and Source Protection Zones. Delineation of Source Protection Zones is time consuming and labour intensive. However, with the advent of newly acquired data (recharge, abstraction, vulnerability) and a GIS driven catchment delineation tool, it was possible to delineate Zones of Contribution quickly for the monitoring points. A database was developed which contains the basic information for the monitoring point, allowing easy rapid retrieval and updating of the information. A smaller subset of the network had Source Protection Zones defined and delineated based on the methodologies employed by the Geological Survey of Ireland. For each of these Sources, Landspreading Exclusion Zones have been proposed based on a geoscientific risk assessment.

INTRODUCTION

Groundwater is an important resource in Ireland, forming a key element of the hydrological cycle, supplying water to ecosystems and rivers, and accounting for approximately 16% of the total drinking water supplied (EPA). In some parts of the country, for example in North Cork and Roscommon, it accounts for closer to 80% of the water supplied. The boreholes, wells and springs from which the water is abstracted form part of the national infrastructure. The sustainability of our supplies both in terms of quality and quantity is therefore of critical national importance and so the protection of our sources of drinking water and the areas that contribute water to the sources is sensible and logical. Thus determining the catchment to a groundwater supply should form part of any water management strategy.

There are a number legislative reasons, apart from ‘common sense’ as to why zones of contribution and source protection zones are being defined and delineated, mainly: the Water Framework Directive (WFD), Groundwater Directive (GWD) and Good Agricultural Practices for the Protection of Waters Regulations (GAPS) (S.I. 101 of 2009), and the Drinking Water Regulations.

Since the 1980’s, the Geological Survey of Ireland (GSI) have undertaken a considerable amount of work developing Groundwater Protection Schemes (DoELG, EPA, GSI, 1999) across the country, which partly fulfill (and in some cases exceed) the requirements of the WFD. The “Establishment of Groundwater Source Protection Zones”, a project funded by the EPA, represents a continuation of this work and assists in the compliance with EU requirements.

This project is being undertaken by a joint consortium of TOBIN Consulting Engineers, CDM Ireland Ltd., O’Callaghan Moran, Dr. Robert Meehan (Consultant Geologist), Jenny Deakin and Dr. David Drew (Trinity College Dublin); together with the assistance of the Geological Survey of Ireland. This paper serves to outline the why the work is being done and what is being done on behalf of the EPA, and includes some insights into the process and lessons learned. It is not a paper on the methods or techniques of source protection zone delineation.

TERMINOLOGY

There are several terms used for the areas around springs and wells, which can lead to a certain degree of confusion. The most widely used terms include safeguard zones, zones of contribution, source protection areas, source protection zones and capture zones, which are detailed as follows:

⇨ **Zone of Contribution (ZOC)** is the catchment area that contributes water to the well or spring (Misstear, 2006). It is a simple, intuitive, basic hydrogeological definition that is considered to be the best term for general use.

⇨ **Capture Zone** is a common term present in the literature and is equivalent to the ZOC.

⇨ **Safeguard Zone** is a specific Water Framework Directive term that encompasses the same area as the ZOC.

⇨ **Source Protection Areas** and **Source Protection Zones**: The Geological Survey of Ireland developed this terminology and the methodology for delineating the zones. Two Source Protection Areas (SPA) are delineated:
- Inner Protection Area (SI), designed to give protection from microbial pollution.
- Outer Protection Area (SO), encompassing the remainder of the zone of contribution (ZOC).

Source Protection Zones are obtained by integrating the Source Protection Areas with the groundwater vulnerability categories, as shown in Figure 1.

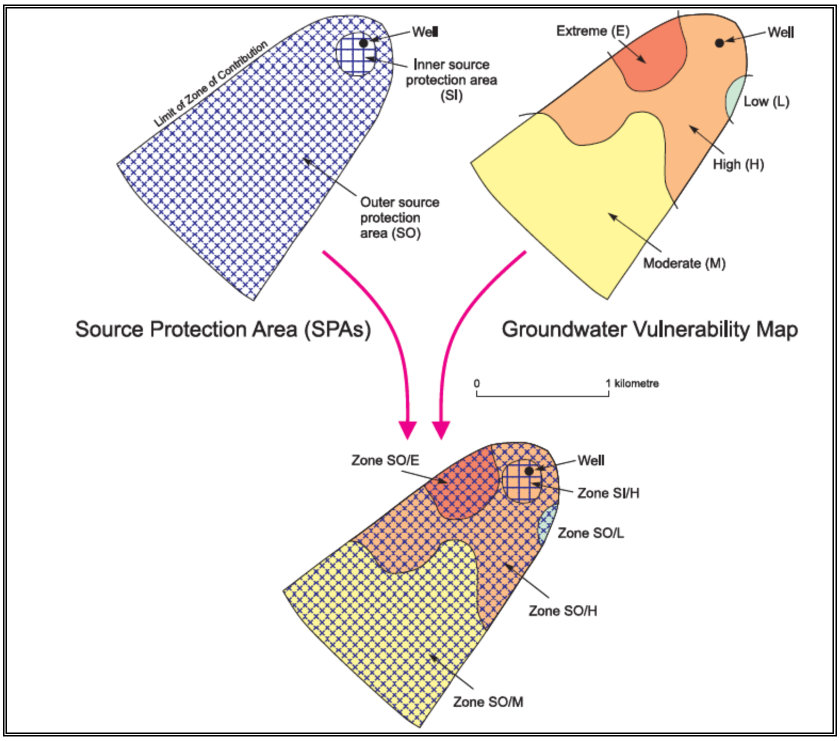

Figure 1: Source Protection Zones (extracted with permission from DoELG, EPA, GSI, 1999)

The ZOC and the SPA account for the 'horizontal' movement of groundwater, while the SPZ includes the complete pathway, both vertical and horizontal, to the abstraction point. Whereas the objective of delineating ZOCs is to define approximate areas that contribute water to an abstraction point, the objective of SPZs is to geo-scientifically characterise the risk to groundwater within the ZOC of a given source.

While these three terms essentially encompass the same total area, there are notional differences and they should be used appropriately. It is recommended that for general usage the simplest, most basic hydrogeological term "ZOC" is used; and that "SPA" and "SPZ" are appropriate when considering protection of groundwater sources. The term "safeguard zones" is only used with reference to implementation of the WFD (Daly, 2009).

SOURCE PROTECTION IN THE CONTEXT OF THE WATER FRAMEWORK DIRECTIVE

Article 7 of the Water Framework Directive (WFD) requires member states to establish "safeguard zones" for those bodies of water, including groundwater, utilised in the production of drinking water. This can be facilitated by firstly delineating the ZOC to determine the extent of the catchment area contributing to the source, and subsequently by developing SPZs that prioritise that catchment area in terms of hydrogeological risk to the source.

One of the key objectives of the WFD is to protect water bodies and water-dependent ecosystems from pollution. In terms of water quality, the primary measure of success is determined from water quality data, whereby monitoring data are compared to water quality objectives (thresholds and criteria) and water bodies are assigned a particular status by EPA. All water bodies, including groundwater, must achieve at least "good status" by year 2015.

Adequate and representative monitoring is therefore a pre-requisite for successful WFD implementation. For groundwater, this includes a good conceptual or demonstrated understanding of the zones of contributions (ZOCs) of individual monitoring points, i.e. where does the water originate from, and what surface or hydrogeological features may influence groundwater quality. With this background, this Project (Establishment of Groundwater Source Protection Zones) is closely tied to two distinct, yet related, WFD implementation programmes:

- EPA's national groundwater monitoring programme - formally submitted to the European Commission in December 2006; and
- Programmes of Measures (POMs) - included in the Draft River Basin Management Plans (RBMPs) dated June 2009 and prepared individually for each of the six River Basin Districts (RBDs) in Ireland.

The EPA is involved in many key components of WFD implementation in Ireland, including environmental monitoring and reporting. Since the establishment of the WFD-required monitoring network in December 2006, the EPA has implemented a sampling and analytical programme involving approximately 280 sampling points. The monitoring network mainly comprises drinking water abstraction points, represented by production wells or springs.

In the context of Programmes Of Measures (POMs), monitoring groundwater quality and understanding water quality data trends and patterns is a key component in arguing for or justifying specific measures. The nature of POMs depends on the specific problem identified from water quality data, and would typically be associated with land use activities within the ZOC of a source or abstraction point. The establishment of ZOC / SPZs could impact on the types of POMs that are implemented in given areas under subsequent River Basin Management Plans.

To gain or have scientific (and political) credibility, it is important that ZOCs and SPZs be defined with the greatest possible "accuracy". This is always tricky in the hydrogeological sciences, as there will invariably be parameter uncertainties associated with any given ZOC or SPZ. As such, ZOCs and SPZs are delineated through consistently applied and thorough methodologies. The approach has been developed within a standard environmental risk framework where the SPZs serve to highlight and prioritise the potential pathways to the supply. SPZs will further assist in the development of potential management practices or controls on activities ("measures") that may take place within the ZOC of a given source.

The delineation of the ZOC / SPZ is therefore very important for those who will be engaged in river basin management under the WFD. Understanding the context and linkages between hydrogeological settings, groundwater flow and land use activities within ZOCs are fundamental to the interpretation of groundwater quality data, and consequently to EPA's status classification of associated groundwater bodies. The establishment of ZOCs and SPZs for drinking water abstraction sources is an important step towards continued WFD implementation in Ireland.

WHAT IS CURRENTLY BEING DONE?

Each of the monitoring points included in the EPA groundwater quality monitoring network has now had ZOCs delineated with the "EPA ZOC tool", discussed in the next section, which represent an understanding of the areas that, conceptually, contribute water to each of the points. Where the GSI and individual RBD consultants had previously defined ZOCs, these were updated and revised with newly available information. This element of the work was for the most part a desk study.

More detailed site work to develop Source Protection Zones (SPZs) was subsequently carried out for a subset of monitoring points, building on the source protection work already completed by the GSI. Finally, for sources that had SPZs delineated, alternative landspreading exclusion zones, in accordance with the Good Agricultural Practices Regulations (S.I. 101 of 2009), were proposed.

A database was developed which contains all the relevant information on each of the monitoring points. It allows for rapid retrieval and updating of information. From the database 'Site Folders' can be generated and printed.

DELINEATION OF ZONES OF CONTRIBUTION (ZOC)

To ensure consistency across the RBDs, a simple GIS based ZOC delineation method was used to update existing ZOCs that had previously been defined by the GSI or the RBD consultants. However, the process had to use a quick approach that led to an approximation of the ZOC. The "EPA ZOC tool" essentially uses recharge, abstraction rates and a flow direction field based on topography as inputs. The "EPA ZOC tool" generates a shapefile of an area (m^2) that is defined by dividing the abstraction rate (m^3/day) by the recharge rate (m/yr). The initial ZOC is similar in shape to a typical well head protection area and is automatically aligned in the direction of the topographic gradient. The ZOC shapefile requires manual alignment and scaling to allow for hydrological, geological, and aquifer properties.

The new, updated ZOCs primarily reflect changes that have been made to the national groundwater recharge map, as well as any changes that are made to abstraction rates following the review of databases in this Project. The national groundwater recharge map (GSI) was updated in 2009 on the basis of new vulnerability mapping by the GSI.

A major challenge using the "EPA ZOC tool" is ensuring that the ZOC shapes make hydrogeological sense. Without detailed field studies and monitoring, all the ZOCs are spatial approximations based on the best available knowledge of the configuration of the source or scheme (number and location and abstraction rates), interpretations of topography, assumed groundwater flow direction, bedrock, structure, aquifer types, groundwater vulnerability, available chemistry, borehole records and flow data. The fact that they are generated from "automatic" GIS processing routines implies that results can be misrepresented or erroneous, and hence run contrary to conceptual models of groundwater flow.

The ZOC delineation is mainly a desk study 'rapid assessment'. However, site visits were carried out in a number of cases, along with consultation with local authority staff and hydrogeologists involved with the site or who had knowledge of the site.

The ZOCs represent the best spatial estimate of groundwater flow to the monitoring point. Ideally every site should be visited to confirm location, source type, borehole construction, abstraction rate, well head protection. As such 'gaps' are likely to remain within the database, which in a sense is a work in progress. It is through the working up of the ZOC to SPZ that improves the basic information, gathers more site specific relevant information, thereby improving the conceptual 3-D understanding of the hydrogeology of the source.

The results from the ZOC delineations were used as inputs to update and complete Site Folders for all sources in the monitoring network, which comprise the basic information about the source.

Advantages of this approach:

- The main advantage is the rapid construction of a simple approximated topographic / geological model of the ZOC;
- The "EPA ZOC tool" highlights issues – particularly where for instance the topographic area is too small for the known abstraction or discharge, and raises questions such as: Is the location right? Is the source right? Is the abstraction rate right? Are there other sources in the scheme? Are the streams losing / perched? What are the mapping requirements? Is the source actually drawing from the bedrock?
- A consistent approach means that the subjective elements of this kind of work can be minimised.

Disadvantages:

- The approach is not replacement for detailed field studies;
- There is limited assessment of the vertical dimension;
- There is limited dialogue if any with the 'person in the know' (often the caretaker);
- Vital information, such as the well has been abandoned and a new supply found elsewhere, might be overlooked;
- Some sites are simply too difficult to assess without more detailed site investigation;
- It is difficult, if not impossible, to appreciate surface water / groundwater interaction.

