INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS – Irish Group

Water Resource Management: The role of hydrogeology

Proceedings of the 34th Annual Groundwater Conference
Tullamore, Co. Offaly, Ireland

15th and 16th April 2014

www.iah-ireland.org

INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS
(IRISH GROUP)

Presents

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Tullamore Court Hotel
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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, Treasurer, Burdon Secretary, 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 provides 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

2014 Conference Objective

As with previous years, the 2014 IAH (Irish Group) Groundwater Conference can be expected to benefit hydrogeologists, engineers, local authorities, consultants, planners, environmental scientists, public health officials, professionals and practitioners from a variety of sectors involved with groundwater.

2014 is the 34th Anniversary of the Annual IAH (Irish Group) Groundwater Conference. This year's theme is entitled '*Water Resource Management: The role of hydrogeology*'. The two-day event is being held at the Tullamore Court Hotel and combines an impressive array of national and international speakers with exhibits, poster presentations, fine dining and a social evening. In general the conference will be broken down into the following main areas.

1. Drilling & Well Design
2. Hydrogeochemistry & Water Quality
3. Groundwater Dependent Ecosystems
4. Karst Challenges
5. Mining
6. Management of Groundwater Supply Sources

IAH (Irish Group) President David Drew will initiate proceedings with an introduction and welcome address. David will chair the opening session on ‘Drilling and Well Design’ and will begin by introducing the keynote speaker – Neven Kresic, from AMEC Environment & Infrastructure, Inc., USA, who co-chairs the karst commission of the International Association of Hydrogeologists. Neven will talk about his experiences of well drilling and testing in karst. David Ball, consultant hydrogeologist, was lead author on a recently published EPA technical report advising best practice in borehole construction and wellhead protection, and this forms the basis of his talk.

Delegates are encouraged to engage with the various panels of speakers within regular fifteen minute question and answer sessions. After a coffee break, Session 2 will commence with Kevin Cullen from Verde Environmental Group who describes a project which has mapped the geochemical signatures of bottled drinking waters across Europe. This is followed by Catherine Coxon (Trinity College Dublin) who will provide a comprehensive review of Irish research data on karst groundwater quality and what implications this has for education and management in the future.

Before lunch, postgraduate students who have agreed to present posters at the conference will be invited to summarise their ongoing research in the field of hydrogeology.

After lunch, we will commence Session 3: Groundwater Dependent Ecosystems. We are pleased to welcome Antonio Chambel of University of Evora, Portugal, and elected Vice-President (Programme and Science Coordination) of the International Association of Hydrogeologists. Antonio will analyse a coastal lagoon in Portugal which is improving understanding of such sites as groundwater-fed ecosystems. Sarah Kimberley will then present findings of her reseach undertaken with Trinity College Dublin, which looked at damage suffered by groundwater dependant terrestrial ecosystems in Ireland. In the third talk after lunch Gareth Farr of the British Geological Survey continues on this theme, looking in detail at the methods that have been used to assess such damage, and some of the experiences in the UK to date.

In the final session of Day 1 Pat Byrne and Matt Craig of the Environmental Protection Agency will present a soon to be published guidance document entitled ‘Guidance on the Authorisation of Direct Discharges to Groundwater.’ This will be followed by Owen Naughton of Trinity College Dublin who will summarise recent reseach, funded by the Office of Public Works, in which he looked into groundwater flooding from karst aquifers.

At the end of a hard day, weary delegates will be treated to an evening of whiskey tasting and a light meal as part of a tour at the redeveloped Tullamore Dew Distillery. The social event will provide plenty of opportunity for attendees to catch up with friends and make new acquaintances.

Day 2 of the conference shall begin with a session on mining. The session opens with Simon Sholl from Schlumberger Water Services who will explain a new method being used to monitor groundwater at mine sites. Billy O’Keeffe from the Exploration and Mining Division in the Department of Energy, Communications and Natural Resources will then present a recently published guidance document on managing discharges to groundwater during exploration drilling.

The final session consists of four talks that will consider different aspects of groundwater supply sources. Conor Lydon of WYG will provide an update on the Rural Borewell Scheme which is underway and aims to provide isolated rural dwellers in Northern Ireland with a sustainable groundwater supply. This will be followed by Monica Lee of the Geological Survey of Ireland who will outline their collaboration with the National Federation of Group Water Schemes which aims to provide a complete set of preliminary source protection plans to Group Water Schemes in Ireland. The final two presentations come from the west of Ireland. Martin Lavelle (Galway County Council) intends to enlighten us with some of the practical issues faced by local authorities in trying to provide potable water, on a consistent basis, in a karstic landscape. This talk will be complemented by Pamela Bartley of Hydro-G who will share her experience of how a hydrogeologist can apply solutions at groundwater abstraction points and the importance of adhering to existing guidelines.

Prior to lunch on Day 2 a final Q&A session will be followed by a closing address by the IAH (Irish Group) Conference Secretary, Colin O'Reilly.

Following the positive feedback and the success of the first technical workshops held last year in Tullamore, the IAH (Irish Group) has decided to continue with the workshops for 2014. It is hoped that this will continue to take place in an informal and interactive environment where delegates can learn (and critique) different field techniques, and bring their own experiences to the debate. Peter Conroy will consider in detail the grouting stage of borehole construction and the potential difficulties that may be encountered on site. Paul Wilson from the Geological Survey of North Ireland/British Geological Survey will then talk about the application of the bailer test as a quick, simple and consistent method of assessing the performance of a borehole.

2014 IAH (Irish Group) Committee:

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

Sources of photographic imagery on the proceedings cover courtesy of Eleanor Burke, John Dillon, Orla O'Connell, Colin O'Reilly,

The proceedings for the 34th Annual Groundwater Conference 2014 will also be made available digitally on the IAH-Irish Group website within the next six months.

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The IAH (Irish Group) would also like to acknowledge the support of the following members and organisations whose staff have worked on the committee of the IAH-Irish Group throughout the year and helped to organise the conference:

David Drew

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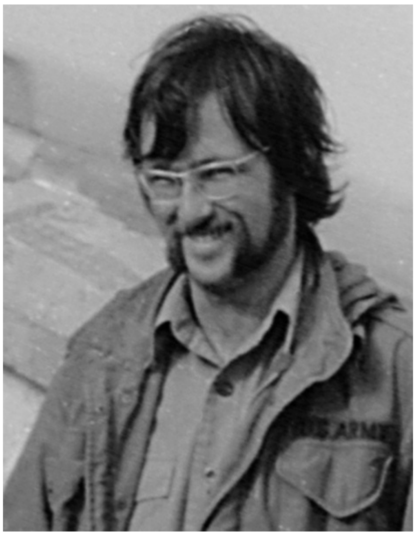

Eugene Daly

12th September 1947 - 28th January 2014

The IAH (Irish Group) wishes to honour the numerous important contributions made by Eugene Daly to hydrogeology in Ireland.

Eugene was a founder member of the Irish Group and was our President from 1994 - 1997.

He was a pioneer. After returning from hydrogeological training in the US, his drive in the Groundwater Section of the Geological Survey started the process of using modern scientific methods and equipment to explore and understand the groundwater resources of Ireland. Eugene, with Bob Aldwell, fought to establish the foundations of modern hydrogeology in Ireland in the early 1970's, and gain recognition for our profession.

Eugene encouraged many aspiring hydrogeologists throughout his life, and was instrumental in recruiting several of the subsequent leaders. He made many contributions to the IAH through field trips, workshops, technical meetings and papers at the Annual Conference. He worked extensively on the Nore Basin, and wrote several important hydrogeological reports whilst with the Survey for 23 years, and more recently as a consultant.

He put a lot of energy into setting up the Institute of Geologists of Ireland and served as a Vice President. Throughout his life, he always insisted on high standards, care and scientific accuracy. He was a consummate field hydrogeologist, one who was immensely curious, knowledgeable and enthusiastic.

Eugene was still discussing hydrogeology and hydrogeologists in the days shortly before he died. Members of his profession, both old and young, respect him, and have heartfelt gratitude for his encouragement as a mentor to others throughout his life. He is missed with genuine sadness by all of us.

‘Water Resource Management: The role of hydrogeology’
34th Annual Groundwater Conference

International Association of Hydrogeologists – Irish Group
Tullamore Court Hotel, Tullamore, Co. Offaly: Tuesday 15th & Wednesday 16th April 2014

Programme Day 1, Tuesday 15th April

08:30 - 09:30 *Conference Registration; Tea, Coffee, & Exhibits*

INTRODUCTION
09:30 – 09:40 Welcome and Introduction
David Drew – *President IAH Irish Group*

SESSION 1: DRILLING AND WELL DESIGN

09:40 – 10:25 ‘Experience of well drilling and testing in karst’ – Neven Kresic (AMEC Environment & Infrastructure, Inc., USA)

10:25 – 11:00 ‘How to construct a water supply borehole in Ireland’ – David Ball

11:00 – 11:15 Q & A

11:15 – 11:35 *Tea and coffee*

SESSION 2: HYDROGEOCHEMISTRY AND WATER QUALITY

11:35 – 12:00 ‘The geochemistry of European bottled waters’ – Kevin Cullen (Verde Environmental Group)

12:00 – 12:25 ‘Water quality in Irish karst aquifers’ – Catherine Coxon (Trinity College Dublin)

12:25 – 12:40 Q & A

12:40 – 12:55 Student Poster Presentations

12:55 – 14:00 *Buffet lunch in Tullamore Court Hotel*

SESSION 3: GROUNDWATER DEPENDENT ECOSYSTEMS

14:00 – 14:35 ‘Coastal lagoons and their watersheds as GWDE: A case study in the southwest coast of Portugal’ – Antonio Chambel (University of Evora)

14:35 – 15:00 ‘Assessing significant damage to selected Irish GWDTE types as part of groundwater body classification under EU Water Framework Directive’ – Sarah Kimberley

15:00 – 15:25 ‘Ecohydrological methods for the investigation of significant damage at GWDTE’ – Gareth Farr (BGS)

15:25 – 15:40 Q & A

15:40 – 15:55 *Tea and coffee*

SESSION 4: KARST CHALLENGES

15:55 – 16:25 ‘Guidance on the authorisation of direct discharges to groundwater’ – Pat Byrne & Matt Craig (Environmental Protection Agency)

16:25 – 16:50 ‘Groundwater flooding in Irish karst groundwater flow systems’ – Owen Naughton (Trinity College Dublin)

16:50 – 17:05 Q & A

19:00 *Social event, including light evening meal, at the Tullamore Dew Distillery, sponsored by IAH (Irish Group).*

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International Association of Hydrogeologists – Irish Group
Tullamore Court Hotel, Tullamore, Co. Offaly: Tuesday 15th & Wednesday 16th April 2014

Programme Day 2, Wednesday 16th April

9:00 – 9:30	*Tea, Coffee & Exhibits*
SESSION 5:	**MINING**
09:30 – 09:55	‘New method for groundwater monitoring at mine sites’ – Simon Sholl (Schlumberger Water Services)
09:55 – 10:20	‘Exploration drilling – Guidance on discharge to groundwater’ – Billy O’Keeffe (Exploration and Mining Division, Department of Energy, Communications and Natural Resources)
10:20 – 10:35	Q & A
10:35 – 10:55	*Tea & Coffee*
10:55 – 11:20	‘Rural borewell scheme exploratory drilling providing qualitative and quantitative hydrogeological information in Northern Ireland’ – Conor Lydon (White Young Green)
11:20 – 11:45	‘Source protection for Group Water Schemes – working with local communities’ – Monica Lee (Geological Survey of Ireland/National Federation of Group Water Schemes)
11:45 – 12:10	‘Difficulties in maintaining post-treatment quality in public water supplies’ – Martin Lavelle (Galway County Council)
11:45 – 12:10	‘Groundwater as a source of public water supply: technical problems with existing sources and drilling solutions sought’ – Pamela Bartley (Hydro-G)
12:35 – 12:50	Q & A
12:50	Conference closing address: Colin O’Reilly (*Conference Secretary* – IAH Irish Group)
13:00	*Buffet lunch in Tullamore Court Hotel*

TECHNICAL WORKSHOP

13:55	‘Well Grouting’ – Peter Conroy
14:35	‘The Bailer Test’ – Paul Wilson (GSNI)
15:15	End of Workshop

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SESSION I

EXPERIENCE OF WELL DRILLING AND TESTING IN KARST

Neven Kresic
(AMEC Environment and Infrastructure, Inc.)

ABSTRACT

The unique heterogeneity of karst aquifers is the main factor behind a general challenge of completing successful water supply wells including finding their optimal location, drilling a well borehole without major interruptions, developing a well, and predicting sustainable withdrawal rates. Although some related problems may be less pronounced or absent in young carbonates with high matrix porosity, there still remain enough related uncertainties in karst aquifers of all origins to complicate lives of hydrogeologists and well drillers alike. In addition to an unspecified amount of luck, the following aspects of well drilling in karst should be considered to avoid unsatisfactory results: likely presence of preferential flow paths in the aquifer, applicable drilling methods, possibility of circulation loss, well completion design, well development, and applicable well (aquifer) testing methods.

SELECTION OF WELL SITES

Selecting the best site for a well or wellfield in karst is considered by some to be an art or a matter of luck, by some to be as simple as finding a recommended dowser, and by some (such as well drillers) as a natural part of collecting fees for drilling holes into the subsurface as deep as possible. Although a few groundwater professionals may agree with some of these statements (except, arguably, relying on a dowser), siting and then designing wells for public supply or large irrigation projects is very complex and should result from thorough considerations of multiple design elements. The most important, however, is to first consider all indirect and direct methods for locating preferential flow paths. A well located in, or adjacent to a preferential flow path in karst will have much higher probability of being successful than a well located at random. The exceptions are wells drilled in younger carbonates with high matrix porosity such as Floridan aquifer in the United States. In such aquifers virtually all wells will have significant yields, while those close to or in preferential flowpaths will be exceptionally productive.

WELL DESIGN ELEMENTS

Well depth, diameter, and construction methods vary widely and there is no such thing as a "one size fits all" approach to well design. Answers to just about any question regarding well design can be found in the classic 1000-page book *Groundwater and Wells* by Driscoll (1986). Another exhaustive reference book on well design is *Water Well Technology* by Campbell and Lehr (1973). Various public-domain publications by United States government agencies provide useful general information on the design and installation of water supply and monitoring wells (e.g., U.S. Environmental Protection Agency (USEPA), 1975, 1991; U.S. Bureau of Reclamation (USBR), 1977; 1991; Lapham et al., 1997). Well design, installation, and well construction materials should conform to applicable standards. In the United States, the most widely-used water well standard is the ANSI/AWWA A100 standard, but the authority to regulate products for use in, or contact with drinking water, rests with individual states which may have their own standard requirements. Local agencies may choose to impose requirements more stringent that those required by the state (AWWA, 1998).

The general design elements of water wells in any porous media include the following: (1) drilling method; (2) boring (drilling) and casing diameter; (3) depth; (4) well screen; (5) gravel pack; (6) well development; (7) well testing; and (8) selection and installation of the permanent pump. In most karst aquifers, however, wells can often be installed without screens and gravel (sand) packs and completed as open borings which significantly reduces well costs and eliminates many issues associated with well screen design, well efficiency/loss and overall well maintenance. In cases of large public supply wells it still is desirable to install coarse screens that will maintain borehole stability and prevent mechanical damage to the pump caused by collapsing cavities and dislodged rocks.

Figures 1 and 2 illustrate some of the more common well designs which can vary widely based on project-specific requirements and can be combined in a single well. Continuing advances in drilling and well installation technology allow for elaborate designs such as under-reaming (widening of borehole below already installed and grouted casings), use of temporary casings for drilling in unstable conditions, telescopic screens, multiple screen intervals with or without continuous gravel packs, and slanted wells.

Figure 1 – Some basic well types. (a) Deep well with multiple cemented casing for bridging unstable and undesired formations, completed as open borehole in the deeper karst aquifer; (b) Well cased through overlying unconsolidated sediments, and with multiple coarse screen intervals in the karstified aquifer separated by blank casing to isolate unconsolidated sediments and unstable intervals; (c) Well completed as open borehole in stable karstified bedrock and with casing cemented in place, extending through regolith and upper portion of the bedrock. (d) Well in unconsolidated (non-karstic) sediments with telescoped screen and gravel pack, and with riser pipe cemented in place (Kresic, 2013).

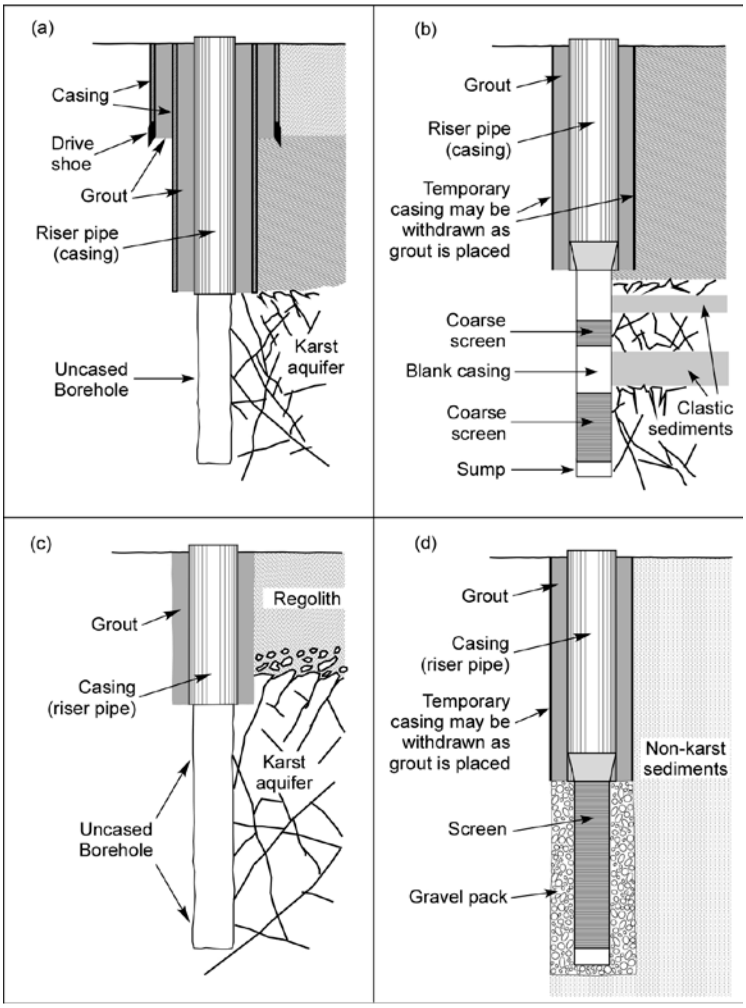
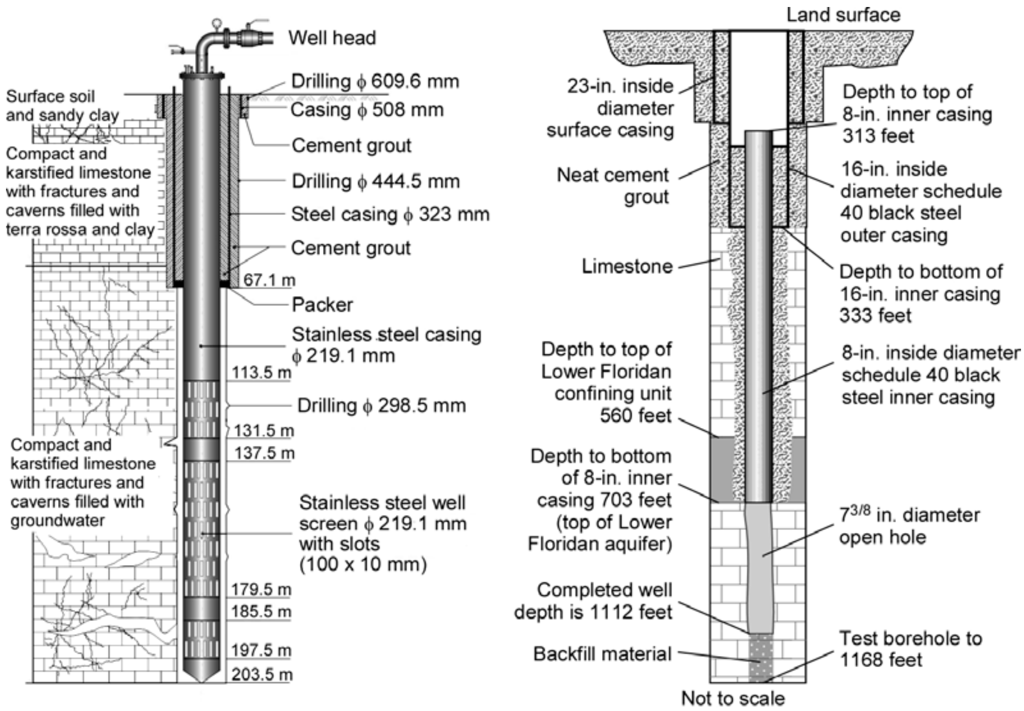


Figure 2 – *Left*: Construction of well RAJ-2 in the village of Rajkovic near Mionica, western Serbia yielding 2 L/s with drawdown of 112 m. (From Djokić et al., 2005). *Right*: Well-construction diagram for well HAAF 11 (36Q392) completed in the Lower Floridan aquifer, Hunter Army Airfield, Chatham County, Georgia. (From Clarke et al., 2010.)



The expected well yield, the well depth, and the geologic and hydrogeologic characteristics of the porous media (rock) all play an important role in selecting the drilling diameter and the drilling method. Deep wells, or thick stratification of permeable and low-permeable porous media, may require drilling with several diameters, and the installation of several casings of progressively smaller diameter, called telescoping casing. This is done to provide stable and plumb boreholes in deep wells, and to bridge difficult or undesirable intervals (e.g., flowing sands, highly fractured and unstable walls prone to caving, thick sequences of swelling clay). The cost of drilling increases progressively with the drilling diameter and it is important to balance this cost with other design requirements, some of which may be desirable but not always necessary. For perspective, a public supply, high capacity well, that is several thousands feet deep (e.g., 1000 meters), may easily cost well over one million U.S. dollars. Such wells are drilled with large drill rigs and may utilize special large-diameter drilling bits and include large diameter casing (24-48 in.).

Whenever possible, a well design should be based on information obtained by a pilot boring drilled prior to the main well bore. Geophysical logging, coring, and sample collection in the pilot boring provide important information on depth to, thickness, and permeability of the water-bearing intervals, and physical and chemical characteristics of the groundwater. Unknown geology and hydrogeology of the formation(s) to be drilled may result in the selection of an improper drilling technology, sometimes leading to a complete abandonment of the drilling location due to various unforeseen difficulties such as flowing unconsolidated materials (silts/sands), collapse of boring walls, or loss of drilling equipment in karst cavities.

Common experience from different countries is that down-hole hammer drilling techniques are now widely utilized for drilling wells in karst, although classic rotary drilling with different fluids, including air, is still prevalent. Problems of seizing up of the hammer and circulation loss are often dealt with by using hydraulic oil alternatives (e.g. vegetable oil), lowering of the drilling fluid density, and using organic additives including exotic ones such as walnut shell chips.

WELL DEVELOPMENT

Proper well development will improve almost any well regardless of type and size, whereas without development an otherwise excellent well may never be satisfactory. There are various methods of well development and their selection depends primarily on the applied drilling technology and the formation characteristics. However, availability of the equipment and driller's preference in many cases play unjustifiably more important roles. It is often impossible to anticipate how a well will respond to certain types of development and how long it will take to achieve adequate development. Since a lump-sum basis for well development may result in unsatisfactory work, it is better to provide for development on a unit price per hour basis and continue until the following conditions have been met (modified from AWWA, 1998): (1) Content of sand and fines should average not more than 5 mg/L for a complete pumping cycle of 2-hour duration when pumping at the design discharge capacity; (2) No less than 10 measurements should be taken at equal intervals to permit plotting of sand content as a function of time and production rate and to determine the average sand content for each cycle; (3) There should be no significant variation in specific capacity during at least 24 h of development.

General methods of well development are pumping, surging, fracturing and washing, each of which has several variations (USEPA, 1975). It is recommended that at least two methods be applied for best results. One of the less effective – but commonly used – methods is over-pumping the well. Here, water flows in one direction only – towards the well. The flow velocities are generally not fast enough to remove much of the fine material plugging the formation. During over-pumping, a surging action can be created by periodically shutting off the pump and allowing the water in the pump column to flow back into the well. This is more effective than over-pumping. However, water will re-enter the most permeable parts of the formation or those that have been least damaged during well construction. Thus, the portion of the formation that requires the most active development is largely excluded (Johnson Screens, 2007). Injecting water into the well and then pumping it back from the well (backwashing) will result in movement of water through the wells screen and borehole walls in both directions, thus increasing the effectiveness of development.

Pumping with compressed air, or airlift is probably the most common method of well development. Driscoll (1986) provides a very detailed discussion on various airlift techniques, including determinations of quantitative parameters required for proper airlift design. Unfortunately, many well drillers and contractors always apply the same airlift method (one they are familiar with) without regard for site-specific conditions.

As with unconsolidated formations, all drilling methods including air-drilling cause some plugging of fractures and other openings in karstified formations. Therefore any material that clogs openings in such formations should be removed during well development. In many cases the best method is the water jetting combined with airlift pumping. Inflatable packers can isolate the productive zones (fractures/voids) supplying water to the well and increase efficiency of their development.

Well yield in carbonate rocks can sometimes be increased considerably if one or more well stimulation methods are applied. Well stimulation is considered as a second level of development which can increase well performance beyond that obtained through traditional methods. Hydrofracturing is used to stimulate both new and old wells in consolidated rock formations. In hydrofracturing, water at extreme pressures can be injected into the entire well or into discrete intervals sealed by packers. The injected water removes sediment from the existing fractures and cavities and creates new fractures resulting in an increased permeability of the formation adjacent to the well.

Blasting with explosive charges lowered in an uncased borehole in consolidated rock is sometimes used to increase well specific capacity. Similarly to hydrofracturing, this method enlarges the existing fractures and creates new ones resulting in an increased hydraulic conductivity of the formation. However, blasting with explosives should be applied with utmost care and only after considering many factors including legal requirements and environmental impacts.

Acid can be used for well stimulation and formation development in limestone and dolomite aquifers. Acid dissolves carbonate minerals and enlarges voids and small fissures in the formation adjacent to the borehole. Acid can also be forced into discontinuities away from the well resulting in dissolution and removal of a larger volume of the native material. This increases the overall hydraulic conductivity of the aquifer around the well and may result in a significant increase of the specific capacity of the well.

Probably the only design element of the entire well that cannot be guaranteed by any well driller, hydrogeologist, or engineer is the well yield and its long-term sustainability. It is not uncommon that, for various reasons, a well that costs several hundred thousand dollars to complete, disappoints all stakeholders by actually producing just a fraction of its designed capacity. However, many such surprises can be avoided by following well-established hydrogeologic principles of karst aquifer evaluation and testing, and, of course, well design and installation itself. And the best kept secret is that a little bit of luck is always needed when installing a well in karst for any use.

WELL (AQUIFER) TESTING

In a karst aquifer the inflow of water to a pumped well would most likely occur from discrete intervals where the borehole intersects preferential flow paths (fractures and solution openings) within the surrounding rock. The presence of such intervals may be indicated by various methods of geophysical logging, and their actual flow contribution may be measured and calculated using borehole flow meters (flowmeters). The flowmeters can be utilized in various ways, with or without pumping of the well. During well pumping packers can be used to isolate portions of the open borehole for a more precise characterization. Classic packer tests, during which water is injected under pressure into discrete intervals, can be performed to determine permeability of the tested intervals. Finally, long-term pumping tests are utilized to assess the amount and quality of water available for extraction from the aquifer.

Simultaneous use of geophysical logging tools and flowmeters is the best available method for in-situ characterization of fractured rock and karst aquifers (Paillet, 1994; Paillet and Reese, 2000). Integration of new geophysical methods with conventional logging techniques can be used to define flow zones, lithology, structure, and their relations. Borehole flowmeters, whether vertical or horizontal, are used to identify and quantify water producing zones in a well. Some horizontal flowmeters can measure the direction of flow through the borehole and thus identify intervals that either contribute water to or take water out of the borehole. Flow logging tests between boreholes (cross-hole tests) can indicate the degree of connectivity of preferential flow intervals (zones) beyond individual borings while transient tests can be used to estimate hydraulic properties such as transmissivity and storage coefficient.

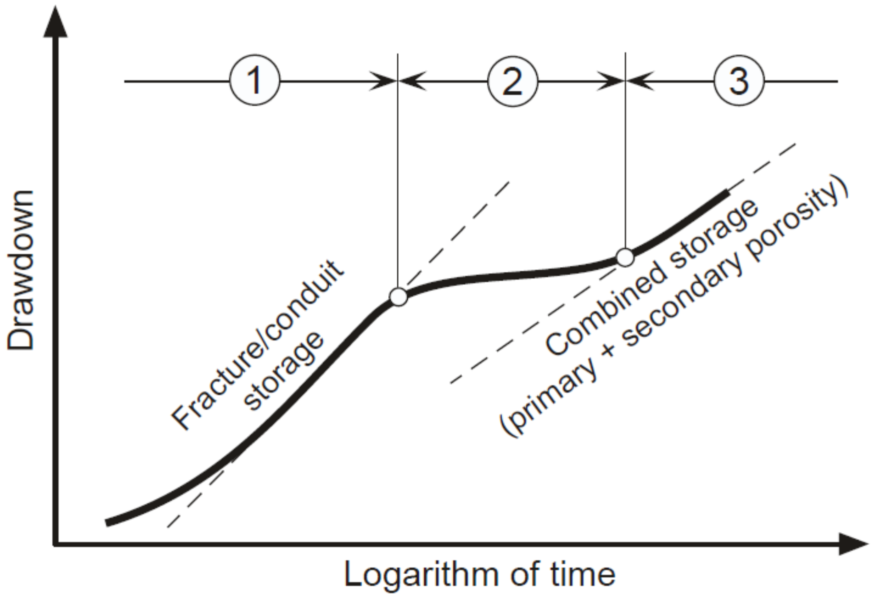
Most methods for characterizing aquifer parameters with pumping tests, including in karst, are based on quantitative analysis of the observed drawdown versus the time of pumping at the well. The drawdown is measured in one or more observation wells (piezometers) and/or the pumping well itself. Because the time of pumping is explicitly included in mathematical formulas describing relationship between the rate of pumping, the drawdown and the aquifer parameters, these methods are called "transient" or time-dependent. Virtually all such methods routinely applied today are in one way or another based on the pioneering work of Theis (1935) and involve some type of grapho-analytical solution during which the observed time-drawdown data are matched against theoretical model curves ("type curves"). Unfortunately, practicing hydrogeologists often forget the following key assumptions of the Theis equation and apply the method indiscriminately including in karst: the aquifer is homogeneous, isotropic, confined, of unlimited extent, with an impermeable horizontal base, horizontal flow lines, and receives no leakage (recharge) of any kind. It is obvious that hydrogeologic conditions in the field often prevent direct application of the Theis method not only in karst but in integranular porous media (sand and gravel) and fractured rock as well.

As discussed by Kresic (2007), various analytical methods have been continuously developed to account for situations such as presence of leaky aquitards with or without storage and above or below the pumped aquifer, delayed gravity drainage in unconfined aquifers, aquifer anisotropy, and presence of fractures. Regardless of the aquifer type and the nature of aquifer porosity, it is important to understand that often more than just one type curve or analytical method may be fitted to the observed time-drawdown curve. Selection of the “correct” one therefore depends on the thorough overall hydrogeologic knowledge of the aquifer in question. This is particularly true in the case of karst aquifers because of the absence of any mathematically rigorous and widely agreed-upon analytical method of aquifer pumping test analysis. As is the case with numeric groundwater modeling of karst aquifers using EPM approach, analyses of aquifer pumping tests in karst are mostly performed by applying and combining methods developed for intergranular and simple fractured rock aquifers.

A typical time-drawdown curve in response to groundwater withdrawal from a well completed in a karst aquifer is shown in Figure 3. Given enough pumping time and presence of all porosity types, the time-drawdown curve would show three distinct segments. The first portion of the curve, with a uniform slope, indicates a quick response from a well-connected network of secondary porosity, which may include large dissolutional openings and/or fractures. Drainage of this type of porosity, in the early stages of the test, is characterized by storage properties generally similar to that of confined aquifers. Unconfined intergranular aquifers often exhibit similar early response to pumping, due to sudden change in hydraulic pressure, with the storage coefficient significantly less than 1 percent ($1x10^{-3}$ or less). The flattening of the curve (curve portion 2) indicates that the initial source of water in large solutional openings (channels, conduits, cavities in general) and/or fractures is being supplemented by water coming from another set of porosity. This additional inflow of water starts when the fluid pressure in the conduits and/or large fractures decreases enough, resulting in the hydraulic gradients from the smaller fractures and fissures towards the larger fractures and/or karst conduits. This is a transitional portion of the curve. Again, a similar response often happens in unconfined intergranular aquifers when additional water, due to gravity drainage, starts reaching the lowered water table (this is called delayed gravity response to pumping).

As the sources of water from different sets of secondary porosity features, possibly including water from the primary (“matrix”) porosity attain similar level of influence, the drawdown curve exhibits another relatively uniform slope (curve portion 3). Such rock formation is often referred to as a dual-porosity formation because of the distinct hydraulic characteristics of different types of porosity present (Barenblatt et al., 1960). An exact determination of individual storage properties of different porosity types in a karst aquifer is beyond capability of common aquifer pumping tests. Aquifer storage parameters related to the effective matrix porosity may be approximately determined by laboratory tests, which would ideally include core samples from multiple locations and depths within the aquifer.

Figure 3 – Theoretical response of the time-drawdown curve caused by effects of specific porosity of karst aquifers. (1) Drawdown due to the initial drainage of secondary porosity (fractures and solution cavities/conduits). (2) Transitional drawdown. (3) Drawdown due to stabilized drainage of all porosity types, including matrix porosity. (From Kresic, 2007)



Depending on the duration of the test and the characteristics of the karst aquifer system itself, the time-drawdown curve may exhibit additional changes due to some factors external to the main portion of the aquifer being pumped. These factors may include influence of impermeable boundaries, recharge, or constant head boundaries. Given the nature of karst aquifer storage properties, and almost inevitable external influences, time-drawdown curves of long-term pumping tests in karst aquifers exhibit a wide variety of shapes, often non-typical. It is therefore critical that such tests be analyzed using overall geologic and hydrogeologic knowledge about the tested aquifer, rather than formally applying a "type curve formula" typically developed for non-karstic porous media (Kresic, 2013).

REFERENCES

AWWA (American Water Works Association), 1998. AWWA standard for water wells; American National Standard. ANSI/AWWA A100-97, AWWA, Denver, Colorado, various paging.

Barenblatt, G.E., Zheltov, I.P, and Kochina, I.N., 1960. Basic concepts in the theory of seepage of homogeneous liquids in fissured rocks. *Journal of Applied Mathematics and Mechanics*, v. 24, p. 1286-1303.

Campbell, M.D., and Lehr, J.H., 1973. Water Well Technology. McGraw-Hill Book Company, New York, 681 p.

Clarke, J.S., Williams, L.J., and Cherry G.C., 2010. Hydrogeology and water quality of the Floridan aquifer system and effect of Lower Floridan aquifer pumping on the Upper Floridan aquifer at Hunter Army Airfield, Chatham County, Georgia: U.S. Geological Survey Scientific Investigations Report 2010 -5080, 56 p.

Djokić, I., Canić, V., and Cekić, M., 2005. Results of hydrogeological investigations in the Valjevo-Mionica karst region, Rajković-Kjluč area (Western Serbia). In: Stevanović, Z., and Milanović, P. (eds.), Water Resources and Environmental Problems in Karst – Cvijić 2005, Proceedings of the International Conference and Field Seminars, Belgrade & Kotor, 13-19 September 2005, Institute of Hydrogeology, University of Belgrade, Belgrade, pp. 597-602.

Driscoll, F.G., 1986. Groundwater and Wells. Johnson Filtration Systems Inc., St. Paul, MN, 1089 p.

Kresic, N., 2013. Water in Karst: Management, Vulnerability, and Restoration. McGraw Hill, New York, 708 p.

Kresic, N., 2007. Hydrogeology and Groundwater Modeling, Second Edition. CRC Press/Taylor & Francis Group, New York, Boca Raton, 807 p.

Lapham, W.W., Franceska, W.D., and Koterba, M.T., 1997. Guidelines and standard procedures for studies of ground-water quality: selection and installation of wells, and supporting documentation. U.S. Geological Survey Water-Resources Investigations Report 96-4233, Reston, Virginia, 110 p.

Paillet, F.L., 1994. Application of borehole geophysiscs in the characterization of flow in fractured rocks. U.S. Geological Survey Water-Resources Investigations Report 93-4214, Denver, Colorado, 36 p.

Paillet, F.L., and Reese, R.S., 2000. Integrating borehole logs and aquifer tests in aquifer characterization. Ground Water, v. 38, no. 5, p. 713-725.

USBR, 1977. Ground Water Manual. U.S. Department of the Interior, Bureau of Reclamation, Washington, D.C., 480 p.

USEPA, 1975. Manual of water well construction practices. EPA-570/9-75-001, Office of Water Supply, Washington, D.C., 156 p.

USEPA, 1991. Manual of small public water supply systems. EPA 570/9-91-003, Office of Water, Washington, D.C., 211 p.

HOW TO CONSTRUCT A WATER SUPPLY BOREHOLE IN IRELAND

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ABSTRACT

The applied science of hydrogeology started with work on groundwater supplies and through this there developed a better understanding of groundwater resources and flow systems. Early and middle career hydrogeologists appear to want to develop increased competency and confidence in developing good groundwater supplies, but it is daunting. The following paper acknowledges some of the difficulties, describes how some classic borehole designs are not appropriate in hard, fracture or conduit controlled, bedrock aquifers in Ireland, and draws attention to recently published Irish guidelines that will help hydrogeologists to carry out successful water supply borehole drilling programmes in the future.

INTRODUCTION

The foundation of hydrogeology lies in the development of water supplies from underground water resources. The need to provide water supplies eventually demanded a better understanding of groundwater resources. Water resources and the exploitation of springs and construction of wells and boreholes formed the start, and foundation of our science and profession.

It is assumed by the hydrogeology profession, and those outside our profession, that all hydrogeologists understand how to develop groundwater supplies, dig wells and drill boreholes. It is assumed that this basic skill is so common place and ordinary, that little attention is paid to it. We all think that we know how to drill a borehole. Text books on our subject describe the design of a borehole, how it works and how we can use it to understand the hydrogeology of an area or aquifer. We have all had lectures on the subject at college. Perhaps, we have had so much information on these fundamentals. Perhaps they are regarded as ‘old school science’, passé, uninteresting. The attitude of hydrogeologists over the last 20-30 years has been to move on and discover new areas of research, discovery and work. Examples would be, contaminated land, modelling, GIS, geothermal, nuclear waste disposal, source protection, septic tanks, and one of the latest, measuring fluctuations in the thickness of saturation in aquifers and the depletion of groundwater resources by measuring gravity differences over time from space. These branches of our discipline are worthwhile and exciting, but for successful work in these areas there is an assumption that the basics of groundwater flow and groundwater storage are well understood by the investigators, and that the boreholes, which provide the samples or the data for the models, have been constructed appropriately, and this can be critically evaluated by the hydrogeologist using the data.

The problem we have is that the teaching and books primarily and almost exclusively describe a hydrogeological world that is simple and that rarely exists in the real world. We learn the principles of groundwater flow, storage, hydrochemistry and borehole design and operation using examples of aquifers that are composed of porous media. This is a reasonable start. Using porous media makes it possible to explain the basics. For example, when it comes to imagining or explaining the effect on water levels in an aquifer caused by pumping a borehole, we all have the image of a cone of drawdown in cross section, curving up and away from the pumping water level borehole. This image is useful, because it is simple. I confess that I use it frequently when explaining flow to a borehole to clients, even though I know it is not realistic. I believe we have a problem as a profession because many of us appear to continue to want to believe in aquifers made up of porous media, even when it is obvious that the ancient bedrock has no intrinsic porosity. We have resorted to describing the interconnected fractures or conduits through the rocks as ‘equivalent porous media’. This may be reasonable at a large scale of several square kilometres, but it is also a concept used at a borehole, or well field, scale. We seem to want to do this because we do not have any alternative simple formulae or mathematical models that can deal adequately and easily with site specific heterogeneity in hard rocks and limited data. We are almost held hostage by our desire for a world framed by porosity. We are taught to believe in porous aquifers, therefore we seek them out even if, in our heart of hearts, we know they don’t exist, and this conceptual understanding is not supported by field evidence.

There is a second related problem. In modern hydrogeology we seem to have insufficient opportunity to test our perceived, or taught, conceptual understanding of groundwater flow in the field. There has been a progressive movement to believe that we can understand, map, predict and write about groundwater from the office. Some training programmes seem content to produce graduates with a belief that new data is unnecessary, and we can understand the groundwater flow system using existing data and modelling. Employers also seem to buy into this notion that hydrogeologists can do profitable work from the office, with the occasional foray out into the field to measure water levels or take a water sample. I am surprised that many firms of consultants do not insist that all exploration and water supply borehole drilling is supervised full time by an experienced hydrogeologist. I am surprised that consultants and clients still employ contractors to carry out pumping tests.

A third problem is lack of confidence or fear. It is daunting to go onto a drilling site for the first time, and try to lead and direct a water supply borehole drilling project. It is even more daunting in Ireland, when you definitely know that the rocks that you will encounter during the drilling will not be porous, predictable, isotropic aquifers, where you can put into practice the theory that was learnt during training. You will find, also, that it is rare to work on bedrock aquifers that are young, soft, porous, un-weathered and un-deformed when you work elsewhere in the world. Large parts of all continents are underlain by fractured rock aquifers. Therefore, for a variety of reasons, hydrogeologists appear to have moved away in the last 30 years from developing a high level of competency in directing, designing and constructing water supply boreholes. This move has occurred at the same time as hydrogeologists and regulatory authorities have realised that our existing water supply boreholes were not located and or constructed properly.

There is hope for a change in modern hydrogeology. I am finding a growing group of un-daunted, bright early career hydrogeologists and mid career hydrogeologists who have realised that they are not satisfied with the desk GIS modelling hydrogeology. They are realising that trying to make sense of groundwater flow in models using assumed values for aquifer characteristics, perhaps based on old data, is not working. They are wanting to come out into the field to drill, or learn about drilling, so that they can experience at first hand the heterogeneity and complexity of real groundwater systems at a borehole scale.

The purpose of this paper is to provide a little guidance to those who want to engage with water supply borehole drilling, and to draw attention to the fact that several borehole design criteria for porous aquifers are irrelevant in hard rock fracture or karst conduit hydrogeology.

Drilling boreholes for water supplies or groundwater investigations is an opportunity to learn about the real characteristics of the hydrogeology in the area. In fact, it is the only way to get a real understanding because it is the only direct way of investigating the subsurface.

Drilling is not what you expect if you just see boreholes as the straight lines, sharp boundaries, parallel sided holes with perfectly straight casing shown in borehole logs or drillers reports. Boreholes are often gentle helixes. There are slight off-sets, areas where drilling has scoured a wider hole. Drilling can sometimes be monotonous when there is no water inflow to the hole, but when you get into productive zones in hard bedrock the rock and cavity conditions can change radically and repeatedly in short distances. Drilling using an air-hammer is fast. Sometimes we are going down at a metre a minute. Therefore, when conditions change, analysis and decisions have to be rapid. Never be afraid to tell a driller to stop, pull the rods back a few metres, or slow down so that you can observe, measure and get time to think.

Every borehole is a story. Each metre drilled is like turning the pages of a book, and as with any engaging story, the reader tries to predict the outcome, but there are surprises along the way and the end is often different. With a borehole, you are trying to read Nature's story and anticipate what she will provide. Nature's story was not written by man or specified by a materials scientist.

Drilling water supply boreholes is complicated because the subsurface does not conform to our preconceptions. In order to design, drill and complete a borehole properly, and to a high standard, it is essential that the hydrogeologist writes the specification for the drilling, assesses the tenders, controls the contract and is present on-site next to the rig at all times. It is essential that the hydrogeologist and the driller work together. Their skills are complementary. He provides the machine, equipment and materials and he knows how to work with them. You have a conceptual understanding of how groundwater flows, and you have been taught about how the earth was formed. You can interpret events and observations as they occur, and you can convert this information into better or amended borehole design and construction. You can observe a change in the tone or beat of a down-the-hole-hammer, the change in colour or size of the drill cuttings or a slight whiff of hydrogen sulphide and translate these observations into a change in weathering, the presence of oxidising water in shales, and imminent karst conduit. The driller may have seen, heard or smelt the same, but drillers are primarily intent on operating their machine to make a hole, and do not necessarily understand the significance of certain changes in terms of groundwater potential, source protection or decisions about borehole design.

There are times when nothing seems to be fitting with the pre-conceived conceptual model of the subsurface at a site, and at those times it is essential to go back to basic principles of good design and the fundamentals of groundwater flow in a heterogeneous hard bedrock aquifer.

The first principle is that you are trying to drill and construct a water supply borehole where the water pumped from the borehole does not require treatment. There is no point in setting out to drill a contaminated water supply borehole. Therefore, the water has to be of good quality with regard to the intended purpose, and the quality has to be sustained. Quantity is not the foremost issue. A second or third borehole can be provided, if the quantity from a single hole is less than required. It is advisable, for a public supply, at the outset, to think in terms of a wellfield of more than one borehole, to avoid being reliant on a single source with no back up.

To get good quality groundwater in Ireland it is essential to develop a conceptual understanding of recharge, groundwater flow in heterogeneous Irish bedrock, and to consider groundwater movement in the vertical perspective. In other words, think in cross section view rather than plan view.
I prepared Drinking Water Advice Note 14 for the EPA in September 2013. In this document I used illustrations to try to explain groundwater flow in Ireland within the context of water supply borehole drilling, design and construction. All the illustrations are in the vertical dimension. Boreholes are vertical.

Below, I summarise the key considerations whilst constructing a water supply borehole; starting with the fundamental of recharge and flow.

RECHARGE AND STRATIFICATION OF GROUNDWATER FLOW

To explain stratification of groundwater flow it is necessary first to go back to a porous media. Consider a porous uniform aquifer underlying a river valley with hills on either side. Groundwater flows from under the hills into the valley and discharges into the river. The recharge reaching the water table under the hills raises the water table level that creates a gradient down to the river in the valley. As the recharge from under the hills flows towards the river by gravity, then subsequent recharge through the surface of the lower slopes and the valley floor is incrementally added to the top of the water on its journey flowing from the hills to the river. Therefore, the recharge that filled the upper part of the saturated zone below the hill, is pushed further down into the aquifer by the subsequent recharge added to the top of the saturated zone through the lower slopes and the valley floor. (I will illustrate this in the lecture). This is important because it means that groundwater that has travelled further will tend to be found lower in the aquifer. Groundwater that has travelled further will tend to have travelled for a longer time. Therefore, there will have been greater opportunity for die off, filtration or breakdown of pathogens. Therefore, a borehole that draws upon all the groundwater flowing through the aquifer (what I term an old-style borehole) will tend to draw upon a blend of recent recharge (and recent pathogens) in shallow groundwater along with older water that that has travelled further, that is found deeper in the aquifer. A borehole that is designed to draw upon water from deeper in the aquifer, and deliberately excludes shallower water is therefore more likely to produce water that contains less or no pathogens or other surface discharged pollutants.

This basic principle of stratified groundwater flow in a homogeneous isotropic porous media also applies in heterogeneous fractured and karst solution weathered aquifers.

Although groundwater flows downhill under the influence of gravity to the discharge point or zone, the gradients are generally low except near the summits in highland areas. Over much of the country, and in particular the Midlands, the horizontal distance from the point of recharge to the point of discharge is much greater than the vertical distance. Normal groundwater gradients have a fall of a few metres over a distance of several hundred metres or a kilometre. Recharge water does not ‘burrow’ down to replace water already deep in the groundwater system. Instead it adds to the top of the saturated zone and tries to take the shortest route to the discharge zone. Flow is principally sideways. Fractures, weathered zones or conduits that permit sideways or sub horizontal flow form preferential flow paths, that are kept open by groundwater movement. One of the common preferential flow paths is the transition zone at the base of the boulder clay and the top of the bedrock. Other conduits develop in the weathered zone in the top few metres of the bedrock.

My experience of drilling has frequently shown distinct zones consisting of several conduits, individual conduits, caves, and permeable deep weathered zones at different depths in the bedrock. These zones often provide relatively large quantities of water. The permeable zones or conduits are often separated by large thicknesses of rock with negligible yield of water. Iron staining on the natural fracture or weathering faces from rock drilled in these zones, indicate that groundwater has moved through this rock and is still stored in the rock, but there is no evidence of significant volumes of groundwater movement. It is normal to drill through tens of metres of this rock and find no appreciable increase in flow from a borehole, yet a close look at the cuttings reveals that water has moved through micro fractures or bedding planes.

The evidence from drilling indicates that the distinct deep well developed flow paths are near horizontal. I have drilled high yielding well fields with several adjacent boreholes, and found evidence of laterally continuous high permeability zones. The evidence from drilling strongly indicates that these deep distinct zones are senescent and do not have a large groundwater flow under modern natural conditions. There are small and large gaps in the rock, and in limestones there are large and small caves, but the openings often contain very fine soft clay, that is also commonly bright yellow in colour. Airlift surging and pumping of the borehole and, in particular, aggressive surging, just above and below these zones causes turbulent flow, in and out of the conduits, that easily dislodges and disturbs the clays. I use airlift pumping to encourage water to flow rapidly along the conduits and scour and clean the yellow mud from the conduit. The flow rate invariably increases. I get the impression during this work that my pumping is cleaning out very old conduits that were developed long ago, when water levels, sea levels and perhaps climates were different. I sometimes find fossils from other formations and even robust plant remains, such as flattened slightly pyrite mineralised wood or fir cones, are brought up to the surface by the airlift pumping. I suspect that many of the deeper, distinct, laterally continuous, permeable zones formed a palaeo-drainage base level for groundwater flow when sea levels were much lower. Groundwater was stored and trickled down through the micro-fractures and joints in the rock mass above, to be collected and concentrated in near horizontal high permeability basal drainage zones. Drilling down into them and pumping seems to be re-energising a groundwater flow system that has nearly gone to sleep.

The position or depth below ground level of zones of permeability varies depending upon topography, rock type, structural geology and proximity to sea level or a local base level for groundwater drainage. When I am drilling near the coast, I expect to find a permeable zone just above modern sea level, and often another much more permeable zone about 30 - 45 metres below modern sea level. Inland, in the Midland’s plains, there is a permeable zone at the top of the bedrock, then a moderately permeable zone around 30 metres below ground level, followed by a more permeable zone between 60 and 75 metres depth and occasionally a more discontinuous permeable zone around 90 to 110 metres. These depths are not hard and fast, or proven by a detailed statistical analysis, they are just a working rule of thumb that I find useful during supervision and direction of water supply borehole exploration drilling.

Therefore, though our bedrock is hard, impermeable and full of heterogeneity created by our long structural geology history, it appears that groundwater has tried to develop enhanced pathways at different levels under different conditions.

Figure 1 Sections illustrating the heterogeneity of Irish groundwater flow systems and the difference between an old style borehole and the modern water supply borehole

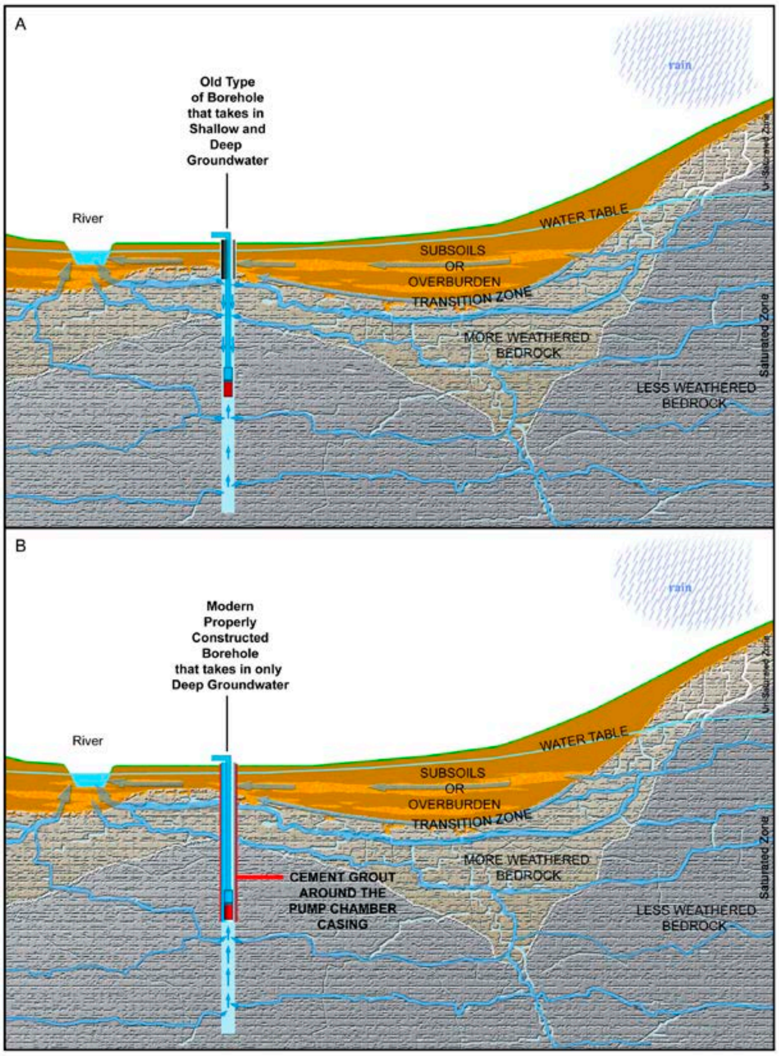


Figure 1 shows a section that tries to represent the typical heterogeneity of a groundwater flow system in Ireland. The vertical scale is exaggerated, but it illustrates recharge on the valley sides, and a downward gradient under the hill. It shows deep groundwater flow from off the section to the right. It shows essentially lateral flow through the bedrock, the transition zone and the overburden under the lower ground. It shows upward gradients near the river that drains the groundwater system.

Section A shows an old-style water supply borehole, where steel casing is supporting the upper hole through the overburden. This borehole, with or without a screen draws upon the upper shallow groundwater flow and the deeper groundwater flow. This borehole draws upon groundwater indiscriminately. Section B shows a modern water supply borehole, that is designed to deliberately exclude the shallow water that is more likely to be made up of recent recharge and be more vulnerable to contamination. It has a solid wall PVC pump chamber casing down to a significant depth, and the space, or annulus, around the casing has been sealed by a cement grout. The modern borehole taps into the deeper groundwater that has followed a longer travel path. The modern borehole has been designed and constructed in accordance with EPA, IGI and international standards. This design is described in more detail at the end of the paper.

In the past everyone was worried about not being able to get enough water. By clients making the unreasonable demand that drillers are responsible for finding and providing water, drillers seldom sealed off any available water, for fear that they would not encounter sufficient water at greater depth. The modern emphasis is on sustainable quality, rather than quantity. Now, it is realised that well fields of more than one borehole can be constructed to get the required yield, and it is better to invest in constructing several proper water supply boreholes at the outset, rather than paying for high on-going costs trying to treat polluted water.

Groundwater from a modern high quality borehole should not require treatment to correct for man-made contamination or pollution. Chlorine should not be necessary to treat a problem with the groundwater, but merely to protect the high quality raw water as it moves through the distribution system to the consumer. Chlorine should be necessary only for protection of the water, not to treat polluted water from a poorly designed and constructed old-style borehole. Unfortunately, most domestic boreholes and many local authority boreholes are still sited and drilled by drillers alone.

WELL EFFICIENCY, FRACTURE FLOW, WELL SCREENS AND GRAVEL PACKS

Early in my career, I used to think that every borehole must have a well screen. I learnt about well hydraulics within the context of porous aquifers. I have found it is a characteristic of human nature to resist relinquishing basic concepts that we learned and accepted at an early age.

In Ireland we do not have porous bedrock aquifers. Therefore, to design a water supply borehole properly we need to look realistically at how water is moving in the ground and how we can get to flow efficiently into a pumping borehole.

In classic borehole design theory, a good yield can be obtained if there is a large infiltration area through which water can evenly flow from the aquifer. A classic design also requires that a screen and gravel pack are installed to hold back the aquifer material from collapsing

into the hole, yet let the water flow easily into the hole. There is a lot of discussion about screen slot size, slot shape, open area, screen entrance velocities and gravel pack grain size in hydrogeology text books. Most of this is irrelevant in hard impermeable fractured bedrock aquifers in Ireland and similar conditions elsewhere in the world.

The assumption that the aquifer is a porous media leads hydrogeologists and engineers to believe that a wide diameter borehole will yield more water, with lower well losses and hence greater hydraulic efficiency, than a narrower diameter borehole.

This belief does not apply in Ireland, because the flow of water into the borehole is controlled usually by the open width of the fracture or conduit intersected by the borehole. It is also important to note that the flow from the fracture or conduit is controlled by the narrowest width of the fracture perhaps some distance from the borehole. In other words, a borehole may intersect a fracture or conduit that has a 100mm open face at the borehole wall, but the flow along the fracture may be restricted by say a 5mm wide constriction in the fracture perhaps ten metres from the borehole. The yield from the fracture will be the same regardless of the width of the borehole.

An argument is sometimes made that a wider diameter borehole may intersect more of a fracture. This can be true if the fracture is horizontal and on all sides of the hole, but not true if the fracture is vertical. Video camera evidence often shows that water is entering a borehole from a single fracture or joint on one side of a borehole. The parting in the rock is not found on the other side of the borehole.
It is sometimes argued that a wider diameter borehole will encounter more fractures. This is true, but how much greater are the chances of hitting a previously undiscovered fracture with an 8 inch or a 10 inch borehole than with a 6 inch borehole. The wider diameters are only improving the chances by one or two extra inches.

It is important to recognise that the diameter of the open hole (producing section) in a borehole does not influence the efficiency of the borehole, if the diameter of the borehole is wider than the fracture or conduit feeding water into the borehole. If a 150mm (6”) borehole encounters a fracture or conduit has an open area equivalent to say a 50mm diameter pipe, then the turbulent flow, with friction head losses, will be in the fracture before the water enters the 150 mm borehole. Making a borehole 200, 250, 300 or even 600 mm in diameter does not change the fact that the fracture supplying the water is still only the equivalent of 50mm pipe.

Un-weathered bedrock in this jurisdiction is old and hard. It is usually very competent. Drilling with the down-the-hole-hammer method and large surging air pressures is violent and aggressive. If the hole remains stable and open during this brutal process, it is very unlikely that it will collapse during the more gentle pumping process. Therefore, it is not necessary to install well screens to support the borehole sides.

It is sometimes argued that a well screen should be installed in case a single piece of loose rock falls into the borehole and traps the pump. In a modern borehole the pump should not be in the producing section. It should be inside the pump chamber casing above the producing section.

A well screen should not be installed in the producing section of a bedrock water supply borehole because it presents a barrier to flow coming out of the fracture or conduit. Water flowing out of a 50mm conduit will find its route restricted by 1mm wide slots in a screen. The situation is made even worse by installing an artificial gravel pack. The fine pore spaces between the gravel grains will further restrict the flow out from the fracture, and also block the openings into the screen slots. As the pack material is installed in the annulus around the screen gravel can flow into the fracture and conduit. A large open karst conduit can accommodate numerous bags of gravel.

The aim in good borehole design is to let water flow freely into the borehole. Installing well screens and gravel packs does the opposite; it impedes flow and reduces well efficiency and sometimes can reduce the yield of the borehole by 80 - 90 per cent.

An important part of borehole construction is to open up the water bearing fractures and conduits. Well development is always required to create an efficient water supply borehole. This is normally carried out by airlift pumping and surging, with the rig at the time of construction, and perhaps later using a compressor and airlift system comprising an airline and an eductor pipe. A hydrogeologist with a compressor and airlift system is a lot cheaper than a drilling rig, crew and hydrogeologist.
Airflift surging to clean out fractures and conduits depends upon air and water being able to freely surge in and out of the fracture or conduit. This induced turbulent flow cleans out sediment in the

fracture or conduit. Installing a well screen and gravel pack muffles the effectiveness of the airlift surging.

It is sometimes argued that a screen and gravel pack should be installed to prevent a turbidity problem in the pumped water. The argument is made that the gravel pack traps fine sand, silt and clay. This is true, but it is counterproductive because a gravel pack clogged with fine sediment also stops water freely reaching the borehole.

A gravel pack to prevent a silt problem will further reduce the yield of the borehole.
It is obvious that the way to prevent a future long term turbidity problem is to spend the effort to clean out the fractures and conduits before pumping takes place. My experience has shown that if I can get a high flow rate with aggressive airlift pumping where the water at the end is still slightly hazey with suspended clay, then later, with gentle constant rate pumping at lower pumping rates the water supply will have no suspended clay or silt.

THE DESIGN AND CONSTRUCTION OF A MODERN WATER SUPPLY BOREHOLE IN IRELAND

The IGI produced the ‘Water Well Guidelines’ in 2007. The guidelines consisted of three separate booklets loose within a cover. We produced the guidelines principally to providing a document for people who wanted to develop a domestic water supply from groundwater. The EPA and the GSI endorsed and helped in publishing these documents. The guidelines described and illustrated, in straightforward terms, the principals and practice for the recommended design, construction, protection and pumping of a modern water supply borehole in Ireland. The documents describe both bedrock and sand and gravel boreholes. The EPA wished to bring these guidelines into legislation for both domestic boreholes and public water supply boreholes, but it was realised that trying to write an Act of the Oireachtas or a Statutory Instrument, would be complicated and take a long time. Instead, I was asked to write and illustrate, what became EPA Drinking Water Advice Note No.14. This advice note is entitled “Borehole Construction and Wellhead Protection” and was published in September 2013.

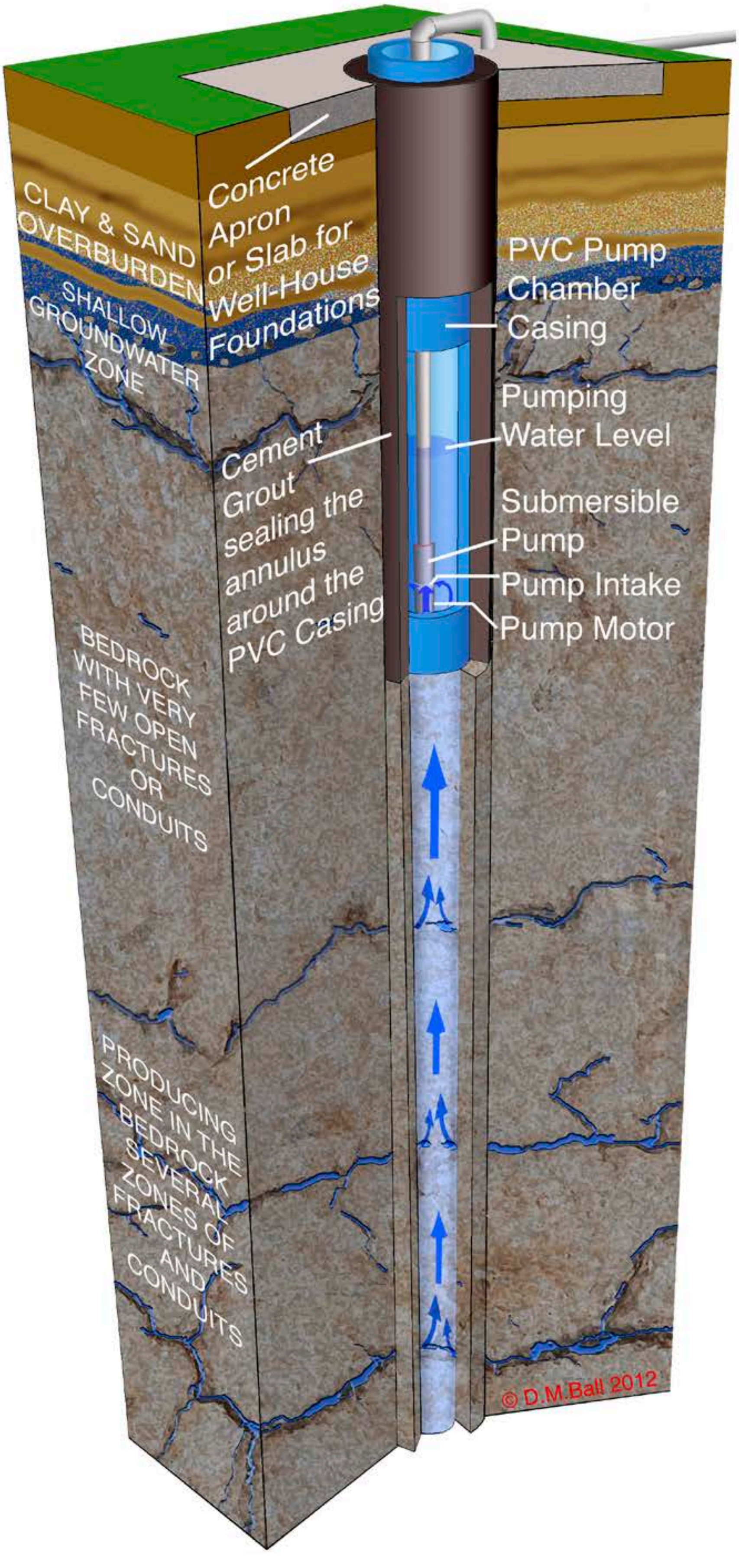
The purpose of Advice Note 14 is to “inform and instruct all Water Service Authorities and private regulated water suppliers to apply the IGI Guidelines when assessing the construction of existing drinking water supply boreholes, and also apply the Guidelines when commissioning the construction of new drinking water supply boreholes.”

It is a clear and mandatory instruction by the EPA Office of Environmental Enforcement to apply the guidelines. The advice note adds to the guidelines by setting out best practice for the design construction and protection of a drinking water supply borehole for Water Service Authorities.
The IGI Guidelines and the EPA Advice Note are designed to be explanatory. They aim to inform, and explain in detail why it important to do, or consider, certain things. I encourage you to read, use and publicise these documents. They have been written to inform, but also as documents to support hydrogeologists and engineers who want to carry out proper water supply borehole development work.

I cannot summarise all the text and illustrations in these documents in this paper, but I will summarise the bedrock water supply borehole design in the Advice Note.

Figure 2 illustrates a water supply borehole that has been designed to draw upon unpolluted water from deep in the bedrock. Drinking water supply boreholes must be designed and constructed with a water well grade PVC pump chamber casing from the surface down to sufficient depth to seal off shallow groundwater in the subsoil and the upper bedrock.

Figure 2 Design for a bedrock water supply borehole in Ireland



The pump should not be placed below the pump chamber casing because, it is unprotected and because there is a risk that a deep draw down could bring shallow groundwater under the casing and cement grout seal. The pump chamber casing should be of sufficient depth to allow sufficient drawdown above the pump placed within the casing. This is difficult to estimate in advance. It is sometimes advisable first to drill an exploration or pilot borehole and carry out a pumping test, before deciding on the final design of the production borehole. The IGI guidelines provide practical descriptions of different ways to convert a pilot borehole into a full production borehole. It can be seen that the principal of the borehole design is to selectively draw upon deeper groundwater and prevent the borehole drawing upon shallow groundwater that is more likely to contain contaminants. The flow arrows in the diagram show water flowing up to the pump intake visible in the cutaway section. This flow of cool water passes the motor at the bottom of the electric submersible pump. A common problem with pumps in old style boreholes is that they are placed below the inflow of water from the lowest fracture. The intake for the water is above the motor. The water around the motor is stagnant and the pump overheats.

An important consideration with this borehole design is to understand and harness the stratification of groundwater flow and the separation of zones of high permeability or fractures and conduits described above in this paper. With this borehole design it is easier for groundwater to flow up to the pump than it is for shallow groundwater to flow down through the rock. It is important to understand water pressure changes and upward and downward gradients in bedrock aquifers. This will be illustrated in the presentation of this paper.

Finally, you may find it useful to note that there are several references in the EPA Advice Note that an experienced hydrogeologist should supervise and direct the work on site

SESSION II

THE GEOCHEMISTRY OF EUROPEAN BOTTLED WATER[1]

EurGeol Kevin T. Cullen PGeo.
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ABSTRACT

This article and the related presentation to the IAH Conference, 2014 draws extensively from the publication, Geochemistry of European Bottled Waters which was produced in 2010 by the EuroGeoSurveys Geochemistry Expert Group and edited by Clemens Reimann and Manfred Birke.
The effective administration of the EU requires comparable data at the European scale. Providing representative groundwater samples at the European scale at any meaningful sample density would be prohibitively expensive. The collection and analysis of bottled waters across Europe provides a first impression of (or 'proxy') for the natural concentration of, and variations in, the determined chemical elements and additional parameters in groundwater at a European scale.

Samples of bottled water were purchased by the EuroGeoSurveys Geochemistry Expert in supermarkets in 2008. The samples were forwarded to the Federal Institute for Geosciences and Natural Resources in Berlin and analysed for 63 elements and 7 other parameters. A total of 1,785 individual brands were collected in both glass and PET containers. A final dataset of 884 samples was used for mapping, constructing CP-Plots (Cumulative-Probability) and statistical graphics.

The resulting datasets are presented on a series of maps together with statistical data for the majority of elements and parameters. While recognising and highlighting the statistical limitations of the dataset, the authors of the Atlas believe that the range of element concentrations represented is reasonably representative of the range of elements naturally present in European groundwater.

The Author acknowledges the Editors' permission and the assistance of Alecos Demetriades and highly recommends the publication.

INTRODUCTION

This article and the related presentation to the IAH Conference 2014 is based upon and draws extensively from the publication, *Geochemistry of European Bottled Waters* which was edited by Clemens Reimann[2] and Manfred Birke[3]. This European Atlas was published in 2010 by EuroGeoSurveys Geochemistry Expert Group and it's editors have kindly allowed me to reproduce here some of their maps, diagrams and findings of the continental scale study. In particular, I would like to thank Alecos Demetriades[4] of the Expert Group for his cooperation in providing original copies of the published maps and charts for inclusion in this article and related presentation.

When Alecos mentioned the *Geochemistry of European Bottled Waters* publication during an EFG[5] conference-call I persuaded myself to source a copy from Amazon.com. I bought the book more with the intention of adding it to my own library of reference books rather than expecting to find its contents of much value or interest to me being based on a small island somewhat removed from

[1] EuroGeoSurveys Geochemistry Expert Group, Norway
[2] Geological Survey of Norway
[3] Federal Institute for Geosciences and Natural Resources (BGR), Germany
[4] Institute of Geology and Mineral Exploration, Athens, Greece
[5] European Federation of Geologists Hydrogeology Expert Group

continental Europe. In fact, the Atlas and accompanying text is an outstanding piece of work and of relevance to every hydrogeologist, regardless of where he or she is working.

NEED FOR AN ATLAS

The effective administration of the EU requires comparable data and information at the European scale. While each country has its own national data base on groundwater chemistry a directly comparable dataset on natural groundwater chemistry at the European scale is not available. A pan-European harmonised dataset of natural groundwater chemistry would contribute significantly to an understanding of groundwater across Europe. Such a dataset would support the implementation of the EU Water Framework Directive and assist in the sustainable management of European groundwater resources.

Providing representative groundwater samples at the European scale at any meaningful sample density would be prohibitively expensive. In addition, issues such as the uneven distribution of wells, varying groundwater chemistry with depth and geology, groundwater residence time and variable well construction might suggest that the production of a comparable groundwater geochemistry database at the European scale is an impossible task. However, such a comparable data base is urgently required by regulators for setting of action levels or for judging the impact of contamination on the natural environment.

The collection and analysis of bottled waters across Europe provides a first impression of (or 'proxy') for the natural concentration of, and variations in, the determined chemical elements and additional parameters in groundwater at a European scale. In doing so it is important to remember in reviewing the data that bottled water may be treated to remove certain unstable elements such as iron, manganese, arsenic and sulphur, that bottle material can contaminate the stored water and that some deep mineral water sources may be 'saltier' than 'normal' shallow groundwater.

While recognising and highlighting the statistical limitations of the dataset, the authors of the Atlas believe that; *the **range** of element concentrations represented in this Atlas (from the very 'dilute' bottled waters of Norway to the mineral-rich waters of Eastern Europe) is reasonably representative of the **range** of elements naturally present in European groundwater (though for some specific elements, the mineral waters may represent the 'high end' of this range).*

The Atlas is an excellent starting point for gaining some impression of European groundwater chemistry. The Atlas provides a much needed basis on which to build a more comprehensive European data base that describes the natural groundwater chemistry to be found in the diverse range of hydrogeological environments found across continental Europe.

BOTTLED WATERS

For the consumer today, and for all practical purposes, there is little difference between Natural Mineral Waters and Spring Waters. (The producers of certified Natural Mineral Waters and their advertising companies would of course want us to think differently.) Critically, both types of bottled waters at the source must be free from pathogenic microorganisms and the total bacterial content must comply with strict criteria.

Natural Mineral Waters do not necessarily need to satisfy normal drinking water quality requirements; they need only satisfy the limits established for Natural Mineral Waters by EC Directive 2003/40/EC. This allows for the sale of mineralised waters whose mineral content or individual elemental composition exceeds that permitted under the EU drinking water regulations. However, and most importantly, all the maximum admissible concentrations refer to natural content, i.e. the contents should not be the result of *'contamination at source'*.

According to Directive 96/70/EC, the term ‘spring water’ shall be preserved for a groundwater which is intended for human consumption in its natural state, and bottled at source, which satisfies many of the same conditions as for mineral waters regarding microbiological purity, source protection, labelling and treatment. Spring Waters must satisfy the normal limits for the quality of drinking water (EU Directive 80/778/EEC) and which may be more stringent than the ‘Mineral Water’ limits.

There are significantly higher catchment and aquifer monitoring requirements associated with the certification of a source as a Natural Mineral Water. However, some producers of Spring Waters do not consider that the added costs of certification and promotion are balanced by the financial return from increased sales.

METHODOLOGY

Samples of bottled water were purchased by the EuroGeoSurveys Geochemistry Expert in supermarkets in 2008 with the aim of getting as good a coverage over the continent as possible. A total of 1,785 individual brands were collected in both glass and PET containers. Using a selection process that;

i) Allowed only one bottle per site
ii) Non-carbonated (still water) was preferred over carbonated water
iii) Clear PET-bottles were preferred to all other battle materials and colours and
iv) Samples sold in glass bottles were only left in the dataset when no other bottle was available.

The rigorous selection process resulted in a final dataset of 884 samples that was used for mapping, constructing CP-Plots (Cumulative-Probability) and statistical graphics.

The samples were shipped to the laboratory of the Federal Institute for Geosciences and Natural Resources (BGR) in Berlin, Germany, where they were kept refrigerated until they were analysed. Chapter 4 of the Atlas describes the sample preparation, the analytical techniques used, the quality control measures adopted, detection limits etc.

The possible impact of bottle material on the water quality was investigated and it was concluded that;

i) Bottled water cannot be used to establish the natural concentration range and variation of Sb.
ii) Ce, Cr, Pb, and Al concentrations in bottled water can be seriously influenced by glass bottles
iii) The majority of elements used in the production of the geochemical maps are not seriously influenced by contamination from bottle materials.

The effect of the limited amount of treatment that can be afforded to bottled waters was also investigated. It was concluded that the distribution data for Fe and Mn are not representative of the natural water composition. It was also concluded that dilution of some samples had likely taken place to maintain the U value below a certain value in the absence of any EU maximum admissible concentration for U in drinking water.

INTERPRETING THE DATA

Groundwater typically begins its hydrological journey as rainfall which falls to the ground and infiltrates into the soil. Rainfall is naturally slightly acidic and contains a range of solutes derived from sea spume or spray. Precipitation near the coast has a higher concentration of Na and Cl and the effect of this can be seen in slightly elevated levels of these elements in shallow groundwater close to the sea. Away from the coast the rainfall becomes less and less mineralised as it passes over the continent with the result that groundwaters in some hardrock aquifers on the continent may have conductivity values of less than 20uS/cm. These groundwaters can have a lower mineralisation than present in rainfall along the Irish coast.

The infiltrating rainfall steadily picks up mineralisation as it firstly passes through vegetation and the shallow soil before mixing with deeper groundwater in aquifers below the water table. As it moves through the aquifer, acid-base reactions release Ca and Mg into the groundwater. Other minerals other than carbonates can also be hydrolysed by the groundwater with the release of Na and K. Other reactions can also take place such as redox reactions that release Fe and S. The solute content of the groundwater continues to rise as does the pH with the final solute content reflecting the availability of leachable elements in the host aquifer.

Each groundwater will have a somewhat unique hydrochemical fingerprint that reflects the balance of all these processes during its evolution, its residence time in the aquifer, and the mineralogy of the rocks and sediments that it has come into contact with since infiltration. Therefore, it can be reasonably expected that uncontaminated groundwaters and which display no impact from human activities should reflect regional geological setting of their host aquifers.

The Atlas produced by the EuroGeoSurveys Geochemistry Expert Group demonstrates that there are enormous natural variations in the concentrations of chemical elements in natural uncontaminated groundwater across continental Europe. These variations are related to many of the geological settings found within continental Europe and are indicative of the range of elements naturally present in European groundwaters.

MAPS

The study carried out by the EuroGeoSurveys Geochemistry Expert Group involved the analysis of the bottled waters for 63 elements and 7 other parameters including, Total Alkalinity, pH, Electrical Conductivity, Nitrate (NO_3), Nitrite (NO_2), Ammonium (NH_4), and Sulphate (SO_4). The resulting datasets are presented on a series of maps together with statistical data for each element and parameter. There are no maps for Antimony, Scandium and Nitrite as the data for these was considered to be unrepresentative of European groundwater.

The reader is referred to the publication, *Geochemistry of European Bottled Waters* for a complete review of the project results.

The distribution maps and extracts of the related commentary for a number of the major ions and trace elements included in the project are reproduced here as examples of the material contained in the publication.

CALCIUM (CA)

‘The map of Ca in bottled water shows high local variation and no clear regional features. It is interesting to note the exceedingly low Ca concentration of many waters on the Scandinavian market. Scandinavia does not have the cultural heritage of ‘true’ mineral waters enjoyed by countries such as Germany, France or Italy. It could be argued that many of the hydrochemically ‘immature’ bottled waters sold in Scandinavia should not even be regarded as mineral waters, due to their very low total mineralisation. The majority of bottled waters with very high Ca concentrations come from Germany, where the water is abstracted from gypsum or a limestone containing formations.‘

MAGNESIUM (MG)

‘The map of Mg in bottled water shows high local variation. Low Mg concentrations are observed in Fennoscandia (lack of carbonate lithologies and a preference for shallow, poorly mineralised groundwaters. Higher, more typical, concentrations are observed in North Western Europe (e.g. Britain and Ireland) were carbonate rich shallow sedimentary aquifers are utilised. The highest values are observed in Central and Eastern Europe, where there is a greater tendency for hydrothermal water or deeper brackish formation waters to be bottled. The waters with the highest Mg concentrations (up to 4,010mg/l) are derived from Hungary and the Czech Republic. The high values are usually related to deep structural waters or wells in ultramafic rocks. In France the strong Mg anomalies in the Massif Central are related to the occurrence of Mg rich amphibolites and weathered Tertiary basalts.’

SODIUM (NA)

'The map of Na in bottled water shows high local variation. The boxplots indicate a predominance of high Na values in the water from Northeastern Europe. This is a surprising observation (but similar to chloride). Given the high Na concentration in seawater, one might expect to see higher Na concentrations in coastal areas, rather than far inland. There are two possible explanations: it could be related to culture, a tendency towards stronger, more mineralised 'tasty' mineral waters in Eastern Europe. However, it could also be related to geology and the deep sedimentary basins that are exploited in these areas for mineral water, giving rise to Na rich brines. The Carpathian Mountain Range as well as the Dinarides are marked by high Na values. The wells with the highest Na concentrations (up to 8,160mg/l) occur in Slovakia and Hungary. Wells are abstracting water from Hercynian granites in France and Portugal also show somewhat increased Na concentrations.'

ALUMINIUM (AL)

The map of Al in bottled water shows no clear regional features, though several of the areas with high values are clearly related to either geology (France, Italy, Portugal), or a combination of geology and climate (the combination of high rainfall and rocks with a low buffering capacity may lead to low pH and generally elevated Al concentrations in Norway). In Italy two different geological sources of high Al are observed: the occurrence of Tertiary volcanic rocks in central Italy and the presence of peraluminous granitic rocks in Calabria. Many high Al values are observed in Germany, where mineral water is traditionally sold in glass bottles. This pattern may thus be caused by the preferred packaging material rather than geology, or it may reflect the prevalence of naturally carbonated mineral waters whose naturally low pH and aggressivity may be sufficient to mobilise Al from the ground.'

ARSENIC (AS)

'Many of the high values displayed on the map occur in parts of Europe that are known for the occurrence of sulphide mineralisation. Furthermore, the alkaline volcanic provinces in Italy are marked by higher than usual As values. The highest value (89.8ug/l) was reported in a Polish mineral water, abstracted in an area with known mineralisation. Even four samples from Germany are above the drinking water standards, one is abstracting the water from Palaeozoic rocks, another drawing water from a major fault zone.'

BERYLLIUM (BE)

'On the map Be concentrations are generally higher from Southern Europe than in Northern Europe, which is somewhat surprising, considering the lower pH of many of the northern waters and the general geology of Scandinavia (predominance of granitic rocks). The young soils in Northern Europe may not be sufficiently weathered yet to release Be or there are simply just not enough wells in Scandinavia to obtain a representative view. Furthermore , a number of clearly geology related hotspots can be seen on the maps; key sources for enhanced Be in the bottled water are young evolved granitic intrusions (like in N- Portugal, France) or young volcanic rocks (like Italy, Canary Islands). In Germany the crystalline rocks of the Black Forest are indicated. The highest value (641ug/l) was reported from Portuguese, well (Hercynian granite).'

CADMIUM (CD)

On the map, Cd concentrations are slightly higher in the waters from Southern Europe than in those from Northern Europe. Several of the anomalies visible on the map are linked to the occurrence of young volcanic rocks (e.g. Italy). The high values observed in Germany, are linked to a variety of different sources - from structurally controlled deep wells to shallow wells in young sediments with little protection against contamination. The one high value in Norway is related to a deep well in mineralised green stones. The highest value (1.13ug/l) was reported from a Slovakian well (young volcanic rocks in a mineralised area). It should be noted that even this concentration is well below the EU drinking water limit of 3ug/l.'

SILICON (SI)

'The map of Si in bottled water shows some clear regional features. High values occur in connection with young volcanic activity and rocks on the Canary Islands, in Central Southern Italy and on Cyprus. High values in the Massif Central in France are also linked to the occurrence of volcanic rocks. The Carpathian Mountain Chain and the Dinarides are marked by many elevated values. In Germany, high values occur in connection with Tertiary volcanic rocks in the Rhine/Eiffel region and 'Oberlausitzer Bergland' and with altered gneisses in the Black Forest, Fichtelgebirge and Frankenwald areas. The highest values (up to 58.9mg/l) occur in Spain (Canary Islands) and Italy and are related to active volcanism.'

VANADIUM (V)

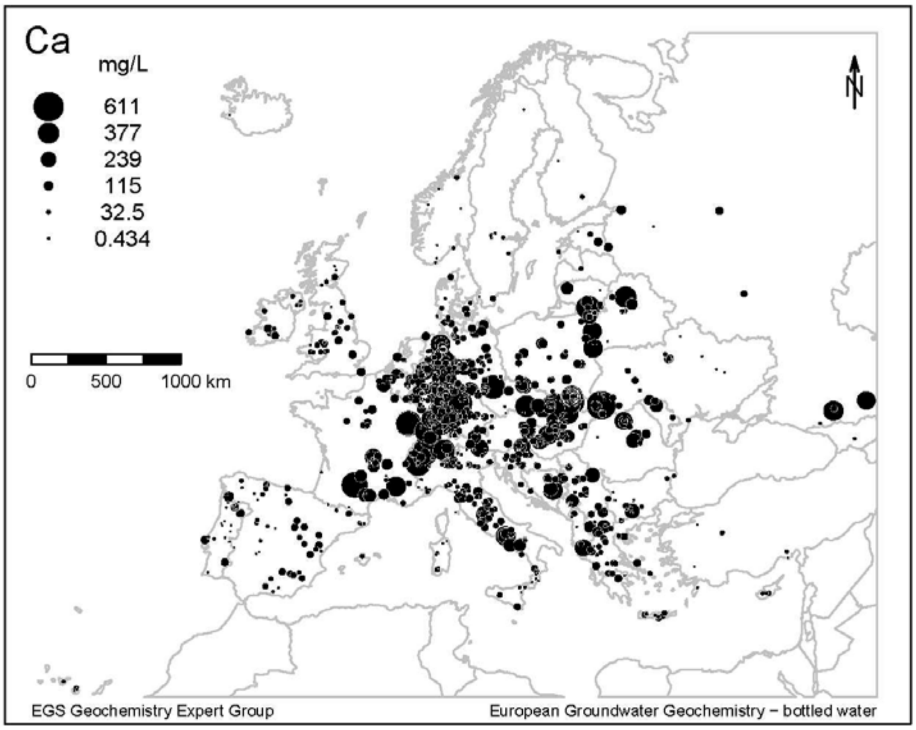
On the map, the active volcanic areas in Europe are clearly marked by V anomalies (Iceland, Canary Islands, Cyprus, Italy) in the bottled water dataset. In France, V anomaly coincides with the Massif Central, possibly linked to volcanic lithologies. In Northern Ireland, the influence of the Palaeogene basalts is visible. In Northern Hellass the V anomalies are associated with ophiolites and possibly bauxite. The highest value (48.9ug/l) was reported from an Italian bottled water and is linked to the alkaline volcanic province.'

COMMENT

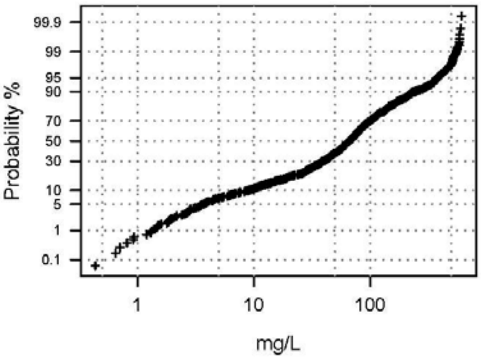
The authors of the publication, Geochemistry of European Bottled Waters highlight all the limitations associated with promoting the bottled water dataset as being representative of the chemistry of European groundwaters.

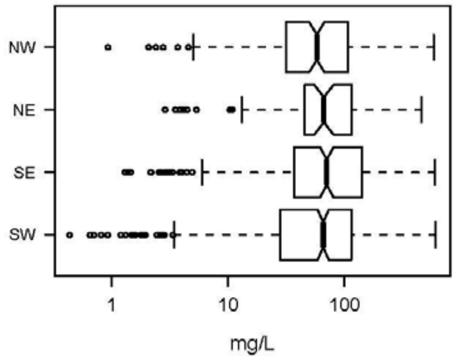
'The very idea of a truly representative European groundwater dataset is highly elusive. What does 'representative' mean when one is talking about a three dimensional, dynamic medium? Groundwater quality varies with time and depth – how could one hope to pin it down on a two-dimensional map? At any given point on a map, for example, the chemistry of deep groundwater may be very different from that of shallow groundwater.'

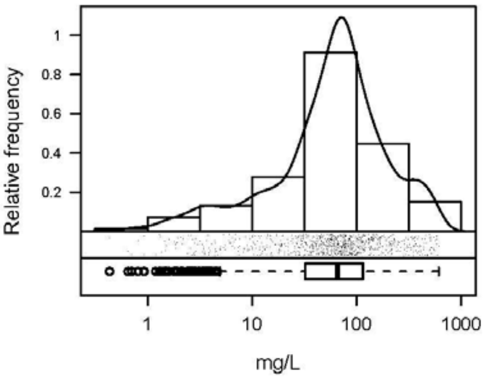
While recognising all the acknowledged statistical and sample distribution limitations of the dataset, the analysis of so many bottled waters from all over Europe in a single laboratory with a high degree of quality control for such a wide range of elements and parameters has provided a value insight into the chemistry of natural European groundwaters in terms of median values and variation.



Drinking water standard: none defined



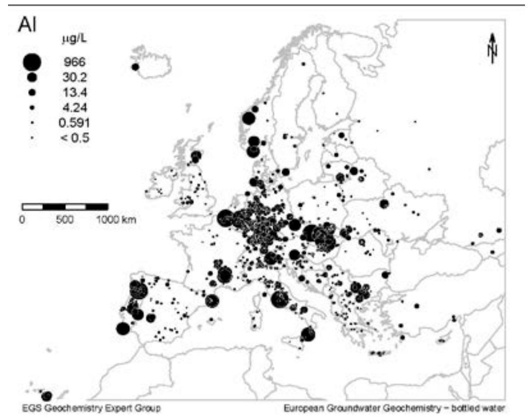


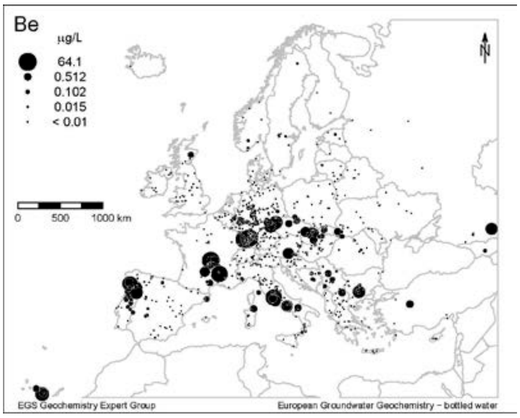


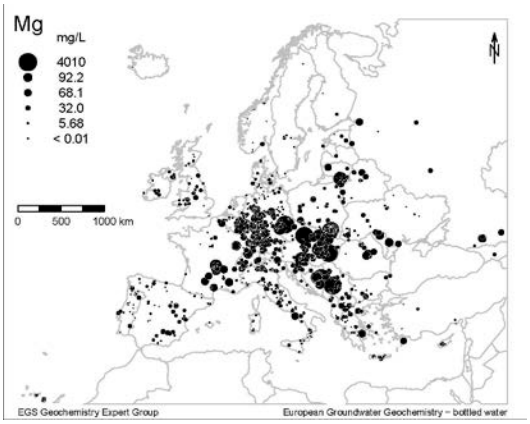
Analytical Method: ICP−OES
Detection limit: 0.01 mg/L
0 (0.00%) < detection limit

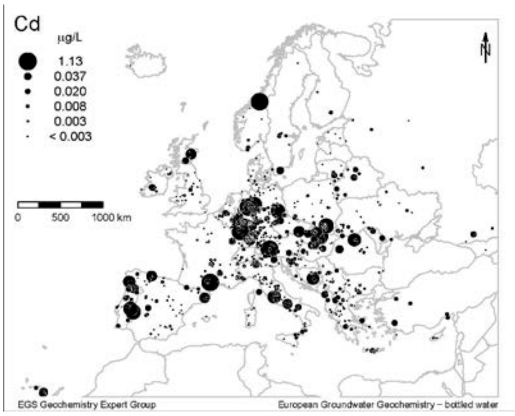
n:	884
Minimum:	0.434 mg/L
5%:	3.71 mg/L
25%:	32.5 mg/L
Median:	65.9 mg/L
75%:	115 mg/L
95%:	377 mg/L
Maximum:	611 mg/L

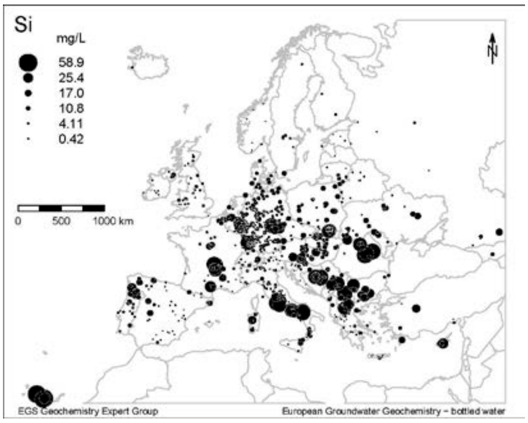
Calcium Ca

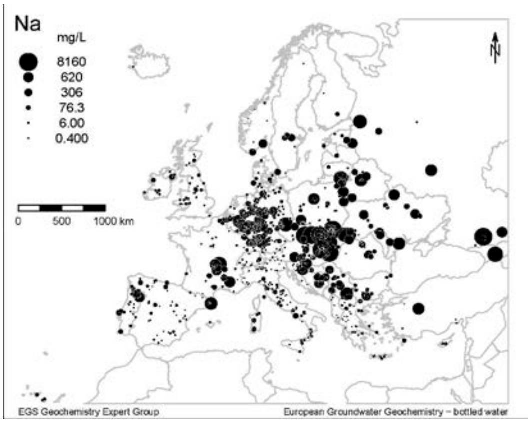

Al
µg/L
966
30.2
13.4
4.24
0.591
< 0.5
0 500 1000 km
N
EGS Geochemistry Expert Group
European Groundwater Geochemistry – bottled water


Be
µg/L
64.1
0.512
0.102
0.015
< 0.01
0 500 1000 km
N
EGS Geochemistry Expert Group
European Groundwater Geochemistry – bottled water


Mg
mg/L
4010
92.2
68.1
32.0
5.68
< 0.01
0 500 1000 km
N
EGS Geochemistry Expert Group
European Groundwater Geochemistry – bottled water


Cd
µg/L
1.13
0.037
0.020
0.008
0.003
< 0.003
0 500 1000 km
N
EGS Geochemistry Expert Group
European Groundwater Geochemistry – bottled water


Si
mg/L
58.9
25.4
17.0
10.8
4.11
0.42
0 500 1000 km
N
EGS Geochemistry Expert Group
European Groundwater Geochemistry – bottled water


Na
mg/L
8160
620
306
76.3
6.00
0.400
0 500 1000 km
N
EGS Geochemistry Expert Group
European Groundwater Geochemistry – bottled water

WATER QUALITY IN IRISH KARST AQUIFERS

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ABSTRACT

This paper provides a review of Irish research on water quality in karst aquifers. Karst regions are particularly vulnerable to pollution due to the occurrence of point recharge, the thin, patchy soil cover found in some karst areas, the presence of epikarst and the occurrence of conduit flow within the aquifer. The main drinking water quality concern is the presence of microbial pathogens including protozoan parasites resistant to chlorination. Ecological quality issues are also significant, involving the transfer of nutrients (phosphorus and nitrogen) through karst aquifers to sensitive ecosystems. Large temporal variations in water quality pose a particular challenge in managing karst water resources. Agricultural best management practices and public education can play an important role in karst groundwater quality management.

INTRODUCTION

Karst water quality is a topic of particular concern here in Ireland because the predominant bedrock aquifer in Ireland is composed of Carboniferous limestone, and much of the limestone is karstified to at least some degree. As seen from Figure 1, regionally important karst aquifers dominated by conduit flow (Rkc) are found extensively west of the River Shannon, while those with a higher proportion of diffuse flow (Rkd) are found particularly in the south of the country.

Awareness of water quality problems in Irish karst aquifers dates back over many years. For example, Naughton (1983) documented the pollution of Teesan Springs in a County Sligo by the disposal of farmyard slurry and silage effluent through a soakpit 500 metres from the springs. Drew (1984) noted that arterial drainage in the Clarinbridge catchment, County Galway, involving channel excavation into the limestone bedrock, resulted in line and point recharge with polluted surface waters, causing a nitrate plume in the karst aquifer. Aldwell *et al*. (1988) reviewed faecal bacterial contamination of Irish karst aquifers by agricultural point sources, with two water quality surveys in karst regions showing more than 50% of groundwater sources to contain *E. coli*. Research on Irish karst water quality has continued up to the present day, as seen from the review below, and this has fed into both outreach material and policy. For example, the vulnerability of karst aquifers to pollution was highlighted in the booklet on the Karst of Ireland (Daly *et al*., 2000) which is available on the Geological Survey of Ireland website, and the vulnerability of karst aquifers has been taken into account in both the Irish groundwater protection scheme (DELG *et al*., 1999) and the groundwater risk assessments for the Water Framework Directive (Working Group on Groundwater, 2005).

Unfortunately, despite this long history of awareness of karst groundwater vulnerability in the Irish hydrogeological community, water quality problems continue to arise, most recently with problems of *Cryptosporidium* in a number of karst springs providing drinking water supplies in Roscommon (Roscommon County Council, 2013). Therefore it is timely to review Irish experience and understanding of karst groundwater contamination and to consider the approaches required to prevent future problems.

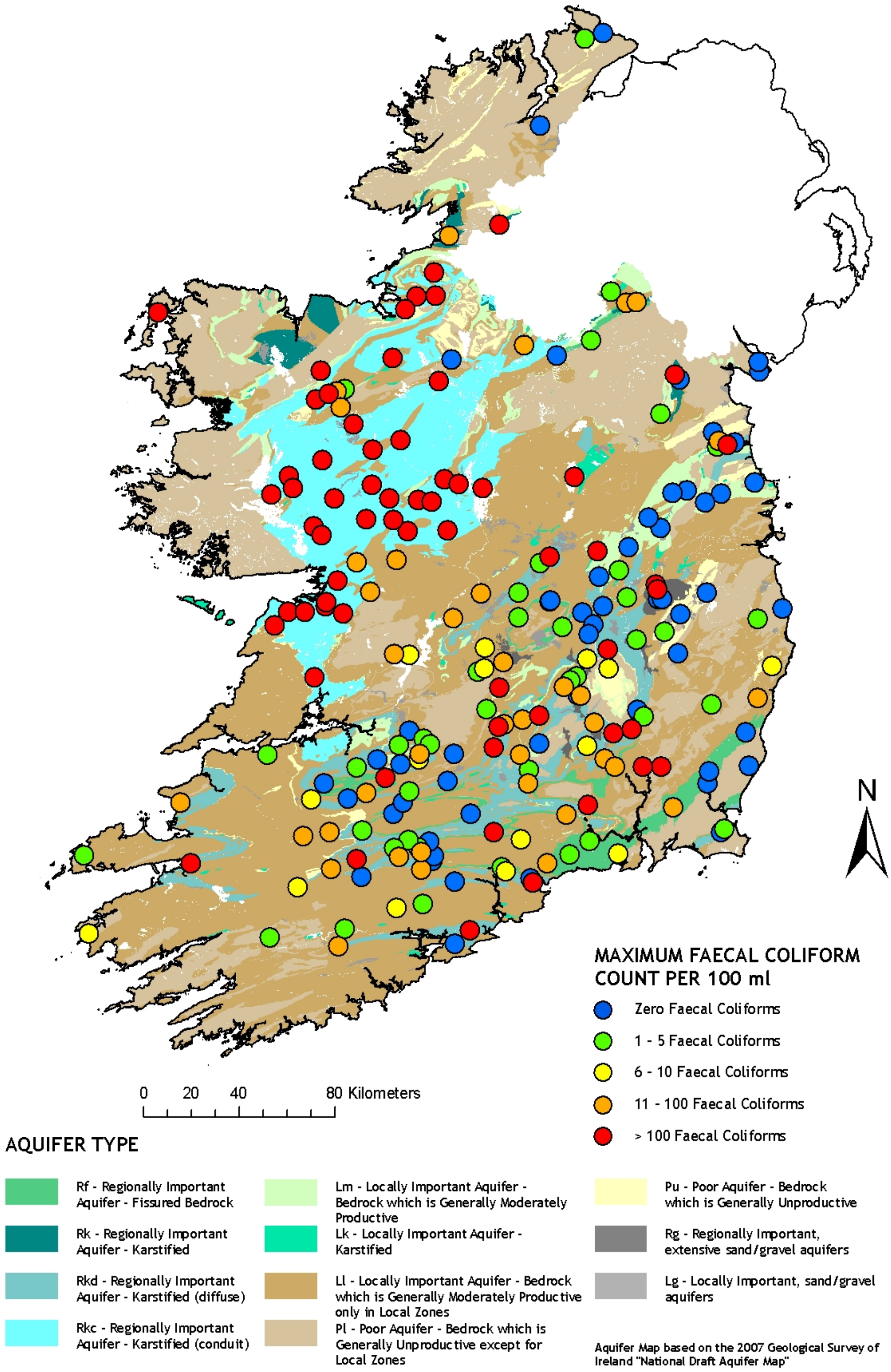

Figure 1: Faecal coliform detections in raw untreated groundwater, 2007-2009 (from Craig *et al.*, 2010)
(Note the high occurrence of faecal coliforms in the western Irish regionally important karst conduit aquifers (Rkc).)