DELINEATING SOURCE PROTECTION ZONES (SPZ)

The objective of the SPZs is to geoscientifically characterise the risk to groundwater within the ZOC of a given source. The approach has been developed within a standard environmental risk framework where the SPZs serve to highlight and prioritise the potential pathways for movement of contaminants to the supply. The GSI developed the terminology and methodology employed as the national standard for defining Source Protection Areas and Source Protection Zones (Daly, 1995; DoELG, EPA, GSI, 1999). To date, the GSI have delineated SPZs for over a 130 sources, all of which are groundwater supply boreholes, wells or springs. The work undertaken within this project represents a continuation of the work. The GSI are involved in reviewing the conceptual models underpinning each Source Protection Zone and in providing expertise and assistance, particularly in any water tracing conducted.

Under this project, fifteen of the EPA groundwater monitoring network were the focus of delineating SPZs (listed in Table 1).

Table 1: SPZ Sites 2009

SITE	COUNTY	MP CODE	DW CODE	SITE USE	SOURCE
Ballyragget (Glanbia) (Ballyconra)	Kilkenny	IE_SE_G_059_10_001	1500PRI4201	Drinking Water (Private)	BH
Ballyhane PWS	Waterford	IE_SW_G_050_24_002	3100PUB1089	Drinking Water (PWS)	BH
Ballyogarty PWS	Waterford	IE_SE_G_146_24_004	3100PUB1018	Drinking Water (PWS)	BH
Bellmount (Crookstown) PWS	Cork	IE_SW_G_002_04_012	0500PUB3102	Drinking Water (PWS)	BH
Carraignabhfear PWS	Cork	IE_SW_G_004_04_005	0500PUB3301	Drinking Water (PWS)	BH
Carraignadoura GWS	Cork	IE_SW_G_005_04_006		Drinking Water (GWS)	BH
Clifden/Clara GWS	Kilkenny	IE_SE_G_078_10_006	1500PRI3002	Drinking Water (GWS)	BH
Cuffesgrange GWS	Kilkenny	IE_SE_G_026_10_007	1500PRI3169	Drinking Water (PWS)	BH
Cullahill GWS (Toberboe)	Laois	IE_SE_G_059_11_004	1600PRI3001	Drinking Water (GWS)	Karst Spring
Doneraile PWS (Shanballymore)	Cork	IE_SW_G_082_04_024	0500PUB1104	Drinking Water (PWS)	Karst Spring
Enfield PWS	Meath	IE_EA_G_002_17_004	2300PUB1010	Drinking Water (PWS)	BH
Knockcroghery PWS (Tobereeoge)	Roscommon	IE_SH_G_091_20_015	2600PUB1003	Drinking Water (PWS)	Karst Spring
Longwood PWS	Meath	IE_EA_G_018_17_010	2300PUB1014	Drinking Water (PWS)	BH
Newtown Cashel PWS	Longford	IE_SH_G_135_14_001	2000PUB1013	Drinking Water (PWS)	Karst Spring
Scarriff PWS	Clare	IE_SH_G_236_03_003	0300PUB1006	Drinking Water (PWS)	BH

SPZ delineation is time-intensive so a limited number of sources were selected based on the available budget. Prioritization depended on a combination and balance of a number of factors, including:

- EPA's qualitative status classification of groundwater bodies (and their grouping);
- Status of vulnerability mapping around the source;
- Hydrogeological complexity and variety (e.g. flow regime: gravel – fissured – karst; geological structures – fault controls);
- Geography;
- Need for groundwater flow modelling;
- Presence/absence or dominance of karst;
- Source type - borehole/spring/surface water;
- Water quality;
- Well construction details;
- SPZs previously delineated;
- Need for extensive field work;
- Public, Group or Private; and,
- Available time (in 2009).

Apart from site visits and field walkovers and hydrogeological mapping, the main field work tasks undertaken included test pumping, water level monitoring, and water tracing at two of the karst springs (Tobereeoge and Shanballymore). The water tracing was done with the assistance of David Drew, TCD, and Caoimhe Hickey, GSI.

SOME TIPS AND LESSONS LEARNT

The work within this project drew on a number of 'tips' employed for SPZ delineation.

- A site visit, usually with the caretaker, is a critical step to establishing vital information, including the location and type of source and what it comprises. There are many things to look out for at a borehole source: for instance, turning off the pump after a period of pumping can allow the hydrogeologist to listen at the top of the borehole for cascading water that can occasionally be heard entering the borehole from intervals above the recovering water level.

- It is extremely useful to take temperature and electrical conductivity of streams and rivers in the study area to assess the 'connectivity' of the streams to the groundwater and to constrain the flow boundaries.

- There are close links with the water quality and vulnerability: for example very low nitrate concentrations are often associated with a "Low' groundwater vulnerability setting.

- One of the margins for error going into the project was the estimate of recharge from the GSI's Groundwater Recharge map. A key aspect of SPZ work is to estimate the recharge, with the Groundwater Recharge map used as the starting point. Then, the hydrogeological field mapping leads to a better understanding of the recharge. By and large, the recharge limits put on the poorly productive aquifers is considered to be reasonable.

- Karst scenarios require extra work, above that which is normally required to establish SPZs for typical sources in non-karst environments, and this should not be underestimated. To establish the best estimate for a karst spring ZOC, two key questions need answering: "How big?" and "Where?" Establishing a reasonably long reliable flow record and water tracing are crucial in determining the answers to these questions.

- Establishing a ZOC through the "EPA ZOC tool" is useful in constraining the initial boundaries. However a much greater level of work is required to understand the pathways to the source, which in turn allows for greater confidence in the boundaries.

LANDSPREADING EXCLUSION ZONES

As outlined in the paper, in these proceedings, by Donal Daly, alternative landspreading exclusion zones under the Good Agricultural Practices Regulations (S.I. 101 of 2009) may be delineated around a drinking water source if an appropriate, geoscientifically-based risk assessment is carried out. The EPA has concluded that the assessment should be based largely on the same basic information used in delineating SPZs – vulnerability, aquifer category, and inner and outer source protection zones. It is arguable that without the delineation of a SPZ, it is unlikely that a scientifically-based decision could be made on alternative setback distances.

Brief, short reports based on the Source – Pathway – Target model for environmental risk management were prepared for each of the SPZ sources, outlining the basic hydrogeological information and the justification for the proposed alternative landspreading exclusion zones.

For the majority of the borehole sources, the risk close to the source is not high, and the evidence from the monitoring data is that the water quality is good. Thus it is justifiable that the proposed landspreading exclusion zones is simply a 30 m radial buffer. For two of the boreholes where there are moderately thick subsoils, and there are no recorded counts of either total or faecal coliforms, and it is known that there is good borehole construction, a smaller 10 m buffer has been proposed.

For the karst springs, it is proposed that that no landspreading should take place in the "SI/X" zone – the Inner Protection area that has rock at or close to surface. This is because:

a) The risk to groundwater is generally very high due to the vertical pathway providing no or very limited protection against microbial contamination (signified by the "Extreme" ("E" and "X") vulnerability category;
b) The groundwater within this part of the ZOC can reach the springs within 100 days, and could therefore carry live bacteria to the source. It is therefore defined as being within the Inner Protection zone (SI);
c) High total and faecal coliforms counts (>10 counts per 100ml) are recorded in most of the untreated water samples.

In all cases the SI/X zones occur both inside and outside the statutory set back distance.

FUTURE

The Landspreading Exclusion Zone reports and the Source Protection Zone reports will be submitted to the relevant local authorities for review and comment. It is anticipated that a further 25-35 points in the EPA monitoring network will have SPZs delineated this year. It is also likely that other drinking water supplies will have ZOCs and Site Folders prepared. It is anticipated that the ZOCs and SPZs will be hosted on the GSI website and the ZOC site folders will be available for download from the EPA.

CONCLUSIONS

- The majority of the groundwater monitoring points in national EPA network have ZOCs defined. A smaller subset of sources had SPZs defined and delineated and also had landspreading exclusion zones proposed in a separate report. This work is going ongoing, and will be publicly available.
- Establishing a ZOC through the “EPA ZOC tool” is useful in constraining the initial boundaries. However, a much greater level of work is required to understand the pathways to the source, which in turn allows for delineation of the SPZ.
- The SPZ reports can mean that the statutory landspreading exclusion zones can justifiably be reduced. However, they may be increased, in certain limited circumstances.

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EPA GUIDANCE ON PROTECTING GROUNDWATER SUPPLIES

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ABSTRACT

The Environmental Protection Agency (EPA) collates and verifies all monitoring results under the Drinking Water Regulations (2007), provides guidance and advice to local authorities in relation to drinking water supplies and has an enforcement role in relation to the provision of safe and secure public drinking water supplies. Recently the EPA has adopted a Drinking Water Safety Plan (DWSP) approach to ensure the provision of safe and secure drinking water supplies in Ireland. The primary objectives of a DWSP, in protecting human health and ensuring good water supply, are the minimisation of contamination of source waters, reduction or removal of contamination through appropriate treatment processes and the prevent of contamination in the distribution network. In line with the DWSP approach new guidance is being developed on catchment management, borehole construction and wellhead protection as well as for Cryptosporidium monitoring and will be available later on in 2010.

ROLE OF THE ENVIRONMENTAL PROTECTION AGENCY

The Environmental Protection Agency (EPA) has a multi functional role with respect to drinking water supplies in Ireland. The EPA collects and verifies monitoring results for all water supplies in Ireland covered by the European Communities (Drinking Water) (No. 2) Regulations, 2007 as required under Section 58 of the Environmental Protection Agency Act, 1992. This involves the collection of results on an annual basis from local authorities and the carrying out of audits on selected local authorities to verify the information that has been submitted. The EPA is also responsible the provision of advice and assistance to local authorities both on a formal basis (e.g. the preparation of guidance documents) and on an ongoing basis. In March 2007, new Drinking Water Regulations were published by the Department of Environment, Heritage and Local Government. While these Regulations made no changes to the monitoring required and only one minor change to the quality standards to be achieved, they significantly changed the role of the EPA in relation to drinking water. Since 2007, the powers assigned to the EPA include a responsibility to:

- Ensure local authorities are taking the appropriate action to ensure that public water supplies comply with the relevant quality standards;
- Review the actions taken by local authorities in public water supplies where there has been a breach of a standard or any other risk to human health;
- Review and approve monitoring programmes to ensure that adequate monitoring is carried out by local authorities;
- Audit local authority water treatment plants;
- Publish guidance on how local authorities are to implement the Regulations.

The local authority, in turn, has been designated as the supervisory authority over private water supplies (including group water schemes) and has similar responsibilities to the EPA in relation to these supplies.

The Regulations do not provide the EPA with powers to prosecute a water supplier for supplying water that is not clean and wholesome. In general, the powers available to the EPA under the 2007 Regulations relate to the performance of the local authority in respect of any (EPA) Direction. The Regulations require local authorities to notify the EPA of failures to meet the quality standards following which the EPA can direct the local authority to take corrective action. Only where the corrective action, as directed, is not taken can a prosecution be considered for failing to comply with the terms of a Direction. In other words the EPA may prosecute a local authority only if it fails to comply with an EPA Direction (EPA, 2009).

DRINKING WATER QUALITY

The most recent published report on the *Provision and Water of Drinking Water in Ireland* is for the years 2007-2008 (EPA, 2009). The report outlines the safety of drinking water supplies in Ireland by comparing the results of almost 240,000 monitoring tests from 2007 carried out on 952 public water supplies (PWS), 830 public group water schemes (PuGWS), 588 private group water schemes (PrGWS) and 888 small private supplies (SPS).

E. Coli was found on at least one occasion in 52 PWS in 2007. 15 of these 52 PWS were from groundwater and 6 from springs (i.e. 40% of the detection of E. Coli was in groundwater sources). The EPA does not report separately on the quality of drinking water based on its source water and so you are referred to the full report for an in depth analysis of the drinking water quality.

Using data on the 970 Public Water Supplies from 2008, 412 are derived from groundwater and 152 are from springs so 58% of the PWS are from groundwater derived sources. However, when you look at the volume provided by groundwater derived sources, it is only 12% of the total volume provided by PWS. So there are a large number of low volume PWS supplies that are derived from groundwater.

The majority of the groundwater derived PWS have disinfection treatment only. The EPA requires that all disinfection systems have associated continual monitors and alarms installed and responded to; this is expected to improve the overall security of drinking water supplies and reduce E. Coli detection. Currently, approximately 93% of all PWS supplies have chlorine monitors with alarm and dial out in place.

DRINKING WATER ENFORCEMENT

Incident Notification

Where there may be a potential danger to human health due to the failure to meet a parametric value as specified in Part 1 of the Schedule or due to the presence of some other substance or micro-organism, Regulation 9 of the *EC (Drinking Water) (No. 2) Regulations*, 2007 requires the local authority to:

- Firstly, consult with the Health Service Executive to determine whether there is a potential danger to human health;
- Restrict or prohibit use of water or take other actions to protect consumers, if such a danger to human health exists;
- Ensure consumers are informed of the above actions; and
- <u>Ensure that the EPA is promptly notified.</u>

The EPA's role in relation to Regulation 9 is to ensure that where there is a risk to human health that where necessary it directs a local authority (in consultation with the HSE) to take the appropriate action to prevent, limit, abate or eliminate the risk to human health.

Regulation 10 places specific legal obligations on the water supplier (LA) and the supervisory authority when a non-compliance with the parametric value has been detected as a result of routine compliance monitoring, operational monitoring or monitoring following a customer complaint. The water supplier is required to:

- Immediately investigate the cause of the failure;
- Carry out remedial action as soon as possible;
- <u>Notify the supervisory authority;</u>

- Prepare and implement an action programme for the improvement of the quality of the water so as to secure compliance;
- Ensure that consumers are informed or the corrective action where the non-compliance is non-trivial.

The EPA as the supervisory authority is required to:

- Ensure that the water supplier takes remedial action as soon as possible;
- Give priority to enforcement action having regard to the extent of the non-compliance;
- Direct the water supplier to prepare an action programme;
- Review and amend, as necessary, the action programme;
- Issue guidance in relation to the nature and timing of remedial, enforcement or other relevant action.

During 2008 there were 413 issues for which the EPA opened new investigation files; and in 2009, 407 investigation files were opened.

Auditing of Public Water Supplies
The majority of the audits carried out by the EPA are of drinking water supplies where notifications have been received or supplies on the remedial action list (RAL). Sixty-five audits were carried out on public water supplies in 2008 with that number increasing to 114 in 2009. An audit report with recommendations is issued to the water supplier (LA) and a programme to implement the recommendations is put in place. Follow up enforcement action may be taken to ensure that corrective actions are taken, where necessary, on the implementation of audit recommendations.

Enforcement Actions
In relation to notifications received under Regulation 9 or 10 of the Regulations, the EPA assesses the corrective actions proposed by the LA and if the actions are appropriate then no further action is warranted by the EPA, however where the corrective action is not deemed to be satisfactory the EPA can issue legally binding Directions or may carry out an audit of the treatment plant. In 2008, forty-four legally binding Directions were issued to locally authorities under Regulations 9, 10 or 16; the number has reduced in 2009 to 28.

Remedial Action List
The EPA carried out an assessment of all public water supplies from surface water sources in 2008, it identified 339 public water supplies that required detailed profiling to ensure that the supply is providing clean and wholesome drinking water. These supplies have been given high priority in terms of enforcement action and infrastructural funding. The list of supplies is not exhaustive and supplies will be added or removed as information becomes available from EPA audits, the HSE and the DoEHLG. In general, supplies will not be removed until the LA demonstrates that appropriate actions have been taken to ensure that compliance with the requirement to provide clean and wholesome drinking water is secured and the risks of failure have been minimized.

Currently, 130 supplies have been removed from the original RAL (339) with an additional 59 supplies having been added, leaving the total number of supplies at 268.

Provision Of Guidance
The EPA provides different forms of guidance in relation to drinking water, some of it is directly associated with the Regulations and is binding on local authorities, some is in the form of advice notes, which arose as a result of issues that have come to light during our enforcement work and finally guidance that should be considered as best practice such as the Water Treatment Manual series, which local authorities should have regard to in the performance of its functions.

Binding Guidance

The EPA has just produced detailed guidance for local authorities to assist in the implementation of the provision of the 2007 Drinking Water Regulations entitled **The European Communities (Drinking Water) (No. 2) Regulations, 2007: A Handbook on Implementation for Local Authorities**. The drinking water handbook assists Water Service Authorities (WSA) with the implementation of the Regulations. It is a comprehensive document and provides guidance on the following:

1. 2007 Regulations;
2. Standards for Drinking Water Quality;
3. Monitoring of Drinking Water Quality;
4. Guidance on Sampling;
5. Guidance on Analysis;
6. Procedures for Non-compliance with Standards;
7. Drinking Water Complaints;
8. Incidents and Emergencies;
9. Annual Reporting of Monitoring Results and Other Information to the Agency;
10. Drinking Water Safety Plans;
11. Water Treatment and Related Matters;
12. Distribution Network and Related Matters; and
13. Audits of WSAs by EPA.

Prior to the publication of the Handbook the EPA had published a series of guidance booklets contained in Table 1. These booklets have now been incorporated into the handbook and have been removed from the EPA website.

Table 1: Drinking Water Regulation Guidance Booklets

	Title	Issue Date
Drinking Water Regulations Guidance Booklet No.1	Guidance for local authorities on Regulation 9 and Regulation 10 of *EC (Drinking Water) (No.2) Regulations* S.I. No. 287 of 2007	November 2007
Drinking Water Regulations Guidance Booklet No.2	Annual reporting of drinking water monitoring results	February 2009
Drinking Water Regulations Guidance Booklet No.3	Guidance for local authorities on the Remedial Action List for public water supplies	October 2009
Drinking Water Regulations Guidance Booklet No. 4	Risk Screening Methodology for *CRYPTOSPORIDIUM*	January 2008

Advice Notes

In addition, to the binding guidance above the EPA has issued guidance in the form of advice notes where it has identified a specific need for guidance. Table 2 contains a list of published guidance all of which are available on the EPA website www.epa.ie.

There are four additional advice notes in preparation at the moment; these are advice notes on Drinking Water Safety Plans, catchment management for PWS derived from groundwater, wellhead protection and borehole construction, and Cryptosporidium monitoring. It is expected that these will be finalized in Q2 2010.

Table 2: Drinking Water Advice Notes

	Title	Issue Date
DW Advice Note No. 1	Lead Compliance Monitoring and Surveys	April 2009
DW Advice Note No. 2	Action Programmes to restore the quality of drinking water impacted by lead pipes and lead plumbing	April 2009
DW Advice Note No. 3	*E. Coli* in Drinking Water	November 2009
DW Advice Note No. 4	Disinfection By-products in Drinking Water	November 2009
DW Advice Note No. 5	Turbidity in Drinking Water	November 2009
DW Advice Note No. 6	Advice Note No. 6 Restoring Public Water Supplies Affected by Flooding	November 2009

Water Treatment Manuals

In the *Environmental Protection Agency Act*, 1992, it is stated that "the Agency may, and shall if so directed by the Minister, specify and publish criteria and procedures, which in the opinion of the Agency are reasonable and desirable for the purposes of environmental protection". These criteria and procedures in respect of water treatment have been published by the Agency in a number of manuals under the general heading of Water Treatment Manuals (Table 3). These manuals set out the general principles and practices that should be followed by those involved in the production of drinking water.

The EPA is in the process of revising the Water Treatment Manual - Disinfection and expects to have it published by end of the year. The revised manual will provide updated and best practice guidance to local authorities on the whole area of disinfection including chlorination and UV (as well as any other alternative disinfection processes) and will incorporate the Drinking Water Safety Plan approach to improving the safety and security of drinking water supplies.

Table 3: Water Treatment Manuals

Title	Issue Date
Water Treatment Manuals – Filtration	1995
Water Treatment Manuals – Disinfection	1998
Water Treatment Manuals – Coagulation, Flocculation and Clarification	2002

DRINKING WATER SAFETY PLAN APPROACH

In 2008, the EPA adopted a drinking water safety plan (DWSP) approach to ensuring that drinking water is both "safe" and "secure". The EPA contends that the most effective means of consistently ensuring the safety of a drinking water supply is through the use of a comprehensive risk assessment and risk management approach that includes all steps in the water supply from catchment to consumer (Figure 1). A DWSP encompasses this approach and is based on the World Health Organisation (WHO) components of:

1. **Risk assessment of water supplies from catchment to consumer** – Identification and assessment of all risks in the catchment, treatment plant and distribution network up to the consumer's tap that may result in a risk to health and/or breach of required standard.
2. **Effective operational monitoring –** Inspection of the catchment, reservoirs, treatment plant and distribution network to detect pollution, equipment failure or chemical dosing faults; followed by prompt and effective corrective actions where problems have been identified.
3. **Effective management** – Competent management of the supply during normal and abnormal conditions, regular and accurate reporting of treatment plant operations and personnel trained and resources to deliver clean and wholesome drinking water.

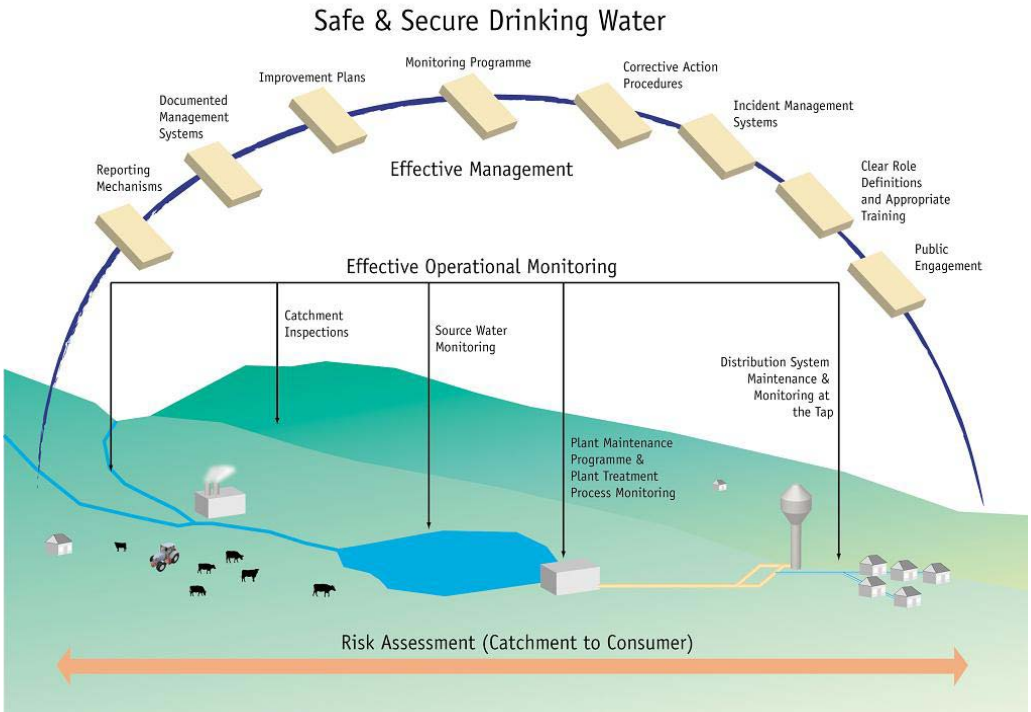

Figure 1: Essential Components of a Drinking Water Safety Plan

The EPA issued guidance for local authorities on developing DWSPs by means of a circular letter in September 2009. The letter advised that the WHO had published guidance on the implementation of the Drinking Water Safety Plan approach (*Drinking Water Safety Plan Manual – A Step-by-step risk management for drinking water suppliers)* which is available for download on the WHO website and that the WHO guidance is recommended by the EPA and should be progressed in each local authority area.

In addition to the WHO guidance, the revised Handbook on the Drinking Water Regulations includes a chapter on DWSPs and the EPA intends to issue an advice note, which will include guidance on hazard identification, risk assessment matrices and a template for improvement plans. A pilot project is underway with Galway City Council and the EPA on the development of a DWSP. The Water Services Training Group will provide training in this area in the future.

CATCHMENT MANAGEMENT

One of the important aspects of the DWSP approach is catchment management; it is also a fundamental element of the River Basin Management Plans that have been developed under the Water Framework Directive (WFD). Historically groundwater had been out of sight out of mind in terms of protection but with the introduction of the Groundwater Protection Schemes guidance in 1999 and the extensive vulnerability mapping as part of the WFD that has changed. The Geological Survey of Ireland (GSI) and consultants have been producing Groundwater Protection Plans and Source Protection Zones for many local authorities. The objective of source protection zones is to provide protection by placing tighter controls on the activities within all or part of the zone of contribution (ZoC).

Zones Of Contribution

The area surrounding a pumped well that encompasses all areas or features that supply groundwater recharge to the well is defined as the zone of contribution of a well (ZoC). The Hydrometric and Groundwater Section of the Office of Environmental Assessment (OEA), EPA have commenced a programme to delineate the zone of contribution (ZoC) for all the groundwater monitoring points in the National Groundwater Monitoring Programme. Of the 280 groundwater monitoring points in the Programme, 152 are public water supplies (PWS) and 34 are group water schemes (GWS). By mid 2010 it is intended that all of the groundwater monitoring points will have a ZoC and in some cases a source protection zone (SPZ) delineated. Some additional 40 PWS have ZoC or SPZ delineated by the GSI or consultants. The Environmental Enforcement (Water) Section of the Office of Environmental Enforcement (OEE) carried out a desk-based risk screening of all groundwater derived PWS and in 2009 identified an additional 20 PWS to be included in the ZoC delineation programme being lead by OEA. Therefore, approximately 38% of the PWS will have ZoC or SPZ delineated by mid 2010. It is intended that work in this area will continue if funding is available.

Reports on the ZoCs that are being delineated at present for the EPA will be made available to all relevant local authorities and will be placed on the EPA website.

Catchment Management Advice

In light of the delineation of ZoCs of groundwater supplies, it was considered appropriate that advice be produced on catchment management and the use of the ZoCs for public water supplies. The EPA intends to issue guidance in the form of an Advice Note to all local authorities later on this year. The Advice Note will contain guidance on hazard mapping including farm surveys, on-site wastewater treatment systems (OSWTS) and buffer zones for landspreading. This will tie in with the guidance issued on DWSPs.

WELLHEAD PROTECTION AND BOREHOLE CONSTRUCTION

In considering the catchment management aspects of the DWSP approach, it was deemed essential that the third dimension be addressed. This third dimension is the well itself, whereby it may be the pathway for contaminants to enter the water supply. It is critical that the well/borehole is properly constructed and also has the appropriate wellhead protection in place.

The Institute of Geologists of Ireland have produced Well Drilling Guidelines (2007) and these are endorsed by the EPA, however, it was considered that additional advice was needed in this area and so a study has been commissioned, which will produce the following;

- An advice note for local authorities outlining the hazards/risks associated with poor borehole construction and wellhead protection. It will refer to the IGI Guidelines as best practice guidance but will build on it in relation to the assessment of existing water supplies;
- A checklist document will be developed for EPA inspectors, which will assist in the assessment of borehole construction and wellhead protection while on-site;
- A leaflet outlining the importance of good well construction and protection for the general public.

This work is to be completed by end of Q2 and will be published in the form of an Advice Note at that time.