RISK ASSESSMENT

Assessing risk of water quality problems involves a consideration of source, pathway and receptor factors (Daly, 2004). Each of these is discussed below in relation to Irish karst water quality:

Potential sources of groundwater contamination in rural Ireland include diffuse agricultural sources (e.g. spreading of fertilizers) and particularly point sources (e.g. farmyards and septic tank systems) (DELG *et al*., 1999). Our karst uplands such as the Burren plateau in Co. Clare and the Geevagh and Bricklieve uplands in Sligo have relatively low intensity agriculture, but are nonetheless prone to localised contamination from point sources. For example, Drew (1996) recorded that silage clamps on bare limestone pavement in the Burren plateau resulted in contamination of karst springs in summer, at times of maximum water demand. Our karst lowlands have greater diffuse pressures due to more intensive agriculture, particularly in the southern dairying region, and although the effects can be mitigated by pathway factors such as thicker subsoils, problematic concentrations of nutrients (nitrogen and phosphorus) have arisen from both diffuse and localised sources. Faecal microbial contamination may arise from both human and animal waste, including effluent from poorly sited, constructed or maintained onsite domestic wastewater treatment systems, farmyard runoff, badly stored manures and slurries, and landspreading of manures and slurries in vulnerable situations; this is discussed further below.

The pathway factor is the key issue in relation to karst water quality. Karst aquifers are particularly vulnerable to chemical and microbial contamination (Field, 1989; Smith, 1993). Figure 2 shows a schematic diagram of pollution pathways in karst aquifers. An important factor is the occurrence of point recharge, whereby water can enter the aquifer directly via sinking streams, bypassing the protective cover of soil and subsoil. Dolines (karstic closed depressions) also provide point recharge, with the extent to which the water bypasses the protective layers depending on the type of closed depression (Mellander *et al*., 2013). The potential for contaminant entry via karst features is recognised in the Irish groundwater protection scheme, with a zone within 30 metres of such features being classified as extreme vulnerability (DELG *et al*., 1999). The risk is greatest where point sources of contamination coincide with karst features and point recharge, for example where cattle enter sinking streams or where animal carcasses are dumped into karstic closed depressions. The aquifer is also vulnerable where the soil cover is thin and patchy (for example the karst areas of Clare and south-east Galway with limestone pavement and rendzina soils). Within the aquifer, the presence of a highly weathered epikarstic zone can allow rapid lateral movement of diffuse contaminants to vertical shafts. It also provides temporary storage for contaminants, which may then be released from this zone by flood pulses (Field, 1989). Finally, the presence of conduit flow within the aquifer allows rapid transfer of contaminants with little opportunity for attenuation by adsorption, ion exchange, chemical breakdown or microbial die-off. The short underground residence time also means that very little time is available for remedial action to avoid contamination of water supplies.

The nature of the receptor will determine which water quality parameters are of concern, and at what concentrations. A key concern is drinking water receptors: karst springs and boreholes provide both public and private water supplies, and transfer of contaminants through the aquifer to these supplies may have implications for human health. A more recent concern, particularly with the advent of the EU Water Framework Directive (2000/60/EC) and the emphasis on integrated management of groundwater and surface water resources, is the influence of karst groundwater quality on aquatic ecosystems such as rivers, lakes and groundwater dependent terrestrial ecosystems including turloughs. Nutrient transfer through karst aquifers to such ecosystems is discussed in greater detail below. Ecosystems within the karst aquifer are another type of receptor of which there is growing awareness, on which groundwater contaminants may have a direct impact.

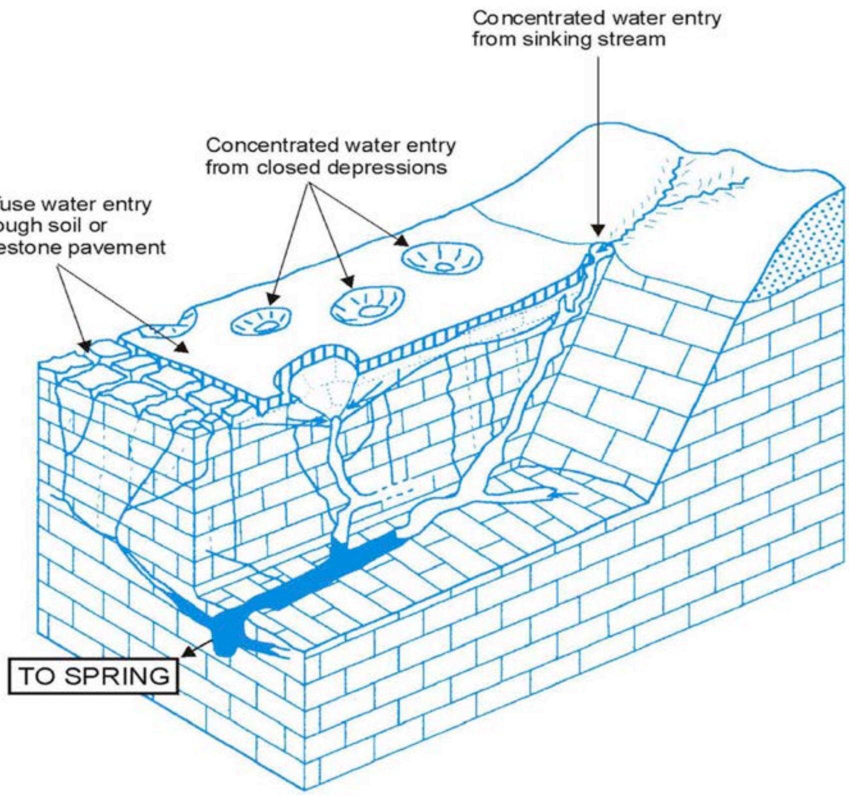

Figure 2: Possible contaminant entry routes to karst groundwater
(From Working Group on Groundwater, 2005, after Gunn)

KARST GROUNDWATER CONTAMINANTS

Nitrate contamination of karst aquifers in rural areas can arise from both diffuse sources (spreading of inorganic and organic fertilizers and release of soil nitrogen due to land use change) and point sources (e.g. badly stored farm wastes and septic tank effluent). The nitrate ion is highly soluble and mobile, so nitrate pollution is found in many free draining hydrogeological situations but karst aquifers are particularly vulnerable. A review of international case studies of nitrate contamination in karst aquifers can be found in Coxon (2011); nitrate in Irish karst aquifers is discussed further below. However, a key aspect of karst groundwater contamination is that it may not be restricted to the more mobile constituents such as nitrate and chloride. Solutionally enlarged joints and sinking streams enable suspended sediment to enter the aquifer. This may cause problems of turbidity in drinking water supplies, but it is particularly important because it can provide a method of entry for a range of adsorbed contaminants including phosphorus, pesticides and viruses. Phosphorus in Irish karst aquifers is reviewed below.

Pesticides occur as contaminants in many aquifer types, but karst aquifers are particularly at risk due to rapid movement through solutionally widened fissures, and also due to the thin soils and subsoils associated with many karst areas. Less mobile pesticides that are readily sorbed to colloidal particles may enter karst aquifers via sinking streams. In Ireland, although pesticides in groundwater have not resulted in any groundwater bodies being placed at poor status, pesticide occurrences have been recorded by the EPA national groundwater quality monitoring programme at a number of locations and data analysis indicated an association with karst aquifers and with springs (McManus *et al.*, 2011). Detailed investigations at a karst spring in County Kilkenny (with a catchment including arable land used for winter wheat and spring barley cultivation) detected five pesticide parent products and five pesticide degradation products (McManus, 2012).

Antibiotics are another group of synthetic organic compounds which may occur as contaminants in karst groundwaters, originating both from septic tank effluent and from animal manures, and giving rise to concern about the spread of antibiotic resistance. Dolliver and Gupta (2008) studied leaching and runoff losses of antibiotics from land application of pig and cattle manure in a karst area of Wisconsin, U.S.A.: compounds detected included chlortetracycline, monensin and tylosin, with detections occurring mainly during the non-growing season (November to April) following autumn application of manures. There is a need for similar investigations of antibiotics in Irish karst aquifers.

Finally, a key feature of karst groundwater quality is the high incidence of microbial contamination.

FAECAL MICROBIAL CONTAMINATION

Microbial pathogens are a particular problem of karst aquifers because the lack of filtration within the aquifer and the short underground residence times mean that if organisms manage to pass through or bypass the unconsolidated material overlying the aquifer, they are almost certain to appear in water supplies. The presence of conduit flow within the karst aquifer can allow viable organisms to travel for hundreds of metres or even several kilometers from the point of entry. The microorganisms involved include bacteria, viruses and protozoan parasites. Sources of microbial contaminants include point sources such as badly stored animal manures and slurries, farmyard runoff and septic tank effluent. Diffuse sources, i.e. landspreading of manures and slurries, may also give rise to contamination where the soil cover is thin and karst features are present, as seen for example in the Walkerton pollution incident in Ontario, Canada in 2000, when 2,300 people became ill and seven people died following karst groundwater contamination which appears to have come from landspreading of cattle manure in a vulnerable location (Daly, 2003).

As noted in the introduction, faecal bacteria have been reported from Irish karst aquifers for several decades. Very considerable improvements in water treatment in recent years have greatly decreased the detection rate of faecal bacteria in treated groundwaters used for public supply, with Hayes *et al*. (2013) reporting a 92% reduction in *E. coli* exceedances in public water supplies and an 89% reduction in exceedances in private group scheme supplies since 2005. However, problems remain particularly in small private supplies, with 11% of these supplies having *E.coli* detections in 2012. Raw water quality remains relevant both because of the large number of private drinking water supplies receiving no treatment or inadequate treatment, and also because of concerns that chlorination does not remove all microbial contaminants, as discussed below. Faecal coliform counts in untreated waters in the national groundwater quality monitoring network are shown in Figure 1, and it can be seen that high counts of >100 faecal coliforms per 100 ml are particularly common in the western Irish Rkc aquifers. As noted by Thorn & Coxon (1992), peak bacterial numbers often coincide with flow peaks, and international karst research (e.g. Pronk *et al*., 2007) has also demonstrated a relationship with turbidity and sediment peaks.

A growing concern with faecal coliform contamination is the presence of verotoxin-producing *E. coli* (VTEC) such as *E. coli* O157 and O26; these can cause dysentery and haemolytic uraemic syndrome, which can be fatal. The Walkerton incident in Canada mentioned above involved *E coli* 0157:H7, combined with *Campylobacter jejuni*. Ireland has the highest reported incidence rate of VTEC in the EU, and exposure to private well water was a risk factor in 47% of cases in 2012 (HPSC, 2013). Viruses are another potential concern: many viruses are retained in soils by adsorption to soil particles, but they may gain entry to karst aquifers while adsorbed to such particles and further work on these contaminants is needed here in Ireland.

Finally, a major concern in karst aquifers is contamination by protozoan parasites, particularly *Cryptosporidium parvum*, which causes acute gastroenteritis, and persistent and potentially fatal disease in immunocompromised individuals. This organism has caused disease outbreaks even where water supplies are treated by chlorination, because it forms an oocyst which is resistant to chlorination; successful removal requires physical barrier treatment e.g. flocculation and filtration, or ozonation. *Cryptosporidium* has been detected in karst groundwater supplies in several countries in the last decade, and here in Ireland it has given rise to problems in the karst springs which provide the water supply for the town of Ennis, County Clare (Page *et al*., 2006, p.64). Within the last year, outbreaks of cryptosporidiosis were traced to several karst springs in County Roscommon used for public water supply, with the Boyle / Ardcarne, Castlerea, Roscommon Central (Ballinagard) and Killeglan water supply schemes all requiring boil water notices (Roscommon County Council, 2013).

CONTAMINATION BY NUTRIENTS

NITRATE

As noted above, nitrate is a common contaminant in many free draining hydrogeological situations, but the characteristics of karst regions make them particularly vulnerable, and passage of this highly soluble and mobile anion to the saturated zone can be very rapid. Because the Carboniferous Limestone has negligible primary porosity, nitrate from diffuse agricultural sources passes rapidly through the fracture network: Richards *et al*. (2005) recorded a vertical travel time of only 34 days through 23 metres of unsaturated zone (0.8m of soil over fractured limestone) at a site in north Cork, while nitrate entering via sinking streams can pass through some karst aquifers by conduit flow in a matter of hours or days.

The synclinal Carboniferous limestone valleys in the south of the country have intensive dairying agriculture, and as a result there are significant nitrogen inputs from both diffuse and point sources. Nitrate in karst aquifers in this region can sometimes exceed the 50 mg/l NO_3 (11.3 mg/l NO_3-N) limit from the Drinking Water Directive (98/83/EC). For example, Bartley & Johnston (2006) investigated nitrate in a karst aquifer beneath a dairy farm in north Cork and found concentrations above the drinking water limit, reaching three times the limit in the groundwater beneath where dairy soiled water irrigation was being carried out in an extremely vulnerable zone. Further research at the same north Cork site by Huebsch *et al*. (2013) investigated factors influencing nitrate concentration over an eleven year period. A decrease in nitrate over the period to below the drinking water limit was attributed to changes in farm management practices (see the concluding section below).

In karst regions with less intensive agriculture such as the Burren plateau and parts of the western limestone lowlands of Clare and east Galway, the shallow, patchy rendzina soil cover and high degree of karstification increases the risk of leaching. However, this is counterbalanced by the lower agricultural intensity and lower nitrogen inputs in these regions, with the result that nitrate concentrations are significantly lower (see Craig et al., 2010, Map 2.2). Therefore nitrate is not a concern from the perspective of drinking water quality, but it may be of ecological concern at much lower concentrations in some situations. Nitrogen can be the limiting nutrient to eutrophication of freshwater bodies in some circumstances and it is generally the limiting nutrient in estuarine and coastal waters: The Surface Water Regulations, (SI 272 of 2009) set an Environmental Quality Standard (EQS) for good status of coastal water bodies of 2.6 mg N/l (at salinity 0 psu). The entry of nitrogen to Galway Bay from submarine and intertidal groundwater discharge has been studied by Cave & Henry (2011).

PHOSPHORUS

Rivers, lakes and turloughs on the western Irish limestone lowlands have close interactions with karst groundwater (Coxon & Drew, 2000), and phosphorus is generally the limiting nutrient to eutrophication in these ecosystems. Investigations of groundwater phosphorus (P) in this area (Kilroy & Coxon, 2005) showed that mean total phosphorus (TP) concentrations in both springs and boreholes were greater than the 20 □g/l threshold
Phosphorus Regulations (SI 258 of 1998), with some sites also exceeding the more recently set groundwater threshold value of 35 □g/l Molybdat
9 of 2010), which is based on the Irish EQS for good status of river water bodies (from the Surface Water Regulations, SI 272 of 2009). The combination of ecologically significant groundwater phosphorus concentrations and the high contribution of groundwater flow to surface waters in karst

regions resulted in 101 groundwater bodies occupying 13.3% of the area of Ireland being designated as of poor chemical status for the Water Framework Directive (Daly, 2009).

Kilroy & Coxon (2005) found that dissolved reactive phosphorus (DRP) was the dominant P component in karst springs, but particulate P and dissolved unreactive P increased to a greater degree than DRP during periods of high rainfall. Research by Mellander *et al*. (2013) at Cregduff springs, Co. Mayo, showed that in this catchment the soils have an unusually high capacity to retain P and buffer against P leaching, and over 90% of the mapped dolines (karst depressions) are sediment floor dolines, which they classed as low P risk; these factors help to explain why total reactive P concentrations at the spring remain below the groundwater threshold value for most of the time. However, karst aquifers remain at risk from high P loads from point sources such as badly stored farm wastes or from point recharge via sinking streams. Kilroy & Coxon (2005) provided a case example of passage of a contaminant plume attributed to release of silage effluent to a sinking stream, resulting in an increase in TP concentration in the downgradient spring from 42 to 1,814 □g/l within 24 hours. Such localised pollution incidents are of particular concern in the summer months when groundwater constitutes a high proportion of river flow.

POTENTIAL IMPACT ON ECOSYSTEMS WITHIN KARST AQUIFERS

Ecosystems within the karst aquifer, including organisms in caves and within the network of solutionally widened fractures, are currently receiving increased attention (although as noted by Knight & Penk (2010), the first record of Irish groundwater Crustacea dates back to Kinahan in 1863). Knight & Penk's 2010 survey of Irish groundwater Crustacea in Ireland includes the northernmost record in Europe for the genus *Niphargus*, from the Carrowmore cave system in County Sligo. Wood *et al*. (2008) document the effect of organic pollution on cave invertebrates in Derbyshire; it is likely that such ecosystems will need to be taken into account in future groundwater management decisions.

CONCLUSION: EDUCATION AND MANAGEMENT IMPLICATIONS

While the water quality problems reviewed in this paper are not unique to karst areas, they tend to occur with particular severity because of the distinctive characteristics of karst. In particular, the occurrence of concentrated recharge in closed depressions and swallow holes, combined with the presence of conduit flow and flow in solutionally widened fractures in the aquifer, render karst regions particularly prone to groundwater contamination. As noted in the introduction, this scientific understanding has already been taken on board in Irish environmental policy and management: the national groundwater protection scheme takes special account of karst aquifers, and an awareness of their vulnerability also fed into the Water Framework Directive groundwater body risk assessments. Furthermore, the Good Agricultural Practice regulations giving effect to the EU Nitrates Directive (SI 31 of 2014 and earlier versions) include a ban on landspreading of manure within 15m of karst features such as swallow holes and collapses, and a ban on manure storage within 50m of such features, while spreading of soiled water is strictly limited in karst regions where the depth to bedrock is less than a metre. Nevertheless, some challenges of education and management remain.

Approaches to preventing karst water pollution may vary in different karst regions. In the Burren karst plateau, the BurrenLIFE project aimed to develop a community based model for sustainable agriculture, providing a model for conservation of distinctive karst flora and fauna and prevention of rural depopulation while minimising any adverse impacts on the environment (BurrenLIFE, 2010). Where karst aquifers are overlain by a thicker mineral soil cover and have more intensive agriculture, particularly in the southern dairying region which is likely to see greater intensification in line with the Harvest 2020 policy, the level of implementation and effectiveness of the current karst-specific Good Agricultural Practice regulations should be evaluated, and development of additional agricultural Best Management Practices specific to karst situations may be needed. On the north Cork dairy farm studied by Huebsch *et al*. (2013), decreases in nitrate in the underlying karst aquifer over an eleven year period were attributed to altered timing of slurry applications, reductions in inorganic nitrogen fertilizer usage, the movement of the dairy soiled water irrigator to a less karstified area of the farm and a change to minimum cultivation rather than ploughing during pasture re-seeding operations, and the authors concluded that implementation of the Nitrates Directive had helped to improve the water quality on the study site. They suggested that mapping of high risk and low risk leaching areas within a farm could be an effective management approach for positioning of the dirty soiled water system and also noted that increased grazing intensity should be strategically positioned on less vulnerable areas.

Improvements to domestic wastewater treatment systems (in terms of siting, construction and operation), and the installation of mains sewerage in urban fringes with high densities of unsewered housing, should bring about a decrease in microbial contamination. The current national programme of septic tank inspections is being made according to an EPA risk assessment which takes karst into account in the determination of pathway susceptibility (EPA, 2013). (It should be noted however that because the risk ranking is also based on unsewered housing density, the areas of low risk from MRP and pathogens in the western Irish karst lowlands in EPA (2013) Figure 14 cannot be taken to mean low risk of faecal contamination of an individual domestic well). Agricultural best management practices can also result in reductions in faecal bacterial numbers, as noted in a U.S. study by Boyer & Pasquarell (1999). However, while improved management of both human and animal waste can improve the microbial quality of karst waters, it is not feasible to eliminate such contamination. Shutting down of raw water intakes at times of maximum risk, during the passage of flood waves at karst springs, may be a useful precaution. However, reliable prediction of contamination incidents can be problematic. Although turbidity is often highly correlated with microbial pathogens, Auckenthaler *et al*. (2002) warn that in some karst systems, microbial numbers may rise several hours before turbidity, so they suggest that spring discharge may be a safer warning parameter. Because of the difficulty of prediction and potentially serious public health consequences, adequate drinking water treatment remains essential.

Finally a renewed and vigorous programme of public education about the vulnerability of karst systems would be beneficial for all Irish karst regions, to increase awareness of the need for adequately constructed and maintained domestic wastewater treatment systems and to prevent problems of poor waste management and illegal waste dumping. A combination of increased public awareness and pollution prevention measures can ensure that the ecological quality of rivers, lakes, turloughs and coastal waters fed by karst groundwater is maintained or improved. From a drinking water perspective, preventative measures to achieve better quality raw water quality need to be combined with adequate investment in water treatment, particularly for protozoan parasites, to ensure consumer safety. Irish hydrogeologists can potentially play an important role in public education and in highlighting the need for prevention measures.

REFERENCES

Aldwell, C.R., Thorn, R.H. & Daly, D. (1988) Point source pollution in karst areas in Ireland. In Daoxian, Y. (ed.) *Karst Hydrogeology and Karst Environment Protection* (Proceedings 21st IAH Congress, Guilin, China), Geological Publishing House, Beijing, Part 2, 1046-1051.

Auckenthaler, A., Raso, G. & Huggenberger, P. (2002) Particle transport in a karst aquifer: natural and artificial tracer experiments with bacteria, bacteriophages and microspheres. *Water Sci. Technol*. 46: 131-138.

Bartley, P. & Johnston, P. (2006) *Eutrophication from Agriculture Sources, Final Report, Nitrate Leaching – Groundwater* (2000-LS-2.3.1.3-M2). Environmental Protection Agency, Wexford.

Boyer, D.G. & Pasquarell, G.C. (1999) Agricultural land use impacts on bacterial water quality in a karst groundwater aquifer. *J. Am. Water Resour. Assoc*., 35: 291-300.

BurrenLIFE (2010) BurrenLIFE – Farming for conservation in the Burren (Final technical report). Downloadable from http://www.burrenlife.com/reports-publications.php

Cave, R. & Henry, T. (2011) Intertidal and submarine groundwater discharge on the west coast of Ireland. *Estuarine, Coastal and Shelf Science*, 92: 415-423.

Coxon, C. (2011) Agriculture and karst. In van Beynen, P.E. (ed.) *Karst Management*, Springer, Dordrecht, 103-138.

Coxon, C. & Drew, D. (2000) Interdependence of groundwater and surface water in lowland karst areas of western Ireland: management issues arising from water and contaminant transfers. In: Robins, N.S. & Misstear, B.D.R. (ed.) *Groundwater in the Celtic Regions: Studies in Hard Rock and Quaternary Hydrogeology*. Geol. Soc. Lond. Spec. Publ. 182: 81-88.

Craig, M., Mannix, A. & Daly, D. (2010) Groundwater Quality. In McGarrigle, M., Lucey, J. & Ó Cinnéide, M. (eds.) *Water Quality in Ireland 2007-2009*. Environmental Protection Agency, Wexford, pp. 15-40.

Daly, D. (2003) The Walkerton Pollution Incident (2000) – A Summary. *GSI Groundwater Newsletter*, 42: p.13.

Daly, D. (2004) *Groundwater at risk in Ireland – Putting geoscientific information and maps at the core of land use and environmental decision-making*. John Jackson Lecture, 2004: Occasional Papers in Irish Science and Technology, 28. Royal Dublin Society, Dublin, 32pp.

Daly, D. (2009) Groundwater – the 'hidden resource'. *Biology and Environment, Proc. R. Ir. Acad*. 109B: 221-236.

Daly, D., Drew, D., Deakin, J., Ball, D., Parkes, M. & Wright, G. (eds.) (2000) *The Karst of Ireland*. Karst Working Group / Geological Survey of Ireland. http://www.gsi.ie/Programmes/Groundwater/Karst+Booklet/

DELG / EPA / GSI (1999) *Groundwater Protection Schemes*. Department of Environment and Local Government, Environmental Protection Agency and Geological Survey of Ireland, 24pp.

Dolliver, H. & Gupta, S. (2008) Antibiotic losses in leaching and surface runoff from manure-amended agricultural land. *J. Environ. Qual*. 37: 1227-1237.

Drew, D. (1984) The effect of human activity on a lowland karst aquifer. In: Burger A, Dubertret L (ed), *Hydrogeology of karstic terrains - case studies,* International Contributions to Hydrogeology, vol 1. Heise, Hannover, 195-201.

Drew, D. (1996) Agriculturally induced changes in the Burren karst, western Ireland. *Environmental Geology*, 28: 137-144.

EPA (2013) *A risk based methodology to assist in the regulation of domestic waste water treatment systems*. Environmental Protection Agency, Wexford.

Field, M.S. (1989) The vulnerability of karst aquifers to chemical contamination. In: Moore, J.E., Zaporozec, A.A., Csallany, S.C. & Varney, T.C. (ed.) *Recent Advances in Ground-Water Hydrology* (Tampa Florida, 1988). Am. Inst. Hydrol., Minneapolis, 130-142.

Hayes, N., Ní Eidhin, C., Page,D. & Flynn, D. (2013) *The Provision and Quality of Drinking Water in Ireland, A Report for the Year 2012*. Environmental Protection Agency, Wexford.

HPSC (2013) Annual Report 2012, Health Protection Surveillance Centre. https://www.hpsc.ie/hpsc/AboutHPSC/AnnualReports/File,14421,en.pdf

Huebsch, M., Horan, B., Blum, P., Richards, K.G., Grant, J. & Fenton, O. (2013) Impact of agronomic practices of an intensive dairy farm on nitrogen concentrations in a karst aquifer in Ireland. *Agriculture, Ecosystems and Environment*, 179: 187-199.

Kilroy, G. & Coxon, C. (2005) Temporal variability of phosphorus fractions in Irish karst springs. *Environmental Geology*, 47: 421-430.

Knight, L. & Penk, M (2010) Groundwater Crustacea of Ireland: a survey of the stygobitic Malacostraca in caves and springs. *Biology and Environment: Proc. R. Ir. Acad*., **110**B, 211–235.

McManus, S-L. (2012) *Pesticide occurrence in Irish groundwater*. Unpublished Ph.D. thesis, Trinity College Dublin.

McManus, S-L., Richards, K.G., Grant, J., Mannix, A. & Coxon, C.E. (2011) *Are physical site characteristics allowing pesticide products to leach to groundwater in Ireland?* Geophysical Research Abstracts Vol. 13 EGU2011-8259-4, EGU General Assembly, Vienna.

Mellander, P-E., Jordan, P., Melland, A., Murphy, P., Wall, D., Mechan, S., Meehan, R., Kelly, C., Shine, O. & Shortle, G. (2013) Quantification of phosphorus transport from a karstic agricultural watershed to emerging spring water. *Environmental Science & Technology*, 47: 6111-6119.

Naughton, M. (1983) Pollution at Teesan Springs, County Sligo - a case study. *Irish Journal of Environmental Science*, 2 (2): 52-56.

Page, D., Wall, B. & Crowe, M. (2006) *The quality of drinking water in Ireland, a report for the year 2005*. Environmental Protection Agency, Wexford.

Pronk, M., Goldscheider, N. & Zopfi, J. (2007) Particle-size distribution as indicator for faecal bacteria contamination of drinking water from karst springs. *Environ. Sci. Technol*., 41: 8400-8405.

Richards, K., Coxon, C. & Ryan, M. (2005) Unsaturated zone travel time to groundwater on a vulnerable site. *Irish Geography*, 38: 57-71.

Roscommon County Council Water and Sanitary Services (2013) Notices for consumers and boil water notices, http://www.roscommoncoco.ie/en/Services/Sanitation/

Smith, D.I. (1993) The nature of karst aquifers and their susceptibility to pollution. In: Williams, P.W. (ed.) *Karst Terrains, Environmental Changes, Human Impact*. CATENA suppl. 25: 41-58.

Thorn, R.H. & Coxon, C.E. (1992) Hydrogeological aspects of bacterial contamination of some western Ireland karstic limestone aquifers. *Environ. Geol. Water Sci*., 20: 65-72.

Wood, P.J., Gunn, J. & Rundle, S.D. (2008) Response of benthic cave invertebrates to organic pollution events. *Aquatic Conservation: Marine and Freshwater Ecosystems* **18**, 909–22.

Working Group on Groundwater (2005) Methodology for risk characterisation of Ireland's groundwater. Water Framework Directive River Basin District Management Systems, Guidance Document No GW8.

STUDENT POSTER PRESENTATIONS

Ecohydrology of Dune Slacks in Ireland

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ABSTRACT

Dune slacks are hollows in sand dune systems where the water table is close to the surface in summer and groundwater rises to cause flooding in winter. These seasonal wetlands support biological communities which are distinct from those of the surrounding dry dunes, both during the aquatic and dry phases. Humid dune slacks are an Annex I habitat (2190) under the EU Habitats Directive (92/43/EC); sites identified as protected areas under this directive are also protected as groundwater dependent terrestrial ecosystems (GWDTEs) under the EU Water Framework Directive (2000/60/EC) and the status of their associated groundwater bodies must be determined. They have been mapped and assessed in Ireland on the basis of their vegetation in accordance with Article 17 of the Habitats Directive. However, the relationship of biological communities, especially invertebrate communities, to water quality or depth and duration of flooding in dune slacks has not been explored in Ireland.

This project will seek to relate hydrological functioning of dune slacks to their biological communities. In a large-scale survey, environmental factors including water quality, water depth, flooded surface area, vegetation structure and dune slack morphology will be recorded from twenty-four dune slacks in Donegal, Mayo, Kerry and the east coast and compared with their vegetation, mollusc and water beetle communities. A more focussed investigation of the effects of specific land management regimes will be carried out in county Donegal. Here, biological communities of sites which are under different management intensities such as extensive pasture and golf courses will be recorded. Shallow boreholes will be installed in the dune slacks so that water level measurements can be made and water samples can be taken for analysis over a period of twelve months. The resulting data will be used to determine how water quality and flooding regime affect the biological communities of dune slacks under differing management methods.

The results of this project will help to inform conservation policy and future monitoring programmes.

HYDROGEOCHEMICAL CONTROLS ON THE OCCURRENCE OF NITRATE IN GROUNDWATER

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ABSTRACT

Investigation into the distribution of hydrogeochemical variables and their relationships with the occurrence of nitrate (NO_3^-) provides better insights into the prediction of the environmental risk associated with nitrogen use within agricultural systems. The research objective was to evaluate the effect of hydrogeological setting on agriculturally derived groundwater NO_3^--N occurrence. Piezometers (n=36) were installed at three depths across four contrasting agricultural research sites: Johnstown Castle (JC), Solohead (SH), Oak Park (OP) and Dairy Gold (DG) in Ireland. Groundwater was sampled monthly for chemistry and dissolved gases, between February 2009 and January 2011. Mean groundwater NO_3^--N ranged 0.7 to 14.6 mg L^{-1}, with site and groundwater depth being statistically significant ($p<0.001$). Unsaturated zone thickness and saturated hydraulic conductivity (K_{sat}) were significantly correlated with dissolved oxygen (DO) and redox potential (Eh) across sites. Hydrogeological settings significantly influenced groundwater NO_3^- occurrence and suggested denitrification as the main control. Denitrifiers were available at all sites and depths but their activities were controlled by site hydrogeology. Groundwater NO_3^--N occurrence was significantly negatively related to DOC and methane and positively related with Eh and K_{sat}. Reduction of NO_3^--N started at Eh potentials <150 mV while significant NO_3^- reduction occurred <100 mV. Indications of heterotrophic and autotrophic denitrification were observed through elevated dissolved organic carbon (DOC) and oxidation of metal bound sulphur, as indicated by sulphate (SO_4^{2-}).

SPATIAL AND TEMPORAL VARIATION OF GROUNDWATER NITRATE IN TWO CONTRASTING CATCHMENTS

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ABSTRACT

There are large uncertainties in the closing of nitrogen (N) budgets in agricultural catchments. Hillslope hydrologic systems and in particular near-stream saturated zones are active sites of N biogeochemical dynamics. Spatial and temporal heterogeneity in catchment landuse, hydrology and groundwater physico-chemistry gives rise to hotspots and hot moments of N attenuation. The efficiency of nitrate removal in groundwater versus the production, consumption and movement of gaseous denitrification products (nitrous oxide and dinitrogen) remains poorly understood. The focus of this study is to develop a holistic understanding of N dynamics in groundwater as it moves from the top of the hillslope to the stream. This includes saturated groundwater flow, exchange at the groundwater-surface water interface and hyporheic zone flow. Using a combination of groundwater flow modelling (Visual Modflow-Flex), high density spatial and temporal sampling and push pull tracer techniques; it is aimed to contribute to the wider understanding of N dynamics, in terms of the environmental parameters affecting N attenuation, spatial and temporal variability in denitrification rates and gaseous emissions along the hillslope flow path. This project is being undertaken in two ca. $10km^2$ Irish catchments, characterised by permeable soils. Multi-level monitoring wells have been installed at the upslope, midslope and bottom of each hillslope. The piezometers are screened to intercept the subsoil, weathered bedrock and competent bedrock zones. Groundwater/stream water samples for nitrate (NO_3-N), nitrite (NO_2-N), ammonium (NH_4-N) and total nitrogen are collected on a monthly basis while dissolved gas concentrations are collected seasonally. Groundwater NO_3-N profiles from monitoring data from 2010-13 in both catchments differ markedly. Although the two catchments had similar 3 year mean concentrations of 6.89 mg/L (arable) and 6.24 mg/L (grassland), the grassland catchment was more heterogeneous. In the arable catchment, temporal and spatial variability was low, exerting a comparable influence on groundwater NO_3-N (temporal SD: 1.19 mg/L vs. spatial SD: 1.185 mg/L). In the grassland catchment, the temporal variability was low (temporal SD: 0.997 mg/L), while spatial variability was 300% greater (spatial SD: 3.63 mg/L). Mean stream water concentrations in the arable catchment (6.66 mg/L) closely reflected the shallow groundwater, whereas in the grassland catchment, the mean stream concentrations (4.77 mg/L) were lower than mean groundwater levels, suggesting a greater buffering capacity of the groundwater pathway. Elevated NO_3-N concentrations in the midslope and upslope of the grassland catchment decreased to close to the limits of detection at the hillslope bottom. Mean dissolved oxygen concentrations and redox potentials in the near stream saturated zone range from 37% to 40% and from -10 to -113 mV, respectively, indicating an environment which is conducive to denitrification.

QUANTITATIVE ANALYSIS OF FAULT AND FRACTURE SYSTEMS AND THEIR IMPACT ON GROUNDWATER FLOW IN IRISH BEDROCK AQUIFERS

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ABSTRACT

Fault and fracture systems are the most important store and pathway for groundwater in Ireland's bedrock aquifers either directly as conductive flow structures or indirectly as the locus for the development of dolomitised limestone and karst. Through the quantitative analysis of fault and fracture systems in the broad range of Irish bedrock types, this project is designed to develop generic conceptual models of depth dependency, lithological control and scaling systematics for the different fault/fracture systems in order to link them to observed groundwater behaviour. Quantitative characterisation of the main post-Devonian fracture systems from over 70 outcrop, quarry and mine locations, shows that their geometry and nature varies with lithological sequence and with spatial controls, such as depth and regional variations in deformation style and intensity. We briefly describe how some of the most transmissive structures, Tertiary strike-slip faults, and the most ubiquitous structure, joints, are being linked to critical groundwater parameters, such as transmissivity, storage coefficient and connectivity, at both regional and local scales. We show that structural parameters critical to groundwater flow (including orientation, spacing and aperture) can be used to compute ranges of hydrogeological parameters (fracture porosity and permeability) and in combination with hydraulic data (groundwater levels, volumetric flow and recharge) used to provide constraints on permeability anisotropy and heterogeneity on different scales.

SESSION III

COASTAL LAGOONS AND THEIR WATERSHEDS AS GROUNDWATER DEPENDENT ECOSYSTEMS: A CASE STUDY IN THE SW COAST OF PORTUGAL

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ABSTRACT

The study of groundwater dependent ecosystems opened the opportunity to involve specialists of different areas of knowledge in order to obtain answers for complex interrelations between groundwater and the associated ecosystems. The actual study, carried out in two coastal lagoons of the Portuguese SW coast, showed the high dependency of the marine life and vegetation of the lagoons and associated streams discharging in the lagoons on the fresh water supply of these two lagoons and the high contribution they receive from groundwater in the dry period, which corresponds to more than half of each hydrologic year. Every year, the lagoons are artificially opened to the ocean for a few days to a few weeks, which dramatically changes the inside salinity. The sensitivity of these ecological niches is demonstrated by the strong dependence that some species that are more sensitive to high salinity waters show in relation to the entrance of freshwater resultant from the discharge of the phreatic aquifer of Sines sedimentary Basin. The great biodiversity of these lagoons and its precarious balance is only possible to preserve if the aquifer continue to act as a regulatory factor of the lagoon's salinity. The equilibrium can be changed in the event of overexploitation of the phreatic aquifer, which is not at risk in the near future. In a scenario of climate change the lagoons will benefit from a slow increase in groundwater contribution, due to the rise of sea level, which will be accompanied by a rise in groundwater levels in the aquifer near the sea.

INTRODUCTION

Following a series of previous studies related with biology, geology and water resources, including contamination and prospecting of groundwater resources in the area of a number of lagoons (3) located in the Basin of Sines, a sedimentary basin containing Mesozoic and Cenozoic deposits overlying Paleozoic hard rocks, a project was developed to study (2010-2013) the interactions between surface water, sea water and groundwater and its importance for aquatic ecosystems in lagoon coastal systems. To develop this project an extensive multidisciplinary team of biologists and water specialists were assembled in order to give answers to the behavior of species according to the relations between waters of different origins inside the lagoons.

The main objective of this project was to identify the relations between saltwater, surface water and groundwater in two of the coastal lagoons (Santo André and Melides, Figure 1 and Figure 2) and to evaluate the ecologic responses of the groundwater dependent ecosystems in face of a possible reduction on groundwater recharge in a scenario of climate change or aquifer overexploitation.

In order accomplish with the objectives, the following questions were addressed:

- Does the water balance of the aquifer (recharge/uses) affect the water balance in the lagoons?
- Will the biodiversity of the little river systems and of the discharge zone in the lagoons be affected by the different level of dependency of the rivers in relation to groundwater?
- Will changes in the biodiversity at the interface of groundwater with surface water occur at the same temporal and spatial scale as the ones in the rivers and lagoons?

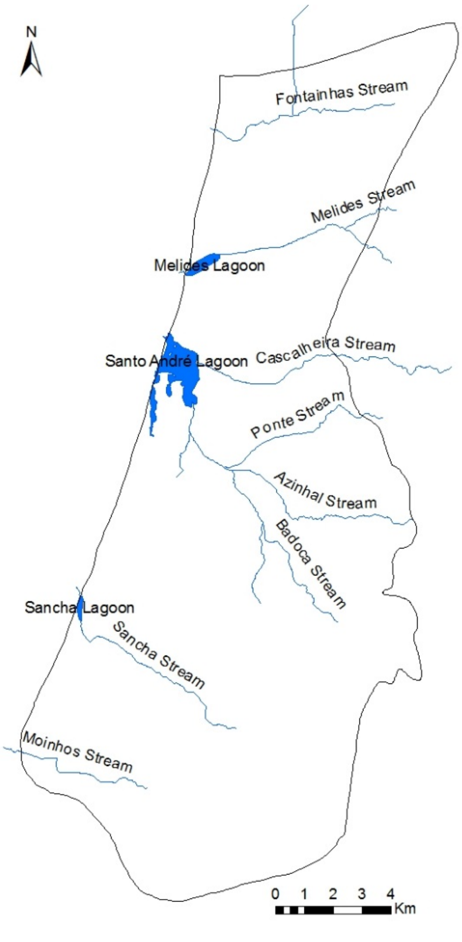

Figure 1. Position of the Santo André and Melides Lagoons in the sedimentary Basin of Sines, including the river network draining to these two lagoons.

Some results of this project can be seen in Correia et al. (2012), Chaínho et al. (2013) or Félix et al. (2013).

THE COMPLEXITY OF WATER INTERACTION IN THE LAGOONS

The system of coastal lagoons in the eastern coast of South Portugal is highly diverse and complex. The Sines sedimentary basin is a complex geologic structure formed since the opening of the Atlantic Ocean, containing formations from Triassic and Jurassic, but lacking the Cretacian in land. Cretacian was detected offshore, which means it was only deposited in the inner part of the basin or, more probably, it was eroded before the deposition of the more recent Miocene and Plio-Pleistocene sediments that overlay the Mesozoic formations.

The evolution of the coast, in a bay environment, had created conditions suitable for the creation of lagoons, which will tend to close in a short time. These lagoons have a maximum deep of a little bit more than 2 m and are very important in the context of the natural areas in Portugal, for flora, birds nesting and feeding and for reproduction of fish on the Portuguese western coast, which happens anytime the lagoons are artificially open to the sea. The evidences show that in the last decades the impact of sedimentation has been incremented, due to deforestation and agricultural use of the lands in the watersheds, creating conditions for a more rapid filling of these lagoons.

The ecological importance of these lagoons had led to the creation of the Natural Reserve of Santo André and Sancha Lagoons (the last one south of Santo André Lagoon and almost filled with sediments nowadays).

The two lagoons defined as targets for this project (Melides and Santo André, Figure 2) were chosen due to their dependence on groundwater, received through little streams which are dependent of the water of the top phreatic aquifer of Sines. Both lagoons are isolated from the sea by sand barriers except for short periods during early spring when the connection to the sea is artificially made by the opening of an inlet in the sand barrier. The reason for this opening is to emulate the natural opening of the lagoons to the sea, which happened naturally in the past (still in the beginning of the twenty century) and also to avoid the eutrophication or contamination of the waters of the lagoons, which can happen through discharges of water from agriculture (rice fields mainly) along the streams that drain into the lagoons. During the period of time that the Lagoons are opened to the ocean, the sea water mixes with brackish water inside the lagoon, influencing the spatial and temporal salinity regime. The inlet provides the main exchange between the lagoons and the sea (discharge of retained materials and recruitment of marine species), and the opening persists for a variable period (ranging from a few days to a few weeks) until the they are sealed off by the sea waves and currents (Cancela da Fonseca et al., 1989; Costa et al., 2003). In 2011 Melides lagoon was opened for 3 days in April and Santo André for 56 days, from March to May (Félix et al. 2013).

Figure 2. Lagoons of Melides (A) and Santo André (B); the sand bar that separates the lagoons from the sea are on the background of both photos.

THE AQUIFER SYSTEM OF SINES

The Basin of Sines contains at least two main aquifers, the top one a phreatic aquifer on marine Miocene and Plio-Pleistocene formations, and the second one an artesian karstic aquifer based in carbonate Jurassic formations. They are separated by the top non-fractured layers of the Jurassic limestones and sometimes also by clay formations of the more recent Miocene sediments. The phreatic aquifer is sometimes defined as multilayered; in many parts drilling has shown clay layers up to 20 m thick. However, it is clear that none of these clay layers are continuous and the location is highly variable along the aquifer, corresponding clearly to lenses in different stratigraphic positions. This is the main reason why the upper aquifer is normally considered, for practical reasons, as a unit, once there is hydraulic connectivity between these water points.

The two aquifers are only separated by the impermeable layer(s) in the aquifer’s north-western part (white area on the left map in Figure 3). In the eastern and south part (grey area) the karstic and the porous upper aquifer are linked. The division is clearly marked in Figure 3 by the main fault F-F defined on the map and in both geological and hydrogeological cuts A-B. The general flux on both aquifers occurs from the eastern limit of the aquifer system to the western part (ocean), as it can also be seen in the hydrogeological profile of Figure 3. The recharge area of the confined aquifer is clearly the northern part of the grey area of Figure 3.

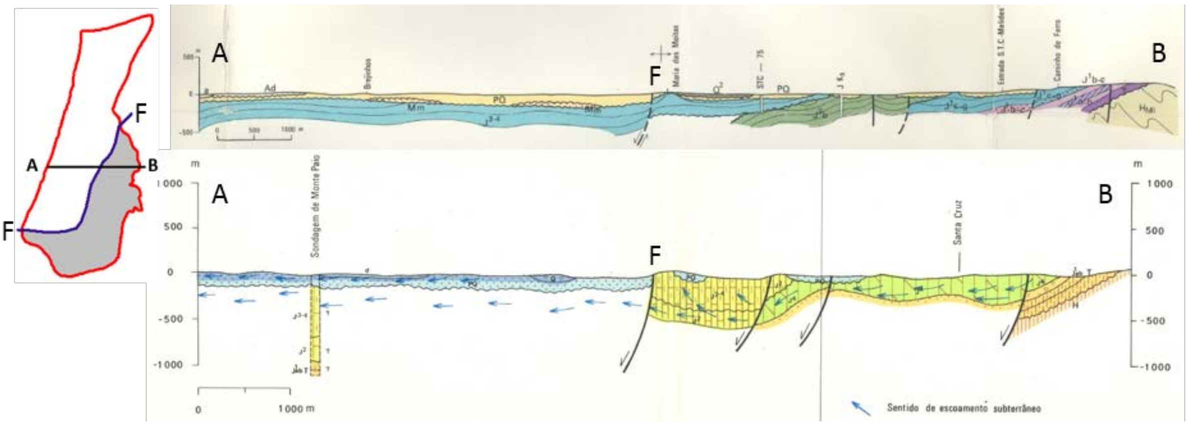

Figure 3. Left: the two areas of the aquifer system of Sines, separated by the fault F-F; in the white area there are two aquifers, the sedimentary porous phreatic aquifer on top and the confined karstic aquifer on the bottom; in the grey area both the karstic and the sedimentary aquifer are linked.
Top profile: Geologic profile of Sines Basin (Oliveira 1984). At right (east) the Paleozoic Formations, over imposed by the Terciary (violet) and Jurassic (blue and green) Formations. On top the sedimentary layers of Miocene and Plio-Pleistocene (yellow).
Bottom profile: Hydrogeologic profile of Sines Basin (Esteves Costa 1994). At right (east) of the fault F both sedimentary (where it exists) and karstic aquifers are linked; at left (west) of the fault, the two aquifers are separated by an impermeable layer and the confined aquifer is between 60 and greater than 100 m deep.

ECOLOGICAL STATUS OF THE LAGOONS

In order to recognize the influence groundwater has in the functioning of the lagoons, the ecological state of both lagoons was analyzed. Principal Components Analysis (PCAs) based on metrics were used for the evaluation of the ecological state, considering benthonic macroinvertebrates, environmental variables and a ratio of non-insects/insects.

The results of this investigation showed:

- That the lotic habitats with finer sediments were associated with a worst ecologic state
- That the better ecologic classifications were associated with a good habitat structure, but also to a minor quality of the existent riparian vegetation.
- A higher taxonomic diversity was associated with the Melides Basin, which seems to depend on a more diverse structure of habitats.
- That the lentic zones have less diversity and dominance of tolerant species, representing less quality.

Inside the lagoons, the study was organized in order to get information near their eastern part, where the influence of streams discharge are more important (at 50 m from the margin) and at 200 m from the eastern margin, where the influence of the streams are less important (Figure 4). Two more points were considered (7, 8) in the Lagoon of Santo André, in this case closer to the sand bar and ocean, but where some influence from the aquifer is noted (Caniços outflow); the same was considered in Melides Lagoon with points 3 and 4.

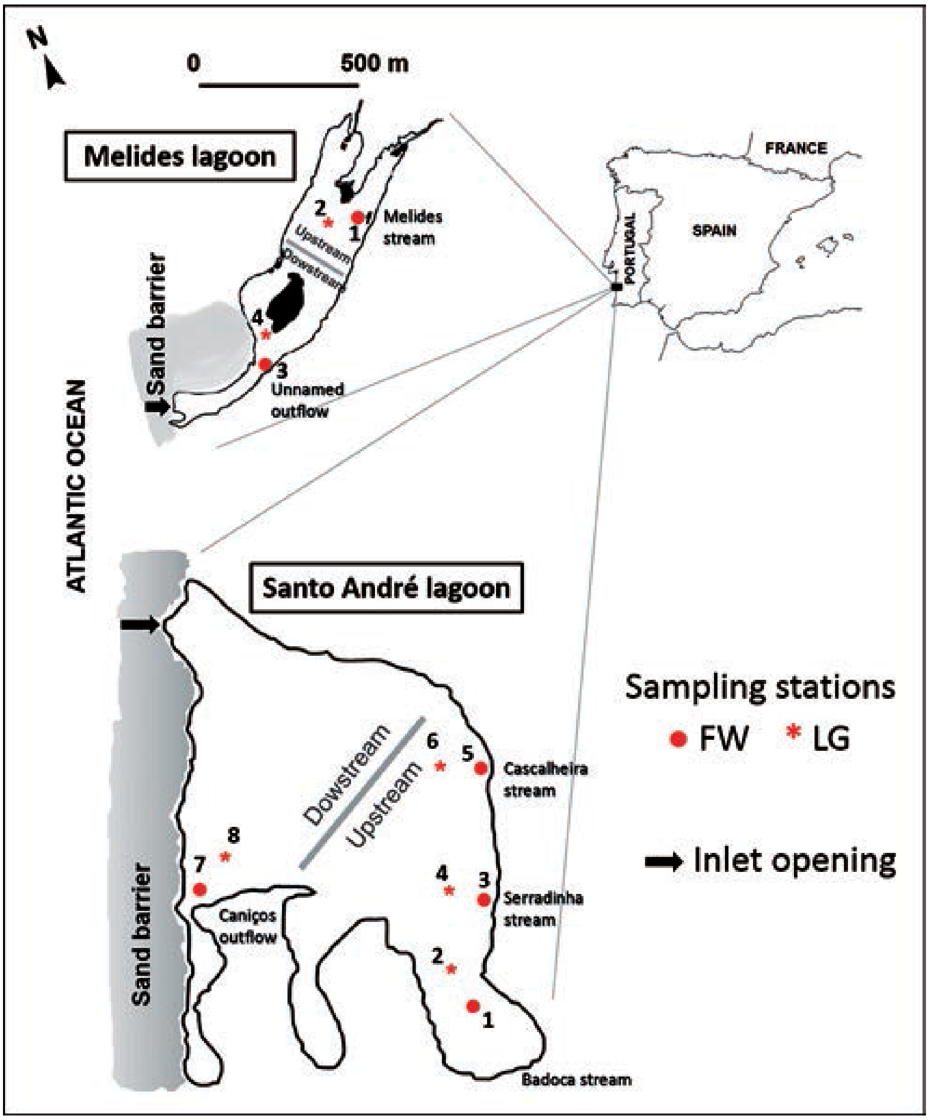

Figure 4. Study area with the location of the sampling stations, with indication of freshwater tributaries and groundwater outflows. FW – direct freshwater influence zone (50 m of discharge); LG – away from the direct freshwater influence (200 m of discharge) (Félix et al. 2013).