ADVICE ON *CRYPTOSPORIDIUM*

Contamination of water supplies with the parasite *Cryptosporidium* presents a significant threat to the safety of drinking water in Ireland. The first outbreak associated with a public water supply in Ireland was in Mullingar in 2002. Improved awareness of the disease and a requirement on notification of the disease to the Health Protection Surveillance Centre has led to increased reporting of the disease and hence more outbreaks of the disease have been detected. Several outbreaks associated with water supplies have occurred in Ireland since 2002, including supplies in Ennis, Roscommon, Carlow, Portlaw and most recently Galway in 2007 (EPA, 2008).

In 2007, the EPA set up a national working group on *Cryptosporidium* and the initial focus of the working group was to review the risk assessment published in 2004 and to provide guidance/assistance to local and other regulatory authorities in implementing the *Cryptosporidium* risk assessment and consequently reducing risk of contamination of water supplies with *Cryptosporidium*. Four subgroups were established to progress the objectives of the Working Group.

1. Subgroup 1 – *Cryptosporidium* Risk Assessment
2. Subgroup 2 – *Cryptosporidium* Monitoring Capacity
3. Subgroup 3 - *Cryptosporidium* Monitoring and Technologies
4. Subgroup 4 - *Cryptosporidium* Incident Outbreak Management

***Cryptosporidium* Risk Assessment**

In 2004, the EPA recommended the use of a *Cryptosporidium* risk assessment, which was based on the Scottish model as outlined in "The Cryptosporidium (Scottish Water) Directions, 2003" as published by the Scottish Executive. Subgroup 1 reviewed the risk assessment and recommended that it be amended following widespread use of the risk assessment and more recent information and research. The amended version of the risk assessment was published in 2008 - *Drinking Water Regulations Guidance Booklet No. 4: Risk Screening Methodology for Cryptosporidium.* This guidance has now been incorporated in to the new drinking water handbook. The risk screening methodology is being used by local authorities on all public water supplies to assist in prioritising supplies that are at a high risk of contamination with *Cryptosporidium* and identify high risk factors, which can be mitigated to reduce the risk associated with the supply. The methodology involves calculating a risk score for the catchment factors and for the treatment, operational and management factors, which is then population weighted to give a final risk score. Where a supply has been identified as high risk the water supplier should develop an action programme to reduce the risk to low. The risk category for each water supply should be reviewed on an annual basis and the methodology re-applied where there is any change to the catchment factors or a change in treatment, operational or management factors. Prior to applying the risk screening methodology an assessment of the catchment factors and the treatment, operational and management factors should be carried out for each source. The risk screening methodology is seen as a pre-cursor to the application of a Drinking Water Safety Plan approach to the management of drinking water.

Monitoring

Subgroup 3 concluded its work and recommended that for an initial period of two years the following minimum monitoring frequencies of treated water be implemented. However, it is noted that each supply must be considered individually and the programme designed to take into account the site-specific risk of the catchments and treatment process. After two years of testing a review of results for each water treatment plant will allow for the monitoring frequencies to be adapted to better suit the characteristics of each supply.

- For plants serving a population greater than 20,000, 24 hour continuous sampling with 1 test to be carried out every week for 52 weeks of the year;
- For plants serving a population less than 20,000, test frequency to be determined depending on risk category and results of plant monitoring.

In 2009, the EPA commissioned a study into the development of appropriate programmes for monitoring *Cryptosporidium* for raw and treated waters used for human consumption for different water supply scenarios. This study recently was completed and the EPA intends to issue its findings in an Advice Note by the end of Q2.

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SESSION V

IMPACTS OF CLIMATE CHANGE ON RAINFALL, RIVER DISCHARGE AND SEA LEVEL RISE

Tido Semmler
Met Éireann

ABSTRACT

The climate has changed worldwide and in Ireland and it is predicted to continue to change. There is certainty about the fact that the global average temperature has already risen by about 0.7 °C from the beginning of the 20th to the beginning of the 21st century while the temperature averaged over Ireland has risen by about 1.1 °C over the same time period. A further increase of 1.8 to 4 °C globally and 2.0 ±1.0 °C over Ireland until the end of this century is predicted according to the Intergovernmental Panel on Climate Change (IPCC) and results of the ENSEMBLES project. The temperature increase has already led to a sea level rise of 18.5 cm over the last 100 years and is predicted to lead to an additional rise of 18 to 59 cm over the next 100 years according to IPCC. However there is uncertainty in the prediction of glacier melt; an acceleration in the glacier melt could lead to a stronger sea level rise of up to 2 m over the next century. Trends are less uniform for rainfall and river discharge. Large regional differences occur and due to a strong natural variability there is uncertainty in the predictions. For Ireland wetter winters and drier summers are likely in the future which would lead to stronger river flows with an increased risk of flooding in winter and weaker river flows with an increased risk of droughts in summer. However, there is no clear trend for summer rainfall from Irish observations over the last 70 years while the observed rainfall slightly increases for spring, autumn and winter.

INTRODUCTION

According to the 4th assessment report of the Intergovernmental Panel on Climate Change (IPCC, 2007) there is very high confidence that human activities have contributed to global warming since pre-industrial times. This is the strongest statement yet by Working Group I of the IPCC, tasked with assessing current scientific understanding of climate change. A summary of current state of knowledge of the climate system by climate scientists, the report states that there is 90% probability that greenhouse gases produced by human activities have caused most of the observed global warming since the mid-20th century. Also future climate predictions have advanced substantially since the previous IPCC report in 2001 and results are more robust. Nevertheless, uncertainty remains especially regarding regional changes and changes in extreme events. Therefore it is important to produce ensemble simulations to sample the uncertainty.

Ireland is no exception and climate change has been observed at Irish observation stations. In the future increases in temperature leading to enhanced evaporation and changes in the seasonal and regional distribution of rainfall can have important influences on groundwater levels in Ireland. Furthermore rising sea levels can influence coastal areas through intrusion of salty water into the groundwater. This paper gives a summary of the results of current research into observed and simulated climate change over Ireland, including estimates of uncertainty in the predictions of future climate.

OBSERVED CHANGES

Instrumental records show a general increase in the global average temperature by about 0.7 °C over the last 100 years (Figure 1). Irish temperatures have even risen by about 1.1 °C over the same time. Due to natural variability the increase has not happened uniformly; periods without increase or even decreases have occurred from 1900 to 1910 and from 1940 to 1970 while periods with a strong increase have occurred from 1910 to 1940 and from 1970 to today.

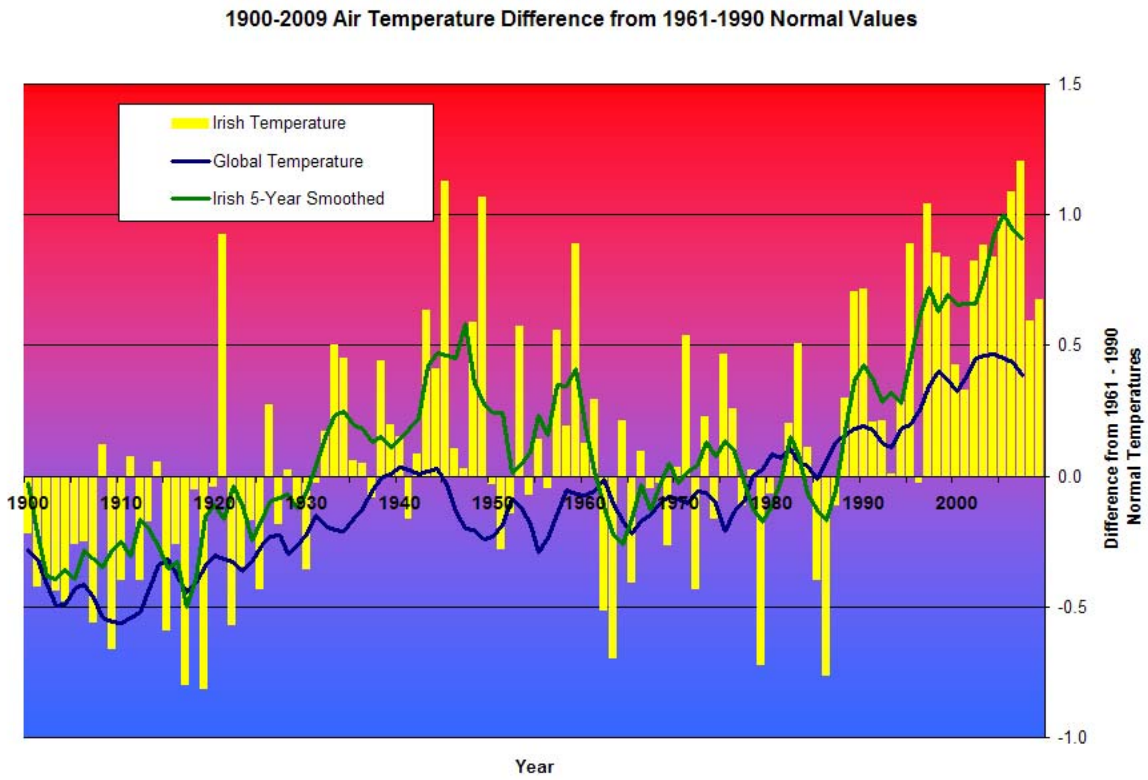

Figure 1: Irish and global mean temperature anomalies. The reference period is 1961-1990. For the Irish temperature data an average over the five stations Malin Head, Armagh, Birr, Valentia and Dublin Phoenix Park has been calculated. For the smoothed Irish temperature data a running 5-year mean has been applied. The global temperature data are taken from the Climate Research Unit (based on Brohan *et al.*, 2006)

A large amount of the additional energy has gone into the oceans leading to a global sea level rise of 18.5 cm over the past 100 years (Figure 2), which corresponds to 1.85 cm per decade. From 1993 to 2003 the global sea level rise has accelerated to 3.5 cm per decade according to satellite altimetry data. Around Ireland the observed trend for 1993 to 2006 is similar to the global trend although there are some important regional differences. The sea level rise is estimated to be between about 2.5 cm per decade in the east to about 4.5 cm per decade in the north-west of the country. These numbers take into account the isostatic adjustment of the Earth's crust. While coasts on the northern and eastern seaboard are rising by about 0.7 cm per decade coasts on the western and southern seaboards are subsiding at a rate of about 0.5 cm per decade.

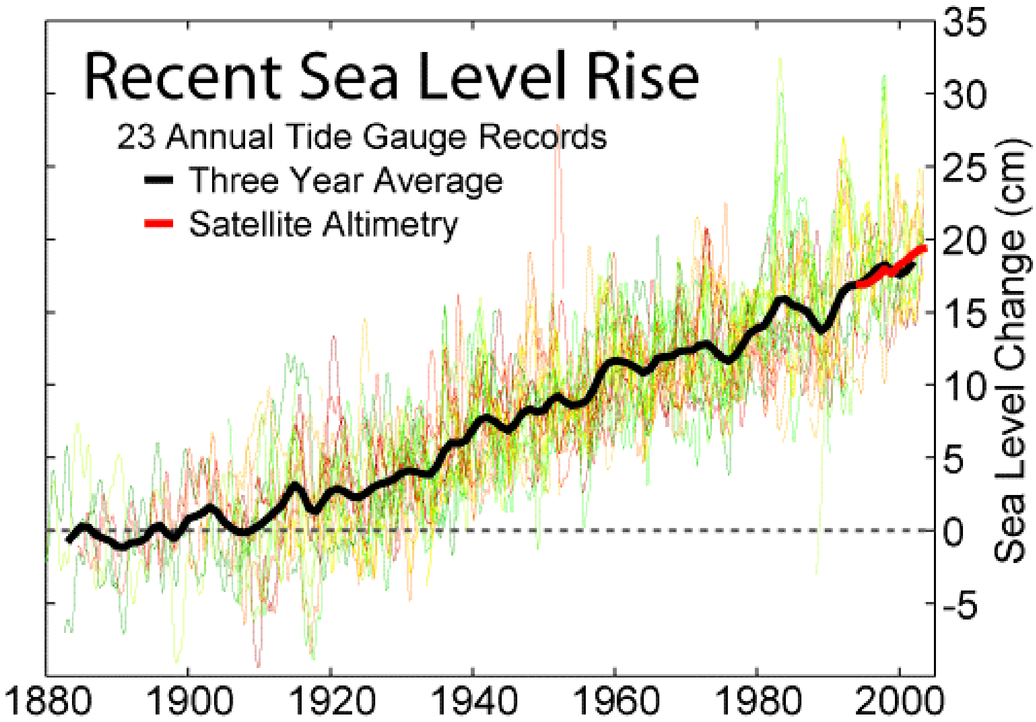

Figure 2: Global sea level rise as an average over 23 geologically stable tide gauge records according to Douglas (1997) updated with records until 2004 and complemented with satellite altimetry data

While temperature and sea level have increased all over the globe with larger temperature increases over land than over sea, changes in rainfall are much more variable. Even though the water vapour content has increased globally due to the larger water vapour holding capacity of warmer air, there are areas with increases in rainfall and other areas with decreases. Globally there is a tendency towards a higher frequency of heavy rain. Over Ireland the rainfall has generally increased in all seasons, bar the summer, over the last 70 years (Figure 3). It is interesting to note that summer rainfall has decreased in the first half of the period while it has increased afterwards, reaching the same level at the end of the time period as occurred in the beginning. The annual increase averaged over the country amounts to 7%. Regarding extreme rainfall events it is not possible to make any statements if they have changed over the past or not. Some stations show increases and some others show decreases in the frequency of days with more than 10 mm/day and more than 20 mm/day. Results also depend on the investigated time period.