RELATIONS BETWEEN GROUNDWATER AND SURFACE WATER IN THE LAGOONS

The study also showed that the direct interaction between the groundwater of the phreatic aquifer and the water of the lagoons is probably null or near null. The lagoons in the past were much more extensive than today and its basins have suffered subsidence in the past. So, the bottom and actual margins of the lagoons are formed by fine sediments (muds) reaching more than 20 m deep, going up to 40 m in some cases. This geological environment does not permit an extensive direct contact between the water in the lagoons and the phreatic aquifer. However, as a previous study showed, the streams discharging in the lagoons are totally dependent on groundwater during most part of the year. Therefore, the lagoons receive groundwater indirectly through these streams (Chambel & Monteiro 2007; Monteiro et al. 2008), and the entrance of groundwater in the lagoons is not diffuse, but concentrated in the mouth of each stream in the eastern margin of both lagoons. Modelling of the phreatic aquifer (Figure 5) shows the flux simulation for this aquifer using two different transmissivity values, 90 and 1,000 m^2/day (Chambel & Monteiro 2007). These two transmissivities were used as there is limited data for the upper aquifer and due to the fact that the transmissivity values change a lot from place to place in addition to clay intercalations in the sedimentary porous sequence and strong lateral variation of the sedimentary facies. Modelling was performed using Feflow (Wasy 2002). The calibration of the model was done using a set of phreatic levels measured in wells in the upper aquifer. The stream beds were also used to refine the model, which permitted the final definition of the flow lines.

Only 6 piezometers have monitored the levels in the phreatic aquifer since 1983 (ARH Alentejo 2012). In a single campaign in 2010, 330 wells where identified in both aquifers (phreatic and confined), with records of the phreatic levels and field physical-chemical parameters (EC, pH and temperature) collected. From 2010 on, under the remit of the present project, the data was collected from around 100 shallow wells in the phreatic aquifer during the wet and dry seasons. The general phreatic gradient is E-W, following the topography, which attains more than 180 masl in the eastern part of the aquifer and is at sea level on the aquifer western part. Therefore, water discharges mainly at the ocean. However, locally, there are very different flow directions, mainly near the few streams crossing the area. Streams present influent and effluent sectors, depending on location and hydrologic period of the year. Stream reaches with an elevation lower than 20 masl tend to be permanent and feed the coastal lagoons throughout the year.

Figure 6A shows one of the places where groundwater discharge can be directly observed on the bottom of the stream bed. Figure 6B and C show a special situation in this drainage system: a well targeting the confined aquifer discharging its artesian flow of 30 L/s to one of the streams flowing to the Lagoon of Santo André for a number of years. Apart from the natural discharge of the phreatic aquifer, this is the only other source of water entering the system of this lagoon during the dry period.

DISCUSSION AND CONCLUSIONS

This project showed that the coastal lagoons of Melides and Santo André are strongly dependent on groundwater, except for the rainy period, where there are some contribution of surface runoff and direct precipitation over the lagoons and not through the bottom of the lagoons. But groundwater contribution is done mainly by the groundwater discharge on the streams flowing to the lagoons. The salinity of the lagoons is mainly due to the yearly opening of a link between the lagoon and the ocean and to some strong evaporation during summer times. The direct link between sea water and the water in the lagoons beneath the sand barrier separating the lagoons from the ocean doesn't seem possible as the water level in the lagoons is always higher than sea level, except during the short period of the artificial opening to the sea.

The study of the ecology of the lagoon systems showed that inside the lagoons there are (Chainho et al. 2013):

A – Marine species that do not tolerate very low salinities
B – Marine species that can withstand very low salinities
C – Freshwater species that withstand brackish environments
D – Freshwater species that tolerate only very low salinities

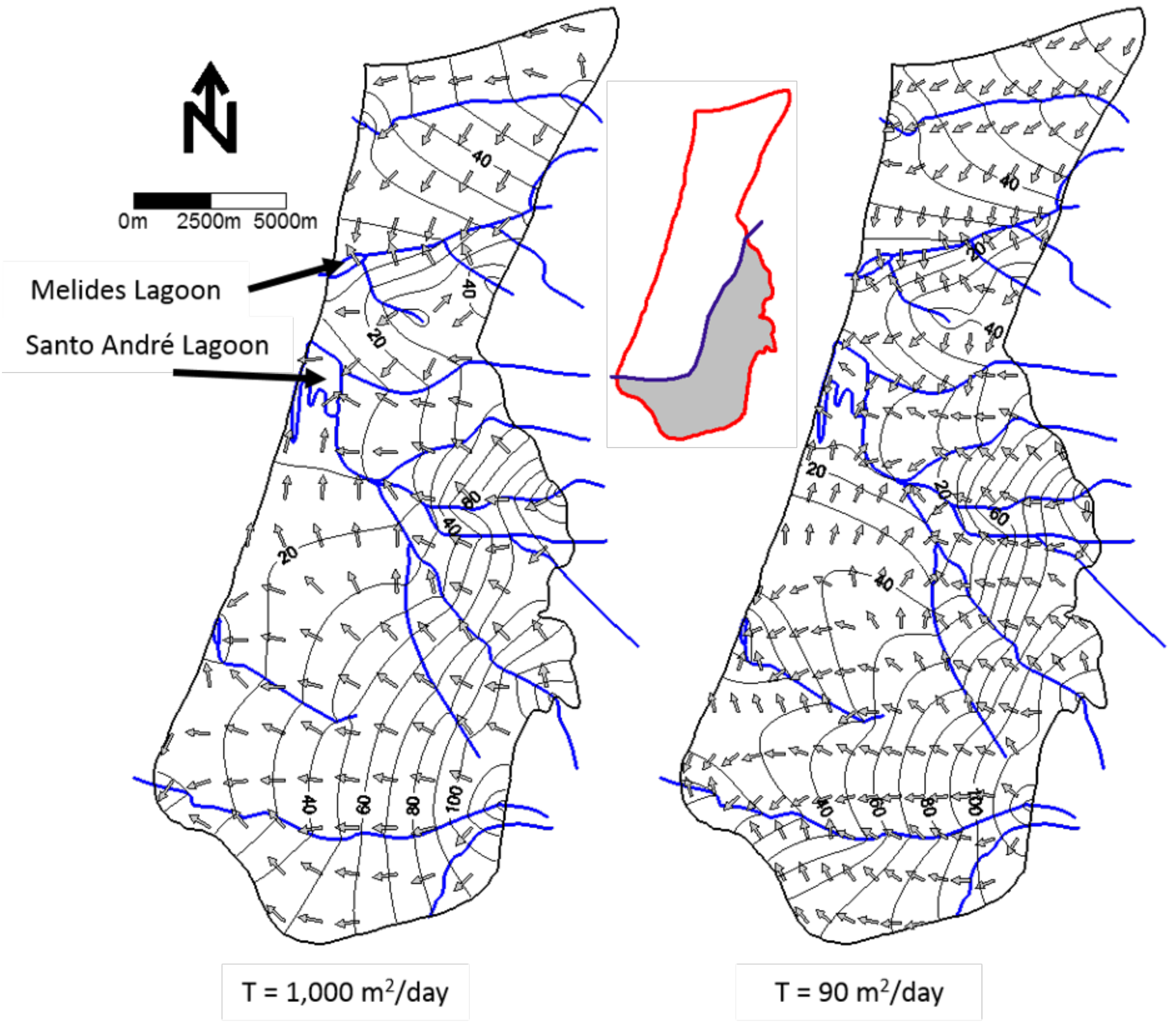

Figure 5. Analysis of the flow in the phreatic aquifer of Sines, assuming two different values of transmissivity (Chambel & Monteiro 2007). The results show that the flow is mainly to the ocean, but locally it flows to the little streams intersecting the territory, discharging part of groundwater indirectly to the lagoons. The central map shows the area where two separate aquifers exist (white area) and the area where there is hydraulic connection between them.

Figure 6. A: One of the points of groundwater discharge in the streams. B: The only artificial discharge to streams directly from the confined aquifer (30 L/s or natural discharge, during the year). C: The stream after receiving the recharge of the confined aquifer (image B).

During and following the opening of the lagoon to the sea, the freshwater species which cannot tolerate high salinities take refuge on the lower parts of the discharging streams, until the environmental conditions inside the lagoon permit their return, which happens slowly after the closing of the inlet between the lagoon and the ocean. In contrast, the marine species that are not so tolerant to very low salinities avoid the areas near the discharge of the streams and take refuge in the area near the dune ridge that separates the lagoon from the sea.

The system is so sensitive that if the abstraction of groundwater increases, there will be a shortage of fresh water into the lagoons, changing the natural spatial and temporal salinity inside. The ecology of the lagoon would change drastically, as all the species less tolerant to salinity would disappear if their freshwater refuge also disappeared. In that case, even if the reaction was to avoid the opening of the lagoon to sea water, the evaporation that occurs would tend to concentrate the internal salinity of the lagoon during the dry period and the pollutants would probably cause irreversible damage to all the life in the lagoon. The consequences could be a great reduction of biodiversity and the survival of only the most resistant species. However, for the moment the abstraction in this aquifer is clearly a reduced percentage of the infiltration that occurs, so in the near future there will be no risk of overexploitation.

In contrast, as the aquifer discharges to the ocean in the area of the lagoons, resulting in an increase in the sea level related with climate changes, the consequence will be an increase also in the groundwater level in the upper aquifer, which will permit an increment of the groundwater discharges to the lagoons in the future.

ACKNOWLEDGEMENTS

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REFERENCES

ARH Alentejo 2012. Planos de Gestão das Bacias Hidrográficas Integradas nas Regiões Hidrográficas 6 e 7, Região Hidrográfica 6. Portugal. Volume 1. Relatório [River Basin Management Plans of the Hydrographic Regions 6 and 7, Hydrographic Region 6. Portugal. Volume 1. Report]. Administração de Região Hidrográfica do Alentejo, 456 pp.

Cancela da Fonseca L, Costa AM & Bernardo JM 1989. Seasonal variation of benthic and fish communities in a shallow land-locked coastal lagoon (St. André, SW Portugal). *Scientia Marina,* 53: 663-669.

Chaínho P, Félix PM, Correia MJ, Fernandes CM, Costa JL, Chaves ML, Stigter T, Hugman R, Salvador N, Costa L, Domingos I, Lopes V, Castro J, Cruz T, Costa A, Monteiro JP, Chambel A, Nunes L, Silva A, Pinto JR, Vidal AM, Pinheiro I, Costa MJ, Cabral, HN, Ribeiro L, Cancela da Fonseca L 2013. Projecto GroundScene (PTDC/AAC-AMB/104639/2008): Biodiversidade de lagoas costeiras e respectivas bacias hidrográficas como ecossistemas dependentes de águas subterrâneas [Project GroundScene (PTDC/AAC-AMB/104639/2008): Biodiversity of coastal lagoons and respective hydrographic basins as groundwater dependente ecosystems]. III Encontro da Rede BRASPOR “Ambientes Costeiros e Bacias Hidrográficas”, Ponte de Lima, Portugal.

Chambel A, Monteiro JP 2007. Sistema Aquífero de Sines – Estudo Hidrogeológico do Sistema Aquífero e Definição dos Perímetros de Protecção das Captações de Água Subterrânea das Águas de

Santo André SA, com recurso a Modelação Matemática [Hydrogeologic Study and Definition of the Wellhead Protection Areas of the Águas de Santo André SA using a Mathematical Model]. Águas de Santo André SA, Tecnical Report, 83 pp.

Correia, MJ, Costa JL, Chainho P, Félix PM, Chaves ML, Medeiros JP, Silva G, Azeda C, Tavares P, Costa A, Costa AM, Bernardo J, Cabral HN, Costa MJ, Cancela da Fonseca L 2012. Inter-annual variations of macrobenthic communities over three decades in a land-locked coastal lagoon (Santo André, SW Portugal). Estuarine, Coastal and Shelf Science 110: 168-175.

Costa AM, Cancela da Fonseca L & Cristo M 2003. Annual cycle of the benthic community of a coastal lagoon: Lagoa de Melides (Grândola, SW Portugal). *Revista de Biologia*, 21: 71-89.

Esteves Costa F (coord.) 1994. Carta Hidrogeológica de Portugal 1/200.000, Nota Explicativa das folhas 7 e 8 [Hydrogeologic Map of Portugal at Scale 1/200,000 and Explaining Note of Sheets 7 and 8]. Instituto Geológico e Mineiro, Departamento de Hidrogeologia, Lisboa, 87 pp.

Félix PM, Correia MJ, Chaínho P, Costa MJ, Henrique N. Cabral, HN, Domingos I, Costa JL, Cancela da Fonseca L 2013. Influence of streams discharges on the structure of fish communities of Portuguese land-locked coastal lagoons. Cah. Biol. Mar., 54: 427-435.

Monteiro JP, Chambel A, Martins J 2008. Conceptual and Numerical Flow Model of the Sines Aquifer System (Alentejo, South Portugal). International Groundwater Symposium, International Association of Hydraulic Engineering and Research (IAHR), Istanbul, Turkey, 18-20 June 2008. p 38 (abstract) and doc. elect. CD-Rom, 9 pp.

Oliveira JT (coordination) 1984. Carta Geológica de Portugal na Escala 1/200.000 e Notícia Explicativa da Folha 7 [Geologic Map of Portugal at Scale 1/200,000 and Explaining Note of Sheet 7], Serviços Geológicos de Portugal, Lisboa, 77 pp.

Wasy 2002. FEFLOW version 5.0. Finite Element Subsurface Flow and Transport Simulation System. Institute of Water Resources Planning and System Research Ltd, Berlin, Germany.

ASSESSING SIGNIFICANT DAMAGE TO SELECTED IRISH GROUNDWATER-DEPENDENT TERRESTRIAL ECOSYSTEM (GWDTE) TYPES AS PART OF GROUNDWATER BODY CLASSIFICATION UNDER THE EU WATER FRAMEWORK DIRECTIVE

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ABSTRACT

Groundwater-dependent terrestrial ecosystems (GWDTEs) are wetlands that receive significant inputs from groundwater. Irish examples of GWDTEs include alkaline fens, petrifying springs and turloughs. The EU Water Framework Directive requires the assessment of groundwater-related damage to GWDTEs as part of groundwater body classification. In Ireland, forty-eight groundwater bodies were identified as at risk of failing to meet the WFD objective of good status owing to potential groundwater-related damage to GWDTEs. More detailed status tests to confirm this damage were applied to only two of these groundwater bodies owing to a lack of ecohydrogeological data. Turloughs and fens are the focus of this short paper which presents outputs from a recent project which used up-to-date information and site visits to assess significant damage to GWDTE sites within the forty-eight 'at risk' GWBs. The GWDTE evaluation process adopted a tiered approach with sites categorised depending upon the quality and accuracy of available information. Four out of six turloughs studied in detail (Tier 1) showed moderate to strong evidence of nutrient enrichment from groundwater. Floodwater sampling of the fourteen turloughs assigned to Tiers 2 and 3 is recommended prior to more detailed ecohydrogeological investigations. Most fens were assigned to Tiers 2 or 3 and there is still a high degree of uncertainty with regard to the extent of groundwater-related damage at most fen sites. There is a still a lack of GWDTE ecohydrological monitoring in Ireland and GWDTEs cannot be fully integrated into the groundwater body classification process in the absence of consistent, long-term ecohydrological data.

INTRODUCTION

GWDTES AND THE EU WATER FRAMEWORK DIRECTIVE

The EU Water Framework Directive (WFD) (2000/60/EC) heralded an integrated, catchment-based approach to water management, with ecological objectives at its core. The overall WFD objective is to achieve 'good' status for all surface and coastal waters and groundwater (Kallis and Butler 2001). Groundwater bodies (GWBs), a distinct volume of groundwater within an aquifer, are the main WFD groundwater management unit (EC 2003). Groundwater body status is determined by a series of chemical and quantitative tests, one of which relates to groundwater-dependent terrestrial ecosystems (i.e. groundwater-dependent wetlands) (GWDTEs) (EC 2008). Whilst the WFD does not set specific objectives for wetlands, it recognises their significance within the hydrological network and they are afforded a level of protection via water bodies (EC 2003). This additional layer of protection is welcome given the high degree of wetland loss in many European countries over the past few decades (EC 2007). For a GWB to achieve good status, groundwater level alterations or pollutants must not result in any significant damage to GWDTEs.

GWDTE TYPOLOGY IN IRELAND

The WFD explicitly supports the objectives of the EU Habitats Directive (92/43/EEC) which established the Natura 2000 network of important ecological sites. To this end, the WFD required Member States to create a Register of Protected Areas to include all waters and wetlands designated for conservation under other EU legislation. Twenty-one terrestrial ecosystems on the Irish Register of Protected Areas were identified by the Irish National Parks and Wildlife Service as GWDTEs (i.e. directly dependent to some extent on groundwater) (EPA 2005). The most commonly occurring GWDTE types in Ireland are alkaline fens (Natura 2000 code 7230), species-rich *Cladium* fen (7210), petrifying springs (7220), transition mire (7140), active raised bog (7110), turloughs (3180), flushes in blanket bog (7130) and wet heath (4010), alluvial forests (91EO), machair (71AO) and humid dune slacks (2190).

GWDTES AND GROUNDWATER BODY CLASSIFICATION IN IRELAND

Each GWDTE site has a unique and complex relationship with the underlying GWB whereby making the process of assessing significant damage extremely challenging. Like many other Member States, Ireland is therefore struggling to incorporate GWDTEs into the GWB classification process. This process began during the first River Basin Cycle (2003-2009) of the WFD when GWBs were initially delineated based on aquifer flow regimes, geological boundaries and flow line boundaries (EPA 2005). In contrast to other EU countries, Ireland delineated GWBs specifically for GWDTEs considered to be at risk from pressures acting on groundwater. These GWBs are essentially zones of groundwater contributing to particular wetland sites. In 2004, GIS-based risk assessments (WFD Working Group on Groundwater 2004) were applied to 132 GWBs containing GWDTEs as part of the WFD Article V Characterisation and Risk Assessment of River Basin Districts. This process identified forty- eight GWBs at risk of failing to meet the WFD objective of good status owing to potential groundwater-related damage to GWDTEs. The GWDTE chemical status test was not applied to any GWDTEs at risk from nutrient pressures. The main challenge in this regard is the lack of groundwater nutrient threshold values which are used to identify sites requiring further more detailed investigation (Blum et al. 2009). Similarly, the quantitative status test was only applied to two of the twenty-three GWBs with GWDTEs at risk from quantitative pressure owing to a lack of site-specific ecohydrogeological information. This paper presents outputs from a recent project which used current best available information and preliminary site visits to identify GWDTE sites within the 48 ‘at risk’ GWBs where evident ecological damage is probably linked to groundwater-related quantitative and/or nutrient pressures. For the sake of brevity, this paper’s focus is placed on the most common Irish GWDTE types, namely turloughs and alkaline fens.

METHODS

GWDTES OCCURRING WITHIN AT RISK GROUNDWATER BODIES

Accurate, reliable spatial data were not available for all GWDTE types during the Article 5 WFD water body characterisation process. In most cases however, GWBs were delineated for a single, discrete GWDTE site where the GWB is named after the GWDTE site of interest. The majority of exceptions relate to GWBs delineated for fen habitats. Some GWBs delineated for GWDTEs do not appear to contain any fen habitat and others contain more than one fen site of potential interest. The most up-to-date spatial datasets were obtained from the NPWS and used to identify GWDTE sites that were the focus of the previous risk assessments.

OVERALL APPROACH TO ASSESSMENT OF SIGNIFICANT DAMAGE TO GWDTES

Given that the scope and accuracy of information available on individual GWDTEs and GWBs varies enormously, the GWDTE evaluation process adopted a tiered approach. Sites were categorised depending upon the quality and accuracy of available information identified during an initial desk-based appraisal. Three tiers were defined, representing low, medium and high confidence levels in the extent of ecological damage within the GWDTE and the linkages to GW-related quantitative and/or qualitative pressures.

Tier 1: Sites which have the highest quality information available, both in terms of GWB pressures and ecological condition, which allow the impact of GW pressures on the ecology of associated GWDTEs to be determined with a relatively high degree of confidence (e.g. Clara Bog).

Tier 2: Composed of GWDTEs which have partial but incomplete site-specific information on GWB pressures and ecological condition. The evaluation of ecological damage and its association with GW pressures was based on the highest quality information available for each site, together with surrogate information from comparable Tier 1 sites where appropriate.

Tier 3: Sites which have very little or no information on the extent of ecological damage and the linkages to GW-related quantitative and/or qualitative pressures. Any conclusions drawn on the level of ecological damage and GW linkages will be at a low confidence level.

Given the number and scale of the study areas and the proposed timescale of the project, site visits focused on Tier 3 classified GWDTEs. The objective of the site visits was to provide a basic assessment of GWDTE ecological condition.

ECOLOGICAL ASSESSMENT OF GWDTES

Turloughs

Turloughs are karstic depressions characterised by intermittent flooding from groundwater and the lack of a surface outflow. Twenty of the forty-eight ‘at risk’ GWBs are associated with turloughs, all of which were at risk from nutrient enrichment of groundwater rather than quantitative pressures. Phosphorus is the key limiting nutrient in the floodwaters of the majority of turloughs (Cunha Pereira et al. 2010). Sites were grouped into three tiers based on the nature of the available ecological information: 1) Turloughs studied in detail as part of the NPWS/TCD Turlough Project titled *Assessing the Conservation Status of Turloughs* based in Trinity College Dublin; 2) Turloughs with detailed but outdated vegetation information (Goodwillie 1992) and 3) Turloughs lacking any reliable ecological data. Table 1 presents a summary of data used to assess significant damage to the twenty turlough sites.

Table 1 Summary of available ecological information used for the significant damage assessments of turloughs.

Tier	No. of turloughs	Summary of available data
1	6	Mean seasonal floodwater Total Phosphorus (TP) (µg l^{-1}), maximum Chlorophyll *a* in floodwater, presence/absence of algal mats, presence/absence of positive and negative aquatic invertebrate indicators of nutrient enrichment.
2	7	A simple linear regression model, derived from the relationship between proportions of low nutrient vegetation communities (Goodwillie 1992) and mean floodwater TP across 18 turloughs, was used to predict mean seasonal floodwater TP.
3	7	Proportions of low scoring vegetation communities were estimated in the field as follows: ~ 10%, ~ 25%, ~50%. A simple linear regression model, derived from the relationship between proportions of low nutrient vegetation communities mapped as part of the project *Assessing the Conservation Status of Turloughs* and mean floodwater TP across 22 turloughs, was used to predict mean seasonal floodwater TP.

Alkaline fens

Alkaline fens are typically base-rich basin or flush fen systems with extensive areas of species-rich small sedge communities. Site visits used both positive and negative vegetation-based indicators to assess ecological damage linked to GW-related pressures. Positive indicators were extensive and limited areas of species-rich small sedge vegetation communities. These communities are the high conservation value aspect of alkaline fens and are indicative of low nutrient conditions. Negative indicators included habitat types indicative of nutrient enrichment (Fossitt 2000). These habitats included: Extensive, dense *Reed and large sedge swamps FS1* dominated by Common Reed (*Phragmites australis*) and/or Bulrush (*Typha latifolia*); *Wet grassland GS4* dominated by *Juncus* spp. and/or *Glyceria* spp. extensive, dense *Scrub WS1*; presence of *Raised bog PB1*.

ASSESSMENTS OF NUTRIENT AND QUANTITATIVE PRESSURES WITHIN AT RISK GROUNDWATER BODIES

Nutrient pressures

Diffuse nutrient pressures were assessed using the EPA/GSI 2010 Impact Potential base map. The risk from point sources was assessed by calculating the densities of Domestic Waste Water Treatment Systems (DWWTS) that potentially present a risk of phosphorus input to GW using EPA DWWTS Risk Assessment point data. The location of High and Very High Risk DWWTS relative to GWDTE sites was also inspected. The status of Urban Waste Water Treatment Plants (UWWTP) within GWBs was checked using the EPA Envision Viewer (http://gis.epa.ie/Envision). GW quality, drinking water quality and river chemical monitoring data were obtained from the EPA Groundwater Section.

Quantitative pressures

The hydrological assessment comprised a desk study of the key site attributes obtained from readily derivable information collated for each GWB, and where identified potential drainage and abstraction issues were described and documented. Data sources included: aerial photography, bedrock and aquifer geology, Teagasc/EPA subsoil mapping, OSI surface water channel layers, OPW Arterial Drainage Scheme Environmental Assessments, groundwater and surface water monitoring points within GWB, OPW Preliminary Flood Risk Assessment (PFRA) maps, OPW National Flood Hazard Mapping website (www.floodmaps.ie), and historical 6” and 25” maps.

RESULTS

Turloughs

Two of the six turloughs studied in detail as part of the project *Assessing the Conservation Status of Turloughs* showed strong evidence of groundwater-related significant damage. Both Caherglassaun and Tullynafrankagh had high maximum Chloropyll *a* concentrations (13.5 and 69.4 μg L^{-1} respectively) and abundant negative aquatic invertebrate indicators in the absence of positive nutrient indicators. Diffuse nutrient sources and High Risk and Very High Risk Septic tank scores in the local and wider GWB are probably driving nutrient enrichment at both sites. Potential septic tank inputs are located some distance from Caherglassaun, which has an extensive catchment area, whereas numerous septic tanks occur in close proximity to Tullynafrankagh and warrant further investigation. There is also moderately strong evidence of nutrient enrichment at Lough Coy and Skealoghan. Caherglassaun and Lough Coy GWBs contain a GW quality monitoring point. Poldeelin Spring (WFD Code 07_019) is situated 4km to the east of Caherglassaun turlough and Kilchreest borehole (WFD Code 07_011) is 10km to the north east of L. Coy. GW MRP levels within Lough Coy GWB have generally exceeded the natural background level (NBL) (OCM 2007) for MRP (0.020 mg L^{-1} P) since 1995 whereas concentrations within Caherglassaun GWB have remained below the NBL over recent years (Figure 1). Annual mean GW TP concentrations (2010-2012) range between 0.051 and 0.054 and 0.026 and 0.077 mg L^{-1} within Lough Coy and Caherglassaun GWBs respectively (Table 2). Recent research (Irvine et al. in prep.) recommends a threshold concentration of 0.020 mg L^{-1} of mean total phosphorus in groundwater for turloughs within the Gort chain and, on this basis, Lough Coy in particular warrants more detailed site investigation. It must be noted that both Lough Coy and Caherglassaun turloughs have extremely large GWBs and complex, time-variant interactions occur between GWDTE and GWB; one GW monitoring point located several kilometres from the GWDTE is insufficient to make conclusive statements about the quality of GW feeding these sites.

The majority of the remaining fourteen turloughs had a predicted good or intermediate water quality. Floodwater sampling is necessary to confirm these predictions. One GW quality monitoring point occurs within Rahasane GWB, approximately 20km north east of the turlough. High MRP levels were recorded between 2001 and 2007; however, concentrations are now similar to those recorded in the 1990s. Since 2010, annual mean TP concentrations have ranged between 0.033 and 0.040 mg/l P, exceeding the recently recommended TV for turloughs (0.020 mg L^{-1}) (Irvine et al. in prep.). A dense cluster of Very High Risk septic tanks occurs along the north-eastern fringe of Rahasane which may contribute nutrients to the site. One GW quality monitoring point (WFD Code 03_001) is located within Ballyvaughn GWB approximately one kilometre to the southwest of the site, which is more like an estavelle than a turlough. Temporal trends indicate that annual mean MRP concentrations are typically below the NBL for MRP (0.020 mg L^{-1}) (Figure 1). These data and low TP concentrations (0.01-0.019 mg L^{-1}) reflect the low risk to Ballyvaughn from diffuse sources of nutrients within the GWB.

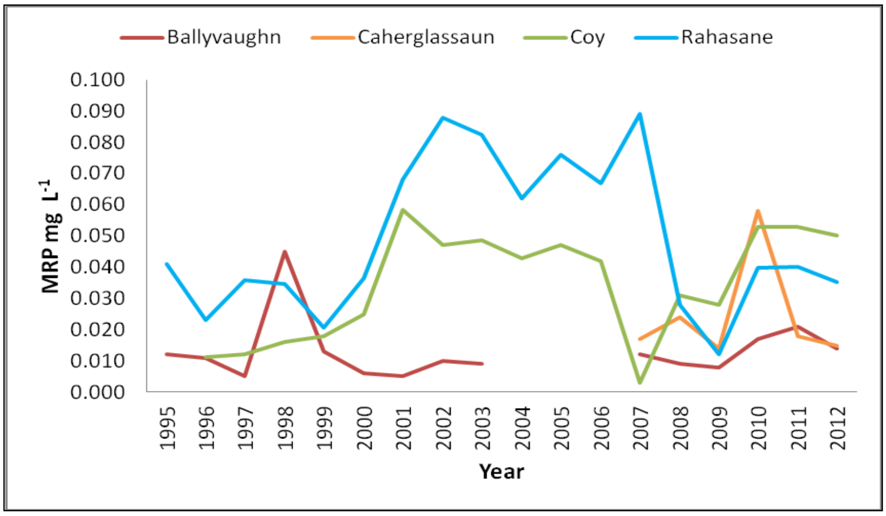

Figure 1 Temporal variation (1995-2012) in annual mean GW MRP concentrations (mg L^{-1}) at GW quality monitoring points (n=1) within Caherglassaun, Coy, Rahasane and Ballyvaughn GWBs.

Year	Caherglassaun	Coy	Rahasane	Ballyvaughn
2010	0.077	0.051	0.040	0.019
2011	0.028	0.047	0.033	0.010
2012	0.026	0.054	0.037	0.010

Table 2 Annual mean GW TP concentrations (mg L^{-1}) at GW quality monitoring points (n=1) within Caherglassaun, Coy, Rahasane and Ballyvaughn GWBs since 2010.

Alkaline fens

In addition to the detailed study undertaken at Pollardstown Fen, detailed ecological information was available for only one fen site, namely Hugginstown Fen. Good GW quality and extensive areas of species-rich small sedge vegetation communities indicate that this fen is in good ecological condition (NPWS 2009). Fens in the Inny GWBs are probably damaged as a result of the Inny arterial drainage scheme. However there is a high degree of uncertainty with regard to the presence and extent of conservation worthy fen habitat within these GWBs. Newtown Lough and Askeaton fens are at risk from both abstraction and diffuse pressures. Newtown Lough is documented as being partially degraded (NATURA 2005) and algal blooms are evident on recent aerial photographs. Inspection of the quantitative pressures within the GWB did not identify any significant regional drainage and abstraction pressures and nutrient enrichment is probably linked to overland flow and inflowing drains. Askeaton is an extensive GWB containing thirteen fen sites in the eastern area. Refinement of GWB for individual SACs is recommended to assess impacts of GW abstraction and nutrients on GWDTEs. Most sites are dominated by extensive stands of Common Reed and the lack of grazing obscures GW impacts. Bunbrosna fen in the Derravarragh_2 GWB is in poor ecological condition, however the lack of grazing and the adjacent plantation forestry are considered to be the main threats to this site. The current GWB encompasses the whole of Lough Owel and delineation of the zone of groundwater contributing specifically to Bunbrosna Fen is required. Fens in the northern portion of the Lough Corrib GWB are apparently more groundwater dependent than the more southerly floodplain fens. The more northerly fens do not show obvious signs of significant damage and the risk to these sites from diffuse nutrients and septic tanks is low. The Oranmore Fens in the Galway Bay Complex GWB are under significant threat from surrounding urban development and numerous septic tanks occur along the perimeter of these sites. Infilling, dumping and lack of grazing make it difficult to assess significant damage at these sites.

Table 3 Summary of assessments of groundwater-related significant damage to turloughs and fens within at risk groundwater bodies.

GWB Name	GWDTE Type	Risk	Tier	Evidence of GW-related significant damage
Caherglassaun	Turlough	Diffuse	1	Intermediate water quality, max Chl *a* > 10 µg l^{-1}, Aquatic Invertebrate Indicators: +ve (absent), -ve (present), elevated GW TP concentrations.
Coy	Turlough	Diffuse	1	Intermediate water quality, max Chl *a* > 10 µg l^{-1}, Aquatic Invertebrate Indicators: -ve (present), elevated GW TP concs, high impact potential from diffuse nutrient sources.
L. Fingal Complex	Turlough	Diffuse	1	None. Good water quality and ecological condition.
Roo West	Turlough	Diffuse	1	None. Good water quality and ecological condition.
Skealoghan	Turlough	Diffuse	1	Max Chl *a* > 10 µg l^{-1}, algal mats present
Tullynafrankagh	Turlough	Diffuse	1	Intermediate water quality, max Chl *a* > 10 µg l^{-1}, Aquatic Invertebrate Indicators: +ve (absent), -ve (present), high risk from local septic tanks.
The Loughans	Turlough	Diffuse	2	Predicted water quality: Intermediate
Glenamaddy	Turlough	Point	2	UWWTP discharges into site. Floodwater sampling is required to confirm predicted intermediate water quality.
Kiltiernan	Turlough	Diffuse	2	Floodwater sampling is required to confirm predicted intermediate water quality.
Rahasane	Turlough	Diffuse	2	Floodwater sampling is required to confirm predicted intermediate water quality.
Shrule	Turlough	Abstraction/Diffuse	2	Floodwater sampling is required to confirm predicted good/intermediate water quality. Quantitative risk minimal and no within-site evidence of impact from quantitative pressure.
Turloughmore (Sligo)	Turlough	Diffuse	2	Floodwater sampling is required to confirm predicted intermediate water quality.
L. Mannagh	Turlough	Diffuse	2	Floodwater sampling is required to confirm predicted intermediate water quality.
Dunmuckrum Turloughs	Turlough	Diffuse	3	Floodwater sampling is required to confirm predicted good/intermediate water quality.
Ballyvaughan	Turlough	Diffuse	3	Small karst feature, probably an estavelle, rather than turlough.
Ballyvelaghan	Turlough	Diffuse	3	Floodwater sampling is required to confirm the predicted intermediate water quality. Extensive, dense stands of *Typha latifolia* indicate a nutrient enrichment issue.
Gortboyheen	Turlough	Diffuse	3	Site not surveyed owing to access difficulties.
Muckinish	Turlough	Diffuse	3	Floodwater sampling is required to confirm the predicted good water quality. Site is not a typical turlough as it is brackish.
Ballinduff	Turlough	Diffuse	3	Floodwater sampling is required to confirm the predicted good water quality.
Cahermore	Turlough	Diffuse	3	Floodwater sampling is required to confirm the predicted intermediate water quality
Newtown Lough	Alkaline fen	Abstraction/Diffuse	2	Significant regional drainage and abstraction pressures not evident. Evident nutrient enrichment probably linked to overland flow and inflowing drains.
Hugginstown Fen	Alkaline fen	Diffuse	1	None. Good water quality and ecological condition.
Askeaton	Alkaline fen	Abstraction/Diffuse	3	Most fen sites dominated by Common Reed. Lack of grazing is likely to obscure GW impacts.
Inny_9	Alkaline fen	Abstraction	3	Poor example of Annex I habitat type. Probably damaged as a result of Inny arterial drainage scheme.
Inny_10	Alkaline fen	Abstraction	3	No negative nutrient indicators present.
Inny_11	Alkaline fen	Abstraction	3	Poor example of Annex I habitat type. Probably damaged as a result of Inny arterial drainage scheme.
Inny_12	Alkaline fen	Abstraction	3	Poor example of Annex I habitat type. Probably damaged as a result of Inny arterial drainage scheme.
Derravarragh_2	Alkaline fen	Abstraction	2	Site is in poor ecological condition however lack of grazing is likely to obscure GW impacts. GW abstraction points are unlikely to impact GW contribution to fen.
Fedamore_4	Alkaline fen	Abstraction	3	Site apparently in good ecological condition. GW abstraction may exert a negative impact on fen.
Lough Corrib Fen	Alkaline fen	Diffuse/Point	2	Floodplain fens to the south are dominated by dense reedbed. Septic tanks may exert a negative on these fens.
Galway Bay Complex	Alkaline fen	Point	2	Infilling and lack of grazing obscure GW impacts. Numerous low risk and medium risk septic tanks on perimeter of sites.

DISCUSSION AND CONCLUSIONS

The overall objective of this study was to assess significant damage to GWDTEs and ultimately prioritise GWDTE sites for more detailed field investigations where the ecology is considered to be significantly damaged due to pressures on the GWB. Of the sites studied as part of the project *Assessing the Conservation Status of Turloughs*, further investigations of Caherglassaun and Tullynafrankagh are an urgent priority. Lough Coy and Skealoghan also show signs of nutrient enrichment from groundwater and warrant further investigation. Floodwater sampling of the fourteen sites assigned to Tier 2 and 3 is recommended prior to more detailed ecohydrogeological investigations.

With regard to alkaline fen sites, baseline habitat surveys are recommended prior to further assessments of groundwater-related significant damage. Many fens sites do not appear to support species-rich sedge dominated swards and it is impossible to ascertain whether these sites previously support this high conservation value aspect of the alkaline fen habitat. Basic surveys would enable EPA and NPWS personnel to target ecological and groundwater monitoring efforts on sites with conservation worthy alkaline fen habitat that show signs of nutrient enrichment. The groundwater flow systems within many GWBs purported to contain alkaline fen habitat are complex and spatially heterogeneous. Thus accurate delineation of fen habitats within 'at risk' GWBs is also a prerequisite for the assessment of quantitative hydrological pressures. This approach would help to prevent further deterioration and loss of this rapidly declining habitat type.

In conclusion, the present project identified a number of GWDTE sites within the 'at risk' GWBs where there is a high confidence that evident ecological damage is linked to groundwater-related quantitative and/or nutrient pressures. The project also identified GWDTEs that do not show signs of groundwater related damage and GWDTE GWBs that need to be re-defined or excluded from the GWB classification process. This represents progress however significant challenges remain. Most turloughs and alkaline fens are associated with a karstic groundwater flow regime which presents major challenges for estimating the areas of groundwater contributing to such GWDTEs and for subsequent GW monitoring. This makes it all the more important that we improve the reliability and accuracy of basic hydroecological and spatial data for GWDTEs in order to effectively target resources at sites with evident GW-related ecological damage.

REFERENCES

Blum, A., Legrand, H., Grath, J., Scheidleder, A., Broers, H., Tomlin, C. & Ward, R. 2009. Threshold values and the role of monitoring in assessing chemical status compliance. *In:* Quevauviller, P., Fouillac, A.-M., Grath, J. & Ward, R. (eds.) *Groundwater monitoring.* Chichester: Wiley.

Cunha Pereira, H., Allott, N. And Coxon, C. 2010. Are seasonal lakes as productive and permanent lakes? A case study from Ireland. Canadian Journal of Fisheries and Aquatic Sciences **67:** 1-13.

EC 2003. The role of wetlands in the Water Framework Directive, CIS Guidance Document No. 18, European Commission, Brussels.

EC 2007. LIFE and Europe`s wetlands: *Restoring a vital ecosystem.* A report by the European Commission.

EC 2008. Groundwater protection in Europe. Brussels, European Commission.

EPA 2005. Article 5: The characterisation and analysis of Irelands River Basin Districts-Summary report on the characterisation and analysis of Irelands river basins. Environmental Protection Agency, Johnstown Castle Estate, Wexford, Ireland.

Fossitt, J. 2000. *A Guide to Habitats in Ireland*, The Heritage Council, Ireland.

Goodwillie, R. 1992. Turloughs over 10 hectares-Vegetation Survey and Evaulation. A report prepared for the National Parks and Wildlife Service.

Irvine, K. Allott, N., Coxon, C., Cunha Pereira, H., Kimberley, S. and Waldren, S. in prep. Water Framework Directive Risk Assessment for Turloughs. Chapter 11 of the final report *Assessing the Conservation Status of Turloughs* for the National Parks and Wildlife Service.

Kallis, G. and Butler, D. 2001. The EU water framework directive: measures and implications. *Water Policy* **3:** 125-142.

Natura 2005. Westmeath Peatlands Study. Final Report prepared for Westmeath County Council and The Heritage Council by NATURA Environmental Consultants.

NPWS 2009. Conservation Statement for Hugginstown Fen cSAC Site Code 404 Co. Kilkenny. A report prepared by the National Parks and Wildlife Service.

OCM 2007. Establishing Natural Background Levels for Groundwater in Ireland. A report prepared for O'Callaghan Moran and Associates for the Southeastern River Basin District Project Team.

WFD Working Group on Groundwater 2004. Methodology for risk characterisation of Ireland`s Groundwater. Guidance Document No. 8, undertaken by the Irish Working Group on Groundwater, 166 pp (report and detailed information available on http://www.wfdireland.ie/wfd-charreport.html).

ECOHYDROLOGICAL METHODS FOR THE INVESTIGATION OF SIGNIFICANT DAMAGE AT GROUNDWATER DEPENDENT TERRESTRIAL ECOSYSTEMS

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ABSTRACT

This paper aims to share the experiences of UK conservation and environment agencies of implementing Water Framework Directive investigations to determine significant damage at GWDTEs (Groundwater Dependent Terrestrial Ecosystems). Much of this work has been driven by various members of the 'Wetlands Task Team' of the Water Framework Directive UK Technical Advisory Group (WFD UKTAG). A three-tier approach to site investigation is described, providing suggestions for useful sources of information to inform both initial and detailed desk studies and subsequent site investigations. Both affordable and more expensive investigative methods are described; their usefulness in terms of contributing to a final site conceptual model are discussed. The need for long term monitoring to characterise baseline conditions and incorporate hydrological extremes (wet years and dry years) is demonstrated. A summary table provides a list of investigative methods scored on their cost, time required and contribution to conceptual understanding. The importance of both hydrogeologists and ecologists working together throughout the process is stressed and the 'ecohydrological site walkover' is proposed as a principal activity for any site investigation.

INTRODUCTION

GWDTEs (Groundwater Dependent Terrestrial Ecosystems) are wetlands which critically depend on groundwater flows and or chemistries (Schutten *et al* 2011). The Water Framework Directive (WFD) requires the identification of GWDTEs, risk screening and assessment of significant damage (Whiteman *et al* 2009). Significant damage can be caused by both quantitative and chemical groundwater pressures that result in unfavourable ecological condition under Habitats Directive (HD) assessments. The magnitude or significance of damage is related to the societal (conservation in UK) importance of the features of the wetland and the degree of change to these features resulting from the pressure (Schutten *et al* 2011).

If during the WFD assessment a groundwater body is classified at 'poor status' due to unfavourable condition at a GWDTE, related to a groundwater pressure, then an investigation is required to characterise the sources and pathways of groundwater mediated pressures. Whiteman *et al* (2009) argue that;

'this means shifting investigative focus somewhat away from, for example, single target water levels to a holistic consideration of water supply mechanisms and delivery or protection of wetland regimes including hydrological (groundwater level, groundwater flow, seepage, surface water flow), hydrochemical and site management'.

WFD investigations in England and Wales have been targeted at high risk sites defined using a tiered risk assessment procedure (e.g Whiteman *et al* 2010). The return of the GWDTE to favourable ecological condition must be realised if the associated groundwater body is to return to 'good status' and European WFD and HD targets achieved. Member States are required to achieve good groundwater body status by 2025. One of the significant challenges faced by practising hydrogeologists and ecologists in the UK in recent years has been assessment and investigation of groundwater impacts upon ecological receptors. Hydrogeologists are required to classify, characterise, risk screen and investigate groundwater mediated pressures that can cause significant damage (see Whiteman et al., 2010) to a range of designated GWDTE. Groundwater body status in England and Wales is assessed through five tests: a) overall resource balance: b) groundwater impacts upon dependent surface water bodies; c) risk of saline intrusion; **d) significant damage to groundwater-dependent terrestrial ecosystems (GWDTEs)** and e) overall groundwater body chemical quality.

This paper describes ecohydrogeological methods used to investigate and conceptualise wetlands that are considered to be at high risk of significant groundwater pressures.

Figure 7 Qualitative and quantitative pressures at GWDTE. Left: enriched vegetation at Dowrog Common (SSSI/SAC), Pembrokeshire Right: Drainage ditch at Cors Hirdre (SSI/SAC). Photograph with kind permission of Dr Peter Jones (NRW).

‘WETLANDS TASK TEAM’

The wetlands community within the UK is supported by the ‘Wetlands Task Team’, part of the Water Framework Directive UK Technical Advisory Group (WFD UKTAG). This team is a partnership of the UK environment and conservation agencies and invited technical specialists. The ‘Wetlands Task Team’ is one of the nine groups that support the UKTAG, providing technical advice and guidance on the implementation and application of the WFD. The WTT reports to the UK wide WFD policy group drawn from UK government administrations. The team comprises staff from Natural Resources Wales, Scottish Environment Protection Agency, Scottish Natural Heritage, Environment Agency, Natural England, Northern Ireland Environment Agency and the British Geological Survey. The free-to-download technical reports produced by the Wetlands Task Team provide a useful resource when considering how to apply the WFD to groundwater dependent wetlands (see UKTAG, 2004, UKTAG, 2012a and UKTAG, 2012b).

BEFORE YOU START

Two questions that need to be satisfied before starting any WFD investigation are:

- Is the site in unfavourable ecological condition (due to suspected groundwater pressures) as reported for the Habitats Directive (Habitats Directive)?

- Have the sites been assessed for significant damage, and scored as ‘at risk’ as part of the EU WFD Classification (Kimberly *et al* 2014 for Republic of Ireland and Farr & Wilson, 2013 for Northern Ireland).

If you can answer ‘yes’ to both of these questions then a WFD investigation is justified and you can move on to the tiered approach to site investigation.

A TIERED APPROACH TO SITE INVESTIGATION

In England and Wales a three tiered approach (Table 1) has been devised and is based upon on pilot work for the WFD by Buss & Hulme (2007) and more recently Brooks *et al* (2008). Investigations are most likely to be undertaken at sites that are in unfavourable ecological condition under the Habitats Directive and sites that have been assessed as being at risk of significant damage under the Water Framework Directive classification. The proposed tiered approach offers a simple step by step guide from initial characterisation (Tier 1), further characterisation (Tier 2) and evaluation and classification (Tier 3). This paper will concentrate on the more practice and field based ‘Tier 3’ that includes site visits and data collection.

	Tier 1	**Tier 2**	**Tier 3**
	Initial characterisation :basic desk study	**Further characterisation** :detailed desk study	**Evaluation and Classification** :site investigation and data collection
Where to look?	Site managers Local experts Published papers Grey literature Other reports Published maps	Ecological maps (NVC mapping) Geological maps Borehole archive Hydrometric archive Water chemistry archive Soil maps Modelling reports Ecology reports Air Pollution Inventory System	Ecohydrological site walk over On-site investigations: -subsurface investigation -Water levels -Water quality -Ecological observations
What to look for?	Local Expert knowledge Anecdotal evidence Known pressures Ecology (NVC mapping) Topography Geology & Soils Hydrogeology / Hydrology	Ecological data and NVC mapping Bedrock & Superficial geology Groundwater levels Groundwater quality Soil types and cover Surface water flow Atmospheric loading data	Surface water inflows Surface water outflows Groundwater discharge points: springs/flushes/seepages Flow rates of all water features Water level controls (.e.g weirs/ditches) Head difference between water features Near-surface geology Distribution of species / communities Enriched vegetation

Table 1 Three tiered approach to risk assessment of significant damage at GWDTE (modified from Brooks *et al* 2008).

TIER 1 AND TIER 2 (DESK STUDY)

Although the desk studies (Tier 1 & 2) are not the focus of this paper it is worth drawing the attention of the reader towards useful information within both the published and grey literature from the UK. During the last decade the effect of groundwater abstractions has been addressed by investigations undertaken as part of the EU Habitats Directive review of consents and other local impact assessment studies (e.g. Whiteman *et al* 2004; Whiteman *et al* 2009; Gellatly *et al* 2012). Further development of eco-hydrological guidelines for specific wetland habitats include: wet woodlands (Barsoum *et al* 2005), humid dune slacks (Davy *et al* 2010) and wet heaths (Mountford *et al* 2005) and more general lowland wetlands plant communities (Wheeler *et al* 2004). The Fen Management Handbook produced by Scottish Natural Heritage (SNH, 2011) is a wealth of practical information and. ‘WETMECS’ or wetlands water supply mechanisms are described in detailed in Wheeler *et al* (2009).