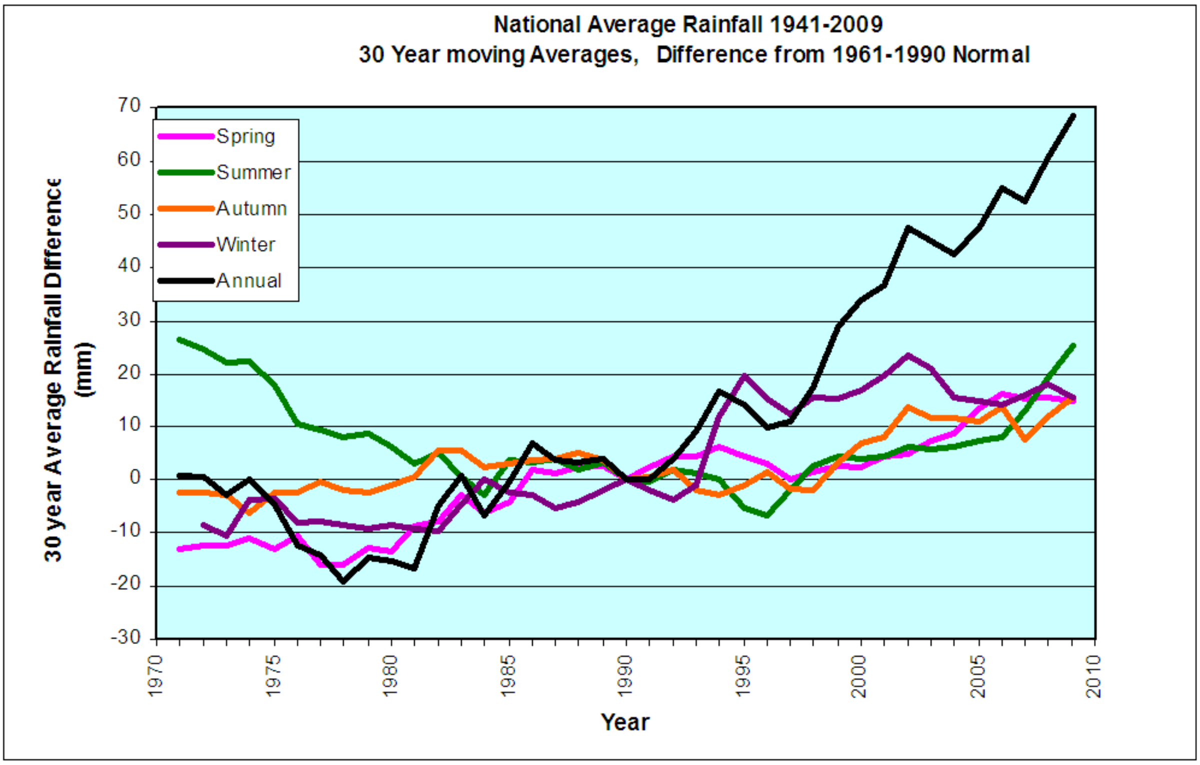

Figure 3: Anomalies of 30 year moving averages of Irish rainfall from 1941-2009 compared to 1961-1990. The value of each year represents the average over the 29 years prior to the year and the year itself, e.g. the value of 2009 is the average over 1980 to 2009

Observed changes in river flow are generally similar to the described changes in rainfall. In the 1970s, 1980s and 1990s low flow events in summer are observed in many river catchments while in the 2000s, no major low flow events occurred. Regarding the extreme high flow events no clear trends can be found.

PREDICTED CHANGES

Temperatures are predicted to continue to rise over the next century. Increases of the global mean temperature between 1.8 and 4 °C are likely until the end of this century (IPCC, 2007). For Ireland increases of about 1.0±0.5 °C for 2021-2050 and 2.0±1.0 °C for 2071-2100, when compared to 1971-2000, are predicted from an ensemble of 12 simulations (Figure 4). Values tend to be slightly higher in the south and east of the country and slightly lower in the north and west. The uncertainty is given as the standard deviation of the 12 simulations. For the first time period (2021-2050) 20 ensemble members are available. Results from the 12 and the 20 member ensembles are very similar. Slightly smaller increases occur in spring and slightly larger increases in autumn. Globally sea levels are predicted to rise by 18 to 59 cm until the end of this century according to IPCC (2007) although some more recent studies suggest a rise of up to 2 m due to accelerated melting of glaciers (Allison *et al*., 2009).

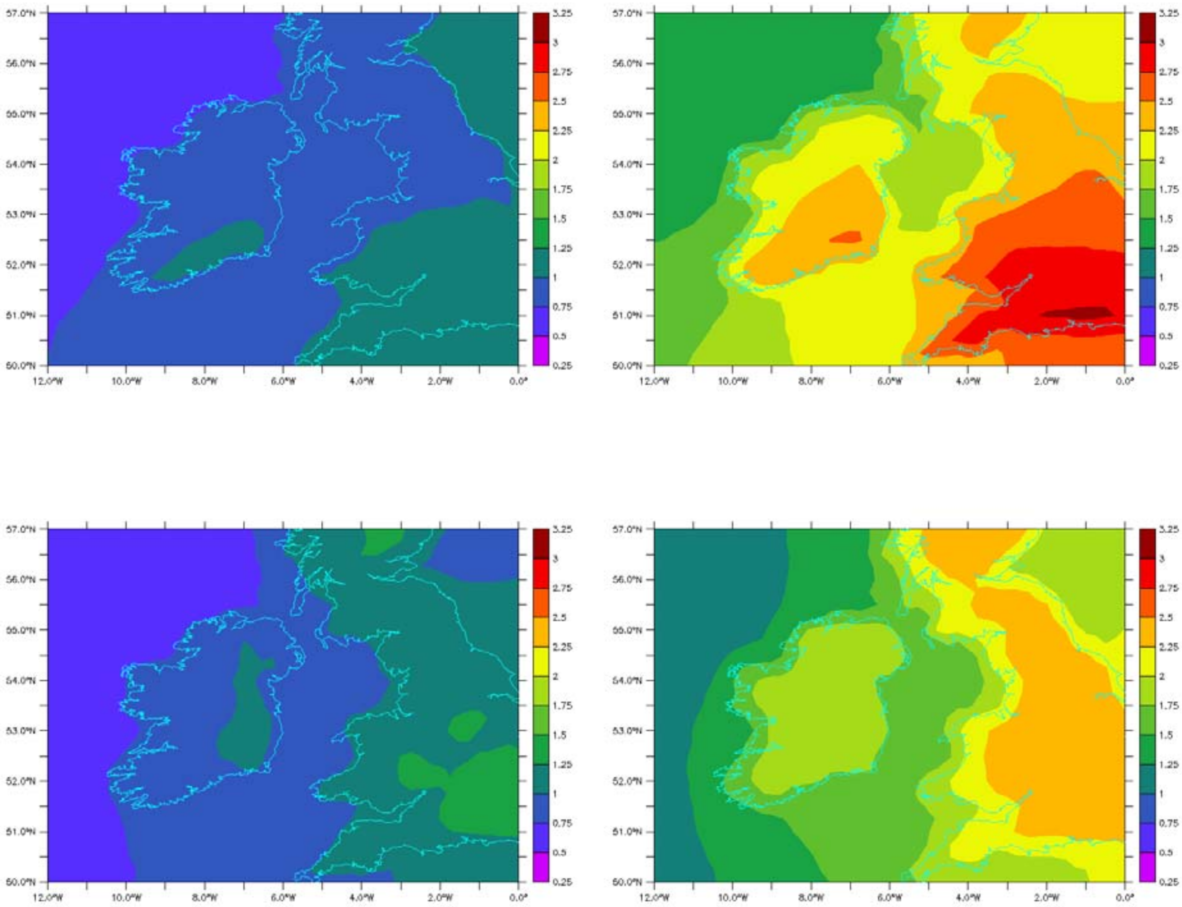

Figure 4: Predicted temperature changes [° C] for summer 2021-2050 and summer 2071-2100 (upper row), winter 2021-2050 and winter 2071-2100 compared to 1971-2000 (lower row)

Predictions in Irish rainfall are more uncertain and the standard deviation of the 12 or 20 regional climate simulations is often comparable with the actual data. On average over the country a decrease of 5±5% in summer and an increase of 6±6% in winter rainfall is simulated for 2021-2050 compared to 1971-2000 data (Figure 5). For 2071-2100 (again compared to 1971-2000 data) the predicted decrease in summer is 15±11% and the increase in winter is 15±7%. The larger decreases tend to occur in the south and the larger increases in the north of the country. The prediction in extreme rainfall events is even more uncertain and has to be further investigated. A preliminary study suggests that 20-year return values of daily, 2-daily and 5-daily rainfall totals increase in large areas of Ireland, especially in the north of the country where some small regions show increases of more than 20%, and tend to decrease in only small areas of Ireland, especially in the south of the country if comparing the values of 2021-2060 with 1961-2000. However, this preliminary study only includes two ensemble members. Work to extend it to 20 ensemble members is in progress.

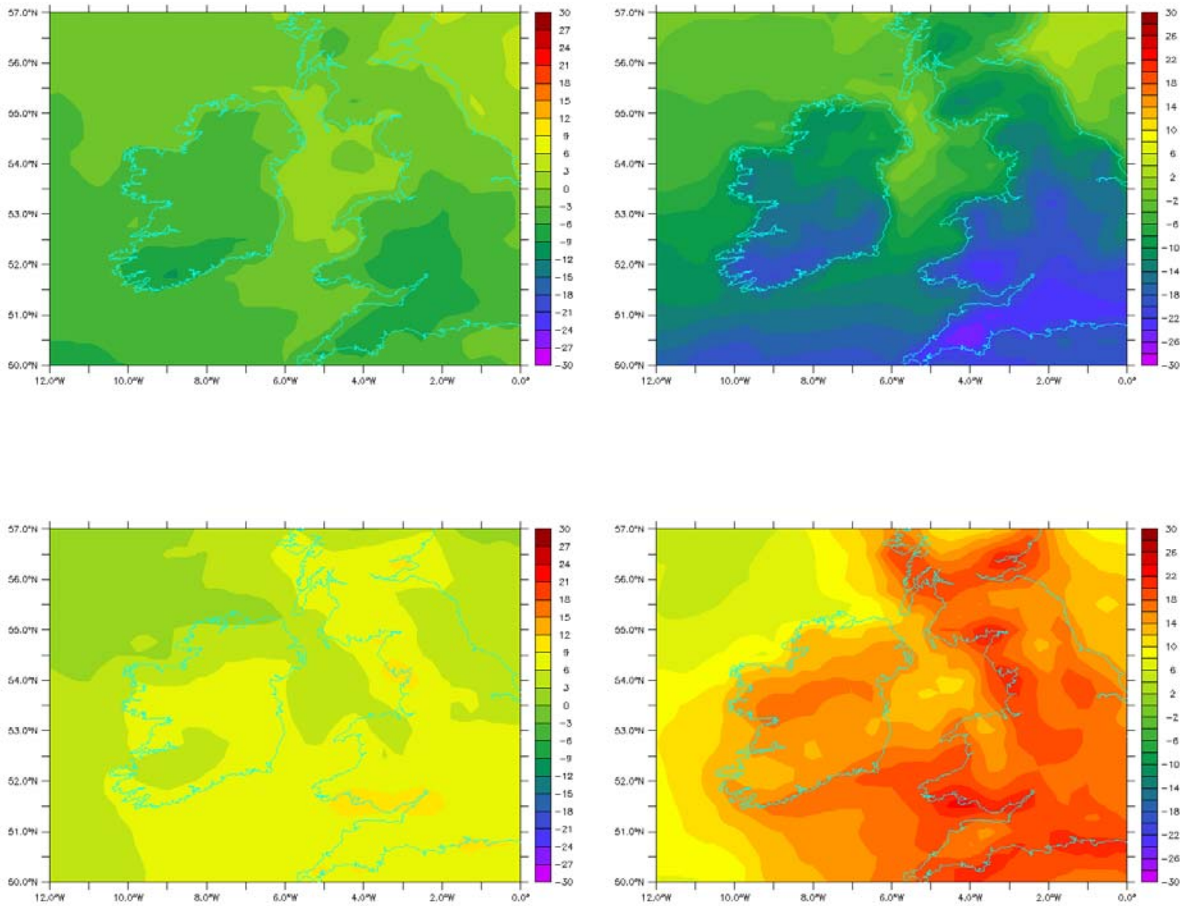

Figure 5: Predicted rainfall changes [%] for summer 2021-2050 and summer 2071-2100 (upper row), winter 2021-2050 and winter 2071-2100 compared to 1971-2000 (lower row)

To estimate the impact of climate change on river discharge, ensemble simulations with the HBV model have been carried out. For each of 13 regional climate model simulations an ensemble of 100 HBV model simulations has been performed giving a total ensemble of 1300 simulations. Figure 6 shows box plots of differences between climatological monthly average values for 2021-2060 and 1961-2000. Even though in the 9 investigated catchments there is a higher probability for more discharge in winter and less discharge in summer, the uncertainty is large, especially in summer. While in January for most catchments all values are above 0 indicating an increase in the monthly river discharge, in the other months generally positive or negative changes are possible according to this prediction.

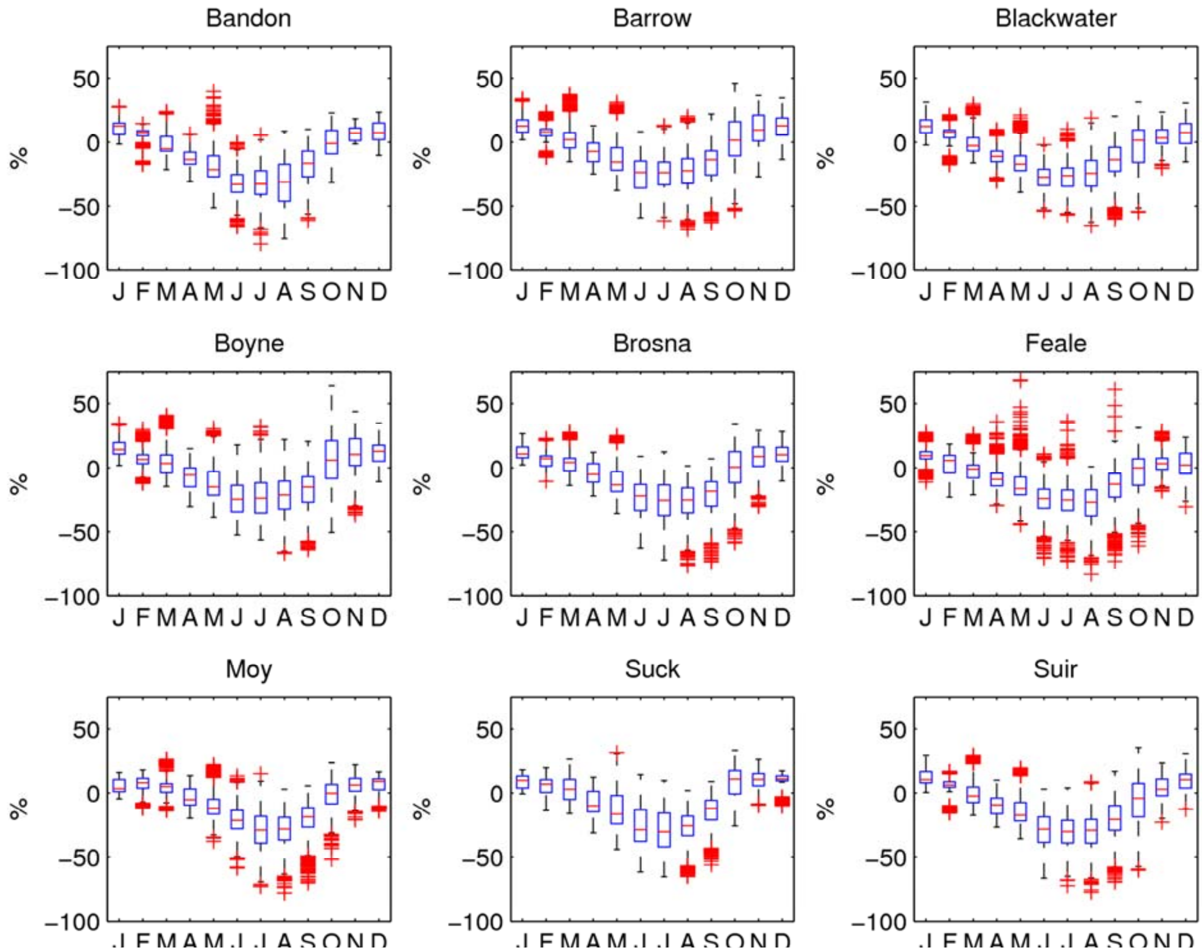

Figure 6: Box plot of the change in stream flow [%] for all 1300 simulations. The median is indicated by a horizontal red line in the blue box, the first and third quartiles by the blue box, minimum and maximum excluding outliers by black bars and outliers (more than 1.5 interquartile ranges lower than the first quartile or more than 1.5 interquartile ranges higher than the third quartile) by red crosses

Since the uncertainty is already large in the monthly mean river flow, no reliable statements on extreme river discharge events can be made. If investigating less extreme events such as changes in maximum daily river discharge per month, simulated changes are very similar to changes in mean daily river discharge.

CONCLUSIONS AND OUTLOOK

Increases in temperature and sea level both globally and around Ireland have been observed and simulations for the future show further increases with very high confidence. Changes in rainfall patterns and river discharge are not as obvious although slight increases have been observed in spring, autumn and winter around Ireland. No statements can be made about observed changes in extreme rainfall and river discharge events. For the future it is more likely that summer rainfall and river discharge decrease rather than increase. However, the uncertainty is large. Most simulations indicate increases in winter rainfall and river discharge. There are some indications that extreme rainfall events could become more intense.
It is important not to only estimate the uncertainty in future climate predictions but also to work on reducing uncertainties. Therefore Met Éireann is involved in the further development of a new earth

system model. Currently the coupled atmosphere-ocean model is already used to perform climate forecasts for the next assessment report of the IPCC. Some of the simulations are run on very high resolutions of 35 to 40 km globally. It has already been shown that extreme events are well represented in these new simulations for present day climate. Therefore it should be possible to make more robust statements on future climate change soon.

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CLIMATE CHANGE IMPACT ON WATER RESOURCES IN IRELAND

David M. Ball
Hydrogeologist

ABSTRACT

Hydrogeology needs to make a stronger input into climate science and assist with the assessment and predictions regarding global warming. The current predictions about climate change are at a large scale, but they can be used to discus the probable impacts on groundwater and surface water resources in Ireland. These impacts and the consequences for our infrastructure and environment can be used to inform planners and decision makers.

INTRODUCTION

Climate change is not new to hydrogeology. We already have a perspective of time and change in our work. We easily appreciate time and change within a sliding scale; from intense rainfall affecting our pumping test results today, to consideration of a frozen environment when the base level for groundwater discharge was 120 metres lower; just 20,000 years ago. We know, from our geology training, that climates were radically different in the past.

Hydrogeologists have also been working on climate change since the 1960's. We started using environmental isotopes in groundwater to fingerprint water, and use this to understand large, long groundwater flow systems. It was not unusual for us to use this research to describe palaeo-climate conditions of temperature and rainfall 10 - 40,000 years ago. Now with more accurate instruments and preparation techniques, we have doubled that range. Yet, even with our awareness and knowledge, we, and our discipline, do not appear to be deeply engaged in the Global Warming scientific community, and groundwater does not appear to form a major component of the climate prediction models. I suggest that this must be corrected.

Groundwater forms the greatest volume of liquid, fresh water on earth. It is becoming greater by proportion, as the ice melts and fresh water flows primarily back into the sea. Groundwater is transient water, just like surface water, or atmospheric water. It is a part of the global water system; yet the oceans, atmospheric moisture and rainfall-runoff are a part of models for the future. Maybe, it does not matter that groundwater is not included because it is hidden, and moves slowly. Maybe, it is not included because a little change in a big resource makes no difference. Though the timescale for response is different, the present position, chemistry, and flow rates are all a representation of a delicate and temporary balance. I have seen little evidence of work done on the effect of climate change on groundwater. Perhaps, groundwater's peripheral consideration is because other scientists, in particular climate scientists, do not understand the parameters of groundwater, and do not feel confident trying to incorporate it into their modelling. If we think that their science of the fast moving atmosphere is complicated, then they probably feel the same about our hidden resource. Maybe we should help them; maybe we should be looking for evidence of climate change in the groundwater world, and volunteering this information to the wider community. Maybe we should be trying to provide some simple explanations and processes that make our world more accessible to others, who might see its importance as a balance to more violent change, but fear to grapple with the hidden and insidious.

As a geologist and natural scientist, I know that climate change has always happened. The world's climate has never been static. It has evolved since its inception. Groundwater is an integral part of the water cycle and the world's climate. It cannot be considered in isolation. My paper is not to argue whether human activity is the principle cause of climate change, but to take the various predictions of climate change, and discus the potential impacts on groundwater, and through the groundwater perspective, on other water resources in Ireland.

SEA LEVEL RISE AND THE URBAN INFRASTRUCTURE

It is widely accepted and understood that global warming climate change will lead to a rise in sea levels. The immediate obvious impact is that coastal areas will become inundated. Most people and decision makers see sea level rise as a problem for the areas that will become flooded by seawater. They think that where these areas are valuable, various forms of sea defence can save them.

Groundwater flows to the lowest accessible base level. Rivers may intercept it, but the ultimate base level is the sea. For groundwater to flow to the sea, there needs to be a minimum gradient to drive the water through the aquifers. This gradient will remain the same if sea levels rise. As a result, when sea levels rise, groundwater levels inland will rise. The rise in level at the coast is transferred inland by groundwater backing up. It is valuable to think of the consequences of this progressive rise of waters inland particularly in urban areas like Belfast, Dublin and Cork.

The historic infrastructure of our coastal towns and cities was often built in relation to groundwater levels. Before waterproof concrete, cellars were not built below the water table. The vaults of banks and the crypts of churches were also built above the water table. Sewers, where possible, were built above the water table so that when they leaked, fluids leaked out, rather than groundwater leaking in. The hidden inland consequences of sea level rise can be illustrated by considering the impact on the sewers. When groundwater levels rise there will be two problems. The first is that the sewers will be carrying more water. There will be an increased base flow of groundwater occupying space in the sewers. Many of the old sewers take runoff from the houses and roads. Therefore, if part of the capacity of the sewer is taken by groundwater there will be less space for storm runoff, and the sewers will be more likely to 'surcharge'; the manholes will lift and sewage and storm water will run down the streets. A second consequence, of sewers gathering an increased amount of groundwater, is the cost. Take Dublin as an example; the City Council pays for sewage treatment in the Ringsend works by the cubic metre. Therefore, a rise in sea levels, leading to a rise in groundwater levels, will lead to an increased cost in sewage treatment (assuming that the sewage works is not under sea water). Therefore, groundwater will transmit the impact of sea level rise inland with often hidden costs.

A consequence of sea level rise will also be felt inland along tidal rivers. At the moment groundwater flows to the flood plains along these rivers. Part of the groundwater flows through the alluvial sediments into the river. The excess emerges as springs along the change of slope between the valley sides and the flood plain. A consequence of raising the base level is that there will be no gradient to drive the water through the flood plain. It will probably revert to a salt marsh. All fresh groundwater will need to emerge through the springs and seepages. The water level will back up and adjacent pasture or gardens in housing estates will be less free draining.

A distressing consequence of sea level rise and inland groundwater level rise will be the impact on graveyards close to the coast. This will affect island communities as well as the mainland.
Coastal septic tanks and percolation areas will be impacted, as sea level rise and water table rise will reduce the thickness of the unsaturated ground below percolation areas. The result will be less opportunity for breakdown of the effluent before it reaches the water table and flows out through the beach. Homes in

areas such as Brittas Bay will be affected; beaches and bathing waters will be affected by greater nutrient enrichment.

SEA LEVEL RISE AND SALINE INTRUSION

Seawater is denser than fresh water. Seawater will flow under fresh water in tidal rivers. It can be seen every day when the tide turns in the Liffey or Lee. Similarly, in karst limestones with large diameter open conduits, fresh water can flow out of upper cavities whilst salt water flows in and out of lower cavities with the tides. The predicted sea level rise will undoubtedly alter the current balance.

Kinvara and the Gort lowlands already have a problem at the end of dry summers. At the moment, if the Kinvara water supply boreholes are not turned off one hour before and two hours after a spring high tide, then seawater can be drawn inland and into the water supply source. It will be very difficult to manage the pumping from the water supply boreholes when sea levels rise even a small amount. The balance between groundwater flowing out to the coast and the level of the sea is very delicate. It is almost inevitable that water supply boreholes will need to be moved inland in such areas of low groundwater gradients and conduit flow through karst limestone. Again, small islands will be particularly affected, for example Inisheer.

Eventually, the habitats in turloughs will be affected. First the levels may not fall to the existing levels and summer feeding grounds will remain under water. Second saline water will flow up the conduits and create inland salt loughs.

A rise in sea level will cause inland flooding. It will, exacerbate the existing problems of flooding in the Gort lowlands but also other areas of Galway, Kerry and the Cork - Waterford synclines.

Finally a rise in sea level will affect the dewatering of quarries close to the coast in limestone with karst conduits. Deep quarries are dewatering the upper karst systems. A rise in sea level and a rise in base level will mean an increase in flow, and in extreme circumstances a flow of salt water back from the coast into the quarry. Quarries discharge their water into adjacent rivers. Sea level rise is a factor to be considered in planning applications for new quarries close to the coasts.

SEA LEVEL RISE AND RIVERS

There is a delicate overall balance in our estuaries and tidal sections of our river between the sea and the flow in rivers. This overall balance will be altered by sea level rise. There will be the impact of salt water further up stream. This may have an impact on the position of river intakes for public water supplies. There will also be an impact on the sediment carried by the rivers. The scouring action of present flows and tides will be altered, and sedimentation will increase in channels. The Shannon will not be affected because there is a large fall in the lower course of the river just before it enters Limerick, but rivers such as the Suir, Nore, Barrow and Slaney will be affected.

CHANGES IN RAINFALL DISTRIBUTION AND INTENSITY – IMPACT ON INLAND WATERS

Predicting the rise in sea levels, and the consequent impact on groundwater levels is relatively straightforward. The main uncertainty is the speed of the rise. The consequences of the rise are relatively easy to predict.

Global warming introduces more energy into the atmosphere and the hydrological cycle. In general, the atmosphere will be more active. The atmosphere can retain more moisture. Evaporation rates will increase and precipitation will be greater. An increase in rainfall will increase throughflow in the combined surface water groundwater drainage system.

The consequences of global warming climate change in broad general terms are predictable. However, the distribution and intensity of these effects will not be even. Climate models have given us predictions about general changes in climate in different parts of the world. They are not, and cannot be, very specific about local effects. Recent work is struggling with understanding and ultimately predicting the nature and extent of extreme weather/climate events. Hydrologists and hydrogeologists think at a local level, and catchment level, rather than at a global level or country level, because the surface and the subsurface are not uniform. We cannot take a 50 km by 50 km grid square and assume that its characteristics are uniform, whereas, a climate scientist can reasonably assume that the changes in the atmosphere across a 50 km square grid will be small at the different levels. At the moment hydrologists and hydrogeologists have little specific information with which to understand and predict the impact on water resources.