TIER 3 (SITE INVESTIGATION AND DATA COLELCTION)

ECOHYDROECOLOGICAL SITE WALKOVER

The 'ecohydrological site walkover' is perhaps the most important stage of any investigation. In simple terms it should involve a hydrogeologist and an ecologist, preferably both of whom should have good local knowledge. The site walkover gives both parties the chance to discuss and share information and to consider how hydrological processes may support and also transmit a range of pressures to ecological receptors. Using the **source-pathway-receptor** concept that is familiar to hydrogeologists, both potential sources and pathways for groundwater mediated pressures should be discussed. During the site walkover the ecologist should identify **receptors** (e.g. vegetation) that are deemed to be in unfavourable condition. In response the hydrogeologist should be looking at the landscape and considering what the **sources** and **pathways** of the pressures could be. An example of the source-pathway-receptor concept is provided: diffuse agricultural pollution (source) is transmitted by groundwater (pathway) leading to nutrient enrichment and unfavourable condition of a designated site (receptor). Information collected during this process should include: notes on the vegetation condition, basic readings of water levels and flow, water chemistry and identification of landuse pressures within the immediate and wider catchment. Even at this initial stage both the hydrogeologist and ecologist are starting to gather information that can be combined to inform an ecohydrological conceptual model. It is important for these thoughts and observations to be recorded. The site walkover will highlight the knowledge gaps that can be answered with further on site investigation.

Figure 8 Ecohydrological site walkovers should include an experienced hydrogeologists and ecologist (only one of each required). This photograph shows a wetlands hydrology training course lead by Dr Rob Low at Cors Bodeilio (SSSI/SAC/NNR), Anglesey.

GEOLOGY AND THE NEAR SUBSURFACE

It is likely that characterisation of the subsurface may form part of any site investigation, providing geological information to inform an initial site conceptual model. In England and Wales a range of methods have been used that range in time, complexity and cost. At the most affordable end is the hand auger (Figure 3). Operated manually it can retrieve small samples of unconsolidated material from 0-3 meters below the surface, however its operational depth can be limited by resistant materials such as clays, cobbles and bedrock. Obtaining detailed cores for stratigraphical logging can be achieved using small portable percussion drills, such as the Dando (Figure 3) that is capable of reaching a depth of almost 20 meters and can cope with stiff clays, larger cobbles and less competent bedrock. Larger percussion and rotary rigs can reach greater depths however their use is limited by their size and ability to gain access to the desired areas within the designated site. Before drilling it would be advisable to address the following questions: is the site or are the designated features too sensitive for a drilling operation? Is the surface competent enough to support and allow safe access for the drill rig?

Not all subsurface investigations need to be intrusive and geophysical methods provide further options for site conceptualisation; a useful example comes from Wybunbury Moss, Cheshire (Environment Agency, 2011 and Brooks *et al* 2011). Large-scale application of geophysics is likely to be more costly than shallow drilling and hand auguring, and it is able to provide information on a wider and more laterally continuous section across a given site. Geophysics can also help inform where best to site intrusive investigations, and in an ideal world one may wish to undertake geophysical surveys prior to any drilling operation, however the cost of this approach is prohibitive. More recently aerial geophysical datasets have been used to infer hydrological and ground conditions over larger areas containing several wetlands (e.g Beamish & Farr, 2013) and for the landscape scale mapping of peat bodies (Beamish, 2014).

Figure 9 Left: Dr Mark Whitman (EA) uses a hand auger to collect information on the near surface deposits at Cors Geirch. Right: Stephen Thorpe (BGS) uses a Dando percussion drill to obtain an 18m core from Tregaron Bog.

WATER LEVEL AND FLOW

For many wetland sites characterisation of both surface and groundwater levels may be required to improve a site conceptual model. Several monitoring options have been used in England and Wales and these vary in terms of cost, time and complexity. Simply installing a dipwell or borehole is not going to answer your questions, it must be designed, located and the data interpreted for it to yield useful information. The siting of any water level monitoring should be carefully considered, groundwater monitoring should where possible be associated with known NVC communities. Several boreholes may be required if you want to infer groundwater flow directions, or 'nested piezos' if you are interested in the vertical movement of groundwater.

The most affordable method of groundwater level monitoring is the ‘dipwell’ (Figure 4). Constructed of 1m length, 50mm diameter PVC pipes that can be joined together, these wells can be installed using basic manual tools to a depth of 2-3m depending on the nature of the subsurface. Geotextile membrane of varying pore size can be fitted to the slotted section to allow for monitoring in a range of environments from peat to fine sands. Shallow dipwells may need to be anchored to stop movement; this can be achieved by using a metal earth anchor attached to a more stable underlying area such as a basal clay or bedrock. Experiences from Dowrog Common (Pembrokeshire) showed that unanchored piezometers and wells experienced movement of several cm’s over the period of one year, lifting concrete headworks from the ground. The solution was to anchor the dipwells to the basal clay unit using a thin metal rod attached to the dipwell. Where surface water features such as ditches require monitoring then the same dipwell casing can be used to construct stilling wells. If deeper monitoring is required then portable drills or larger percussion rigs (Figure 4) can be used to drill boreholes, although access problems, especially where the terrain is very soft, wet or inaccessible are likely to be limiting.

Figure 10 Left: affordable dipwells can be installed by hand to a depth of 2-3meters in unconsolidated materials (Rhoswen Leonard and Janine Guest). Middle: slightly more expensive power hand drill can install to a greater depth and through more competent materials. Right: a large percussion drill can install a borehole or piezometer into more competent material (photograph Dr Peter Jones).

Once the monitoring wells are in place they should be surveyed to a datum, preferably meters above Ordnance Datum (maOD). Groundwater levels can be recorded using a manual dip meter or by installation of a digital pressure transducer. Manual dipping is initially a lower cost option however once time is taken into account then it can become very expensive and data records are likely to show only general trends, with any changes in water level between visits not recorded. Pressure transducers are manufactured by a range of companies and their small size often allows insertion into dipwells <30mm in diameter, some transducers now come with a 2m range allowing a better resolution of data from small scale water level changes often associated with wetlands. The transducers can be set to record both pressure and water temperature at various intervals, and experience from England and Wales suggests an hourly frequency provides a suitable baseline data record for most wetlands. During a pump test at Greywell Fen, Hampshire (Low *et al* 2013) a 15-minute interval was chosen to record sub hour changes in response to changes in the pumping regime. Pressure transducers can also be used for monitoring surface water features such as drains, ditches, lakes and turloughs (e.g. Farr *et al* 2012) .Hydrometric water level data is often plotted as a time series against rainfall events however cumulative frequency curves which illustrate the period of time a water level spends at a set level are also a useful output.

Measurements of surface water flow can be achieved by the installation of weirs or by direct gauging. Water level data from stilling wells can be converted using stage-discharge relationships. Ephemeral springs and diffuse seepage areas are much more difficult to measure. Spring flow can be measured using a simple 'bucket and stop watch' method which provide individual spot readings (Figure 5). Installation of temperature or electrical conductivity meters in springs or areas of diffuse flow can provide records of when discharge occurs or when the sites are dry, however conversion to volumetric flow is not simple. Researchers in the U.S have successfully used electrical resistivity sensors to measure diffuse and ephemeral flow in a range of settings (see Bhanjee & Lindsay, 2011 and references therein) and these techniques could be applied to characterise diffuse or topogenous flow into and within GWDTEs in the UK and the Republic of Ireland.

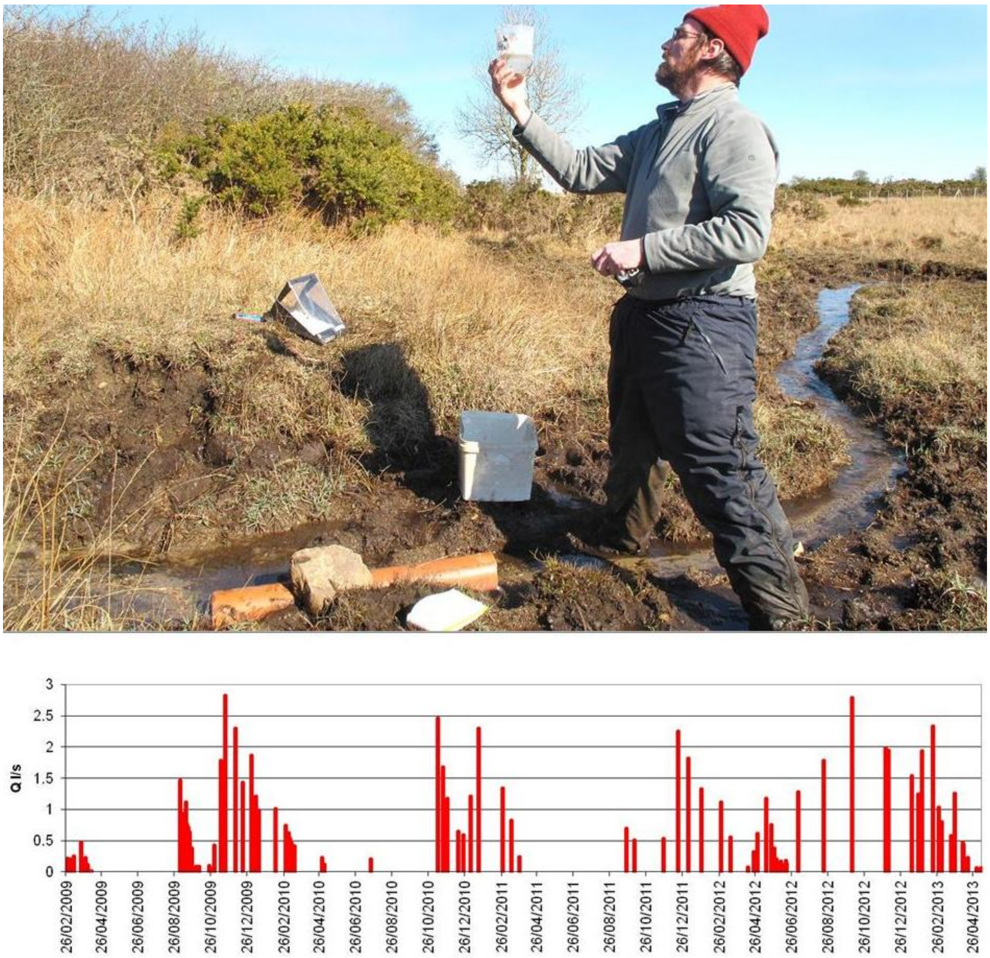

Figure 11 'Bucket and stopwatch' method for gauging flow of an ephemeral spring (data with kind permission of Dr Peter Jones, Natural Resources Wales)

WATER QUALITY

Collecting representative samples is not always straight forward, especially as many wetlands experience diffuse groundwater discharge at their margins and widespread shallow rheo-topogenous flow within the sites. Discrete springs, boreholes or dipwells may provide easier sampling locations. Before simple methods for collection of representative samples are discussed we must consider what determinands should be analysed. Discrepancies between analyses from one wetland to another were identified in early WFD investigations in England and Wales. Analysis may vary in terms of the total number of determinands and also between their limits of detection (LOD). It was decided that a standardised analysis suite should be proposed allowing conservation bodies and environmental regulators to collect comparable water quality information, the standard analysis suite is presented in

Table 2. The proposed suite is not definitive and can be changed and tailored to specific investigations. It covers major ions used to type groundwater (e.g. Ca, Mg, Na, K, Cl, SO_4 and $HC0_3$), field parameters (temperature, electrical conductivity, dissolved oxygen and Eh) and nutrients (nitrate, phosphate etc).

Useful equipment for obtaining representative samples from wells includes: 12v specialist submersible pumps, and the very light and more affordable suction ‘caravan pumps’ which can lift water from just a few meters and are easy to transport across large sites (Figure 6). A stainless steel jug is indispensible for lowering into surface water features and for collecting shallow topogenous flow (Figure 6). Field parameters such as temperature, pH and dissolved oxygen should always be recorded in situ whenever possible and 50mm diameter dipwells allow most multiparameter sondes to be lowered into them. If this is not possible flow through cells should be used or the probes submerged in flowing water on the surface of the site.

	ALKALINITY PH 4.5 - $CaCO_3$	AMMONIA - N	BICARBONATE - HCO_3	CALCIUM - Ca	CHLORIDE ION - Cl	CONDUCTIVITY @ 25C	HARDNESS TOTAL - $CaCO_3$	IONIC BALANCE (ANIONS/CATIONS)	IRON - Fe	MAGNESIUM - Mg	MANGANESE - Mn	NITRATE - N	NITRITE - N	NITROGEN TOTAL OXIDISED - N	ORTHOPHOSPHATE - P	OXYGEN DISSOLVED-SITU	OXYGEN DISSOLVED - SITU	pH IN SITU	Phosphate	POTASSIUM - K	SODIUM - Na	SULPHATE - SO_4	TEMPERATURE	Redox Potential in Situ	Iron Dissolved	Manganese Dissolved
Units	mg/l	mg/l	mg/l	mg/l	mg/l	uS/cm	mg/l	%	ug/l	mg/l	ug/l	mg/l	mg/l	mg/l	mg/l	mg/l	%	pH	mg/l	mg/l	mg/l	mg/l	CEL	Mv	ug/l	ug/l
Limit of detection	5	0	-	1	1	-	-	-	30	0.3	10	-	0.004	0.2	0.02	-	-	-	0.02	0.1	2	10	-	-	-	-

Table 2 Standardised water quality analysis suite for groundwater dependent ecosystems

Figure 12 Left: an affordable 12v ‘caravan pump’ is lowered into a dipwell to obtain a sample. Middle: Les Colley and Adam Daniel use a stainless steel jug to collect water from a ditch Right: Jon Hudson collects a water quality sample from a diffuse seepage zone.

A range of novel groundwater quality techniques have been used to define the sources of nutrients and the travel time (or age) of groundwater reaching springs and seepages at various wetlands (Table 3). These methods complement the more traditional water quality analysis improving the site conceptual model.

Nitrogen and oxygen stable isotopes can help to determine the source of nitrogen dissolved in groundwater. The method works by comparing the ratios of the respective isotopes, ^{15}N to that of air □□^{15} N ‰) and ^{18}O relative to Vienna Standard Mean Ocean Water (□^{18} O ‰). The analysis can help to ‘fingerprint’ various sources of nitrogen including, soil organic matter, inorganic fertilizers and atmospheric deposition.

Chlorofluorocarbon (CFC) and sulphur hexafluoride (SF_6) can be used as tracers to date water up to 50 years old, to infer groundwater mixing and provide indicators of likely groundwater flow mechanisms (Gooddy *et al* 2006). Two results are presented in Table 3 alongside the nitrogen and oxygen stable isotope analysis. The results are used to infer a year of recharge for each sample, showing that in both cases there is a mixing of younger and older water.

Monitoring point	$\delta^{15}NNO3$ (‰)	$\delta^{18}ONO3$ (‰)	Nitrogen Source	CFC-12 pmol/l	CFC-11 pmol/l	SF_6 fmol/l	CFC-12 pmol/l	CFC-11 pmol/l	SF_6 fmol/l	Year of recharge (range of values)
SPRING 1	8.1	4.8	Nitrification of soil organic nitrogen	**2.9**	**4.4**	**2.2**	**0.98**	**0.84**	**0.81**	**1984-2002**
SPRING 2	7.6	3.8	Nitrification of soil organic nitrogen	**3.4**	**5.1**	**2.1**	**1.17**	**0.99**	**0.8**	**1987 - Modern**

Table 3 Examples of novel groundwater quality analysis to inform site conceptualisation at GWDTEs

HYDROLOGICAL EXTREMES AND THE NEED FOR LONG TERM MONITORING

It is important that the classification process for the WFD is based on long term data sets. Individual water quality samples or water level readings are useful as part of one-off investigations and can contribute to baseline datasets, but they do not allow us to identify seasonal or long term variations and trends (Farr *et al* 2014 in press). A complete annual cycle is suggested as the minimum duration of recording for water levels and quality, but even this has obvious risks in that some years are very wet and others much drier. It is difficult to define what is meant by ‘long term’ monitoring. A minimum of 5 years may be required to characterise hydrological extremes e.g. wet years and dry years. To detect longer term changes related to the changing climate even longer records (>20 years), such as those collected from Ainsdale sands (Clarke, D and Sanitwong Na Ayuttaya, 2010) may be required.

CONCLUSIONS

WFD investigations in England and Wales have produced a wealth of information that can be readily used by other regulatory and conservation bodies. Examples from England and Wales include Wybunbury Moss (Environment Agency, 2011) Cors Bodeilio and Merthyr Mawr (SWS, 2010a) and Cors Erddreiniog (SWS, 2010b). We have attempted to score the investigative methods by considering their cost (e.g. equipment, plant hire etc), duration (e.g. labour costs and ongoing monitoring) and ultimately their contribution to a better conceptual understanding (Table 4). The table is provided as a guide only and we recognise that the order may change depending upon the type of pressure and GWDTE, however in all cases the ecohydrological site walkover should remain at, or close to the top of the table.

		Cost			Duration			Conceptual Understanding		
		<£1K	£1-5K	>£5K	1 Month	1 year	> 1 year	Low	Medium	High
Ecohydrological site walkover		■			■					■
Local expert knowledge		■			■					■
Catchment audit		■			■				■	
Ecological surveys	Short term	■			■				■	
(NVC and quadrats)	Long term			■			■			■
Groundwater quality	Short term	■	■		■	■			■	
	Long term			■			■			■
	In situ		■			■			■	
Surface water quality	Short term	■	■		■	■			■	
	Long term		■				■			■
Shallow intrusive investigation (auger)		■			■				■	
Drilling	Shallow		■		■					■
	Deep			■	■					■
Groundwater levels	Short Term	■			■				■	
	Long Term			■			■			■
Surface water hydrology	Short Term		■		■				■	
	Long Term			■			■			■
	Nitrogen/ Oxygen isotopes		■		■				■	
Novel geochemical analysis	Age dating (e.g CFC/SF_6)		■		■				■	
Atmospheric loading (www.apis.ac.uk)		■			■				■	
Historic aerial photography		■			■				■	
Remote sensing				■		■			■	
Geophysical surveys			■		■				■	
Airborne geophysical survey				■		■		■		
Flow/nutrient modelling			■			■			■	
Groundwater model				■			■			■
Multi-disciplinary review		■			■					■

Table 4 Investigative methods for GWDTE in England and Wales, rated in terms of cost, time and ultimately their contribution to an ecohydrological conceptual model. (n.b it has not been possible to discuss all of the methods listed in this table within this paper).

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REFERENCES

Barsoum, N., Anderson, R., Broadmeadow, S., Bishop, H, & Nisbet, T.2005. Ecohydrological guidelines for wet woodland – Phase I. English Nature Research Reports Number 619. http://publications.naturalengland.org.uk/publication/91054

Beamish, D. 2014.Peat Mapping Associations of Airborne Radiometric Survey Data. *Remote Sensing*. 2014; 6(1):521-539. http://www.mdpi.com/2072-4292/6/1/521

Beamish, D, & Farr, G., 2013. Airborne Geophysics: a novel approach to assist hydrogeological investigations at groundwater dependent wetlands. *Quarterly Journal of Engineering Geology and Hydrogeology*. V46: p54-62. http://nora.nerc.ac.uk/500309/

Bhamjee, R & Lindsay, J.B. 2011. Ephemeral stream sensor design using loggers. Hydrol. Earth System Sci., 15, 1009-1021. http://www.hydrol-earth-syst-sci-discuss.net/7/6381/2010/hessd-7-6381-2010.pdf

Brooks, A., Whiteman, M., Skinner, A., Farr, G., Papaioannou, A, Coffey., A, Bellis, L, Jones, P & Low, R. 2011. Investigation of diffuse groundwater chemical impacts on groundwater dependent terrestrial ecosystems in England and Wales: implications for WFD groundwater body classification and programmes of measures. HydroEco, 2011, May 2-5th 2011. http://web.natur.cuni.cz/hydroeco2011/download/presentations/oral/SWFD/6_240_Brooks_HydroEco_2011_final_240.pdf

Buss, U & Hulme, P.2007. Testing UKTAG guidance for the identification of groundwater dependent terrestrial ecosystems in the Ribble Basin. Environment Agency of England & Wales report SCHO0407BMNG-E-P. http://cdn.environment-agency.gov.uk/scho0407bmng-e-e.pdf

Brooks, A., Cohen, A., Evers, S., Hulme, P & Phillips, N.2008. Methodology for the assessment of significant damage at wetlands. Environment Agency Science Report SC030285. https://publications.environment-agency.gov.uk/skeleton/publications/ViewPublication.aspx?id=ef289b1e-1e14-4e73-84f4-8ac1581d2258

Clarke, D and Sanitwong Na Ayuttaya, S (2010). *Predicted effects of climate change, vegetation and tree cover on dune slack habitats at Ainsdale on the Sefton Coast.* UK.Journal of Coastal Conservation, 14, (2), 115-125.

Davey, A.J., Hiscock, K.M., Jones, M.L.L., Low, R., Robins, N.S & Stratford, C. 2010. Ecohydrological guidelines for wet dune habitats. Environment Agency Science Report GEH00310BSGV. http://cdn.environment-agency.gov.uk/geho0310bsgv-e-e.pdf

Environment Agency, 2011. Refining river basin planning through targeted investigations on GWDTE, Wybunbury Moss.

Farr, G & Wilson, P. 2013. EU Water Framework Directive: 2nd cycle characterisation of groundwater dependent terrestrial ecosystems in Northern Ireland. British Geological Survey Commissioned Report for NIEA. CR/14/046.

Farr, G., Hatton-Ellis, T., Jones, D.A., Lambourne, C., Bevan, J & Naughton, O. 2012. Hydrology, water quality and condition of Pant-y-Llyn, Wales' only turlough. Countryside Council for Wales (CCW) Staff Science Report No. 12/8/1.

Farr, G., Whiteman, M., Jones, P. S & Breen, R. 2014. Wetlands and the Water Framework Directive: Key challenges for achieving good ecological status at the Angelsey and Llŷn Fens SACs. In Restoring Alkaline and Calcareous Fens within the Anglesey and Llyn Fens SACs in Wales: Report of the Project Conference 9-11 October 2013. Eds: P.S. Jones, J. Hanson & J. Guest. [in press]

Gellatly, S. C., Raynor, M. E., Green, C. J. E. & Neve, P. S. 2012. Using and refining an existing regional groundwater model to assess the impacts of an abstraction on a nearby wetland. In: Shepley, M. G., Whiteman, M. I., Hulme, P. J. & Grout, M. W. (eds) Groundwater Resources Modelling: A Case Study from the UK. Geological Society, London, Special Publications, 364, 333–351

Gooddy, D.C., Darling, G.W, Abesser, C & Lapworth, D.J. 2006. Using Chloroflurocarbons (CFCs) and Sulpher Hexafluoride (SF_6) to characterise groundwater movement and residence time in a lowland chalk catchment. *Journal of Hydrology*, 330 (1-2). 44-52 http://nora.nerc.ac.uk/4481/

Kimberly, S., Naughton, O., Regan, S. 2014. Assessing significant damage to selected Irish groundwater-dependent terrestrial ecosystem (GWDTE) types as part of groundwater body classification under the EU Water Framework Directive. Proceedings of the 34th Annual Groundwater Conference, 2014.

Low, R., Morris, S., Hutchings, A., Harfoot, H & Goodard, E. 2013. Ecohydrological impact assessment, Greywell Fen, Hampshire. Poster presented at the Groundwater Dependent Ecosystems meeting of the Geological Society of London, Birmingham and Midlands Institute, 27th February, 2013.

Mountford, J.O., Rose, R.J. & Bromley, J. 2005. Development of eco-hydrological guidelines for wet heaths – Phase 1. English Nature Research Reports Number 620. http://publications.naturalengland.org.uk/publication/107007

Schutten, J., Verweij, W., Hall, A., & Scheidleder, A. 2011. Technical Report on Groundwater Dependent Terrestrial Ecosystems (GWDTEs). Technical report to European Working Group C – Common Implementation Strategy Working Group on Groundwater.

SNH (Scottish Natural Heritage). 2010. *The Fen Management Handbook*, (2010), Editors A. McBride, I. Diack, N Droy, B. Hamill, P.Jones, J. Schutten, A. Skinner, and M. Street. Scottish Natural Heritage, Perth. http://www.snh.gov.uk/about-scotlands-nature/habitats-and-ecosystems/lochs-rivers-and-wetlands/fen/

SWS (Schlumburger Water Services). 2010a. Report on targeted risk-based investigations on selected Welsh Groundwater Dependant terrestrial Ecosystems (Cors Bodeilio and Merthyr Mawr). For Environment Agency.

SWS (Schlumburger Water Services). 2010b. Desk Study Cors Erddreiniog Investigations on selected Welsh Groundwater Dependent Terrestrial Ecosystems (GWDTEs). November 2010. 1-274/R3 FINAL. For Environment Agency.

Wheeler, B.D, Gowning, D.J.G, Shaw, S.C, Mountford, J.O and Money, R.P. 2004. Ecohydrological guidelines for lowland wetlands plant communities. Eds Brooks A.W, Jose, P.V and Whiteman M.I. Environment Agency Report. http://www.floodplainmeadows.org.uk/files/floodplain/d96435.pdf

Wheeler, B.D., Shaw, S & Tanner, K. 2009. A wetland framework for impact assessment at statutory sites in England and Wales. Intergrated catchment science programme. Science report: SC030232. http://www.floodplainmeadows.org.uk/files/floodplain/d96435.pdf

Whiteman, M.I., José, P., Grout, M.W., Brooks, A., Quinn, S.A., Acreman, M. 2004. Local impact assessment of wetlands – from hydrological impact to ecological effects. Paper presented at the 2nd

British Hydrological Society International Conference, Imperial College, London, 12-16 July 2004. In: *Hydrology: Science & Practice for the 21st* Century, Proceedings of 2nd British Hydrological Society International Conference, Imperial College London 12-16 July 2004, Volume 2, 198-212.

Whiteman, M.I., Wheeler, B.D., Shaw, S.C., Lewis, T., Grout, M., Tanner, K. 2009. Use of WETMECS typology to aid understanding of groundwater-dependent terrestrial ecosystems in England and Wales. In: Quevauviller, P., Fouillac, A.M., Grath, J., Ward, R. (Eds.), Groundwater Quality Assessment and Monitoring. John Wiley & Sons, pp. 259–272.

Whiteman, M., Brooks, A., Skinner, A. Hulme, P. 2010. Determining significant damage to groundwater dependant terrestrial ecosystems in England and Wales for use in implementation of the Water Framework Directive. Ecological Engineering, 36, p1118-1125.

UKTAG. 2004. United Kingdom Technical Advisory Group on the Water Framework Directive. Guidance on the identification and risk assessment of Groundwater dependent terrestrial ecosystems. Version 5 (21.01.04)
http://www.wfduk.org/sites/default/files/Media/Characterisation%20of%20the%20water%20environment/Risk%20assessment%20of%20terrestrial%20ecosystems%20groundwater_Draft_210104.pdf

UKTAG. 2012a.United Kingdom Technical Advisory Group on the Water Framework Directive. Technical report on groundwater dependent terrestrial ecosystem (GWDTE) threshold values. (Version 8) 23.03.2012.
http://www.wfduk.org/sites/default/files/Media/Environmental%20standards/GWDTE%20chemical%20values_Final_230312.pdf

UKTAG. 2012b. Paper 11b(i) Groundwater Chemical Classification for the purpose of the Water Framework Directive and the Groundwater Directive. (Version 0.8) Feb 2012.
http://www.wfduk.org/sites/default/files/Media/Assessing%20the%20status%20of%20the%20water%20environment/GWChemical%20Classification_FINAL_2802121.pdf

SESSION IV

GUIDANCE ON THE AUTHORISATION OF DIRECT DISCHARGES TO GROUNDWATER

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ABSTRACT

Direct discharges are, with the exception of a few possible exemptions, prohibited under the Groundwater Regulations, yet they are known to occur in Ireland, and where they occur, they are potential contributors to water quality issues.

The recently finalised EPA guidance on the authorisation of direct discharges to groundwater presents an overview of the Groundwater Regulations with respect to these direct discharges; gives examples of different types of direct discharges and provides technical rules when consideration is being given to the authorisation of direct discharges of domestic-type wastewater effluents.

The guidance is intended to contribute towards groundwater and environmental protection initiatives by the EPA and other public authorities and should be a useful reference document for public bodies and private entities alike.

INTRODUCTION

The EPA has recently published draft guidance on the Authorisation of Direct Discharges to Groundwater[1] for consultation. This guidance supplements the 2011 EPA guidance on the Authorisation of Discharges to Groundwater, which focused on the technical assessment of indirect discharges to groundwater.

Direct discharges are, with the exception of a few possible exemptions, prohibited under the European Communities (EC) Environmental Objectives (Groundwater) Regulations, herein referred to as the Groundwater Regulations (S.I. No. 9 of 2010). Nonetheless, direct discharges are known to occur in Ireland, and where they occur, they are potential contributors to water quality issues.

In the Irish context, direct discharges are of particular concern in vulnerable hydrogeological settings, as they significantly increase the risk to groundwater quality and associated receptors in these areas. Areas of extreme vulnerability, in particular those areas that are underlain by karstified limestone aquifers, represent a particularly vulnerable hydrogeological setting in which groundwater can transport pollutants over large (kilometre-scale) distances in hours or days, often with little pollutant attenuation.

Primarily the guidance document presents an overview of the Groundwater Regulations with respect to direct discharges to groundwater; gives examples of different types of direct discharges and provides technical rules when consideration is being given to the authorisation of direct discharges of domestic-type wastewater effluents. The guidance is intended for use by EPA personnel, but may also

[1] http://www.epa.ie/pubs/consultation/proposedguidanceontheauthorisationofdirectdischargestogroundwater.html

be helpful to other public bodies, local authorities and environmental professionals involved in the preparation or review of applications for discharges to groundwater.

WHAT DO THE GROUNDWATER REGULATIONS REQUIRE?

The Groundwater Regulations define environmental objectives that must be achieved to protect groundwater resources and associated receptors from pollution. The relevant local authority is responsible for authorizing and regulating effluent and trade discharges to waters under the Water Pollution Acts, and the EPA is responsible for regulating specified activities that may have significant polluting potential, encompassing Industrial Emissions Directive licensing, Integrated Pollution Control Licensing (previously Integrated Pollution Prevention and Control Licensing), Waste Licensing and Waste Water Discharge Authorisation.

Under Regulation 4 (a) of the Groundwater Regulations, public authorities should take all reasonable steps to *"prevent or limit, as appropriate, the input of pollutants into groundwater and prevent the deterioration of the status of all bodies of groundwater"*

The 'prevent or limit' objective is the key WFD objective when considering the authorisation of discharges to groundwater as it acts as the first line of defence in restricting inputs of pollutants to groundwater and thereby avoiding or reducing pollution.

- The 'prevent' objective relates to *hazardous* substances, whereby all necessary and reasonable measures should be taken to avoid the entry of such substances into groundwater and to avoid any significant increase in concentration in groundwater, even at a local scale.
- The 'limit' objective relates to *non-hazardous* substances, whereby all necessary measures should be taken to control inputs into groundwater to ensure that such inputs do not cause pollution, deterioration in status of groundwater bodies, or significant and sustained upward trends in groundwater concentrations.

Under Regulation 8 of the Groundwater Regulations, direct discharges to groundwater are prohibited. However, certain types of direct discharges *may be permissible* subject to a requirement of prior authorisation.

Under Regulation 14 of the Groundwater Regulations, certain direct and indirect discharges to groundwater may be granted exemptions from measures to prevent or limit the input of pollutants into groundwater, where conditions (technical rules) have been established by the EPA.

In practice, Regulation 14 is broader in scope than Regulation 8 and represents special cases where the need to take all measures necessary to achieve the 'prevent or limit' objectives may be relaxed, but not ignored, under case-specific circumstances.

Technical rules for direct discharges of domestic-type waste water effluent are described in the guidance but technical rules for other types of discharges may be developed in time.

WHAT CONSTITUTES A DIRECT DISCHARGE TO GROUNDWATER?

As defined by the Groundwater Regulations, a direct discharge is a "*discharge of pollutants into groundwater without percolation throughout the soil or subsoil*". As a consequence direct discharges can therefore also occur where the input bypasses the unsaturated zone via natural or artificial open conduits.

Typical examples of direct discharges include:

- Discharges to karst features such as swallow holes and dolines, both directly and indirectly via surface water courses (e.g. streams that sink underground at swallow holes);
- Discharges directly onto bedrock surfaces where subsoils are absent and/or thin;

- Discharges into cesspits and boreholes/wells (e.g. injection wells, mineral exploration wells) that extend down to or below the groundwater table;
- Discharges into mine adits that extend down to or below the groundwater table.

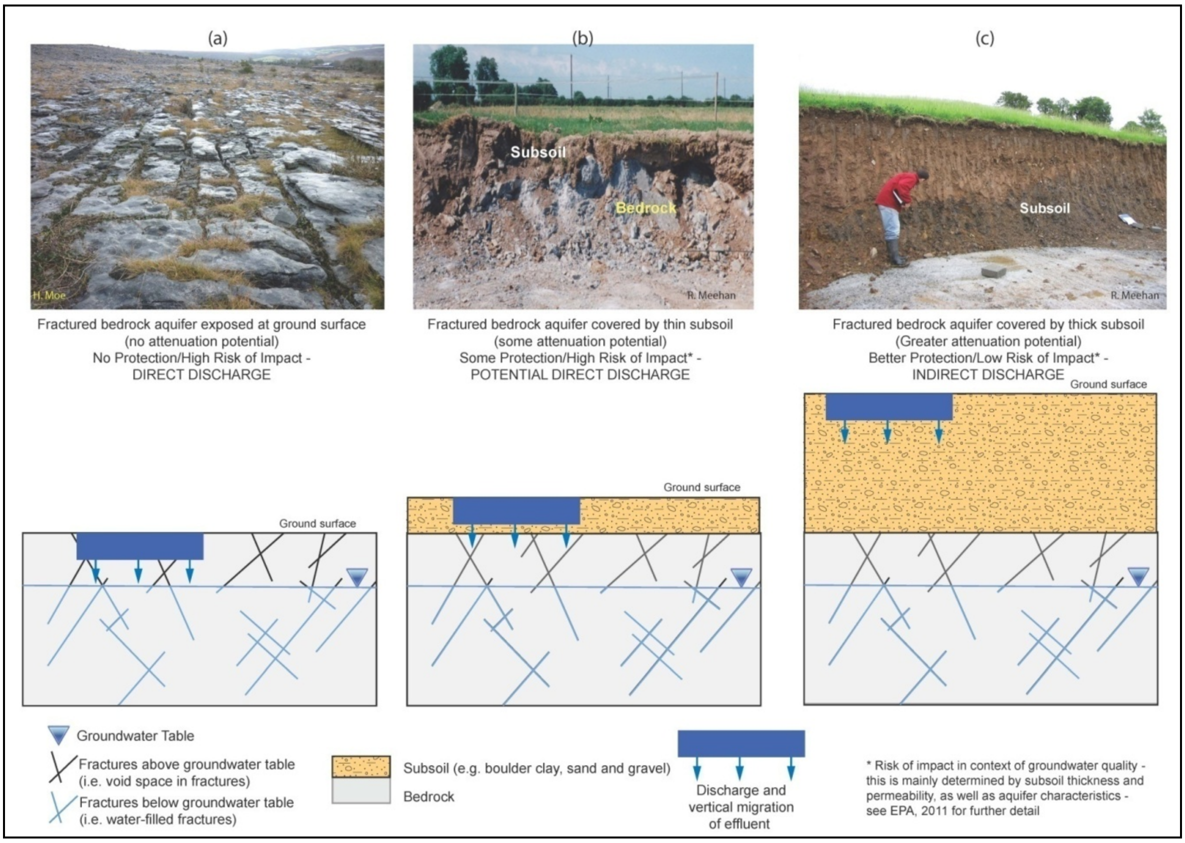

Figure 1 - Presence or absence of soil/subsoil as a protective layer

The presence or absence of soil/subsoil in the unsaturated zone is a key factor when considering the risk of impact to groundwater quality from a discharge. As shown in Figure 1 (a), situations exist where soils/subsoils are mainly absent and karstified or fractured bedrock surfaces are exposed at ground surface. In such settings, bedrock aquifers are extremely vulnerable to pollution despite the presence of an unsaturated zone between ground surface and the groundwater table in the rock beneath. This is because the vertical migration of pollutants to the groundwater table takes place through open conduits or fractures in the unsaturated zone within the rock, without attenuation. There is, accordingly, a high risk of impact to groundwater quality, and discharge activities directly onto bedrock surfaces should be avoided.

In contrast, where subsoils are present in the unsaturated zone, the bedrock aquifer is offered natural protection by the subsoil cover, whereby pollutants are subjected to attenuation processes during percolation through the unsaturated subsoil. The degree of protection offered by the subsoil is a function of its thickness and permeability. Where subsoils are thick (see Figure 1 (c)), the risk of impact to groundwater quality is reduced.

Swallow holes and dolines are surface expressions of underground solution features (i.e. conduit systems). They are particularly common in counties Clare, Galway, Mayo, Roscommon and Cork, but can also be present elsewhere. These preferentially move water and pollutants at high flow rates (hundreds of metres per day) to discharge points such as streams, natural springs or abstraction wells, see Figure 2. As a rule of thumb, a discharge to a doline should be approached as a direct discharge case, as the degree of attenuation is likely to be inadequate/insignificant due to a relatively high hydraulic loading in a small area.

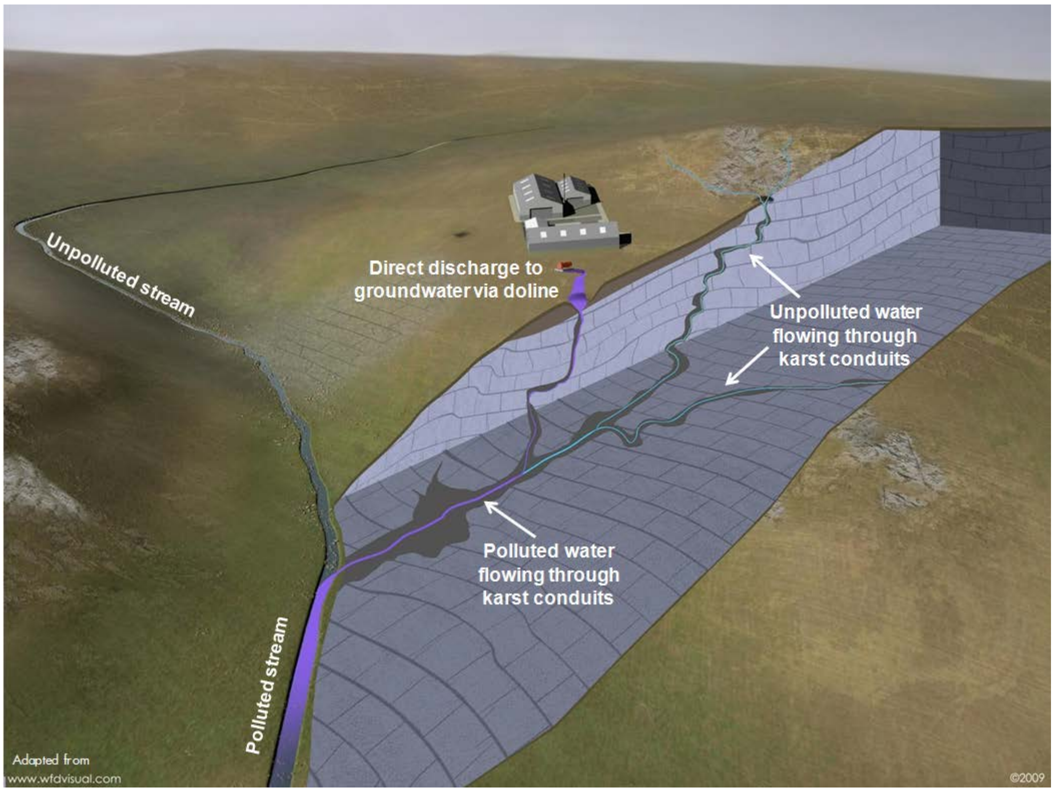

Figure 2 - Flow through karst conduits (Image adapted from WFD Visual)

In karstified aquifers, where a river may flow above ground only to sink below ground and then re-appear at the surface further downstream, the distinction between groundwater and surface water can sometimes be difficult to determine. In the Groundwater Regulations, a river is defined as "*a body of inland water flowing for the most part on the surface of the land but which may flow underground for part of its course*". The implication of this definition is that sinking streams are considered to be rivers in the sections that flow underground. However, the nature of water movement in karstified aquifers is complex and is not just confined to movement via conduits, so the question of "which regulations apply" also becomes more complex. Importantly, recent tracer testing in the Burren, County Clare (Drew, 2012) demonstrated that dyes that were injected into an active swallow hole migrated to down-gradient springs rapidly via conduit flow, but also to down-gradient private wells via slower and more "diffuse" pathways, represented by fractures, in the same karstified aquifer.

The inference of these results is that the consideration of sinking streams at swallow holes solely as "river water" does not always accurately depict the true nature of water movement in karstified aquifers. Therefore due to the potential that a sinking stream can flow diffusely away from conduit systems underground, a stream that sinks should be regarded as groundwater unless defensible evidence exists that diffuse dispersion (through fracture systems) is not important for a given site or setting.

The complexity of water movement that can occur in karstified aquifers, and which may influence the transport of pollutants, is exemplified further in Figure 3, which shows a direct discharge to a swallow hole under two different hydrogeological conditions:

- Following a period of prolonged dry weather when the karst conduit system has been draining and water levels are low; and
- Following a prolonged or significant wet weather event when rainfall and natural recharge has replenished the karst conduit system and water levels are high.

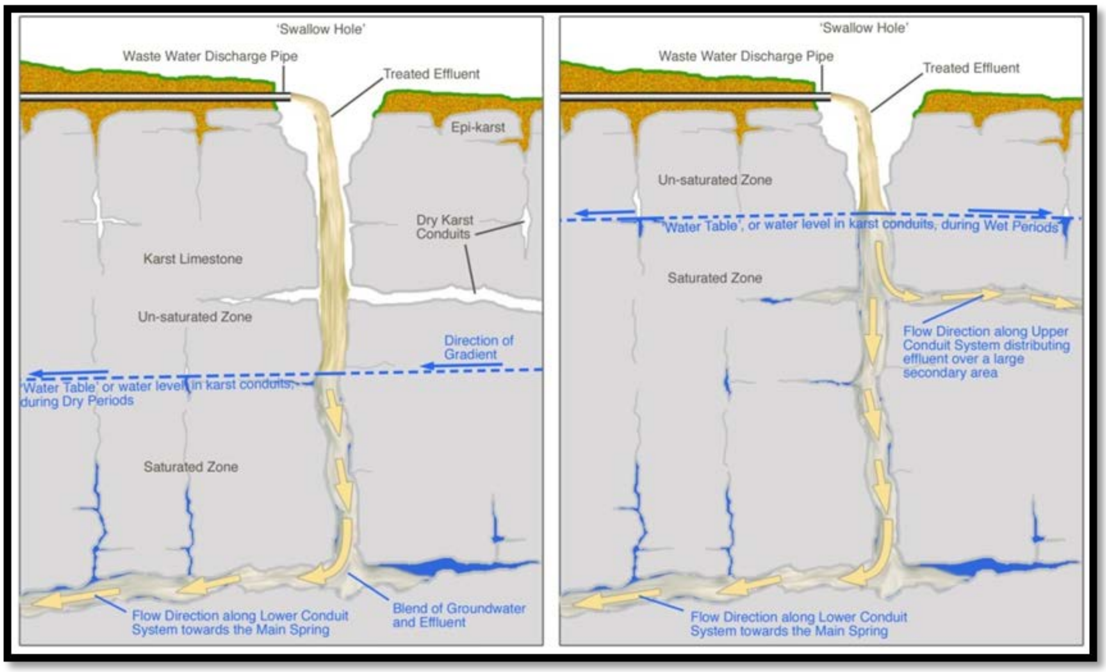

Figure 3 - Influence of water levels and hydraulic gradients on flow through karst conduits (Image source: David Ball)[2]

Figure 3 shows that the discharge effluent may be transported via the upper conduit system to a different outlet (discharge) location than the effluent transported via the lower conduit system. There are several examples of such hydraulic behaviour from tracer studies in western parts of the country. In the Burren case described by Drew (2012), tracer movement in different directions from the same swallow hole can be attributed to the subsurface presence of alternative pathways which become active/inactive according to prevailing hydrological conditions. It also describes how tracer materials (and effluent) can be dispersed into areas that are away from the main flow axis between a swallow hole and down-gradient springs.

Similarly a discharge that enters a swallow hole via a stream, surface drain and/or open-water area has direct and rapid access to groundwater with no attenuation occurring except dilution and mixing. The important question in this instance is whether the associated surface water feature has sufficient assimilative capacity under relevant seasonal conditions (WTSG, 2011) to reduce pollutant concentrations to acceptable concentrations in groundwater (and associated receptors) before entering the swallow hole. For this reason, and as a rule of thumb, a discharge to a surface water drain or open water area that flows to a swallow hole should also be approached as a potential direct discharge to groundwater scenario.

Likewise, where discharges occur directly onto bedrock surfaces, pollutants can move vertically through the unsaturated zone via open fractures to groundwater without any attenuation. Examples include retention ponds without appropriate liner materials that receive urban runoff, poorly designed, constructed and/or maintained domestic wastewater percolation systems and percolation/infiltration areas in open mine areas and unauthorised waste sites.

[2] In the dry weather scenario, the waste water effluent flows down the swallow hole and into a lower (deeper) conduit system that transports the diluted effluent (after mixing with groundwater) towards an outlet point, typically represented by a spring. In the wet weather scenario, water levels are higher and the diluted effluent (after mixing with groundwater) can now flow along both the lower and the upper karst conduit systems.

TECHNICAL RULES WHEN CONSIDERING AUTHORISATION OF DIRECT DISCHARGES OF DOMESTIC-TYPE WASTEWATER EFFLUENT

Technical rules for direct discharge activities are conditions that are defined by the EPA, or another public authority, which, if followed, may exempt the activity from the requirements to undertake measures to prevent or limit the input of pollutants into groundwater.

The overall approach adopted for consideration of direct discharges of domestic-type waste water effluents under Regulation 14 generally is outlined in Figure 4. In short, direct discharges can only be considered if, in order of priority:

a) Discharge options to surface water are precluded;
b) Indirect discharges to groundwater (via percolation) are precluded.

The onus is on the applicant to make the case that a surface water discharge or indirect discharge to groundwater is not technically feasible or is disproportionately expensive and therefore that a direct discharge to groundwater requires consideration.

A direct discharge of domestic-type waste water effluent should only be considered in areas where the physical landscape is the underlying reason why a surface water discharge or indirect discharge to groundwater is not an option. Consequently if a surface water discharge or indirect discharge to groundwater is feasible then the artificial creation of a direct discharge, e.g. by drilling injection wells or removing sub-soil to improve percolation, will not be considered an acceptable solution.

DISCHARGES TO SURFACE WATER

Discharges to surface water should always be considered as a first option in the process. An evaluation as to whether a discharge to surface water is feasible should be undertaken in accordance with the technical rules established in the "*Guidance, Procedures and Training on the Licensing of Discharges to Surface Waters*" (WSTG, 2011).

However, it is acknowledged that surface water options may not always be technically feasible or may involve disproportionate cost. Examples would be where:

a) Technical assessment of the discharge to surface water option demonstrates that the assimilative capacity of the target stream is insufficient to meet relevant water quality standards;
b) Where alternative piping of the effluent to the next largest stream in the same area is considered to be prohibitively costly; and/or
c) Improved treatment of the effluent will either not resolve the assimilative capacity issue or still involve prohibitive costs.

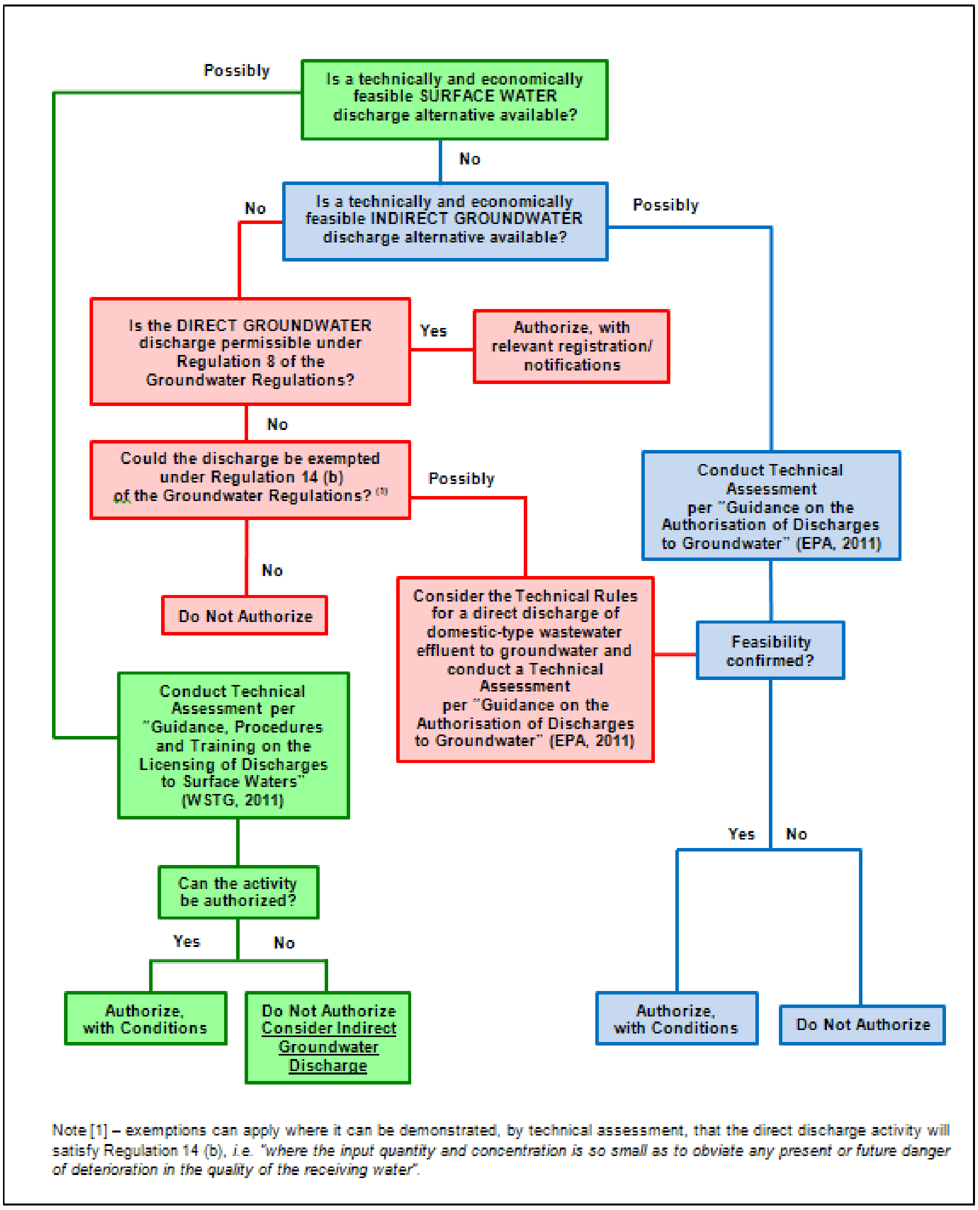

Figure 4 - Assessment process when considering authorisation of direct discharges of domestic-type wastewater effluent

DISCHARGES TO GROUNDWATER VIA PERCOLATION

If discharge to surface water is not an option on the grounds of technical feasibility or disproportionate cost, then discharge to groundwater via percolation (i.e. indirect discharge to groundwater) is the next preferred option. This option should undergo a technical assessment in the manner described in the "*Guidance on the Authorization of Discharges to Groundwater*" (EPA, 2011). This technical assessment should demonstrate whether an indirect discharge is technically feasible or disproportionately expensive, which will allow the EPA or appropriate public authority to make a decision as to whether the discharge can be authorised.