I think it is worthwhile to try to describe potential changes in water resources, even with imprecise predictions, because it causes us to think about processes and systems, and the strength and weakness of our understanding.

Climate change is indicating a re-distribution of rainfall amounts during the year, and an increase in extreme weather events. When considering these changes, it draws attention to our knowledge of the effective permeability of soils and subsoils, and available storage in the overburden and the bedrock. These characteristics determine the change in the amount of immediate run off and the amount of recharge, assuming that slope and vegetation have not changed.

Winter rain and decrease in summer rain with more extreme rainfall events:
The effect of this predicted change is perhaps already being observed. We have just experienced a very wet November-December with widespread flooding followed by a cold dry January-February. We first noticed this pattern of heavy intense rain and periods with little rain in the mid 1990's when we were trying to understand the extreme flooding in the South Galway lowlands.
The impact on surface water and groundwater depends upon the permeability of the overburden and the depth of the unsaturated zone above the water table.
Three examples serve to explain the impacts; low permeability surfaces, and open rock or high permeability surfaces with deep or thin unsaturated zones.

Low permeability surfaces and overburden:
Heavy winter rain will often not be absorbed by the soil and overburden. This will obviously lead to the short-term problem of increased run off and surface water floods. However, it also has a long-term effect. Recharge to the groundwater system will be less through low permeability overburden. This will lead to lower baseflows from groundwater into streams and rivers in spring and summer. Therefore summer river flows will be less, not just because there is less summer rain, but also because there was too much winter rain. The winter rain could not be captured and stored in the groundwater system for later slow release back into the rivers. This will have the following consequences.

Less recharge in winter will mean lower groundwater levels in summer. Water supply borehole pumping rates may need to be throttled back. Pumps may be unwisely lowered in boreholes below the protective pump chamber casing, and shallow contaminated groundwater drawn into the supply. Spring flow will be reduced and water levels in rivers will be less. Changes in rainfall distribution will lead to an impairment of both surface and groundwater sources for drinking water supplies to towns, villages and rural homes.

This impact will be more noticeable in the eastern part of the country, where the overburden is thicker and less permeable.

An increase in the proportion of groundwater in the reduced river flow will alter the chemistry of the rivers. An increase in the calcium bicarbonate level will have no effect, but an increase in phosphates, nitrates, organic carbon from groundwater, added to less diluted treated effluent from waste water treatment plants, will lead to an increase in algal blooms in both rivers and estuaries. We might expect an increase in total fish kills in rivers draining areas with low permeability soils.

Even without algal blooms, the nighttime release of $CO2$ from river plants will be into a smaller volume of flow. There will be an increased, and perhaps mortal stress, on fish in the hours just before dawn when photosynthesis and oxygen production can commence. This may have an adverse impact on small salmonids.

High permeability surface materials, or open rock, but with a thin unsaturated zone:
With these characteristics the ground can absorb the intense rainfall, but the space to store the water is not available. The water fills up the unsaturated zone, the water table reaches the surface, and potential recharge is rejected. The result later in the year is the same as described above, with perhaps a greater risk of groundwater quality affecting river water quality because the travel time for the breakdown of contaminants from septic tanks will be shorter.

In both the above examples, there will be prolonged surface flooding and waterlogging of soils. We may find that we need to extend the ban on landspreading of slurry, or resist the frequent demands for extension of the period for spreading.

High permeability surface materials, or open rock outcrop with a thick unsaturated zone:
These characteristics are often associated with our regionally important bedrock aquifers and important gravel aquifers. The consequences of intense winter rainfall are also dependent on the storage in the aquifer.

These characteristics will enable the increased rainfall to recharge the aquifer. The result will be higher groundwater levels or heads, which in turn will lead to higher discharges of groundwater in winter and spring. The problems that will arise relate to the capacity of the combined groundwater surface water system downgradient of the recharge area. If the down gradient system is not able to take the higher groundwater flow rates, then there will be flooding, such as occurred in Galway, Roscommon east Mayo last year.

A consequence of high flow rates through conduit flow aquifers is the increase in turbidity in spring water used as a water supply. Inadequately constructed boreholes will have more frequent problems with turbidity and contamination by pollutants flushed into the shallow groundwater system.

High recharge rates into karst limestone aquifers will re-flood the upper karst conduits that had been depleted by quarry dewatering, or drainage along road cuttings. This may have an impact on the economic activity in certain quarries or the capacity of SUDS systems associated with our modern roads.

The larger volume of recharge and the faster through flow in the aquifers will provide shorter residence times for water in the aquifers. This will influence the water chemistry of bottled waters. The waters will be less mineralised in winter, and perhaps more mineralised in late summer. This is not necessarily an adverse impact but it will mean that the label on the outside does not describe accurately the content on the inside.

Rivers in late summer will generally receive less base flow from these aquifers. The main flow will have taken place in spring and early summer.

INCREASED TEMPERATURE

Increased temperatures without a radical reduction in rainfall appear to bring benefits, yet could also have negative impacts.

Warmer temperatures will increase plant growth in winter and spring when there is an abundance of rainfall and soil moisture. Therefore agricultural production could increase. Warmer temperatures may mean that there is earlier and more effective uptake of nutrients after landspreading. Increased water uptake by plants will also reduce recharge in spring and perhaps exacerbate problems arising from insufficient recharge in winter. Trees draw water from deep in the overburden and top of the bedrock, therefore increased temperatures and tree growth will deplete shallow groundwater resources. For example spring fed streams in wooded catchments may frequently be dry in late summer. Increasing temperatures will increase demand for groundwater for irrigation. Protected wetlands or special groundwater fed habitats down gradient of golf courses may be desiccated.

Increased atmospheric and soil temperatures will lead to warmer groundwater. An obvious impact will be a benefit to our low-grade geothermal resources. We used to think that groundwater temperatures were constant, but we are finding evidence that they rise in autumn and decline in spring and summer. We may find that groundwater is temporarily warmer in regionally important aquifers than in less productive aquifers with less recharge. Therefore, groundwater is a heat sink, and perhaps removes some of the rise in atmospheric heat and transfers it back into the earth. There is much talk and research into carbon capture, but little investigation of heat capture and storage by groundwater. It is worth bearing in mind that some groundwater flows along a very long travel path. The recovery of the heat ‘captured’ by groundwater reduces our CO_2 emissions. It is a renewable energy source.

Warmer groundwater also brings some uncertainties. Our soil, overburden and bedrock groundwater systems contain an abundant flora and fauna. These small, poorly understood, organisms break down our effluent discharge on, or into, the subsurface. At the moment, we are relying upon them to persist and continue with their invaluable role. We know so little about them that we assume they will remain a constant ecosystem, even when temperature, rainfall, drought and land use will change. Climate change may bring about long-term changes in the subsurface. For example, subsurface microbial action may produce more CO_2, which may increase the speed of karst processes in our limestone aquifers.

CONCLUSION

Climate change is happening and it appears to be an accelerated global warming. Many of the World’s scientists and decision makers are trying to predict or respond to the probable impacts. I suggest that we take every opportunity to become in engaged, because we have a valuable understanding of the largest, liquid fresh water resource on earth. We also have a valuable role in explaining the checks and balances, problems and consequences arising from a change in climate and sea levels. It is not too difficult to predict that certain things will happen as sea levels rise or rainfall changes. I suggest that we owe it to society to explain what we know to decision makers, engineers, planners and the general public. We might allay some fears and draw attention to problems that others have not considered. However, the biggest challenge we face from climate change is verifying and improving our understanding of recharge, groundwater storage and groundwater flow systems. We are making a start within the Water Framework Directive, but we must continue to refine, adjust, change and, maybe, re-think some of our preliminary assessments.

IMPLICATIONS OF CLIMATE CHANGE ON GROUNDWATER RESOURCES

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ABSTRACT

Groundwater is an important natural resource and an essential part of the hydrologic cycle. Worldwide, it has been estimated that more than 2 billion people depend on groundwater for their daily water supply. A large proportion of the world's irrigated agriculture is dependent on groundwater, as are a large number of industries, and groundwater is also critical in sustaining streams, lakes and wetland ecosystems. In many countries, excessive groundwater development, encroachment on recharge areas, uncontrolled urban and industrial discharges, contamination by naturally occurring chemicals and agricultural intensification have compromised the ability of groundwater to help resolve the emerging water management challenges in the 21st century. Furthermore, the Intergovernmental Panel on Climate Change has highlighted the implications of accelerated climate change for groundwater. Changes in rainfall, evaporation and soil moisture conditions leading to changing patterns of recharge and runoff patterns are expected to add to the resource management burden for both groundwater depletion and rising water tables. Therefore, there is an urgent need for new strategies for groundwater governance in order to maintain the availability of high-quality groundwater resources to meet human, economic and ecosystem needs in the face of climate change. However, the added uncertainty of global environmental and climate change may reinforce sound resource management, leading to possible beneficial impacts for groundwater systems, even if the projected climate appears less favourable.

INTRODUCTION

The vast store of water beneath the ground surface has long been realised as an invaluable source of water for human consumption and use. Groundwater development dates from ancient times, as demonstrated by the wells and horizontal tunnels known as qanats (ghanats) or aflaj that originated in Persia about 3,000 years ago. In more recent times, the location of groundwater resources has been crucial in the economic development of rural and agricultural areas such as the Great Artesian Basin aquifer in the outback of Australia (Prescott and Habermehl, 2008) and the High Plains aquifer in the mid-section of the United States (Dennehy *et al*., 2002), and also urban and industrial areas such as the Cretaceous Chalk aquifer underlying the London Basin (Price, 2002) and the Quaternary alluvial aquifer of the North China Plain (Foster *et al*., 2004).

Unfortunately groundwater is often unacknowledged and undervalued resulting in adverse environmental, economic and social consequences. The over-exploitation of groundwater by uncontrolled pumping can cause detrimental effects on neighbouring boreholes and wells, land subsidence, saline water intrusion and the drying out of surface waters and wetlands. Groundwater pollution from uncontrolled use of chemicals and the careless disposal of wastes on land cause serious impacts requiring difficult and expensive remediation over long periods of time. Thus, achieving sustainable development of groundwater resources is a major challenge for the 21st century, in addition to managing the anticipated impacts of climate change on the availability of water resources. This paper highlights these challenges and assesses the actions needed now to protect groundwater from further uncontrolled development and degradation in the face of climate change.

CHALLENGES OF GROUNDWATER RESOURCES DEVELOPMENT

Total global fresh water use is estimated at about 4,000 km^3/year (Margat and Andréassian, 2008) with 99% of the irrigation, domestic, industrial and energy use met by abstractions from renewable sources, either surface water or groundwater. Less than 1% (currently estimated at 30 km^3/year) is obtained from non-renewable (fossil groundwater) sources mainly in Algeria, Libya and Saudi Arabia. With rapid population growth, groundwater abstractions have tripled over the last 50 years, largely explained by the rapid increase in irrigation development stimulated by food demand in the 1970s and by the continued growth of agriculture-based economies (World Bank, 2007). Emerging market economies such as China, India and Turkey, which still have an important rural population dependent on water supply for food production, are also experiencing rapid growth in domestic and industrial demands linked to urbanisation. Urbanised and industrial economies such as the European Union and the United States import increasing amounts of food and manufactured products, while water use in industrial processes and urban environments has been declining, due to both technological changes in production processes and pollution mitigation efforts (WWAP, 2009).

Groundwater is an important natural resource. Worldwide, more than 2 billion people depend on groundwater for their daily supply (Kemper 2004). Aquifers, which contain 100 times the volume of fresh water that is to be found on the Earth's surface, supply approximately 20% of total water used globally, with this share rising rapidly (Figure 1), particularly in dry areas (IWMI, 2007). This rise has been stimulated by the development of low-cost, power-driven pumps and by individual investment for irrigation and urban uses. Globally, 65% of groundwater utilisation is devoted to irrigation, 25% to the supply of drinking water and 10% to industry. Groundwater resources account for more than 70% of the water used in the European Union, and are often the only source of supply in arid and semi-arid zones (100% in Saudi Arabia and Malta, 95% in Tunisia and 75% in Morocco). Irrigation systems in many countries depend very largely on groundwater resources (90% in Libya, 89% in India, 84% in South Africa and 80% in Spain) (Zektser and Everett, 2004).

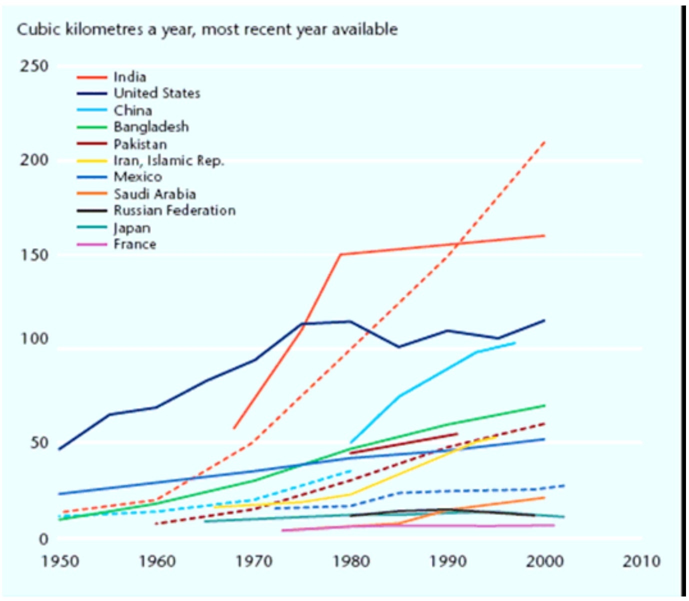

Figure 1: Growth in groundwater use, 1950-2000. Note that countries with two lines have different datasets that do not reconcile. After Margat (2008).