DIRECT DISCHARGES TO GROUNDWATER

Regulation 14(b) allows for exemptions of direct discharges where inputs are "*considered to be of a quantity and concentration so small as to obviate any present or future danger of deterioration in the quality of the receiving groundwater*". In practice, for domestic-type waste water effluents, this relates primarily to effluents that have received a high level of treatment and/or types of effluents where hazardous substances are generally or typically not present.

If a direct discharge to groundwater of domestic-type waste water effluents cannot meet the requirements stipulated in these technical rules then the activity should not be authorized.

Technical Rules for Direct Discharges

Listed below are minimum criteria that must be met/demonstrated by the applicant before the EPA or appropriate public authority would be in a position to review and render a verdict on potential exemption cases for direct discharge of domestic-type wastewater effluents:

- A Tier 3 technical assessment equivalent to those described in EPA, 2011 has to be carried out with inputs from relevant and suitably qualified professionals.
- The treated effluent quality has to meet a high standard to be determined by full technical assessment that is both site and case-specific. Achieving a high standard of treatment means that tertiary treatment, via sand or other filter systems, and possibly also sterilization techniques must be included in the treatment process prior to discharge. Treatment designs and achievable quality standards must be clearly identified and presented.
- Chemical analyses and routine reporting of effluent and groundwater quality must include total Nitrogen, ammonia (NH_3), molybdate reactive phosphorus (MRP), and pathogens (coliforms, *E. coli*), and other parameters that may be considered appropriate on a site by site basis. In the case of trade effluent, other parameters may be needed depending on the nature and composition of the effluent.
- In terms of treatment of nutrients and potential impacts on groundwater quality, ammonia has to be nitrified to the point that after dilution in groundwater, relevant receptor-based water quality standards are met (EPA, 2011). For example, where a receptor includes a down-gradient spring or well used for public or private water supply, the drinking water standard at the spring/well location must be met.
- Individual discharges to karstified limestone aquifers should not be of a magnitude that would cause the status of associated receptors to be failed. In addition, the technical assessment must examine cumulative impacts on groundwater bodies using the methodology outlined in EPA (2011) for groundwater assimilative capacity, so that discharge activity does not result in deterioration of groundwater body status.
- Results of the full technical assessment must be prepared in a hydrogeological report and submitted to the EPA or appropriate public authority for review.

The above conditions imply that a direct discharge activity must include strict source controls and rigorous technical assessment before authorisation can be granted. Source controls involve good engineering design and subsequent good operations and maintenance practice. Rigorous technical assessment involves a detailed hydrogeological study following Tier 3 assessment guidance as outlined in the EPA, 2011 guidance. Decisions shall be made on a case-by-case basis.

A direct discharge activity that falls into this category requires a comprehensive Tier 3 technical assessment (EPA, 2011) involving a karst study, which typically requires extensive hydrometric monitoring over ranges of hydrogeological conditions and the use of dye tracing techniques to demonstrate source-pathway-receptor linkages (EPA, 2011).

A well developed and acceptable scope for technical assessment is one that typically:

a) Identifies all relevant, potential receptors ‘downstream’ of the discharge site;
b) Establishes the infiltration or injection capacity of the direct discharge point under flood conditions;
c) Addresses a sound hydrogeological model of source-pathway-receptor linkages;
d) Quantifies effluent volumes (mean and maximum);
e) Describes groundwater and surface water conditions in sufficient detail to define “representative” discharges and flows under dry and wet weather/season conditions;

f) Accounts for water balances of natural groundwater discharge points (springs) with respect to estimated groundwater recharge and zones of contributions (to discharge points);
g) Is capable of defining expected mixing/dilution of effluents in groundwater under dry weather/season conditions; and demonstrating that the resulting dilution will meet the groundwater quality objectives defined in the Groundwater Regulations; and
h) Adequately monitors appropriate and relevant receptors, including private and public water supplies, for impacts on groundwater quality.

Adequate monitoring programmes must be put in place for both the source and potential receptors. The nature and scope of monitoring will be determined from the Tier 3 technical assessment.

In addition, for swallow holes there should be no significant impacts to local populations, economic activity (e.g. farming), and sensitive ecosystems (e.g. wetlands) along associated surface pathways and/or floodplains. In particular there should be no adverse direct or indirect exposure risk to pollutants in terms of human health and that of associated aquatic ecosystems. Potential impacts, would have to be examined and assessed under both low-flow and flood-flow conditions, and would be subject to existing discharge to surface water guidance (WSTG, 2011).

SUMMARY

Direct discharges to groundwater are, with few exceptions, prohibited under the Groundwater Regulations. They are prohibited because they significantly increase the risks of impact to groundwater quality and because they, typically, contradict the ‘prevent or limit’ objective which forms the core environmental quality objective as set forth by the Groundwater Regulations.

In the Irish context, direct discharges are of particular concern due to the often vulnerable hydrogeological settings, particularly in the karstified limestone aquifers in counties such as Galway, Clare, Mayo and Roscommon. Nonetheless, direct discharges are known to occur, and where they occur, they are potential contributors to water quality issues.

In practice, and in certain physical settings, direct discharges may be unavoidable if there are no ‘suitable’ alternatives, whereby ‘suitable’ is judged on a case by case assessment of technical feasibility and disproportionate costs. Direct discharges to groundwater should only be considered if, in order of priority:

a) Discharge options to surface water are precluded;
b) Indirect discharges to groundwater (via percolation) are precluded.

Where consideration of direct discharges becomes necessary, detailed technical assessment will be needed and must involve relevant and suitably qualified professionals. The technical assessment must be based on a carefully developed scope of work, an appropriate field investigation programme (including monitoring), and judicious analysis of results. Judging ‘suitability’, therefore, is a process which defines the details of the technical assessment that is required and results in a technically-based verdict on whether or not a prior authorisation can be granted or should be denied.

This guidance details the technical rules for direct discharges of domestic-type wastewater effluent to groundwater. They are intended to contribute towards groundwater and environmental protection initiatives by the EPA and other public authorities and to be a useful reference document for public bodies and private entities alike. As such, it supplements and should be read in context of the previously published report by the EPA entitled “*Guidance on the Authorisation of Discharges to Groundwater*” (EPA, 2011).

REFERENCES

Drew, D. (2012). Water Trace from Ballybreen Swallow hole (Kilfenora). Report to Clare County Council, Prepared by David Drew, April 2012.

EPA (2011). Guidance on the Authorisation of Discharges to Groundwater. Version 1, December 2011.

European Communities Environmental Objectives (Groundwater) Regulations 2010 (S.I. No. 9 of 2010).

WSTG (2011). Water Services National Training Group: Guidance Manual: Guidance, Procedures and Training on the Licensing of Discharges to Surface Waters and to Sewer for Local Authorities. Department of the Environment, Heritage and Local Government.

GROUNDWATER FLOODING IN IRISH KARST GROUNDWATER FLOW SYSTEMS

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ABSTRACT

In recent years, driven by the EU Floods Directive, there has been a new focus on the potential risk posed by flooding to human health, the environment and economic activity. During the first implementation phase of the Floods Directive, the Preliminary Flood Risk Assessment (PFRA), groundwater flooding was identified as a significant component of the risk from flooding in the Republic of Ireland. This paper discusses groundwater flooding in the context of Irish geological and groundwater conditions. The combination of predominantly karst groundwater flow systems and relatively high rainfall prevalent in the west of Ireland leaves this region particularly susceptible to groundwater flooding. The main characteristics of groundwater flooding in Ireland are described and the unique difficulties faced in evaluating likely groundwater flood occurrence, both present and future, are highlighted.

GROUNDWATER FLOODING AND THE EU FLOODS DIRECTIVE

Floods are natural phenomena which cannot be completely prevented as they are primarily caused by excessive rainfall; they have the potential to cause fatalities, damage property and infrastructure, and compromise economic development (EU Directive 2007/60/EC). The extensive flooding in November 2009 is one such example, which caused widespread damage throughout much of Munster, Connaught and the midlands and had an estimated insurance cost of €244 million (IIF, 2009). There is a pressing need to improve our understanding of the factors and mechanisms which cause flooding, in all its forms, in order to ensure efficient flood prevention and mitigation in future. This is addressed within the aims of the recent EU Directive on the assessment and management of flood risks (2007/60/EC), known as the “Floods Directive”.

The Floods Directive requires all Member States including Ireland to reduce and manage the risks that all forms of flooding (e.g. fluvial (river flooding), pluvial (direct rainfall), coastal and groundwater) pose through the mapping of probabilistic flood extents and the establishment of flood risk management plans. The Floods Directive was transposed into Irish Law in 2010, with the Office of Public Works (OPW) being the authority responsible for its implementation through the Catchment Flood Risk Assessment and Management (CFRAM) Studies programme. The three main requirements of the Directive are:

- To undertake a Preliminary Flood Risk Assessment (PFRA) by December 2011
- The development of flood hazard and risk maps by December 2013
- The production of Flood Risk Management Plans by December 2015

The first phase of the Directive implementation, the Preliminary Flood Risk Assessment (PFRA), was essentially a screening process which used available and readily derivable information to identify areas where the risks associated with flooding were potentially significant. The evidence-based approach taken in the groundwater PFRA found that groundwater flooding posed a significant flood hazard in Ireland (Mott Mc Donald, 2010).

The second implementation phase requires the development of flood hazard and flood risk maps for areas identified during the PFRA as at significant risk of flooding, known as "Areas for Further Assessment" or AFAs. Flood hazard maps show geographical areas which could suffer flooding under a range of low, medium and high probability scenarios. Flood risk maps combine the probability of a flood event with the adverse consequences of the event (i.e. potential damage) using standardised vulnerability classifications. The complexity associated with mapping groundwater flooding is acknowledged within the Floods Directive, which states that for areas where flooding is from groundwater sources the preparation of flood hazard maps can be limited to floods with a low probability (i.e. extreme events). One of the unique challenges in flood risk management for Ireland identified during the PFRA, and subsequent flood mapping, is the assessment of risks posed by groundwater flooding.

GROUNDWATER FLOODING IN IRELAND

Groundwater flooding is defined as flooding caused by the emergence of water originating from sub-surface permeable strata induced by exceptional and/or prolonged recharge (Morris et al., 2007). Groundwater flooding events in Ireland are centred on the limestone areas of the western lowlands, which extend from the River Fergus in Co. Clare in the south upwards to the areas east of Lough Mask and Corrib in Co. Galway and southern Co. Mayo (figure 1). The prevalence of groundwater flooding in the western counties is fundamentally linked to bedrock geology.

Limestone bedrock underlies almost half the land surface of the Republic of Ireland, with over 90% located in lowland areas of less than 150 mAOD (Drew, 2008; Simms, 2004). Limestones are susceptible to dissolution by water, in a process known as karstification. The passage of water through the limestone dissolves and enlarges flow pathways, creating a complex system of water-carrying fractures and conduits beneath the ground surface. This process has occurred in limestone formations throughout Ireland, but it is in the areas of pure, well-bedded Carboniferous limestone formations found predominantly in the west of the country that karst drainage dominates regional hydrology. As a result of the karstification process, Irish karst aquifers are characterised by a combination of low storage within the rock matrix and high transmissivity due to dissolution pathways. Moreover, due to the low-lying nature of most karst areas, the depth to groundwater is relatively shallow and interactions between ground and surface waters are widespread and complex (Coxon and Drew, 1998). The combination of karst groundwater systems, low elevation and relatively high rainfall prevalent in the west of Ireland leaves this area particularly susceptible to groundwater flooding.

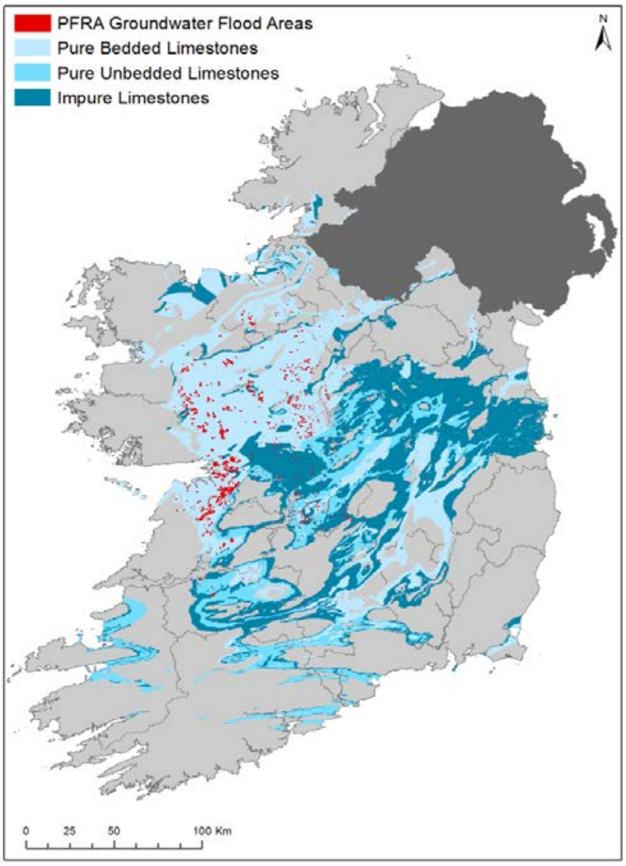

Figure 1: Distribution of limestone geology and PFRA groundwater flood hazard areas

GROUNDWATER FLOODING: SPATIAL DISTRIBUTION

While groundwater flooding can occur in a discontinuous manner across a wide geographical area (figure 2), the principal form of recurrent groundwater flooding is the seasonal inundation that occurs within turloughs (Mott McDonald, 2010). Turloughs are transient lakes, predominantly groundwater-fed, which flood as a result of a combination of high rainfall and accordingly high groundwater levels in topographic depressions in the karst (Naughton et al., 2012). They fill mainly by rising groundwater levels through estavelles, springs, direct precipitation, surface runoff and fluvial input; they ultimately drain to the groundwater system through estavelles and swallow holes (Coxon and Drew, 1986). Filling normally occurs in late autumn due to periods of intense or prolonged rainfall, with emptying typically occurring from March onwards. There are over 400 reported examples of turloughs across 12 counties, with the highest concentrations occurring in counties Galway, Clare, Mayo and Roscommon.

Surface drainage systems are frequently absent in well-developed karst landscapes. In their absence, turloughs play an important role as attenuation devices for both local and regional groundwater and surface water flow. The timing and extent of flooding within turlough basins can vary significantly between sites depending on the nature of and connections to local and regional karstic groundwater systems. Under extreme conditions, levels can exceed normal bounds and flood adjacent receptors such as residential and commercial buildings, agricultural land, and road networks. Furthermore, during extreme groundwater flood events a range of mechanisms beyond simple turlough flooding can cause a significant flood hazard.

Figure 2: Widespread groundwater flooding near Gort, Co. Galway

GROUNDWATER FLOODING: RAINFALL RESPONSE

Groundwater flooding generally requires sustained rainfall over relatively longer durations than other types of flooding; the groundwater aquifer acts as a damping filter that responds to rainfall fluctuations over a relatively long, sometimes even multi-year, timescale (Pinault et al., 2005). Accumulated recharge causes a rise in the groundwater table until it breaches the surface, which can represent a significant flood hazard during extreme events. Groundwater flooding, unlike fluvial (river) flooding, rarely poses a risk to life but instead can cause prolonged damage and disruption due to the relatively long flood duration. In terms of response times to rainfall, there are two end members in the groundwater flooding spectrum;

- Flash flooding involving rapid groundwater surges caused by short-duration (hours to weeks) intensive rainfall events (Bonacci et al., 2006; Marechal et al., 2008)
- Long-duration flooding as a result of the cumulative effect of sustained high levels of rainfall over successive months or years (Finch et al., 2004; Pinault et al., 2005)

The well-developed secondary porosity characteristic of Irish karst results in a relatively short system "memory", with maximum groundwater levels in one year having little or no impact on the following year. At a shorter timescale, the nature of Irish groundwater flooding is diverse; groundwater systems show a continuum of response times ranging from days to months (Naughton et al., 2012). At one end of the flooding continuum are rapidly responding systems in which water level or discharge increases in response to short-duration rainfall events (figure 3 (B)). At the other flooding extreme lie systems which show a seasonal flood pattern, where flood peaks are a cumulative response to rainfall over a number of months (figure 3 (D)).

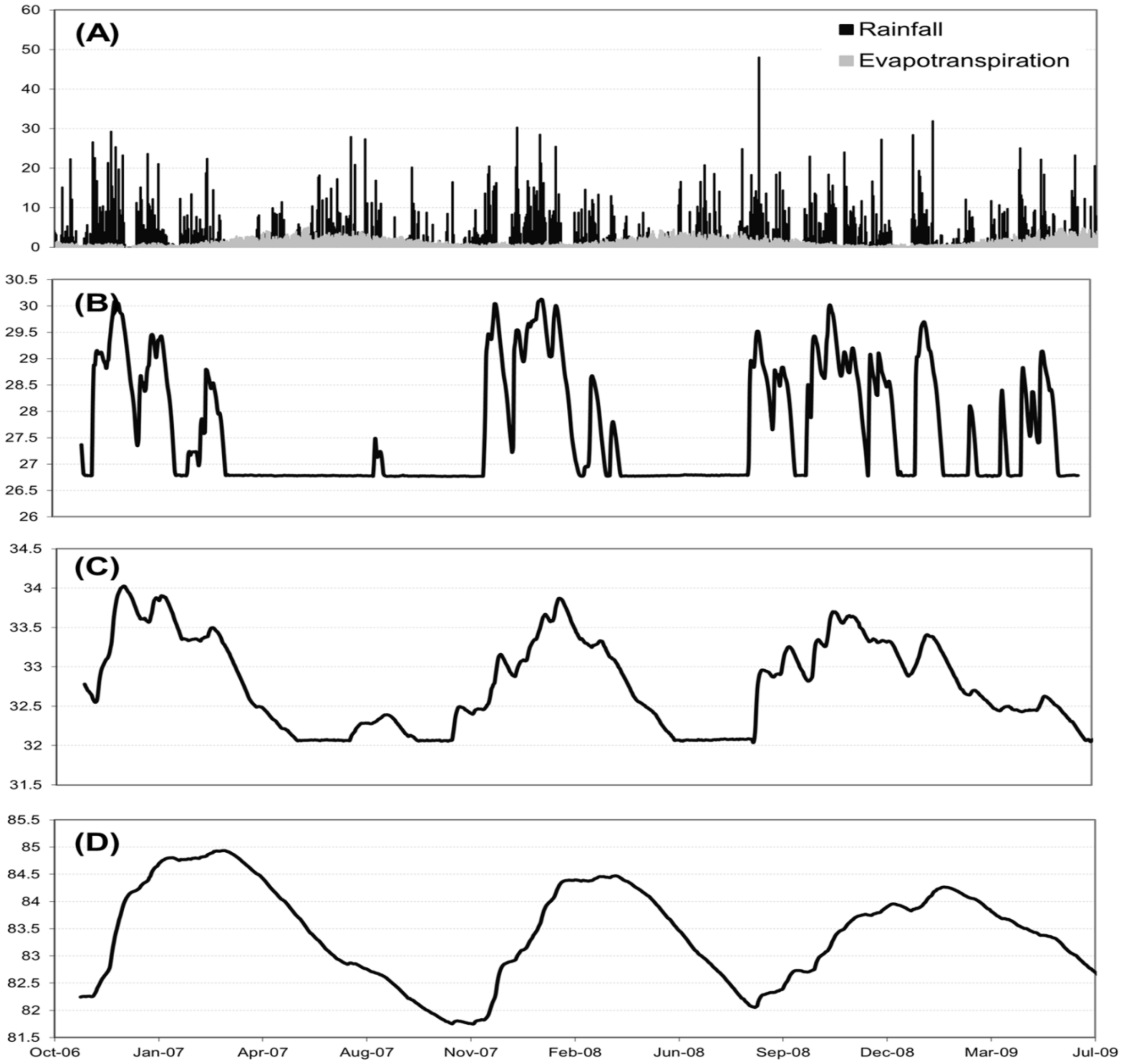

Figure 3: (a) Daily Rainfall and evapotranspiration for monitoring period from 2006 to 2009, and water level time series plot for (b) Turloughmore, (c) Skealoghan and (d) Coolcam (from Naughton et al., 2012).

An example of the variability of turlough flooding behaviour during extreme events is given in figure 4, which shows the normalised water level hydrographs for three turloughs in the Gort Lowlands during the regional flooding of late 2009. All three sites caused localised flooding, forcing extensive road closures and posing a flood hazard to surrounding property. Despite similar rainfall input to the catchments of the sites, the water level hydrographs and time of maximum flood level differ substantially. The peak annual level in Blackrock Turlough, a river-dominated turlough, occurred on the 26th November as a direct result of the heavy rainfall that occurred throughout November. Caranavoodaun Turlough, only 4km away, is primarily fed by a local epikarst system in a smaller catchment and showed a damped response compared to Blackrock. The peak level occurred on the 4th December, lagging behind Blackrock by some 8 days. Termon South Turlough, in contrast, did not peak until much later in the flooding season, on the 6th January 2010. The heavy rainfall of November 2009 contributed significantly to the stored floodwaters within Termon, but the slow drainage characteristic of this turlough meant that flood levels peaked much later in the season following a rainfall event in late December. The flood maximum in Termon turlough is thus a function of cumulative rainfall over a duration measured in months rather than days or weeks.

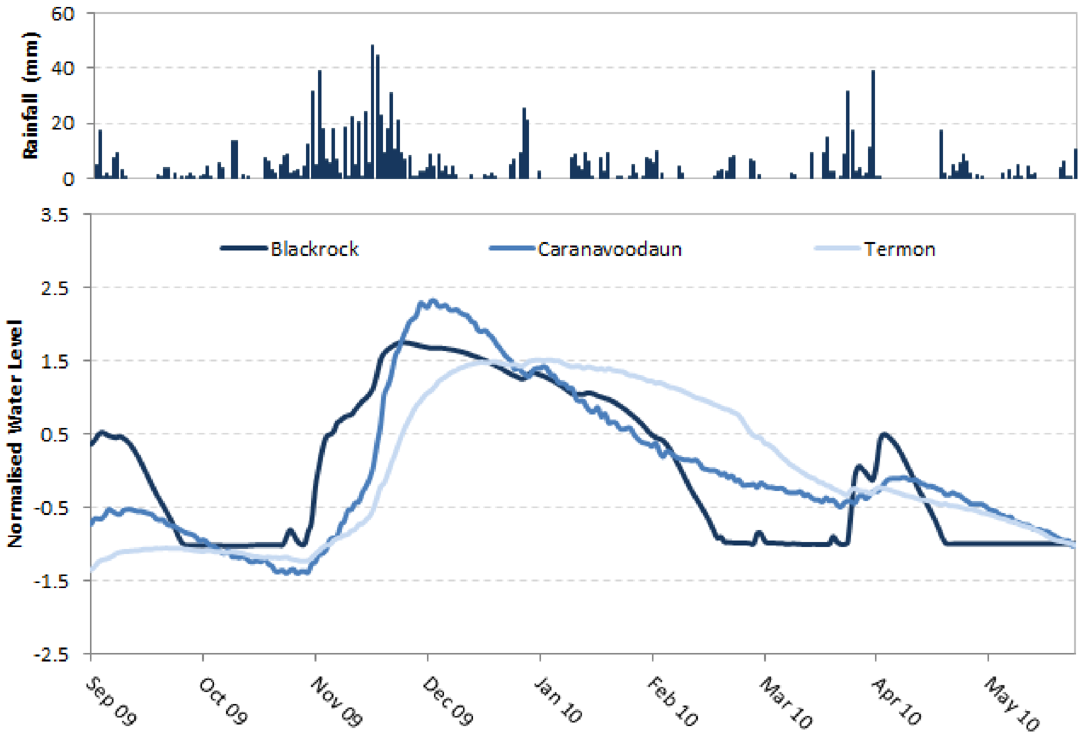

Figure 4: Rainfall record from Gort rainfall station and normalised water level hydrographs for Blackrock, Caranavoodaun and Termon South Turloughs, Co. Galway.

GROUNDWATER FLOODING: MECHANISMS

While the primary form of extensive, recurrent groundwater flooding in Ireland may originate in turloughs, during extreme groundwater flood events a range of mechanisms beyond simple turlough flooding play a key role. Groundwater flooding in karst areas manifests itself in a variety of ways depending on local hydrogeological conditions, often in combination with fluvial, pluvial and coastal flooding. Turloughs may exceed their normal inundation boundaries and pose a flood risk to the surrounding area. In such instances recharge can be derived from local or distal autogenic karst and/or allogenic non-karst catchments. Excess diffuse recharge can overwhelm the storage of an epikarstic aquifer, causing rapid rises in the unconfined water table and consequently flooding of historically dry topographic hollows. Epikarst features, such as shallow fissures and springs, may only become active at exceptionally high groundwater levels and thereby facilitate transfer of groundwater to adjacent topographic depressions.

Excess point recharge (such as sinking rivers) may inundate sinks and swallow-holes which would be capable of accommodating recharge under normal conditions. Such cases would usually have a strong fluvial component. The nature of flooding associated with this mechanism may be:

(a) Backwater flooding of the area upstream of the sink

(b) Overland flow into adjacent areas due to overtopping of the topographic depression containing the sink (figure 5)

This mechanism is one of the primary flood hazards in the Gort Lowlands. Such flooding precipitated the evacuation of six residential properties in Skehanagh, Co. Galway, during 2009 and again in 2014 and forced the closure of the N18 at Kiltartan (figure 5).

Figure 5: Flooding across the N18 National Road at Kiltartan (left) and L4506 Local Road at Corker (right) (images supplied by Galway County Council)

Groundwater springs and risings exceeding normal discharge levels can cause localised flooding around and downstream of the resurgence (groundwater-induced flooding). An example of such groundwater-induced flooding occurred at Four Roads, Co. Roscommon during late November 2009. A combination of pluvial flooding and groundwater-induced flooding associated with excess spring discharge inundated 6 houses, a community centre basement and a playground, as well as causing the prolonged closure of local and regional roads.

GROUNDWATER FLOODING: CURRENT AND FUTURE CHALLENGES

The spatial heterogeneity of groundwater flow within karst systems poses unique difficulties in evaluating likely flood occurrence. Unlike fluvial or coastal flooding, where the general flood locations are relatively simple to identify as they are concentrated along the river network or coastline, zones of groundwater flooding can be much harder to determine. Groundwater flooding predominantly occurs in rural areas and flood receptors tend to be similarly dispersed. This raises difficulties in the implementation of flood prevention schemes, as the scale and cost of such measures required to reduce the flood risk frequently outweighs the economic benefit. Models of hydraulic behaviour within Irish karst groundwater systems have been developed (Gill et al., 2013; Naughton, 2011; Southern Water Global, 1998) which could support the assessment of groundwater flood risk and appropriate mitigation measures, although the hydrological data required to predict behaviour during extreme events is commonly unavailable.

A second challenge to the effective management of groundwater flood risk is finding a balance between the often competing priorities of flood risk management and ecological conservation. Turloughs, the primary form of recurrent groundwater flooding in Ireland, are classified as Groundwater Dependent Terrestrial Ecosystems (GWDTE) and a protected habitat under the EU Habitats Directive. The depth, duration and frequency of inundation strongly influences the vegetation communities present within turloughs, with flood duration primarily controlling plant species survival (Casanova and Brock, 2000; Sheehy Skeffington et al., 2006). The periodic flooding of these depressions is an essential characteristic of the habitat, and any significant modification of the hydrological regime could negatively impact ecological status. Mitigation measures would ideally be solely directed at floodwaters resulting from the ‘overtopping’ of these depressions, while leaving the hydrological regime of the main basins unaltered. However, the imposition of such hydrological restrictions often makes the design of drainage schemes impractical of ineffective, thus the need to find balance between mitigating flood risk and promoting or maintaining good ecological status.

The climate in Ireland is changing in line with global and regional trends and this change is expected to continue in the coming decades (Desmond et al., 2009). It has been suggested that there will be an amplification of the seasonal hydrological cycle in response to the altered seasonal precipitation and an increased occurrence of extreme precipitation events (Steele-Dunne et al., 2008). The distribution of rainfall is also changing, with precipitation increasing in the north and west of the country (McElwaine and Sweeney, 2007). An increase in winter rainfall since the mid-1970s in the Gort lowlands had been identified during the Gort Flood Study, the largest groundwater flooding investigation to date in Ireland conducted following the major flooding of 1995 (Southern Water Global, 1998).

These climatic changes, if they occur, are likely to have a significant impact on groundwater flooding given the strong connection between short- to medium term rainfall and groundwater flooding response in Irish karst flow systems. Higher cumulative rainfall over the winter period could see the frequency of extreme regional flood events, such as occurred during the winter of 2009 and again in early 2014, increase significantly over the coming decades. While substantial progress has been made in elucidating the main groundwater flooding processes and drivers prevalent in Ireland, major challenges remain in terms of flood prediction which must be addressed in order to enable effective future groundwater flood management.

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REFERENCES

Bonacci, O., Ljubenkov, I., Roje-Bonacci, T., 2006. Karst flash floods: an example from the Dinaric karst (Croatia). Natural Hazards and Earth System Sciences, 6: 195-203.

Casanova, M.T., Brock, M.A., 2000. How do depth, duration and frequency of flooding influence the establishment of wetland plant communities? Plant Ecology, 147: 237 - 250.

Coxon, C.E., Drew, D.P., 1986. Groundwater flow in the lowland limestone aquifer of eastern Co. Galway and eastern Co. Mayo western Ireland. In: Paterson, K., Sweeting, M.M. (Eds.), New Directions in Karst. Geo Books, Norwich, United Kingdom, pp. 259-279.

Coxon, C.E., Drew, D.P., 1998. Interaction of surface water and groundwater in Irish karst areas: implications for water resource management In: Brahana, J.V., Eckstein, Y., Ongley, L.K., Schneider, R., Moore, J.E. (Eds.), Gambling with Groundwater - Physical, Chemical and Biological Aspects of Aquifer-Stream Relations. American Institute of Hydrology, St. Paul, Minnesota, pp. 161-168.

Desmond, M., O'Brien, P., McGovern, F., 2009. A Summary of the State of Knowledge on Climate Change Impacts for Ireland. Climate Change Research Programme - CCRP Report 1. Environmental Protection Agency, Johnstown Castle, Co. Wexford, Ireland.

Drew, D., 2008. Hydrogeology of lowland karst in Ireland. Quarterly Journal of Engineering Geology and Hydrogeology, 41: 61 - 72.

Finch, J.W., Bradford, R.B., Hudson, J.A., 2004. The spatial distribution of groundwater flooding in a chalk catchment in southern England. Hydrological Processes, 18(959-971).

Gill, L.W., Naughton, O., Johnston, P.M., 2013. Modelling a network of turloughs in lowland karst. Water Resources Research, 49(6): 3487-3503.

IIF, 2009. IIF 2009 Annual Report. Irish Insurance Federation (IIF), 39 Molesworth Street, Dublin , Ireland.

Marechal, J.C., Ladouche, B., Dorfliger, N., 2008. Karst flash flooding in a Mediterranean karst, the example of Fontaine de Nimes. Engineering Geology, 99: 138-146.

McElwaine, J.G., Sweeney, J., 2007. Key Meteorological Indicators of Climate Change in Ireland. Environmental Protection Agency, Johnstown Castle Estate, Wexford, Ireland. 31 pp.

Morris, S.E., Cobby, D., Parkes, A., 2007. Towards groundwater flood risk mapping. Quarterly Journal of Engineering Geology and Hydrogeology, 40: 203-211.

Mott Mc Donald, 2010. Preliminary Flood Risk Assessments - Groundwater Flooding. Prepared for the Office of Public Works, Dublin.

Naughton, O., 2011. The Hydrology and Hydroecology of Turloughs. PhD Thesis, Trinity College Dublin.

Naughton, O., Johnston, P.M., Gill, L., 2012. Groundwater Flooding in Irish Karst: The Hydrological Characterisation of Ephemeral Lakes (Turloughs). Journal of Hydrology, 470-471: 82-97.

Pinault, J.L., Amraoui, N., Golaz, C., 2005. Groundwater-induced flooding in macropore-dominated hydrological system in the context of climate changes. Water Resources Research, 41(W05001, doi:10.1029/2004WR003169).

Sheehy Skeffington, M. et al., 2006. Turloughs - Ireland's unique wetland habitat. Biological Conservation, 133(3): 265-290.

Simms, M.J., 2004. Tortoises and Hares: Dissolution, erosion and isotasy in landscape evolution. Earth Surface Processes and Landforms, 29: 447-494.

Southern Water Global, 1998. An investigation of the flooding problems in the Gort-Ardrahan area of South Galway. Final Report, April 1998. Produced by Southern Water Global and Jennings O`Donovan and Partners for the Office of Public Works, Dublin.

Steele-Dunne, S. et al., 2008. The impacts of climate change on hydrology in Ireland. Journal of Hydrology, 356: 28-45.

SESSION V

NEW METHOD FOR GROUNDWATER MONITORING AT MINE SITES

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ABSTRACT

The effectiveness of fully grouted vibrating wire piezometers (VWPs) has been understood since the 1960s. However, it is only in the last 15 years that they are routinely used at mine sites around the world. The uptake in use is in response to an increasing awareness in the importance of vertical hydraulic gradients on slope stability and the need for greater understanding of potential environmental impacts caused by dewatering. Fully grouted VWPs can be installed to significant depths (over 1,000 m) at relatively low cost, if piggybacked onto exploration or geotechnical borehole drilling programmes. The data provides a useful conceptualisation tool at all stages of mine life from baseline investigations through feasibility studies, operation and closure. Furthermore, coupled with lithological, structural and abstraction data, VWPs can play a key role in groundwater management at a mine site.

INTRODUCTION

Mining projects have become larger and deeper over the last 15 years, in parallel with a greater focus on environmental awareness. As a result, the industry requires greater understanding of groundwater issues both within concessions and the wider environment.

Mine dewatering can impact both a wide area and significant depth around the workings. Good understanding of the groundwater system is essential to ensure that negative impacts are minimised and managed; and that the dewatering system is efficient – both hydraulically and financially. Furthermore, vertical hydraulic gradients are often higher than lateral gradients at many mine sites, potentially causing a significant impact on slope stability.

The fully grouted vibrating wire piezometer (VWP) has become an indispensable groundwater monitoring tool at many mine sites as a response to this greater awareness in the role of groundwater management. Multi-level piezometers can be rapidly installed at great depths (over 1 km) in a geotechnical or mineral exploration borehole with relatively low incremental cost.

The fully grouted method involves the VWP (or VWPs) being placed at the chosen depth then the entire borehole being filled with bentonite-cement grout. Intuitively, reliable measurement of pore pressure using this method does not seem feasible. However, studies as far back as the 1960s have shown that a fully grouted VWP can record pore pressure with a small error in accuracy and a short hydrodynamic lag time (Mikkelsen and Green, 2003).

This paper summarises installation using the fully grouted method and provides examples where VWPs can provided valuable information through groundwater monitoring at mine sites.

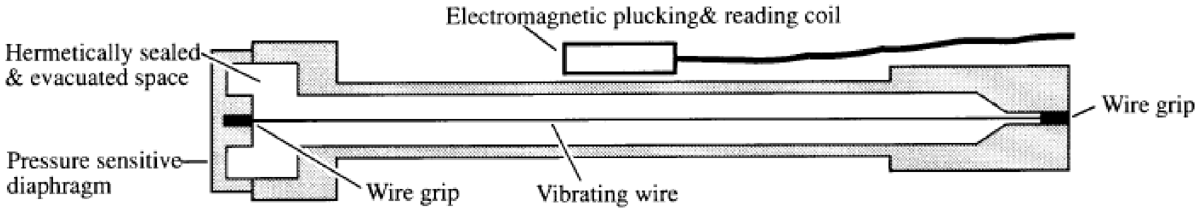
FULLY GROUTED VWP CONCEPTS

VWP SENSOR

A VWP consists of a tensioned steel wire fixed to the main body or the sensor at one end and to a flexible diaphragm at the other (Figure 1). Two opposing electromagnetic coils induce a brief voltage excitation which causes the wire to oscillate. Correspondingly, this oscillation generates an alternating current in the coils. The frequency of the output current varies depending upon the tension of the wire and, hence, the fluid pressure exerted on the diaphragm. This allows the output current to be converted into units of pressure (Geosense, 2009).

VWPs do not have batteries within their main unit but are installed with cables running to the surface. This allows a readout box or datalogger at surface to be used to power and control the VWP at depth. Without the need to change batteries in the sensor unit, a VWP can be permanently deployed (i.e. grouted into place). Furthermore, a datalogger at surface can be used for multiple VWPs allowing several to be installed within the same borehole (Figure 2).

Figure 1 VWP schematic (from Geosense, 2009)



GROUT
The hydraulic conductivity of the grout must ensure that no significant vertical flow occurs within the grout along any part of the borehole. Contreras, *et al.* (2007) demonstrate that the hydraulic conductivity of the grout can be no greater than three orders of magnitude higher than the surrounding formation.

Mikkelsen (2002) provides a mixing ratio to achieve an in-situ hydraulic conductivity of less than 10^{-8} m/s. The proportions by weight are: 1 cement to 2.5 water and 0.3 bentonite. This provides a ‘general purpose’ grout mix because very few installations will ever be completed in formations with a hydraulic conductivity of less than 10^{-11} m/s. Where required, different mixing ratios can be used which have a lower hydraulic conductivity, this usually requires a lower cement-water ratio.

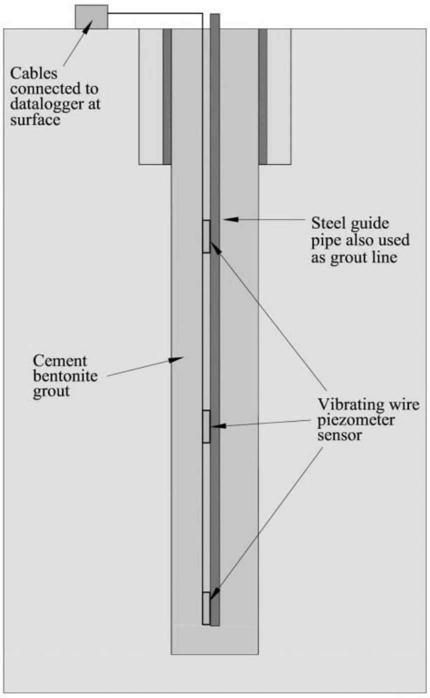
Cement provides the strength to the grout while the bentonite stabilises the mix and reduces the volume change during curing. The mixing of the grout is also seen as an important part of the process. First the water and cement are mixed before slowly adding the bentonite. The result should be a grout consistency similar to heavy cream. Only pure sodium bentonite powder, Type 1 Portland cement and fresh, clean water should be used.

INSTALLATION METHODOLOGY

METHODOLOGY
A number of different methods are available for installing multilevel VWPs. The guide-tremie pipe method provides the greatest control for locating the VWPs at the correct depth and ensuring that the grouting has been completed effectively. Installation is carried out using a rigid sacrificial tremie pipe with each of the VWPs strapped to the outside. Once installed, grout is pumped down through the tremie pipe until it flows from the annulus at surface. Ideally, this is done with an open hole although in collapsing conditions casing can be removed as the grouting progresses. Care must be taken so that the VWPs or cables are not snagged by the casing as it is retrieved.

For mining purposes, VWPs can be installed in exploration or geotechnical boreholes. This significantly reduces the cost of installation but usually means the borehole locations (and angles) have been selected by resource or geotechnical geologists rather than hydrogeologists. HQ boreholes (96 mm diameter hole) are considered to be the optimal size because: a) they are large enough for the installation to not pinch whilst being installed; b) the VWP is relatively close to the borehole wall; and c) the volume of grout required is not excessive.

Figure 2 Typical configuration of fully grouted guide-tremie pipe method VWPs (from Doubek, *et al.*, 2013)



TYPICAL INSTALLATION

1" diameter PVC or galvanised steel is commonly used for the guide-tremie pipe although this depends upon a number of factors including the diameter of the borehole, VWP and cable, whether the borehole is cased, the depth of borehole and the installation depth of VWPs.

A typical mining installation in an uncased, HQ exploration hole is as follows.

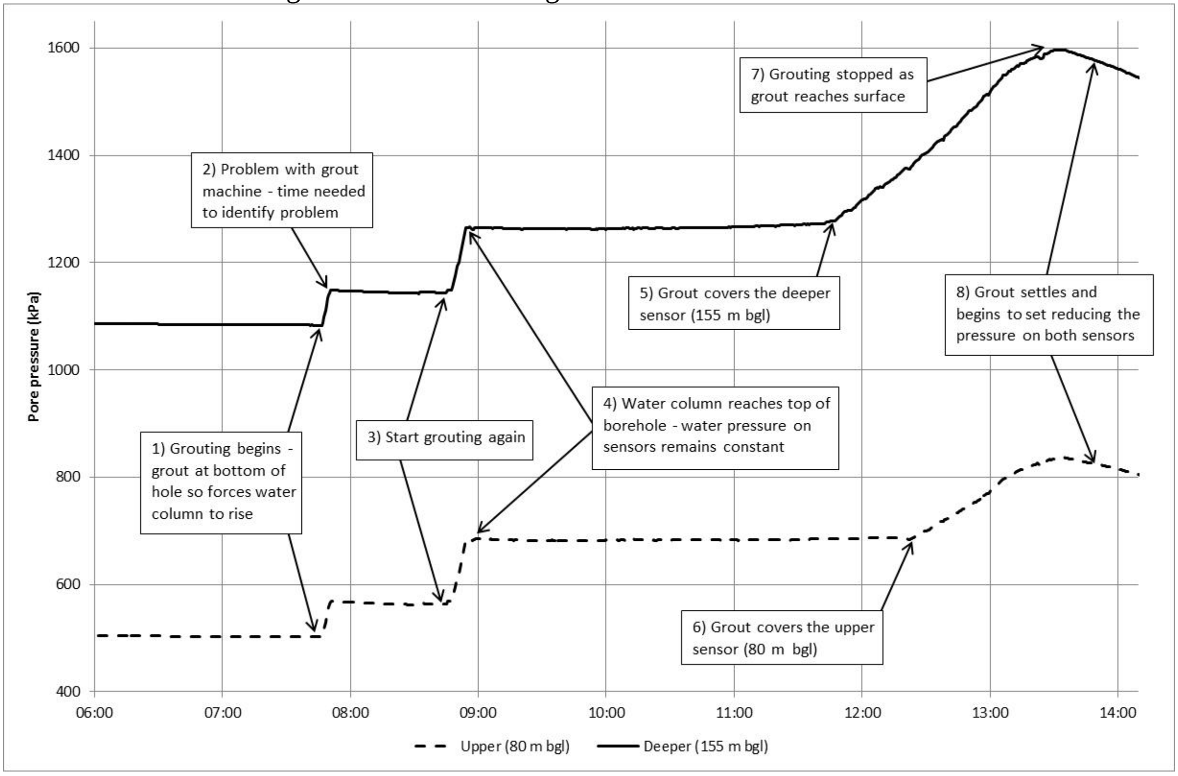
1. Review the borehole log and identify the depths (m down hole) for each of the VWPs.
2. Measure and mark corresponding depths on the guide-tremie pipe ensuring that 15 m to 24 m of slotted pipe is at the bottom of the string.
3. Record the zero readings[1] (temperature and pressure) of each of the VWPs.
4. Connect the VWPs to a datalogger and begin logging at a 1 minute interval.
5. Measure and record the static water level in the borehole using a dip meter.
6. Run the guide-tremie pipe into the borehole attaching the VWP sensors at the corresponding depth marks with a centraliser. Each VWP should be inverted so that the filter end is facing upward; this avoids trapping air bubbles on the filter during grouting.
7. Continuously monitor the datalogger readout to ensure the VWPs are not damaged during installation.
8. Strap the VWP cable tightly to the guide-tremie at 1 m to 2 m intervals.
9. Once the guide-tremie pipe and VWPs reach their specified depths, re-measure and record the water level in the borehole. Cross-check the actual water level measurement with the VWP pressure measurements to confirm the target depths have been achieved.
10. Inject the grout at a rate of around 0.5 L/s until it discharges from the annulus at ground surface. If possible, grouting should be completed without stopping and starting the injection. Depending upon formation losses, up to three borehole volumes of grout may be required. In very permeable formations, grouting may have to be undertaken in stages – waiting for a few

[1] For boreholes where the static water level is deep (over 50 m bTOC), the zero readings must be taken inside the borehole approximately 5 m above the water level. This can be done during the installation process.

days to allow the grout to set between injections. The use of multiple tremie pipes may be used for this purpose, staged at different depths within the borehole.

11. Monitor the pressure recorded by the VWPs throughout the grouting process using the datalogger. This provides details on the grout as it migrates up the annulus and helps identify how much grout is required (Figure 3).

Figure 3 Grouting of two VWPs in one borehole



MINING APPLICATIONS

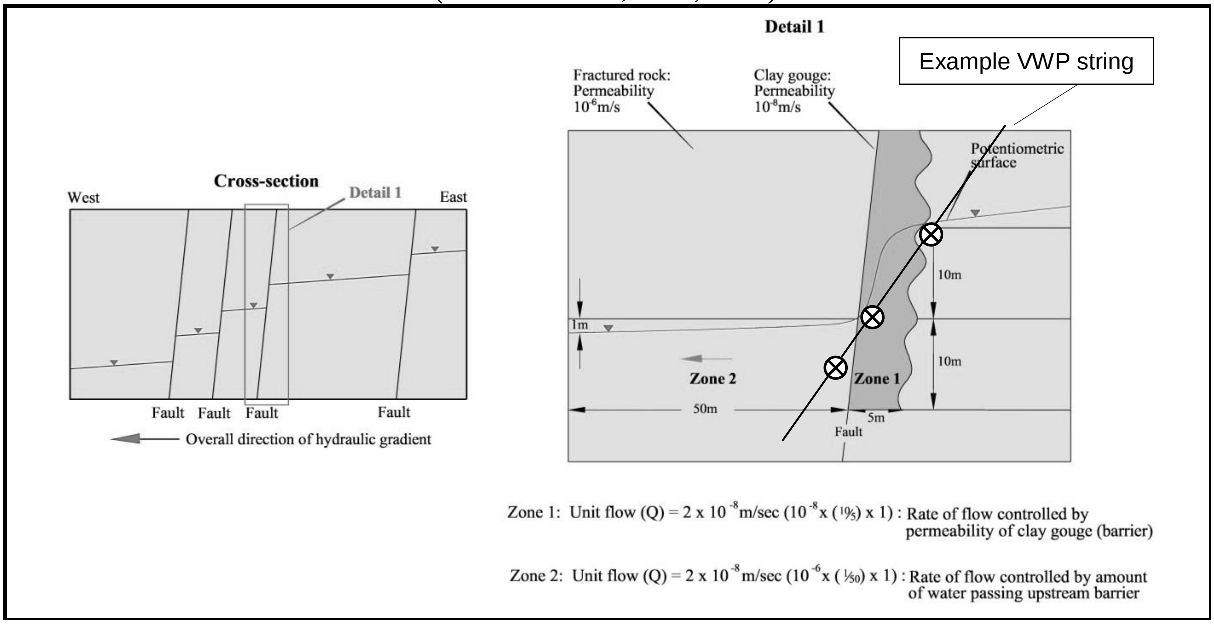
Data from a VWP string can provide important information at all stages of mine life from initial baseline investigations to monitoring rebounding water levels after closure. A number of examples are provided below to illustrate areas where VWPs have provided greater understanding of the groundwater system.

SLOPE STABILITY ANALYSIS

Geotechnical controls on slope stability include geomechanical properties, such as total stress or angle of internal friction, and pore pressure. Of these controls, pore pressure is the only one which can be readily manipulated to decrease the potential of slope failure. Time series VWP data is central to the calibration of numerical pore pressure models as part of a depressurisation study. On an active mine site, they may also provide key data for understanding the role of geomechanical unloading or presence of an over break zone on pore pressure distribution.

Even if a mine is successfully dewatered, residual water pressures may remain in certain areas so that a specific depressurisation programme may be required. For example, steep hydraulic gradients may develop across geological structures producing a ‘stair-stepping’ of pore pressures. This has the potential to create destabilising conditions, particularly for those structures that have an adverse dip into the wall. The installation of a VWP string across such a structure can be used to identify the development of vertical hydraulic gradients in response to dewatering (Figure 4). This will allow the planning and monitoring of a depressurisation programme, such as horizontal drains.

Figure 4 Illustration of the effects of compartmentalisation with example VWP locations (based on Beale, *et al.*, 2013)



DEWATERING EVALUATION

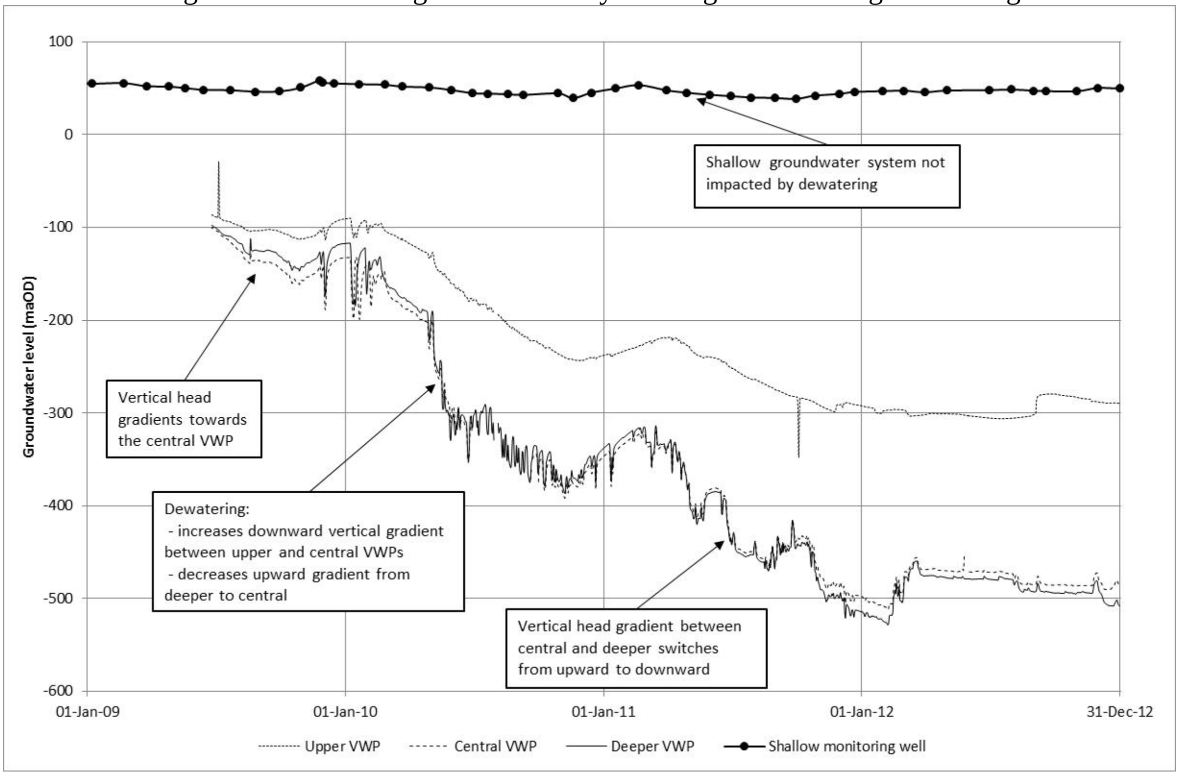
VWPs allow the monitoring of piezometric head at significant depths where conventional piezometers may be impractical. In mining, the workings may be 100s of metres below ground level; this allows the monitoring of groundwater to be completed simply and effectively from surface.

For example, Figure 5 shows the hydrograph from a mine with workings over 1 km below ground level. The VWP string was installed in 2009 to monitor the dewatering of an extension from the main workings. The hydrographs show that:

- prior to dewatering of the extension, the piezometric head at depth was already substantially lower than the shallow groundwater system;
- dewatering has not impacted the shallow groundwater system;
- initially, vertical hydraulic gradients were towards the ‘aquifer’ unit represented by the central VWP;
- the units of the central and deep VWPs are hydraulically well connected;
- the unit of the upper VWP is leaky;
- as dewatering increased, so did the downward vertical hydraulic gradient from the upper to the central VWP;
- in mid-2011, the hydraulic gradient between the central and deep units was reversed from upward to downward.

Used in parallel with lithological data, the location of dewatering pumps and abstraction rates, this information is a valuable tool for identifying: a) potential impacts of the dewatering programme; b) the success of the programme; c) areas where refinement is required; and d) provide information for the design of future dewatering programmes in the area.

Figure 5 Changes in vertical hydraulic gradient during dewatering



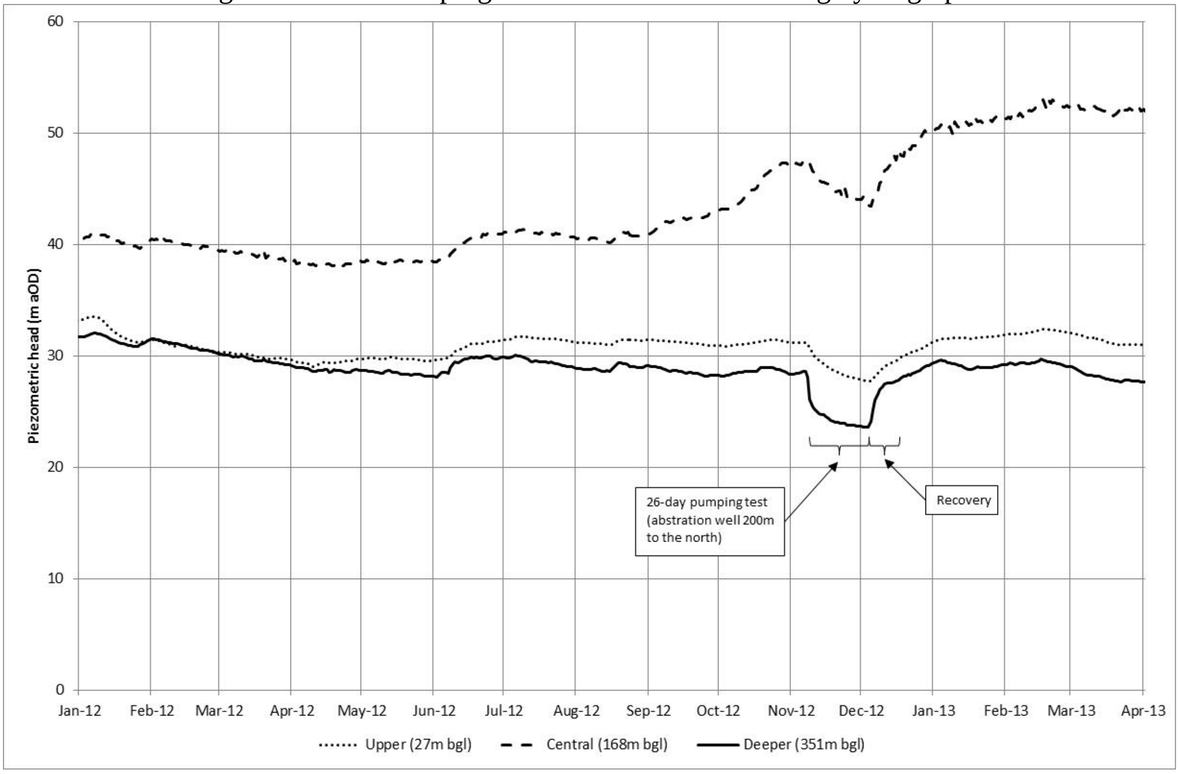
FRACTURE FLOW

VWP strings may be used to provide a greater understanding of transient pore pressure and compartmentalisation of groundwater within geological formations. This is further enhanced if the aquifer system is stressed through, for example, a pumping test. Such studies are beneficial at early feasibility phases of mining projects to greater understand the likely water management (dewatering) requirements and potential impacts to the environment.

Such techniques are useful in karst environments which are dominated by fracture flow. During a preliminary feasibility study, a number of multilevel VWP installations were installed around a proposed mine site. One installation (Figure 6) showed anomalously low pore pressure in the deeper VWP 'aquifer' unit. The other installations showed a consistent upward gradient in all VWPs, with the exception of one installation on a hilltop which was consistently downward. This installation, however, showed an upward gradient from the central to the upper and a downward gradient from the central to the deeper. It has been proposed that this is due to a high permeability fissure or fracture being hydraulically connected to the deeper VWP 'aquifer' unit. The next step is to further investigate this feature locally and regionally to understand its significance for mine dewatering.

The hydrograph (Figure 6) includes the period when a 26-day pumping test was carried approximately 200 m north of the VWP string. The deeper VWP had the greatest drawdown (4.9 m) but was also the fastest to recover. The upper VWP showed the smallest drawdown (2.7 m) and was the slowest to recover. This strengthens the assumption that the deeper VWP is in hydraulic connection with a high permeability feature (which was also in connection with the pumping well). In addition, it identified that all three units being monitored are also in hydraulic connection with one another.

Figure 6 Pumping test observation VWP string hydrographs



LAST WORD

Whilst not a new technology, the installation of VWPs for groundwater monitoring at mine sites has increased significantly in recent years. As the global database expands, so a greater awareness of the benefits they provide increases. In turn, these benefits can be applied to other areas of groundwater management such as source protection for water supplies or contamination plume migration modelling.

Thanks to Geoff Beale and Paul Littlewood for their help researching this paper, providing examples and general technical overview.

REFERENCES

Beale, G., Price, M. and Waterhouse, J., 2013. Framework: assessing water in slope stability. In Beale and Read (eds). Guidelines for evaluating water in pit slope stability. CSIRO Publishing, Australia.

Contreras, I. A., Grosser, A. T. and Ver Strate, R. H., 2007. The Use of the fully-grouted method for piezometer installation. International Symposium on Field Measurements in Geomechanics, Boston, USA. September 2007.

Doubek, G., McKittrick, R., Price, M. and Sholl, S., 2013. Guidelines for field data collection and interpretation. In Beale and Read (eds). Guidelines for evaluating water in pit slope stability. CSIRO Publishing, Australia.

Geosense, 2009. VW Piezometers Instruction Manual. Version 1.4 May 09. Marton Geotechnical Services Ltd, UK.

McKenna, G. T., 1995. Grouted-in installation of piezometers in boreholes. Canadian Geotechnical, Journal 32, p 355-363.

Mikkelsen, P. E., 2002. Cement-bentonite grout backfill for borehole instruments. Geotechnical News, Vol. 20, No.4 p 38-42. December 2002.

Mikkelsen, P. E. and Green, E. G., 2003. Piezometers in fully grouted boreholes. International Symposium on Geomechanics, Oslo, Norway. September 2003.

EXPLORATION DRILLING - GUIDANCE ON DISCHARGE TO GROUNDWATER

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ABSTRACT

In order to comply with the ***European Communities Environmental Objectives (Groundwater) Regulations, 2010 (S.I. No. 9 of 2010)*** *there are now clear requirements to carry out an impact assessment for activities that are discharging to groundwater bodies. This includes prospecting activities such as exploration drilling. EMD have completed a draft guidance document on procedures and risk assessments to be carried out for exploration drilling to meet the requirements of the groundwater regulations. It is anticipated that, as a result of the regulations, authorisation will be required for exploration drilling. The procedures outlined in the document will also assist with the screening process required as part of the* ***European Communities (Birds and Natural Habitats) Regulations, 2011*** *(S.I. No. 477 of 2011) for prospecting activities. The principle objective of the document is to outline the necessary measures to be adopted during drilling to protect sensitive receptors such as groundwater wells and surface water bodies from any potential contamination.*

INTRODUCTION

This document provides guidance on the hydrological and hydrogeological assessments required for exploration drilling programmes taking into account the objectives and requirements of the European Communities Environmental Objectives (Groundwater) Regulations, 2010 (S.I. No. 9 of 2010) (Groundwater Regulations). In the context of pressures on groundwater quality, exploration drilling is not considered to be a major threat. However, there are now clear requirements to carry out an impact assessment for activities that are discharging to groundwater bodies.

Under Regulation 8 of the Groundwater Regulations direct discharges to groundwater are prohibited. Certain types of direct discharge may be permitted subject to prior authorisation "*provided such discharges and the conditions imposed, do not compromise the achievements of the environmental objectives established for the body or groundwater into which the discharge is made*". Exploration drilling is one of the exempt activities listed under Regulation 8.

Under Regulation 14 of the Groundwater Regulations certain direct and indirect discharges to groundwater may be exempted (includes activities listed under Regulation 8) from the environmental objectives for groundwater under conditions (technical rules) that can be established by the Environmental Protection Agency (EPA). Exploration drilling will be subject to such conditions (technical rules).

Regulation 4 places a duty on public authorities to promote compliance with the Groundwater Regulations. Exploration and Mining Division (EMD), in consultation with the EPA, has established the technical rules in this guidance document.

This document outlines the required assessment process and establishes the technical rules required to obtain approval for exploration drilling. The assessment is carried out by the holder of the Prospecting Licence (PL) and submitted to EMD.

Exploration drilling involves both direct and indirect discharges to groundwater:

- Direct discharge is through the drill hole annulus and

- Indirect discharge generally takes place at surface through a percolation area used for the return drill water or though sumps.

The principal risk to groundwater receptors relates to the introduction of microbial pathogens into the aquifer through the drill hole as a result of using surface water streams and rivers as a source of drill water. The principal receptors are domestic, group and public well supplies. In many drilling operations, the objective is to operate a closed loop system where the return water is recycled for reuse with no losses to the aquifer. The return water is collected in a sedimentation tank and then the decanted water is pumped to a second tank for reuse as drill fluid. If the water is returned without loss, then only occasional topping up is required from the surface water source to take account of small losses. In other situations the water can be lost and continuous topping up is required with discharge to the aquifer.

The Environmental Protection Agency (EPA) has published '*Guidance on the Authorisation of Discharges to Groundwater*', outlining detailed technical assessments required to authorise discharge to groundwater –. The Source-Pathway-Receptor (SPR) model is adopted and develops a framework for the process that includes guidance on:

- Risk screening for potential impact to groundwater based on pollutant load;
- Levels of technical assessment for different types of discharge;
- Predicting impact.

The procedures and assessment methodology outlined in the EPA document forms the basis for the assessments carried out in this guidance appraisal document.

The purpose of this document is to provide guidance on the assessment of discharge to groundwater for exploration drilling programmes in order to ensure compliance with the objectives of the Groundwater Regulations (S.I. No. 9 of 2010) by outlining a risk based approach based on the Source-Pathway-Receptor model. A wide range of hydrogeological conditions exist in Ireland. It is, therefore, not possible to be prescriptive, but rather to give an overall consistent approach and outline the underlying procedures that should be followed to protect groundwater aquifers from contamination including:

- Assessment of the risk posed by discharges to groundwater from exploration drilling activities taking into account the likely pollutant concentrations;
- A screening process to be adopted to determine if mitigation measures are required;
- A risk based assessment tool to predict the scale of the likely impacts, and
- Appropriate mitigation options for different levels of exploration activities.

RISK ASSESSMENT APPROACH

The document outlines the essential steps required to carry out a risk assessment and the appropriate mitigation measures for exploration drilling dealing with direct discharge down the borehole and indirect discharge on surface. The main risk is considered to be related to the use of surface water as a drilling fluid and the potential to introduce pathogens into the groundwater body. A simple three step approach is outlined and includes:

1. Assess the nature of the discharge activity and carry out a risk screening to determine the degree of risk posed by the discharge activity on groundwater quality and receptors from direct (Step 1A) and indirect discharge (Step 1B).
2. From the risk screening, determine an appropriate level of technical assessment (Step 2) that is needed to address questions about site suitability and estimation of loading and attenuation.
3. Conduct technical assessment (Step 3).

Step 1 involves assessing the distances to sensitive receptors, the volumes of drill waters required for drilling, the likely pathways to groundwater and surface water and the scale of the exploration activity (number of planned drillholes).

Step 2 involves determining the level of assessment.

• **Negligible risk, no further work necessary, document findings.**
• **Where risk is deemed to be low, a Stage 1 assessment is required;**
• **Where risk is deemed to be moderate, a Stage 2 assessment is required;**
• **Where risk deemed to be high, a Stage 3 assessment is required.**

Step 3, depending on the level of risk, various levels of technical assessments are required relating to the aquifer properties and distances to sensitive receptors.

Negligible	**Consider nature of activity and distances to receptors.**
Stage 1	Basic assessment of aquifer properties. Examine zones of contribution to wells.
Stage 2	Assess travel time and groundwater velocities. Assess receptors and zone of contribution. Assess impacts of abstraction on surface water bodies.
Stage 3	Assess aquifer properties, permeability, flow direction. Abstraction impacts. Develop conceptual model.

Technical Assessment – Step 3
It is not possible to be prescriptive about every situation and the following table is an ***indicative guide only***. In certain instances where an impact cannot be ruled out, monitoring of the well may be required before, during and after drilling. In such situations it is advisable to consult a hydrogeology specialist. The following table lists examples of different exploration activities and the technical assessment requirements for the various levels of assessment.

Level of Assessment	**Nature of Activity and Geological/Ecological Conditions**	**Tests/Calculations Requirements**
Negligible	➢ > 100m from domestic receptor ➢ > 300m from group or public water supply borehole or spring ➢ No sensitive ecological receptors ➢ 1 Drillhole Planned ➢ Poor or locally important aquifer	➢ No receptor ➢ No pathway ➢ No impact
Stage 1 Assessment Low risk	➢ > 100m from domestic receptor ➢ > 300m from group or public ➢ No sensitive ecological receptors ➢ 1 to 3 Drillholes Planned ➢ Poor aquifer	➢ Basic assessment of aquifer properties. ➢ 300m unless local indications indicate otherwise
Stage 2 Assessment Moderate Risk	➢ > 100m from domestic receptor ➢ > 300m from group or public ➢ No sensitive ecological receptors ➢ Regionally important aquifer fissure flow ➢ 1- 10 drillholes planned	Aquifer Properties ➢ Determine permeability and travel velocity ➢ Assess receptors based on high velocities Surface waters ➢ Ensure no significant impact on surface waters from abstraction.
Stage 3 Assessment High risk	➢ Advanced exploration programme > 10 drillholes ➢ > 100m from domestic receptor ➢ > 300m from group or public ➢ Karst aquifer. ➢ Fissure and Conduit flow ➢ Sensitive water course < 100m from percolation zone.	➢ Aquifer Properties o Permeability o Flow Direction o Groundwater gradient ➢ Conceptual Model ➢ Conduct percolation tests if required ➢ Assess significance of abstraction impact on surface waters ➢ Calculate travel velocities based on aquifer properties.

MITIGATION MEASURES

In many instances a closed loop system is operated where drill water is recycled through sedimentation tanks or sumps. This is considered to be best practice and significantly reduces the risks to groundwater and surface water bodies. For some grassroots drilling, drill water is not recycled and may be discharged though sumps or settlement tanks. There should never be a direct discharge to a stream or river. If the assessment indicates a potential impact on a sensitive receptor then various mitigation measures are outlined to reduce the risks. Mitigation measures may, if required, include:

Drill water source can be	Treated water from a mains supply; Clean treated surface waters; Groundwater of a satisfactory quality from an existing well in the aquifer
Additives	Use appropriate additives in broken or fractured bedrock to prevent ingress into the aquifer; Ensure additives are non-hazardous, non – toxic and biodegradable.
Setback distance from wells/spring supplies	At least 100m from a domestic supply and 300m from a public or group water scheme supply. If in karst area with conduit flow then re-evaluate distance based on risk assessment. Holes should also be an adequate distance from sensitive receptors (including SACs and surface water receptors where there is a large component of groundwater flow to surface waters). Setback distances will be site specific.
Site setup	Oil matting Spill kits Proper storage of fuel
Percolation area	Assess aquifer vulnerability (GSI) and ensure adequate protection of groundwater body. Ensure adequate setback from surface water Ensure no discharge to ground via karst features such as dolines and swallow holes. Ensure adequate treatment of discharge through by allowing to percolate through a vegetated buffer zone. Ensure adequate treatment through sumps (sufficient residence time) and modify or recirculate waters as required. If subsoil thickness is not adequate for recharge to the aquifer, or the area is underlain by low permeability soils, an alternative treatment system is required, such as settlement tanks.
Decommissioning	Advice on decommissioning boreholes and backfilling is also outlined.

HOLE COMPLETION

Drill holes must be decommissioned in a manner that will:

- Remove the hazard of an open hole;
- Prevent the borehole acting as a conduit for contamination to enter groundwater;
- Prevent the mixing of contaminated and uncontaminated groundwater from aquifers within the drilled strata;
- Prevent the flow of groundwater from one groundwater body to another; and
- Stop artesian flow.

The objective of decommissioning is to ensure no long term impact on the groundwater system. This means sealing off the different groundwater inflow zones to prevent cross-flow from one level to another. Particularly relevant is the subsoil-bedrock interface s, in most cases; the aquifer requiring protection will be the bedrock aquifer. Shallow groundwater has a higher potential for being polluted. Particular care should also be given to the upper fractured and weathered bedrock zone. This zone should also be backfilled. Backfilling materials should be clean (washed), inert, uncontaminated, excavated materials. Low permeability materials such as clay, bentonite or cement grout and concrete are inserted where impermeable barriers are required (e.g. in the upper weathered zone so that this high permeability zone does not link with any conduits in the borehole). This will prevent mixing between aquifers such as the till aquifer and the bedrock aquifer.

For artesian boreholes, the objective is to confine the flow to the horizon it came from. This could require lowering the groundwater for a period by:

- Pumping the borehole to produce a drawdown;
- Introduction of a precast plug at the appropriate level;
- Use of an inflatable packer and pressure grouting the void space.

The top of the borehole must be sealed to prevent surface water ingress. The borehole is finished with an impermeable plug or cap. The final 2 metres is filled with cement, concrete or bentonite grout with a concrete or cement cap. The final capping level should take into account future farming activities.

SESSION VI

RURAL BOREWELL SCHEME EXPLORATORY DRILLING PROVIDING QUALITATIVE AND QUANTITATIVE HYDROGEOLOGICAL INFORMATION IN NORTHERN IRELAND

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ABSTRACT

A wholesome drinking water supply is a valuable resource, often taken for granted. Whilst less than 1% of water in Northern Ireland is sourced from private water supplies there remain a significant number of people in Northern Ireland that do not have access to public water and as a result do not have an adequate or sustainable drinking supply. This group of society predominantly live in isolated rural areas of the country. The Rural Borewell Scheme is a government grant scheme aimed at offering financial and technical assistance to domestic property owners who do not have a wholesome water supply. Since the project launch in June 2012 sixty-three new borewells have been drilled across the region, with an estimated 110 additional borewells expected by the end of 2015. Most of the wells drilled have been in low productivity hard rock aquifers, often seen as having limited resource potential. The results of the exploratory drilling have yielded additional qualitative and quantitative information augmenting our understanding of hydrogeology of Northern Ireland. It has been 20 years since the completion of the last large scale groundwater exploration work undertaken across the region. A better understanding of groundwater quality of targeted aquifers is possible, with comprehensive testing undertaken at each new supply. Despite variability of raw water quality between supplies, groundwater quality is of a high standard. The most prevalent groundwater quality breaches have been naturally occurring iron and manganese, with 23% of sites requiring additional treatment for iron, manganese, arsenic, aluminium, nickel, selenium and turbidity over and above precautionary filtration and ultra violet installed at all sites. Despite very low yields having been encountered in some regions it has been demonstrated that with careful design even very low productivity boreholes can supply adequate water on the domestic supply scale.

SOURCE PROTECTION FOR GROUP WATER SCHEMES; WORKING WITH LOCAL COMMUNITIES

Monica Lee, Caoimhe Hickey, Taly Hunter Williams (GSI).
Colm Brady, Karen Carney, Barry Deane, Joe Gallagher, Brian MacDomhnaill (NFGWS).

ABSTRACT

In 2011, the Groundwater Section in the Geological Survey of Ireland (GSI), in collaboration with the National Federation of Group Water Schemes (NFGWS) embarked on a pilot study to provide preliminary groundwater source protection plans to Group Water Schemes (GWS) to enable them to more effectively understand, and therefore manage, their own drinking water.

Further to the success of the pilot project and a campaign to acquire funding in the form of a Rural Water Programme (DECLG) grant, a full programme of works was proposed to provide preliminary source protection plans for the remaining c.250 groundwater supplied GWSs.

From the outset, the aim was to provide usable, tailored, GWS-friendly reports (short and standardised; use non-technical language; include a 'sky to source' description of the conception model, which is also represented in an easy-to-understand cross section diagram; and delineate a ZOC that is supported by the available data).

The overall goal, however, was to enable a more comprehensive understanding of the GWS's supply to support their management decisions and ultimately, really enable them to take ownership of their drinking water. This necessitated an on-going interaction with the GWS, both throughout the term of the project, and after. The excellent relationship between the NFGWS and GWSs has undoubtedly facilitated this process, and their continued engagement will remain as one of the supporting pillars to aid good management into the future.

INTRODUCTION

In 2013, the Groundwater Section in the Geological Survey of Ireland (GSI), in collaboration with the National Federation of Group Water Schemes (NFGWS), embarked on a 5 year programme to provide preliminary source protection plans to Group Water Schemes (GWS). This followed on from a two-phase pilot project undertaken in 2011 and 2012.

The ultimate aim was to enable the GWS operators and members to more effectively understand, and therefore manage, their own drinking water. Keeping this aim in mind, and working within the ethos of the NFGWS, the underlying goal is to not just provide *useable information*, but to enable a more comprehensive *understanding* of their supply to support their management decisions and ultimately, really enable them to take ownership of their drinking water supplies.

The NFGWS has worked hard to win recognition for the utility and importance of this programme. Consequently, a Rural Water Programme (DECLG) grant scheme was approved that has enabled this work to be undertaken in a reasonable time frame.

The aim of this paper is to present the background to the project, the role of the organisations and consultants involved, the progress and some of challenges. We also need to answer the question; 'Are we succeeding?'

GROUP WATER SCHEMES

Until the 1950s piped water supplies outside of Ireland's towns and cities were rare. Rural communities relied on buckets and barrels for their daily water needs, drawing supplies from wells, rivers or lakes – a painstaking and laborious chore.

In 1955 a committee comprising representatives of the Department of Local Government and the Local Authorities carried out a comprehensive assessment of water and sewerage services. This Committee identified the non-availability of piped water in rural areas as a major issue and their conclusions formed the basis of a three-pronged strategy launched in 1959 to provide:

- Regional schemes by the Local Authorities.
- Group Water Schemes by local communities where reliable, local sources were available.
- Piped water by individual householders where the other approaches were not feasible.

Group Water Schemes (GWS) flourished in the 1960s/1970s, often through the efforts of local co-operatives and farm organisations and, in time, the sector was providing drinking water to approximately 25% of the rural population.

Currently, approximately 540 GWSs are large enough (greater than 50 people, or 15 domestic connections) to come under the terms of the Drinking Water Regulations (2007). Of these, just under 70% (c.370 schemes) depend either wholly or in part on groundwater sources, while a proportion of surface water supplies are spring-fed. As sources of drinking water, the importance of GWS supplies is also acknowledged under the Water Framework Directive.

NATIONAL FEDERATION OF GROUNDWATER WATER SCHEMES

The National Federation of Group Water Schemes (NFGWS) is the representative and developmental organisation for GWSs in Ireland. Founded in 1997, in response to the ending of water charges on public water schemes, the primary objective of the NFGWS, at its inception, was to secure equality of treatment for the group scheme sector.

The aims of the organisation broadened in light of mounting evidence of poor drinking water quality within the sector. The extent of this problem was highlighted by successive EPA Drinking Water Quality Reports, and by a case brought against Ireland in the European Court of Justice and adjudicated on, in November 2002.

One of the main focuses of the NFGWS during the late 1990s and early 2000s was on ensuring GWSs were provided with the financial resources necessary to install essential water treatment facilities. The Rural Water Programme, agreed between the NFGWS and the DECLG, has been extremely successful in achieving this, and recent EPA Drinking Water Quality Reports reflect the consequent mammoth improvements in GWS drinking water quality.

With the water treatment issue largely addressed, the NFGWS is now concentrating more of its efforts on the improvement of management capabilities and structures of the sector, and on the very important area of drinking water source management and protection.

The NFGWS recognise and promote source protection as a major element of the Quality Assurance[1] system that they roll out to GWSs. The Quality Assurance system is a holistic framework advocating the multi-barrier approach to achieving good quality drinking water. The responsibility to provide

[1] A Quality Assurance (QA) system, based on 'Hazard Analysis Critical Control Points' principles, has been developed by the NFGWS under the auspices of the National Rural Water Services Committee. GWSs are strongly encouraged to implement this system. In essence, QA provides a GWS with a framework for identifying, monitoring and recording present and potential hazards, as well as providing a range of standard operational procedures.

clean water lies with the GWS and its members and, therefore, the NFGWS work very closely with the GWSs to implement the Quality Assurance system in a *bottom up* approach.

GSI AND SOURCE PROTECTION

Groundwater source protection work is well established. Catchment areas, or 'zones of contribution' (ZOC), and source protection zones (SPZs)[2] have been delineated for a number of generally larger, public drinking water supplies. Over the years, this work has been undertaken by the GSI, consultants, and more recently the EPA.

Since the 1980s, the Groundwater Section in the Geological Survey of Ireland (GSI) has been delineating SPZs for groundwater drinking supplies as part of the Groundwater Protection Schemes (DoELG/EPA/GSI, 1999) programme of works. These projects have been in partnership with Local Authorities and therefore the focus has been public drinking water supplies and the Local Authority-managed (publically sourced) GWSs. To date, the GSI have delineated c.130 SPZs.

As the public drinking water supplies and publically sourced GWSs characteristically supply a large number of people, the GSI SPZ work is geared towards having the highest level of confidence possible when delineating the SPZs. Consequently, these are frequently resource-intensive projects, often taking between one and two months to collect, collate and assess all information, delineate SPZs, and to produce the reports. Once delivered to the Local Authorities, the digital SPZ maps are integrated into the Local Authority’s intranet and GIS systems, in order to support decision-making e.g. for planning purposes. In this way, the information is frequently used in a *top-down* approach to management.

More recently (2009-2013), the EPA commissioned the delineation of c.35 SPZs and c.240 ZOCs to characterise the EPA water quality monitoring points, which are used to meet the requirements of the Water Framework Directive.

GWSs AND SOURCE PROTECTION

Source protection work has been undertaken on some privately sourced GWSs, however, this is not a common occurrence as the focus was previously on ‘end-of-pipe’ water quality and solutions.

To fit with the multi-barrier approach advocated in their Quality Assurance system, the NFGWS have been pursuing a more comprehensive programme of source protection works. In 2004 they started looking at catchment delineation and protection for *surface water* supplied GWSs. The following year, they launched the ‘National Source Protection Pilot Project for [surface water] GWSs’, which is a collaboration with the National Centre for Freshwater Studies (Dundalk Institute of Technology; DkIT). The aim was to develop strategies for protecting surface water sources, with a focus on community support, commitment and participation.

In 2010, the NFGWS partnered with the GSI to develop a similar preliminary source protection plan model for groundwater-sourced GWSs. From the outset, it was considered that wholesale application of the SPZ report model would not be appropriate for the majority of these generally smaller supplies and the mainly non-technical GWS stakeholders. The cost of the full SPZ studies was also thought to be prohibitive.

GSI suggested that GWSs would get the most value for money from a predominantly desk-based study delineating ZOCs, which could then be combined with the existing national groundwater vulnerability maps. By doing this, they would have a very good indication of where their water was coming from and, within that delineated catchment area, where the risks of groundwater contamination were likely to be. This information could then be used to target areas to reduce the risk

[2] Please refer to Groundwater Protection Schemes, 1999 (DoELG/EPA/GSI,1999).

of future contamination or, conversely, to undertake hazard mapping in order to identify potential sources of existing contamination. To ensure that these reports and data would be properly utilised, an intrinsic part of the programme has been to fully engage with the GWS personnel at all possible stages of the project.

From the outset it was recognised that, even with relatively short, predominantly desk-based studies, this project would require a number of consultants to complete this work for c.250 GWSs[3] within a reasonable timeframe.

The overall benefits of this approach are that i) the GSI and GWSs were working with a larger group of hydrogeologists, ii) the GWS recognised the necessity of engaging a hydrolgeologist in this type of work, and iii) in addition to the GSI, the GWS would have a hydrogeological contact for future queries and work. For this reason, the GSI were keen to match GWSs with relatively local hydrogeologists wherever possible.

Consequential areas of work arising from this arrangement include the procurement of the consultants and maintaining a degree of standardisation of the workflow and reports for each GWS – roles that the GSI undertook.

PILOT STUDY

In 2011, the NFGWS and GSI initiated a pilot project to test the utility and feasibility of the project. Over the next year, preliminary source protection plans were developed for six GWSs in Counties Limerick and Cork.

GSI developed a report format that combined elements from both the GSI SPZ reports and the EPA ‘site folders’[4]. The aim was to present the information in an easily digestible way, which included tables of data, and a straight forward ‘sky to source’ description of the conceptual model i.e. how rainfall recharges the aquifer and how it flows through the aquifer to the GWS’s well or spring. The report was to be relatively short and to include a cross-section of the hydrogeologist’s conceptual model of the sky to source pathway (see Figure 1a-d for examples).

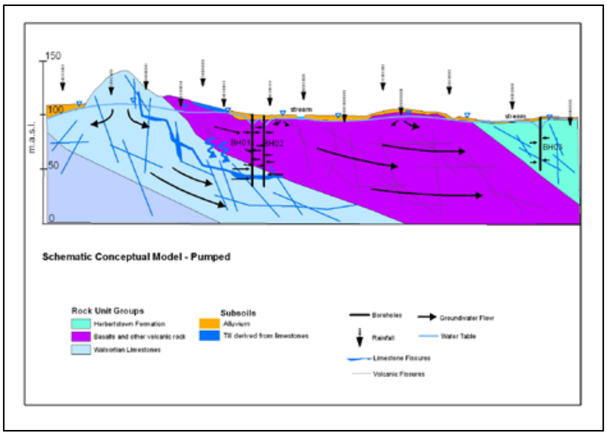

Figure 1a. Ballybricken Cross Section

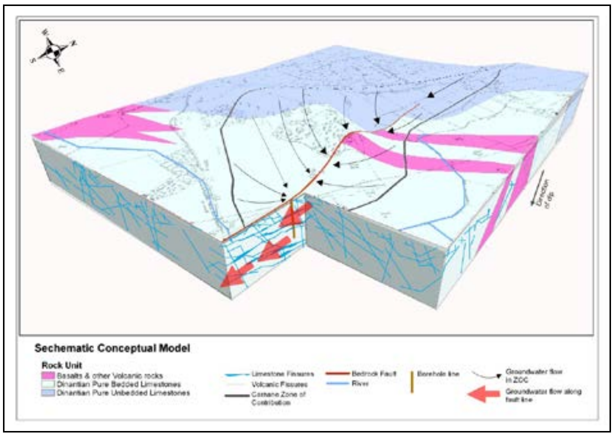

Figure 1b. Carnane Block Diagram

[3] The final number of predominantly groundwater supplied GWS (with greater than 50 people, or 15 domestic connections) has not been fully determined. It is currently assumed to be around 250 GWSs.

[4] The EPA’s national groundwater monitoring programme provides data to assist the classification of status of the groundwater bodies and informs subsequent programme of measures. However this implementation requires 'knowing where the water comes from' thus the EPA funded the delineation of ZOCs around all of their monitoring points. A proportion of these had full SPZ work undertaken. The remaining locations had all available information collated to produce a ‘site folder’ and to enable delineation of an approximate ZOC based on that information. The site folder includes all of the available information, much of which is in tabular format.

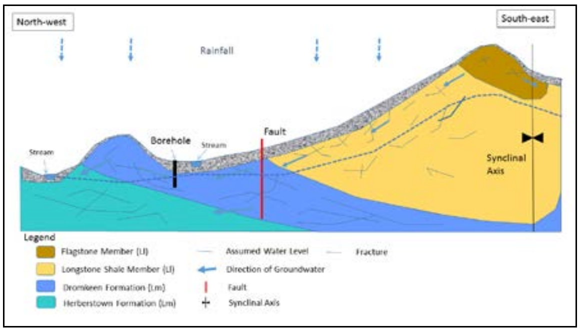

Figure 1c. Caherline Newtown Cross Section

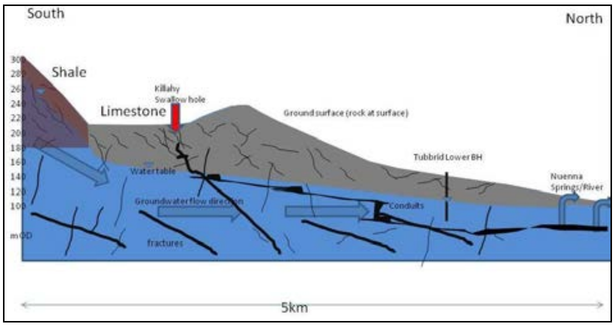

Figure 1d. Tubrid Lower Cross Section

Although predominantly desk-based, the programme of works included a site visit to gain an understanding of the source, the catchment and any existing or likely issues, and also the first opportunity for the GSI to engage with the GWS. Where appropriate, the GWS personnel were asked to help out with site measurement (e.g. water levels, collection of abstraction information if that is absent, locating of known karst features etc.) as this presented an opportunity to talk about groundwater and relevance of the data (refer to Images 1 and 2).

Image 1. Ballinguyroe/Tankardstown GWS and GSI personnel.

Image 2. Caherline Newtown GWS personnel dipping their borehole for water levels.

Engagement with the GWS continued throughout the project, mainly with following up on data requests. On completion of the draft reports, the information was presented to the GWS with the view to discussing the conceptual model, recommendations and any management implications. The GWSs were also asked to comment on the reports, level of engagement, and how both of these might be improved to increase their understanding and therefore usage of the information.

Throughout the entire period, the NFGWS were working alongside the GSI: they initially spoke to the GWSs about the project, arranged and participated in site visits, collected source information (populating source data sheets), reviewed the reporting templates and the draft pilot reports, and arranged the subsequent presentations and meetings.

Once four of the six pilot reports had been completed, two consultant hydrogeologists with previous SPZ experience were procured to complete the remaining reports. The consultants were then able to make additional comments and suggestions on the products and process.

In 2012, the pilot project was expanded into Phase II, which comprised four GWSs in Louth. The same consultant hydrogeologists undertook all stages of work, which enabled the GSI and NFGWS to ascertain the workload and time required, whilst keeping the affordability in mind. From Phase II, the workflow for each GWS was more clearly defined to include:

- Initial engagement between the NFGWS and GWSs;
- Populating a source data sheet (all known information) – this was initially done by the GWS with their NFGWS contact, and then followed up with the consultant;
- Site/catchment visit to gain an understanding of the site, catchment and any existing/potential issues. The sites were visited by the consultant, GSI and NFGWS;
- Discussion with the GWS personnel/other parties;
- Desk study examining all available data, including any hydrochemistry;
- Hydrogeological interpretation of all collated information to delineate a ZOC;
- Review of conceptual model and report by GSI;
- Presentation of report to GWS and discussion of any issues/recommendations.

Again, the underlying theme was to have the GWSs as an active partner in this process to be able to understand the information and inform their management decisions (Images 3 and 4).

Image 3. Site visit with the Seskin GWS, and hydrogeologist, Karen-Lee Ibbottson.

Image 4. Presentation by Coran Kelly to the Nuenna Catchment GWSs (Kilkenny).

THE CURRENT WORK PROGRAMME 2013 – 2018

Further to the success of the pilot phases, the GSI and NFGWS were keen to provide preliminary source protection plans for the remaining GWSs. A five-year programme was proposed, to start in 2013. In response to a strong case made by the NFGWS through the National Rural Water Services Committee, DECLG funding was awarded. The funding was facilitated through the Water Services Programme, which is administered by Local Authorities.

Funding is in the form of grant aid -: 85% of up to a maximum cost of €3,000 can be drawn down by each GWS towards the professional assessment and report. This means that the outlay for a GWS should be a maximum of €450. This fell short of the total costs estimated by the GSI and proposed by the NFGWS, however, it was still thought to be feasible to delineate a ZOC for most of the GWSs or identify where more in-depth studies would be required.

After the 2013 funding was secured, the GSI procured a panel of 10 hydrogeological consultants to undertake the proposed work for that year, which constituted protection plans for c.45 GWSs across five counties.

The programme was jointly managed by the GSI and NFGWS, with both organisations have an on-going, active role in the project. This is primarily due to the interest that both organisations have in this work and the need to maintain consistency in the assessments, ZOC delineations and report production. Furthermore, GSI and NFGWS are keen to support the consultants in order to maximise the use of their time input. So, for example, the NFGWS development officers are collating as much source information as possible in advance of site visits and the GSI are providing data layers, helping with site visits, advice and reviewing all reports.

CHALLENGES

There were a number of challenges with this work. The most apparent were the tight financial and time constraints, which the consultants and GSI endeavoured to work within. These on-going challenges have been further exacerbated by the initial delay in the approval of funding: the NFGWS and GSI planned to start the project at the beginning of 2013 but the funding delays resulted in the project starting mid-2013. Despite excellent progress, the knock-on effect was to push the first year's completion into 2014, which has impacted on the proposed 2014 schedule.

Additionally, there have been teething problems. These have revolved around managing a large number of GWSs (c.45) and consultants (numbering 10): the logistics of collecting data from the GWSs; providing base layers to the consultants; collecting water quality samples to fit with the timeframe of the reports; becoming familiar with the report format; scheduling report reviews to avoid bottlenecks; endeavouring to review and return reports as soon as possible.

Despite these issues, the continual work and patience of the consultants and input of the GSI and NFGWS has resulted in all but a handful of reports being drafted, issued and presented to the GWSs (see Images 3 and 4). Overall, the progress made in the first year has been very encouraging, which has paved the way for continued funding. Therefore, in the past few months, alongside the completion of the first year's reports and presentations, the NFGWS have advanced the data collection process for the Year 2 GWSs.

HOW SUCCESSFUL HAS THIS PROJECT BEEN SO FAR?

The production of reports and presentation of information are significant undertakings and, understandable, frequently the focus of projects such as these. However, for this project, the question that could still be asked is: Has it been successful?

With the full SPZ reports that are issued to the Local Authorities, part of the 'success' is that the SPZ maps are incorporated into their systems as a reference point to inform decisions. Beyond that, our experience is that additional use and further understanding of the reports it is mainly based on whether individuals within Local Authorities have a more active role or interest in groundwater drinking water supplies, and related areas. Given the wide remit and ever-declining resources of Local Authorities, we would have to think carefully about how this work could be used more effectively. Currently, we tend to focus our own limited resources on engaging with those who do have a more active role.

For the GWS project, one of the stated aims is to convey our understanding of the supply, so that the GWS can use this information themselves to minimise the risks of contamination or overuse, or work towards fixing any existing problems. Have we achieved this?

According to the work flow, we are able to tick off a) meeting the GWS, b) acquiring data and c) giving presentations. To determine the 'success', though, we must ask ourselves:

- Does the GWS understand the information?
- Can the GWS more effectively manage their drinking water/catchment area?
- Is there any follow up once the reports have been submitted and presentations given?

From the pilot study and 2013 schemes, the questions asked by the GWSs at the presentations would suggest that, generally, the information has been understood and that the presentations are an important investment of time to clarify anything further. The GWSs do seem to have a better understanding of why certain water quality/quantity issues arise and can now start to think of solutions for these problems. In a number of cases, the GWSs have been looking for a technical assessment, such as the preliminary source protection plans, to implement management strategies for existing issues.

Furthermore, some of the GWSs have either asked the consultant hydrogeologist or GSI for further information, which may lead to more work being undertaken, or have gone back to the NFGWS to investigate potential management strategies that are being used elsewhere, and/or to determine what funding may be available to implement these strategies. In terms of follow up, not only are the GWSs aware of the GSI as a resource, but they also have a consultant hydrogeologist who is familiar with their scheme, and as ever, they have the NFGWS.

Fundamentally, much of the success to date is due to the solid foundation that the NFGWS have built with the GWSs for us to work from. GSI and the consultants are fulfilling a technical role within an already well-functioning programme of community engagement. The project will hopefully run for the next four to five years. The challenges to date have already been discussed and there are likely to be more. However, it will be absolutely essential that GSI and the consultants do not get too distracted with the technical, financial and timeframe challenges and lose sight of the end goal: communicating information to the communities that are drinking the water and using the land. Ultimately it is these people who can change their practices to make a difference, with guidance from us and the NFGWS.

ACKNOWLEDGEMENTS

This paper is produced with permission from the Geological Survey of Ireland (GSI). The GSI would like to acknowledge the work of, and help from, the NFGWS, both in this project and in the production of this paper. We would also like to acknowledge the work and input from the consultants: Donal Crean, Sean Moran (O’Callaghan Moran & Associates); Peter Conroy, Jenny Deakin; Michael Gill, David Broderick (Hydro Environmental Services); Karen-Lee Ibbottson (WaterWise Environmental); Coran Kelly, John Dillon (Tobin); Richard Langford (Parkmore Environmental Services); Robbie Meehan; Orla O’Connell, Aisling Whelan, Maeve Rochford (IE Consulting); Evelyn Smyth (ARUP); Suzanne Tynan (Tynan Environmental).

GROUNDWATER AS A SOURCE OF PUBLIC WATER SUPPLY: TECHNICAL PROBLEMS WITH EXISTING WATER SUPPLY SOURCES AND SOLUTIONS SOUGHT

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ABSTRACT

This paper describes the problematic treatment processes of a number of public water supplies in karstic limestone. The raw water data available at the time of design was monthly for EPA compliance monitoring and it failed to identify the raw water characteristics in that spikes of colour and turbidity were not identified. We then decided to look at potential solutions that might solve our difficulties at low cost. We guessed and hoped that we would be lucky in drilling a borehole on our existing site at mid-Galway and that we might get a time difference between the spring source and the underground source. A time delay of up to 24 hours, if achieved would have allowed us to blend the waters to remove the spikes. Hope springs eternal. We asked Pamela for advice and she has been a partner in our attempts.

We then decided to ride our luck a bit further and drill a borehole at Leenane into what was identified as a major faultline from Lough Corrib to Killary Harbour. Pamela will elaborate our problems in this area due to cementation of the fault line. Connemara conglomerate sandstones are an unusual bedrock type to go looking for public supply scale groundwater abstraction yields. However, colour responses in the mountainous surface water source, in extreme rainfall events, occasionally creates problems for the upgraded water treatment plant. This necessitated evaluation of groundwater in a faulted bedrock zone. The project is on-going in terms of evaluation of long term yield from two of the most successful of six borehole drilled. It appears that groundwater does have the potential to alleviate some of the Local Authority's challenges in managing the efficacy of the WTP infrastructure.

INTRODUCTION

County Galway's water supply is like an Indian Vin da loo, many ingredients, spices and flavours in different sizes with different influences. No one knows where it comes from and no one know where it goes to. We in GCC are now achieving 99.6% compliance with the Drinking Water Regulations across the County and we will have any outstanding THM's sorted out in 2014 to gain 100% compliance. We in the County didn't have a major problem with viruses in previous years, however, we maximised the financial opportunity to ensure we could guarantee top class water.

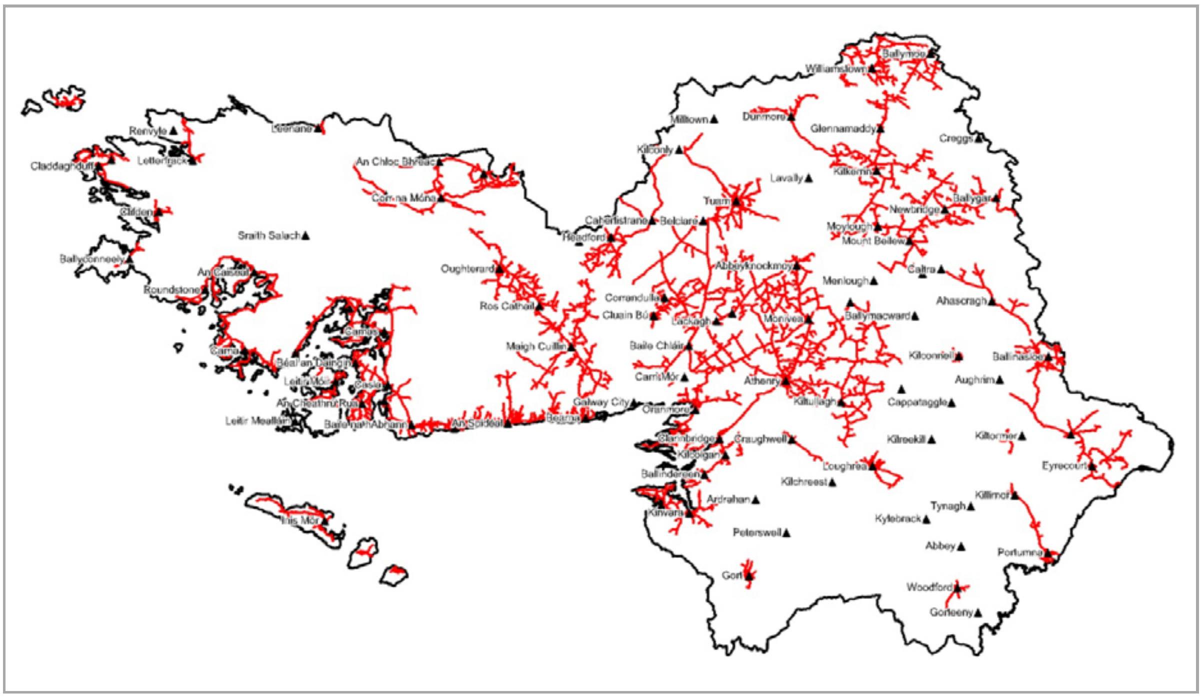

Table 1: County Galway showing public watermains across 34 water treatment plants.