ADAPTATION TO CLIMATE CHANGE

The Intergovernmental Panel on Climate Change has highlighted the implications of accelerated climate change for groundwater (IPCC, 2007). Changes in rainfall, evaporation and soil moisture conditions (Figure 2) leading to changing patterns of recharge and runoff patterns are expected to add to the resource management burden for both groundwater depletion and rising water tables, depending on the region. However, these impacts must be considered in comparison to the stresses placed on groundwater systems by current socio-economic drivers.

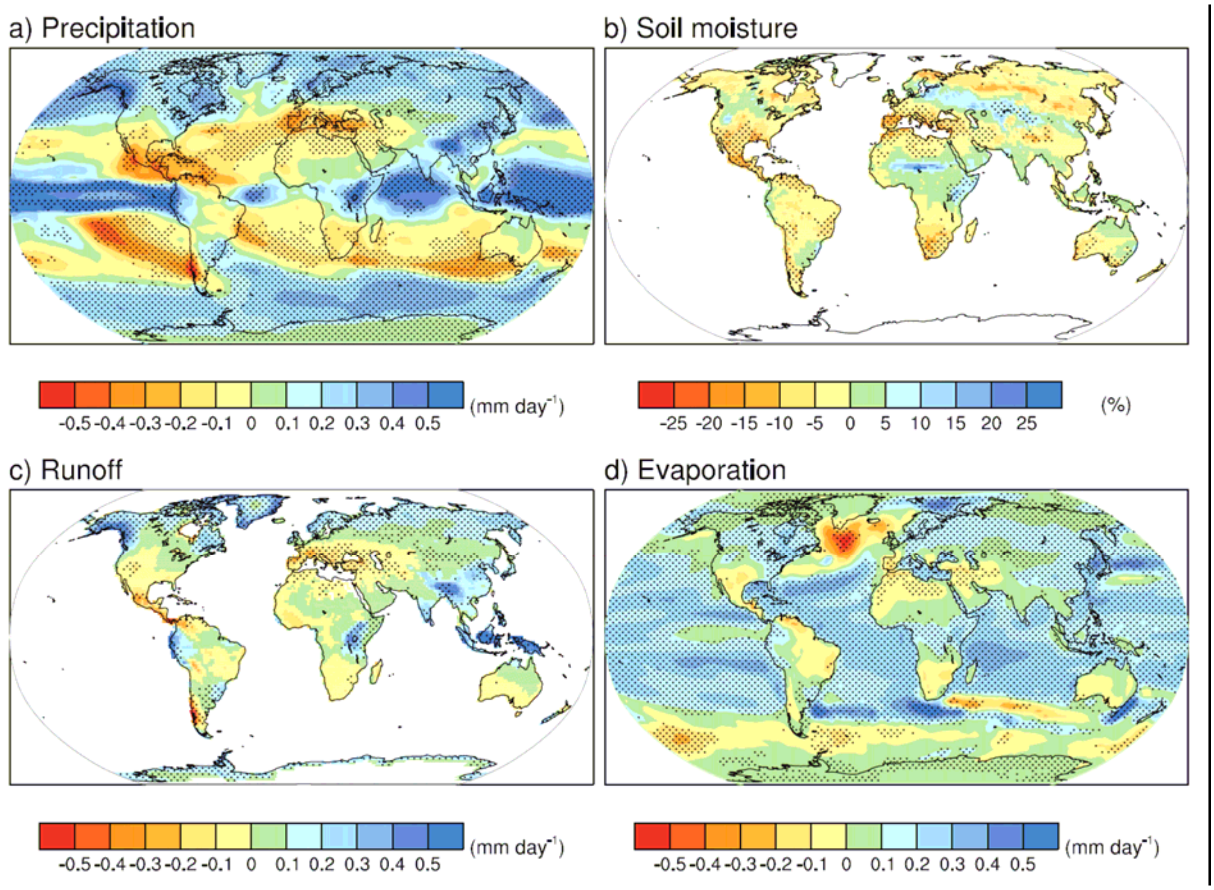

Figure 2: Multi-model mean changes in: (a) precipitation (mm/day), (b) soil moisture content (%), (c) runoff (mm/day) and (d) evaporation (mm/day). To indicate consistency in the sign of change, regions are stippled where at least 80% of models agree on the sign of the mean change. Changes are annual means for the medium, A1B scenario 'greenhouse gas' emissions scenario for the period 2080-2099 relative to 1980-1999. Soil moisture and runoff changes are shown at land points with valid data from at least 10 models. After Collins *et al*. (2007).

Climate change challenges the traditional assumption that past hydrological experience provides a good guide to future conditions (IPCC, 2007). In times of surface water shortages during droughts, groundwater resources are often abstracted as an emergency supply. Under conditions of climate change, this response is likely to be unsustainable, especially in those areas expected to experience an increase in drought frequency and duration. Also, rising sea levels under climate change will further threaten coastal fresh water aquifers, especially those already experiencing salinisation due to over-exploitation.

One of the major challenges facing water resources managers is coping with climate change uncertainties in the face of real-world decision-making, particularly where expensive investment in infrastructure such as well-field design, construction, and testing and laying of pipelines is required. As discussed by Dessai and Hulme (2007), this challenge presents a number of new questions, for example how much climate change uncertainty should we adapt to? Are robust adaptation options socially, environmentally and economically acceptable and how do climate change uncertainties compare with other uncertainties such as changes in demand? Also, what relationships between supply and demand for groundwater exist in particular regions, and how might global change affect these relationships and feedbacks? The answers to these questions leading to robust adaptation decisions will require the development of probability distributions of specified outcomes (Wilby and Harris, 2006) and negotiation between decision-makers and stakeholders involved in the adaptation process (Dessai and Hulme, 2007). For lower income countries, availability of resources and building adaptive capacity are particularly important in order to meet water shortages and salinization of fresh waters (IPCC, 2007).

According to the IPCC (2007), the array of potential adaptive responses available to human societies is very large, ranging from purely technological (for example, deepening of existing boreholes), through behavioural (altered groundwater use) to managerial (altered farm irrigation practices), to policy (groundwater abstraction licensing regulations). The IPCC (2007) argued that while most technologies and strategies are known and developed in some countries (for example, demand-management through the conjunctive use of surface water and groundwater resources), the effectiveness of various options to fully reduce risks for vulnerable water-stressed areas is not yet known, particularly at higher levels of global warming and related impacts. Table 1 summarises supply-side and demand-side adaptation options designed to ensure supplies of water and groundwater during average and drought conditions. As explained by the IPCC (2008), supply-side options generally involve increases in storage capacity or water abstraction. Demand-side adaptation options rely on the combined actions of individuals (industry users, farmers (especially irrigators) and individual consumers) and may be less effective. Indeed some options, for example those incurring increased pumping and treatment costs, may be inconsistent with climate change mitigation measures because they involve high energy consumption.

Examples of current adaptation to observed and anticipated climate change in the management of groundwater resources are few, with groundwater typically considered as part of an integrated water-supply system. The ability of California's water supply system to adapt to long-term climate and demographic changes is examined by Tanaka *et al*. (2006) who concluded that the water supply system appears physically capable of adapting to significant changes in climate and population, albeit at significant cost. Such adaptations would entail large changes in the operation of California's large groundwater storage capacity, significant transfers of water among water users and some adoption of new technologies. In contrast to this example from North America, Ojo *et al*. (2003) discussed the downward trends in rainfall and groundwater levels, and increases in water deficits and drought events affecting water resources availability in West Africa. In this region, the response strategies needed to adjust to climate change emphasize the need for water supply-demand adaptations. Moreover, the mechanisms needed to implement adaptation measures include: building the capacity and manpower of water institutions in the region for hydro-climatological data collection and monitoring; the public participation and involvement of stakeholders; and the establishment of both national and regional co-operation.

FUTURE CHALLENGES FOR GROUNDWATER MANAGEMENT

Three aquifer characteristics determine whether groundwater resources will ultimately prove sustainable: vulnerability to pollution under contaminant pressure from the land surface; susceptibility to irreversible degradation from excessive exploitation; and renewability of storage reserves under current and future climate regimes. These characteristics vary widely by aquifer type and hydrogeologic setting.

The tension between private and public services derived from aquifers remains. More convergent and sustainable resource use will be achieved only through substantial investment in management operations on the ground, working primarily through community consultation and cross-sectoral policy dialogue (WWAP, 2009). Adopting an adaptive management approach (Figure 3) it should be possible to establish acceptable regulations, adopted by all parties, based on a holistic definition of the aquifer system and understanding of the impacts of abstraction and contamination.

Table 1: Types of climate change adaptation options for surface water and groundwater supply and demand (based on IPCC, 2008)

Supply-side	Demand-side
Increase storage capacity by building reservoirs and dams	Improve water-use efficiency by recycling water
Desalinate seawater	Reduce water demand for irrigation by changing the cropping calendar, crop mix, irrigation method and area planted
Expand rain-water storage	Promote traditional practices for sustainable water use
Remove invasive non-native vegetation from riparian areas	Expand use of water markets to reallocate water to highly valued uses
Prospect and extract groundwater	Expand use of economic incentives including metering and pricing to encourage water conservation
Develop new wells, deepen existing wells	
Maintain well condition and performance	Introduce drip-feed irrigation technology
Develop aquifer storage and recovery systems	Licence groundwater abstractions
Develop conjunctive use of surface water and groundwater resources	Meter and price groundwater abstractions
Develop surface water storage reservoirs filled by wet season pumping from surface water and groundwater	
Develop artificial recharge schemes using treated wastewater discharges	
Develop riverbank filtration schemes with vertical and inclined bank-side wells	
Develop groundwater management plans that manipulate groundwater storage, e.g. resting coastal wells during times of low groundwater levels	
Develop groundwater protection strategies to avoid loss of groundwater resources from surface contamination	
Manage soils to avoid land degradation to maintain and enhance groundwater recharge	

A significant challenge for the future development of groundwater sources is to raise political awareness of the issues involved. Unfortunately, increased scientific understanding of groundwater has not yet made a significant influence on resource policy-making or featured prominently in global or national water policy dialogues, with discussion too often on groundwater development rather than groundwater management. Also, governance and practical management are not well funded and, as a consequence, opportunities for utilising groundwater resources sustainably and conjunctively are being lost and insufficient attention is being paid to the inter-relationship between groundwater and land-use planning (IAH, 2006). Often, decisions on groundwater development and management objectives, and the allocation of human, financial and environmental resources to meet these objectives are made by leaders in government, the private sector and civil society, not by groundwater professionals alone. Therefore, hydrogeologists must help inform the decisions of these leaders outside the water domain on such issues as spatial and development planning, and agricultural, energy and climate change policies.

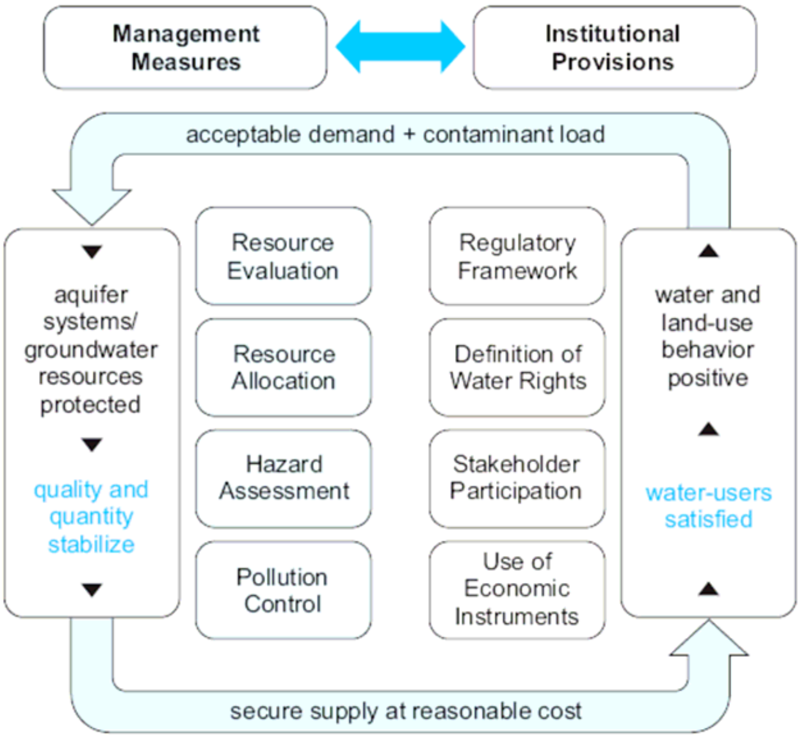

Figure 3: Integrated, adaptive management scheme for the protection of groundwater resources. After IAH (2006).

CONCLUSIONS

Groundwater resources stored in aquifers can be managed given reasonable scientific knowledge, adequate monitoring and sustained political commitment and provision of institutional arrangements. Although there is no single approach to relieving pressures on groundwater resources given the intrinsic variability of both groundwater systems and socio-economic situations, incremental improvements in resource management and protection can be achieved now and in the future under climate change. Future sustainable development of groundwater will only be possible by approaching adaptation through the effective engagement of individuals and stakeholders at community, local government and national policy levels. The added uncertainty of global environmental and climate change may reinforce sound resource management, providing additional social and political impetus for science-based practice. In this way, the anticipation of change may have beneficial impacts on groundwater systems, even if the projected climate appears less favourable.

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