- The area west of Lough Corrib gets its water from high level peat lakes by in large. The water has colour but the colour is relatively consistent. Therefore it is easy to treat. Chemical treatment using coagulation and flocculation achieves compliant water.
- We supply a large area immediately east of Lough Corrib from Lough Corrib. We have slight difficulties with this water as it is almost pristine and it is difficult to use coagulation and flocculation as it is too low in colour for the most of the time.
- The eastern periphery of the County around Ballinasloe and Portumna are supplied from surface waters of the Shannon and Suck rivers. The water quality is regularly consistent and easy to treat.
- The south of the County around Loughrea, Gort and Kinvara are complicated karstic sources and many studies indicate that they should be treated as surface waters. Accordingly we use coagulation and flocculation at these sources.
- We come to the middle eastern karstic area, the subject of this paper. We have sources at Ballymoe[spring: 562m^{3}per day], Williamstown [spring: 529m^{3}per day], Glenamaddy [spring: 318m^{3}per day], Dunmore [spring: 340m^3 per day], Kilkerrin-Moylough [spring], Mountbellew [spring: 1310m^3 per day], Ballygar[borehole: 356m^{3}per day] and mid-Galway [spring: 3,415m^{3}per day]. These spring sources provide high quality water for 95% of the time but in unpredictable periods of heavy rain we get colour and turbidity spikes. In mid-Galway, where we have a rain gauge we can identify a time delay of about 12 hours. The turbidity spike can rise from 0.3 to 4 in a period of 1 hour for a duration of 24-48hours and it can drop back just as quickly. We have installed Ozone, GAC, UV & Chlorination, but this plant cannot handle the spikes and shuts down in severe events. The GAC has to be replaced about once per annum and it can be shorter than this. Replacement costs are around €40,000. On top of this the UV lamps have a life of 8,000 hours while the servicing of compressors, Ozone and monitors raises the costs further. The use of coagulation & flocculation is not possible as we cannot predict which events cause a spike. Thus the dosage rate for chemicals would be haphazard and we might have dose level right by the time the spike decreases with corresponding carryover of Aluminium sulphate. The vulnerability of the drinking water source is created from a wide spectrum of possible pollutants from Agriculture, Septic tanks and Industry. Risk of contamination is always a consideration with an unknown recovery period. How long will it take to wash out, if at all? How much will it cost to tanker in potable

water? How will we supply farmers and their cattle in the interim period? Should we abandon karstic sources for drinking water usage?

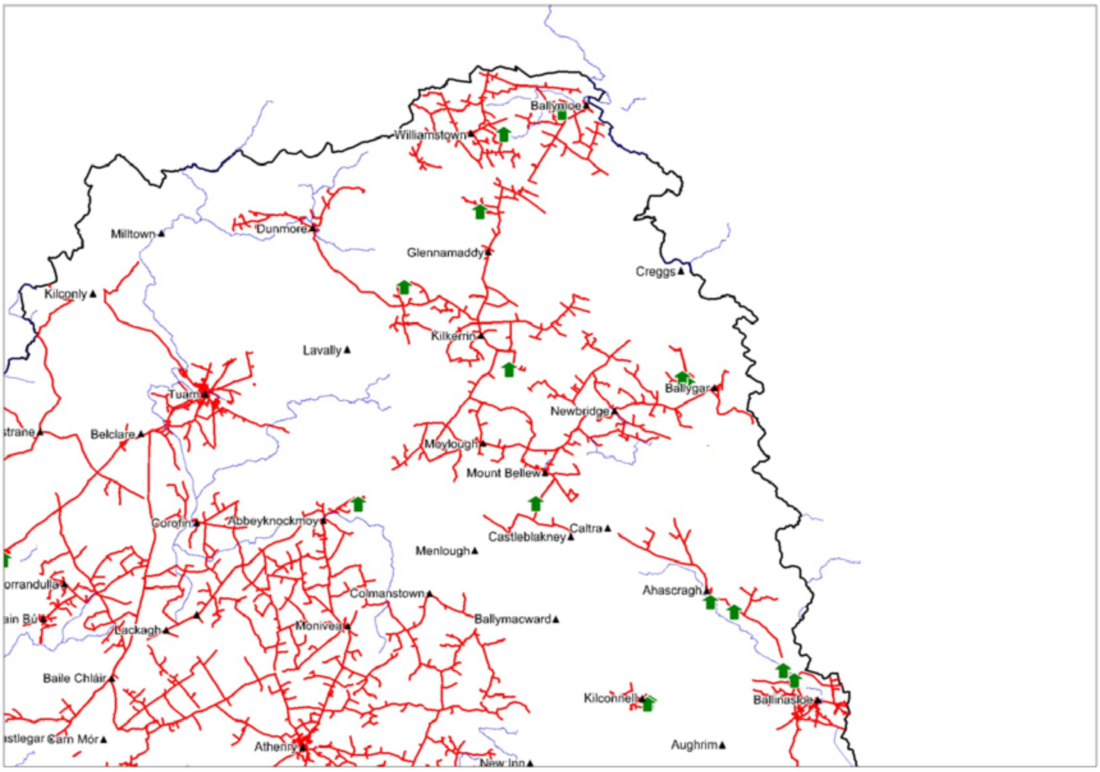

Table 2: Northeastern Karstic area showing watermains and treatment plants.

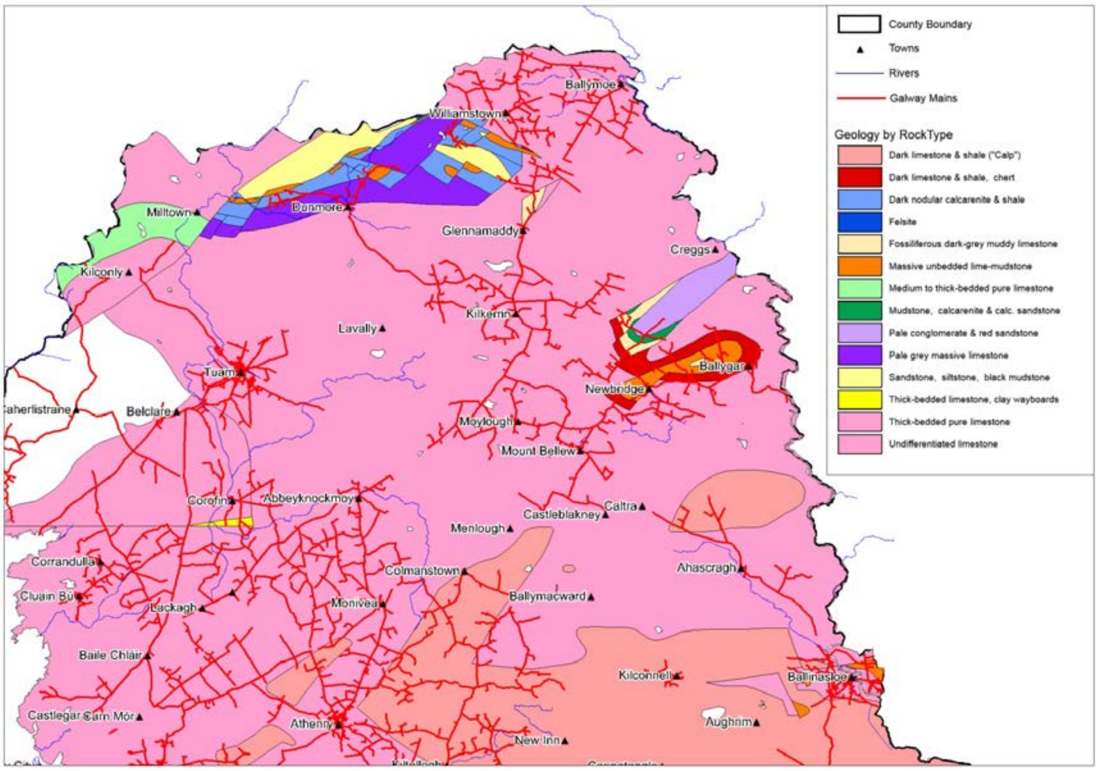

Table 3: Northeastern area showing geology of area

What did we do at the various sites?

Mountbellew Water Treatment System: We have Filters, PAC, UV, Chlorination: We identified issues with turbidity spikes and we engaged divers to go into the spring chamber and hoover out the silt. This was very beneficial. In order to identify the constituents we obtained an X Ray Electron Microscope analysis. This indicated the presence of zinc sulphide as the main component with significant amounts of Silicon. We installed a PAC system that is manually switched on where spikes occur.

Sample 33184 – Water Filter Deposit. The filter was examined and analysed. Tiny deposits of material were attached to the surface of the filter. Analysis of these deposits identified the composition as Carbon, Oxygen, Aluminium, Silicon, Sulphur, Potassium, Calcium, Iron and Zinc. The composition indicated that Zinc "Sulphide" was the main component. Silicon was also present in significant amounts.

Mid-Galway Water Treatment Plant: Treatment here involves Ozone, GAC, UV, Chlorination. We have turbidity spikes that are sufficient to shut the plant. We decided to drill an additional borehole and hope that we would get a time delay between the surface spike and the underground spike**.** The water quality sampling results from the borehole, have given us room for thought. We have noticed that colour and turbidity increase over time, while the result for colour and turbidity is higher in the plastic compared to the glass bottles.

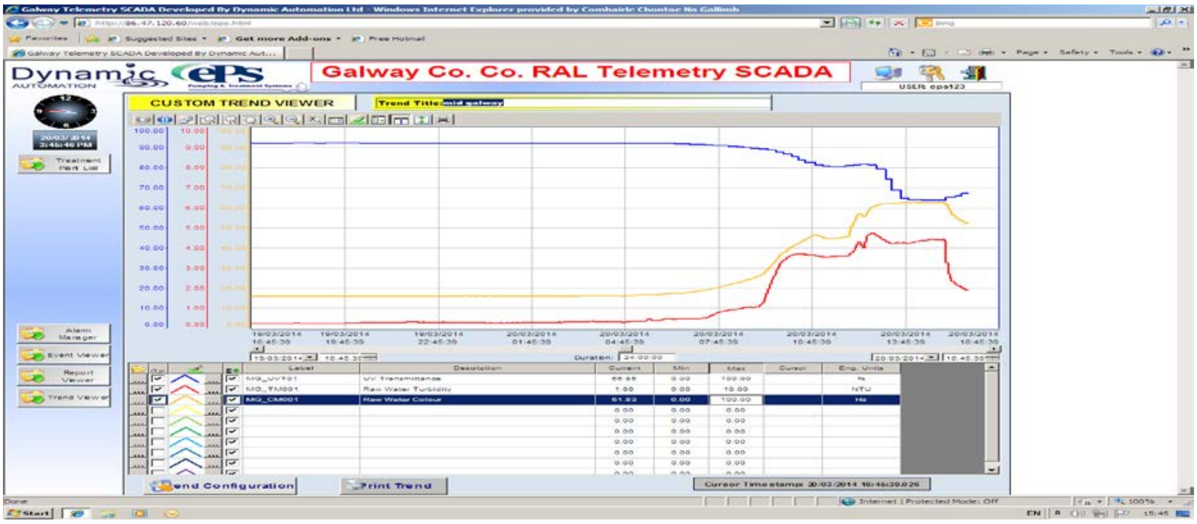

Table 4: Mid-Galway spike of Colour, turbidity and UVT ON THE 20/3/2014

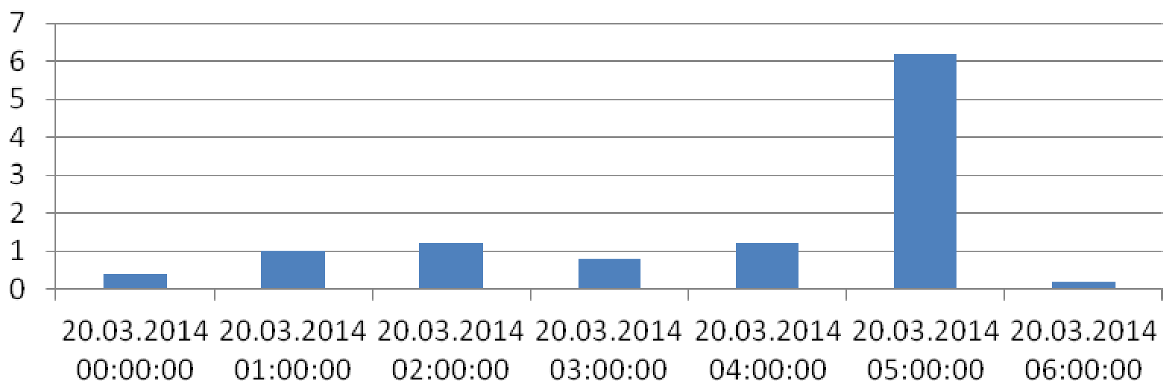

Figure 1 : Rainfall data on the 20/3/2014

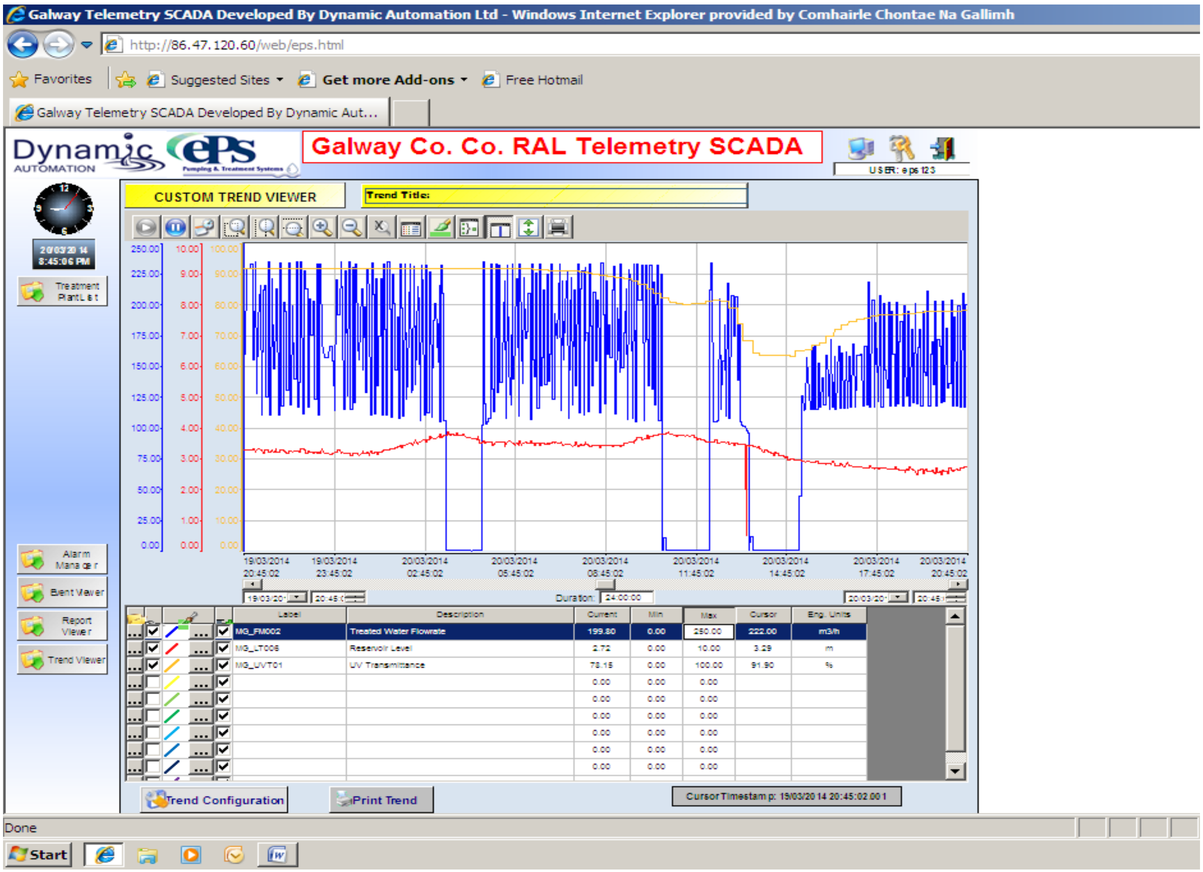

Table 5: Plant reaction to spikes on 20/3/2014.

Mean readings	Mid-Galway Borehole sample 12/3/14	Mid-Galway Borehole sample 28/1/14	Mid-Galway Borehole sample 19/9/13	Mid-Galway 41) 06-13
Aklkalinity		316	350	347
Ph		7	7.1	7.16
Turbidity	3.7	14.8	8.6	0.78
Chloride		33.5	24.5	18.6
Colour Hazen	4.7	88.2	9.8	17
Conductivity		668	665	731
Hardness		334	395	375
Iron		359	1484	86.9
Ortho-Phosphate		<0.01	<0.01	0.046
Temperature				11.6
DO%				69.9
Manganese		137	199	7.05
SS				6.68
TOC		1.2	<1	4.11
Sulphate		8.5	12.8	7.42

Table 6: Water quality comparison between existing source and new borehole.

Kilkerrin-Moylough Water Treatment Plant: Treatment system is Amazon filters, Chlorination. Our source protection report suggests Potential pollution from the Kilkerrin Lehinch Turlough. However Rainfall, Turbidity and Colour printout indicates no reaction to rainfall events.

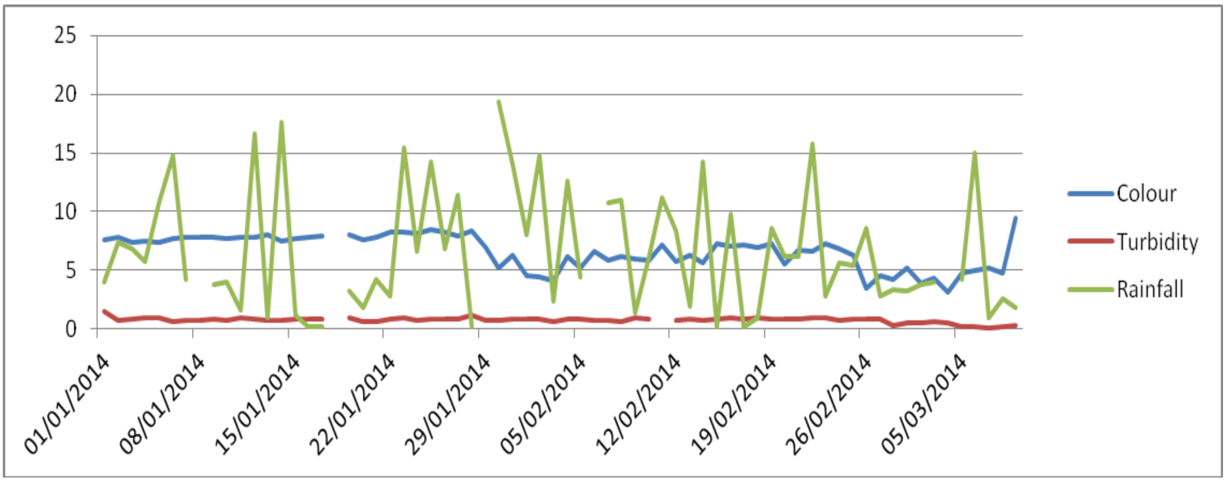

Table 7: Kilkerrin-Moylough analysis of Colour, Turbidity versus rainfall.

The remainder of our treatment plants at **Ballygar, Ballymoe, Glenamaddy, Dunmore-Glenamaddy and Williamstown** are treated with Ozone, GAC, UV, Chlorination. In a number of cases we have engaged divers to hoover out the springs and this has been very satisfactory and has reduced our problems.

I have been looking at the SCADA results over a period for UVT, Colour and Turbidity and noted that some of the treatment plants were similar over a period. This led me to believe that the concept of source protection zones in karstic sources might not be totally reliable. I include a SCADA screenshot for Turbidity for the past 4 months for Mid-Galway, Mountbellew and Dunmore-Glenamaddy. This indicates a correlation of Turbidity spikes. One must question if they are all linked together. Could fault lines be transporting the water over a bigger area?

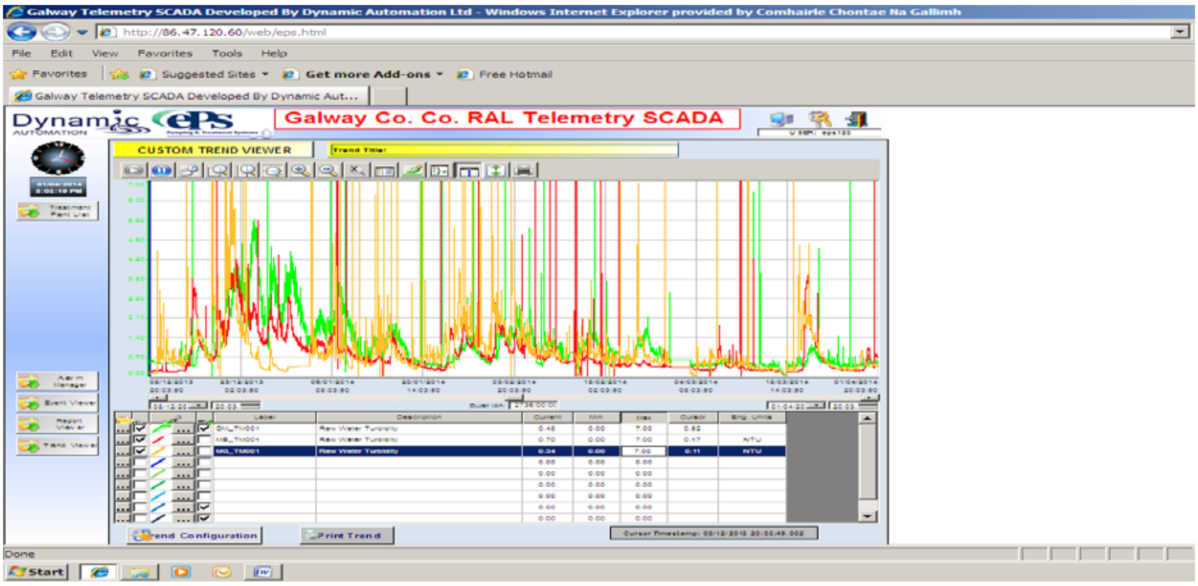

Table 8: Mid-Galway, Mountbellew and Dunmore-Glenamaddy turbidity spikes over the past 4 months.

I then tried to compare colour spikes, but these did not show any correlation. The source in Mid-Galway is from a stream approximately 350 metres downstream of the spring sources and it is possible that colour is impacted by the influence of sediment in the channel.

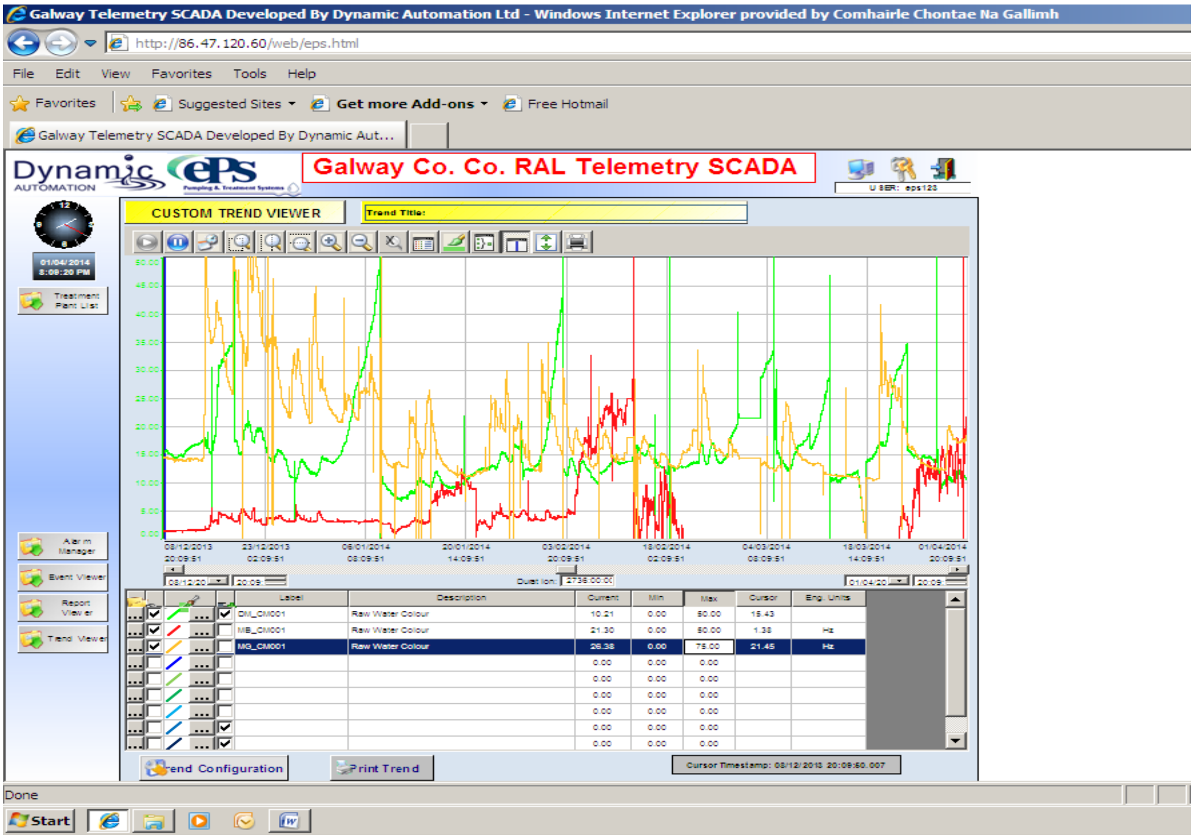

Table 9: Mid-Galway, Mountbellew and Dunmore-Glenamaddy colour spikes over the past 4 months.

Finally, I checked UVT and again compared the results.

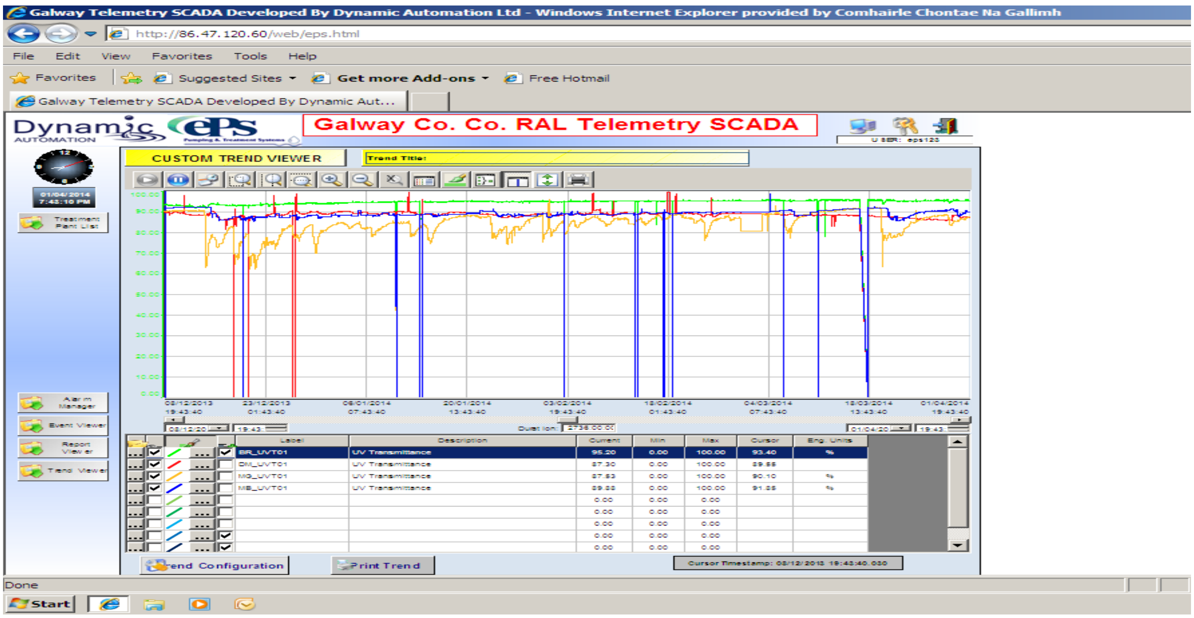

Table 10: Mid-Galway, Mountbellew and Dunmore-Glenamaddy UVT spikes over the past 4 months.

As my interest was raised, I decided to compare the long term compliance sampling results over a number of years. The attached table indicates the number of samples and the period of sampling for each scheme (Number of samples) period of years.

Mean readings	**Mid-Galway (41) 06-13**	**Mountbellew (70) 95-13**	**Glenamaddy (30) 07-13**	**Ballymoe (23) 08-1**	**Williamstown (33) 07-14**	**Dunmore-Glenamaddy (74) 95-13**	**Ballygar (71) 95-13**	**Kilk-Moylough**
Aklkalinity	347	348	347	346	275	317	351	351
Ph	7.16	7.11	7.05		7.1	7.09	7.1	7.06
Turbidity	0.78	0.896	0.88		1.05	1.09	1.58	0.575
Chloride	18.6	17.4	14.9	15	15.3	19.4	18.7	16.8
Colour Hazen	17	6.98	21.4	9.09	27.9	11.8	9.72	9.05
Conductivity	731	714	706	704	574	667	724	720
Hardness	375	376	371	374	300	335	366	372
Iron	86.9	70.8	171	51.8	196	163	161	64.9
Ortho-Phosphate	0.046	0.032	0.032	0.036	0.035	0.029	0.026	0.035
Temperature	11.6	10.5	10.3	10.5	10	10.5	10.7	10.4
DO%	69.9	47.1	54.3	41.7	46.3	40	25.4	28.6
Manganese	7.05	4.63	19.6	2.48	23.7	18.6	65.5	16.5
SS	6.68	7.86	7.87	7.83	7.9	8.42	8.52	7.87
TOC	4.11	3.14			4.69	7.93	6.29	4.15
Sulphate	7.42	7.76	7.42	6.99	6.08	8.43	13.4	6.38

Mean readings	**Mid-Galway (41) 06-13**	**Mountbellew (70) 95-13**	**Glenamaddy (30) 07-13**	**Ballymoe (23) 08-13**	**Williamstown (33) 07-14**	**Dunmore-Glenamaddy (74) 95-13**	**Ballygar (71) 95-13**	**Kilk-Moylough**
Alkalinity	347	348	347	346	275	317	351	351
Conductivity	731	714	706	704	574	667	724	720
Hardness	375	376	371	374	300	335	366	372
Iron	86.9	70.8	171	51.8	196	163	161	64.9
DO%	69.9	47.1	54.3	41.7	46.3	40	25.4	28.6

I narrowed down this list down to schemes with similar results:

Mean readings	Mid-Galway (41) 06-13	Mountbellew (70) 95-13	Glenamaddy (30) 07-13	Ballymoe (23) 08-13	Kilk-Moylough
Alkalinity	347	348	347	346	351
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Ph	7.16	7.11	7.05		7.1	7.09	7.1	7.06
Turbidity	0.78	0.896	0.88		1.05	1.09	1.58	0.575
Chloride	18.6	17.4	14.9	15	15.3	19.4	18.7	16.8
Colour Hazen	17	6.98	21.4	9.09	27.9	11.8	9.72	9.05
Ortho-Phosphate	0.046	0.032	0.032	0.036	0.035	0.029	0.026	0.035
Temperature	11.6	10.5	10.3	10.5	10	10.5	10.7	10.4
Manganese	7.05	4.63	19.6	2.48	23.7	18.6	65.5	16.5
SS	6.68	7.86	7.87	7.83	7.9	8.42	8.52	7.87
TOC	4.11	3.14			4.69	7.93	6.29	4.15
Sulphate	7.42	7.76	7.42	6.99	6.08	8.43	13.4	6.38

Again I narrowed this down to schemes with similar results:

Mean readings	Mid-Galway	Mountbellew	Ballymoe	Kilk-Moylough
Ph	7.16	7.11		7.06
Turbidity	0.78	0.896		0.575
Chloride	18.6	17.4	15	16.8
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Ortho-Phosphate	0.046	0.032	0.036	0.035
Temperature	11.6	10.5	10.5	10.4
Manganese	7.05	4.63	2.48	16.5
SS	6.68	7.86	7.83	7.87
TOC	4.11	3.14		4.15
Sulphate	7.42	7.76	6.99	6.38

Long Term solution to karstic Limestone sources

Accept that the sources are vulnerable. Accept that treatment is variable and that out of compliance results can occur.

Options:

- Provide bankside storage to ride out spike situations, if reasonably practicable.
- Alternatively extend surface water systems to replace karstic sources if economies of scale allow. Water Safety Plans dictate that we have to remove risk and have control on activities. The black art of water divining has to be consigned to the middle ages and we have to revert to pure science. Removal of risk is impossible with groundwater sources.
- Consider extending raw water from Lough Corrib to service all karstic area springs/boreholes west of the Shannon from Mayo to Clare along the proposed M17/M18. Install modular treatment plants for a well defined water treatment at points of demand.

GROUNDWATER AS A SOURCE OF PUBLIC WATER SUPPLY: TECHNICAL PROBLEMS WITH EXISTING SOURCES AND DRILLING SOLUTIONS SOUGHT

Pamela Bartley
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ABSTRACT

This paper describes the process of a water supply engineer/ hydrogeologist being tasked to solve a problematic water supply. One might ask when are problems identified? I offer that it is usually when a water treatment plant has been upgraded and the water supply is causing problems for the efficient and economical operation of the new plant. Why is a hydrogeologist called in? Because while the OPERATOR in a DBO scheme is responsible for the operation of the water treatment plant to deliver potable water, the WATER SOURCE is the responsibility of the OWNER i.e. the GWS. So, when the source is considered unusable or difficult to treat, an engineer/hydrogeologist with water supply borehole construction is called in. I have had the fortune of conducting detailed assessments and remediation drilling at a number of water supply sites in County Galway. In this paper I shall describe a number of experiences that have arisen when I have been called in when problems arise at either a Group Water Scheme's groundwater borehole, whose WTP has been upgraded under the DBO2 programme, or a County Council managed Public Supply borehole. Usually it is at the commissioning phase of the new and expensive water treatment plant: increased monitoring facilities give rise to identification of a problem that was probably always there but never highlighted before. I shall present methods and monies spent on providing solutions.

I will present some drilling & monitoring experiences from some sites at the following locations:

1. *East & South Galway Karst Sites: completions to EPA Guidance Standards[5].*
2. *Connemara Drilling for Public Water Supply: Leenane drilling and pump testing.*
3. *North Tipperary Limestone: Private Commercial Development Water Supply Borehole: starting from scratch, doing it according to Guidance[1] and interesting pump test opportunities.*

Many of the initial problems encountered relate to increasing abstraction rates, as a result of amalgamating GWSs, which caused stress manifestation on the existing borehole. The demonstrator of the stress was either excessive drawdown or intake to the borehole of inert constituents, which caused turbidity problems outside the design influent parameters of the upgraded water treatment plant. I shall discuss drilling experiences at new groundwater supply borehole drilling sites, recommendations for completion with casings and cement grouting (which should ALWAYS be based on trial well experiences at the site) and subsequent field tests conducted to test problematic responses (i.e. turbidity and water level responses) lessons learned from the commissioning and operation of water treatment plants.

Overall, lessons learned are as follows:

(A) One cannot change what mother nature periodically throws in a karst groundwater environment but a continuously slow and steady abstraction rate shall assist in solving most problems;

[5] EPA (2013) EPA DRINKING WATER ADVICE NOTE Advice Note No. 14: Borehole Construction and Wellhead Protection.

(B) Fixing leaks in distribution systems and installing meters at each connection can greatly assist;

(C) Hydrogeologists must spend a considerable effort to insist on doing it to Guidelines – this is usually far more expensive than historical methods of borehole drilling and

TECHNICAL WORKSHOP

WELL GROUTING

Peter Conroy, EurGeol, PGeo., M.Sc., B.Sc., DipGeol.
Hydrogeologist

ABSTRACT

Grouting of the borehole annulus above the producing section of a well is recognised as best practice for borehole construction in two recent Irish guidance documents, i.e. Guidelines on Water Well Construction (IGI, 2007) and Advice Note No. 14: Borehole Construction and Wellhead Protection (EPA, 2013). This technical workshop considers the practical aspects of implementing the grouting procedures recommended by the guidance documents.

The workshop looks at the considerations involved in designing the grouting stage:

- *Health and Safety;*
- *Identifying the aim of the grouting step;*
- *Calculating Grout Quantities;*
- *Specifying Grout Composition;*
- *Casing Considerations; and,*
- *Grouting Equipment.*

The workshop then looks at potential difficulties that may be encountered on site and ways to overcome them:

- *Cement and Weather Conditions*
- *Equipment Blockages*
- *Grouting in under-reamed circumstances*
- *Grout loss to surrounding formation*
- *Monitoring the Grout Level*
- *Assessing completion of the grouting step*

The Bailer Test

Paul Wilson
Geological Survey of Northern Ireland/British Geological Survey

ABSTRACT

The bailer test is a quick, simple and consistent method of assessing the performance of a borehole. It can be easily applied by drillers, hydrogeologists, engineers or borehole owners. The test is not a substitute for a pumping test but could be deployed on boreholes installations where pumping tests are rarely performed. They provide an accurate assessment of whether a borehole will meet the demand required of it, for example low demand domestic or farm supplies. The bailer test can then inform decisions such as whether to install pump chamber casing in a pilot hole, what specification of pump to install and at what level, and what the pumping regime should be to ensure efficient use of a borehole. The bailer test can also be used to inform the design of a pumping test schedule.

The test requires two steel buckets, a length of rope, a stopwatch and a water level dipper. 20-50 bails are removed from a borehole during a ten minute period. Recovering water levels are measured after the bailing period. The maximum drawdown and time for 50% and 75% recovery are determined and these can be compared against simplified tables to assess if the borehole will perform as required. Including preparation time, a bailer test can be carried out in 30 – 60 minutes and does not require the operator to have a high degree of technical understanding.

INTRODUCTION

MacDonald et. al (2008) presented a test method for assessing the performance of boreholes drilled in developing countries for rural village supplies. One of the difficulties in such cases is practically performing a pumping test to determine if the borehole can sustain the demand a hand pump will place on it. Installing a hand pump in a borehole that cannot meet the demand will likely result in functionality problems and could ultimately lead to the abandonment of the borehole. To avoid this, MacDonald et. al (2008) proposed the bailer test method as a simple, robust, quick and consistent method for assessing if a hand pump should be installed in a newly drilled borehole.

Although the bailer test was designed for use in developing countries, it could have valuable application potential in Ireland. It is not common for a pumping test to be performed on a borehole constructed for domestic or farm supply. The appropriate time required and the equipment necessary to perform a proper pumping test can make it difficult to justify the value that a pumping test would provide in low demand cases. However, decisions such as pump chamber casing design, submersible pump specification and installation depth, and pumping regime design to improve efficiency all require a good appreciation of the borehole performance.

METHOD

To perform the bailer test, the following equipment is required. 1.Two steel bailer buckets 2. 20 m of strong rope 3. Gloves 4. Stopwatch 5. Water Level Dipper 6. Clipboard and results table

A bailer bucket is made of a length of steel pipe, of suitable diameter to fit comfortably inside the borehole casing. Typically a pipe of outside diameter 75 – 100 mm (3 -4 inches) works in most cases. The bucket is about one metre long. A bucket longer than one metre will be difficult to handle and will float inside the borehole. Shorter buckets can be made but this will reduce the volume of the bucket and therefore the effective bailing rate. Steel stockists may have an off-cut length of pipe that would be suitable and the off-cut length available can be what controls the length of the two buckets. Both buckets must have the same dimensions. A steel plate is cut and welded to one end of the bucket. The plate and weld are ground to give a smooth edge to prevent damaging the borehole casing. The bucket is tested to ensure it is watertight and re-welded if a leak is found. A steel bar is bent and welded to the inside of the bucket. This is where the rope should be attached. (See **Fig. 1**)

Fig. 1 – Steel Bailer bucket

Each bailer bucket is tied to an end of one length of rope as seen in **Fig. 2**.

Fig. 2 – Preparing to commence the bailer test

The bailer test procedure is as follows. The test can be performed by two people but this could prove taxing. Ideally two teams of two people should do the bailing and one separate person giving instructions:

1. Measure the rest water-level.
2. Lower bailer A into the borehole. Allow the bailer to free-fall down the borehole. This gives it momentum to go below the water-level and fill up with water.
3. When bailer A is full of water, it is withdrawn from borehole as quickly as possible. When the withdrawal begins, the stopwatch should be started.
4. When bailer A is clear of the borehole, bailer B is inserted. Meanwhile bailer A is emptied.
5. 40 bails in 10 minutes is a reasonable number to aim to achieve. This requires a bail every 15 seconds. One person calls every 15 seconds and the bailer inside the borehole should be withdrawn. At the start of the test, this may mean waiting between bails, but near the end of the test as the distance the bailer needs to needs to be withdrawn increases it may be difficult to keep up with this pace.
6. Bailer A does bail number 39. Once bailer A is emptied, pick up the water-level dipper and insert it quickly in to the borehole after bailer B is clear of the borehole on bail number 40.
7. The first water-level measurement is taken 30 seconds after 10 minutes. Water level measurements are taken every 30 seconds for the first five minutes, every minute after that up to 15 minutes and every 5 minutes after that.
8. Water-level measurements are taken until at least 75% recovery is achieved (75% of the first water-level measurement minus the rest water-level)

ANALYSIS

MacDonald et. al (2008) aimed to make the analysis of the bailer test results straightforward but at the same time robust and reliable. They found that large-diameter well analysis using BGSPT software (Barker and MacDonald, 2000) based on the method of Papdopulos and Cooper (1967) was the most promising technique for analysing bailer test data. The data required were simplified to the drawdown at the end of pumping, the time for 50% and 75% recovery, borehole radius, pumping rate and length of pumping to estimate transmissivity.

Validation of the data analysis was also performed. The analysis assumes constant rate pumping but the bailer test involves removal of bails for a period of time, which will likely reduce with time. By applying various realistic and boundary scenarios to the analysis, the test was found to be relatively insensitive to moderate changes in pumping rate. A 65% decline in pumping rate results in a 5% error of transmissivity.

In an effort to simplify the analysis MacDonald et. al (2008) used criteria for typical Sub-Saharan African hand-pumped boreholes. **Fig. 2** shows the relationship between transmissivity and storage coefficient for a 125 mm diameter borehole being pumped rate of 0.145 l/s for 12 hours per day for 6 months, with a maximum allowable drawdown of 15 m.

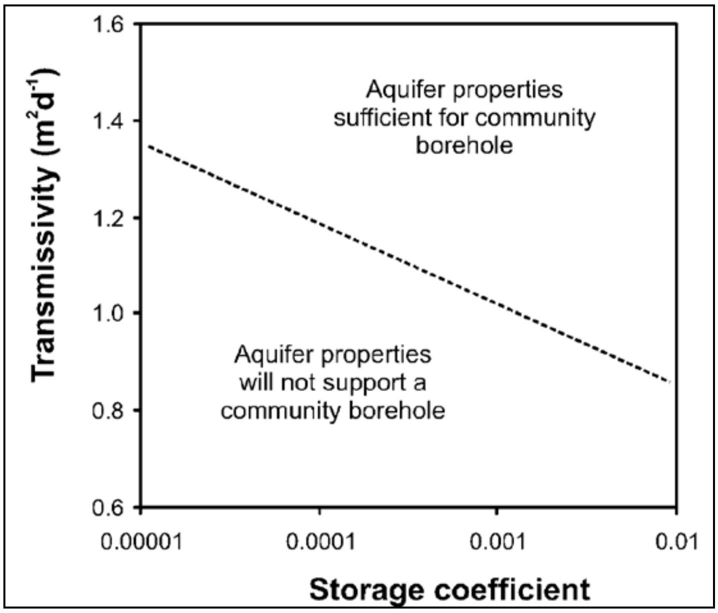

Fig. 2 – Aquifer properties required to sustain a community borehole for 250 people. Pumping rate of 0.145 l/s, 12 hours per day for 6 months.

The method was applied in an area of Nigeria at 15 boreholes. Five-hour constant-rate pumping tests and the bailer test were performed on each borehole. The Theis Recovery method (Kruseman and de Ridder, 1990) was used to analyse the constant-rate test results.

Fig. 3 shows a comparison between the transmissivity derived from the constant-rate test and the bailer test.

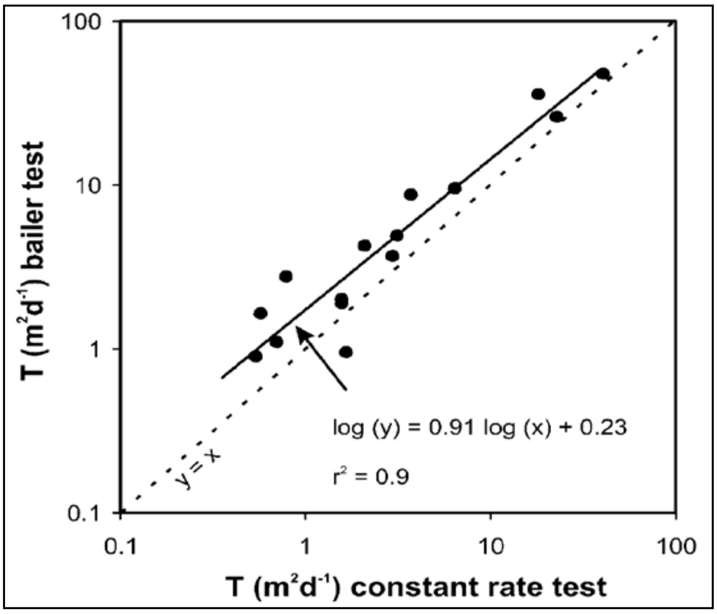

Fig. 3 – Comparison between transmissivity calculated from a bailer test and a longer constant-rate test for a series of borehole in Nigeria

The coefficient of determination r^2 of 0.9 shows a good fit and demonstrates that the bailer test gave comparable results from that acquired by the constant-rate test.

Further simplification was then carried out to address the basic question "can this borehole sustain the demand that will be placed on it?" Look-up **Table 1** was produced for the common case of a hand

pumped borehole. By knowing the diameter of a borehole, the number of bails removed during a 10 minute bailer test, and by determining from the bailer test results the maximum drawdown and the time for 50% and 75% recovery, **Table 1** can be used the determine if a borehole will be successful at sustaining a flow rate of 0.145 l/s for a 12 hour period.
If the maximum drawdown is less than the value in the table and the times for 50% and 75% recovery are less than the values in the table, then the borehole is likely to be successful.
If there is suitable interest, alternative simplified tables could be produced for a range of borehole demands such as for domestic, farm and small industry purposes.

Diameter of the borehole	Pumping rate in litres per minute (Number of standard bails)	7 (16)	11 (24)	14 (32)	18 (40)	21 (48)
100 mm	Max drawdown (m)	3.5	5.3	7.1	8.8	11
	Time for half recovery (mins)	6	6	6	6	6
	Time for three quarters recovery (mins)	14	14	14	14	14
125 mm	Max drawdown (m)	2.9	4.3	5.7	7.1	8.5
	Time for half recovery (mins)	9	9	9	9	9
	Time for three quarters recovery (mins)	21	21	21	21	21
150 mm	Max drawdown (m)	2.3	3.4	4.6	5.7	6.9
	Time for half recovery (mins)	12	12	12	12	12
	Time for three quarters recovery (mins)	28	28	28	28	28
175 mm	Max drawdown (m)	1.5	2.3	3.1	3.8	4.6
	Time for half recovery (mins)	19	19	19	19	19
	Time for three quarters recovery (mins)	46	46	46	46	46

Table 1 – Guidelines for success of rural water-supply borehole using the 10 minute bailer test.

APPLICATIONS

Despite being produced for use in developing countries, the bailer test has practical applications in developed countries including Ireland.

WATER-WELL DRILLERS

Performing a bailer test at various stages of the construction of a borehole could provide valuable information that would enable a driller or supervising hydrogeologist to make better decisions when completing a borehole installation. Air lifting during drilling to estimate the groundwater inflow rate can be problematic since the positive air pressure inside the borehole reduces the groundwater inflow rate (MacDonald et. al, 2005). This can lead to an underestimation of the rate that water can be pumped from a borehole.

Performing the bailer test on a recently drilled borehole provides a more accurate assessment as to whether the demand required will be met. This can inform decisions on how to complete the borehole and what pumping regime to operate.

The bailer test requires 10 minutes of intense activity followed by periodic water level measurements. This could be performed just after a drilling rig is moved off a borehole whilst equipment is being packed away and the site is being restored. As an added bonus, the bailer test will produce water that can be used to wash down equipment and the site. Performing a bailer test should not delay a driller from moving on to a new site.

HYDROGEOLOGISTS

It is common for hydrogeologists to be employed to do a performance assessment on a borehole for which no or limited records are available. Often the only data available may be anecdotal. It is therefore difficult to design and plan a pumping test without having an impression of the transmissivity of the borehole.

During an initial site visit, as well as plumbing the base of a borehole and measuring the rest water-level, a bailer test could be performed to quickly achieve an impression of the borehole performance. The rate of drawdown at a low pumping flow rate could be rapid or minimal, particularly in low productivity fractured bedrock. If the rate of drawdown is rapid, performing a proper pumping test may be difficult and of limited value. In such cases, deploying all of the equipment required to perform a pumping test would be a waste of resources. If the rate of drawdown is minimal and a low flow rate is selected, the value of such a test would be limited since the aquifer may not have been adequately stressed for the results to be analysed.

Performing the bailer test provides the hydrogeologist with the confidence to determine what type of pump to specify, what depth to install it at, the length of rising main and importantly the flow rate to start the pumping test at to adequately stress the aquifer but not induce rapid drawdown that would cause the pump to dry-run. It will even help work out what length of dipper is required.

CONCLUSION

The bailer test is a robust, cheap, quick and reliable method for assessing the performance of a borehole. It is not a substitute for constant-rate pumping tests, but could be easily applied in cases of low demand and low productivity boreholes where a pumping test would normally not be applied.

A bailer test performed during or at the end of a borehole construction will provide valuable information that will enable drillers and hydrogeologists to make more informed decisions.

The bailer test can be performed in less than one hour and requires cheap, robust and simple equipment that can be sourced or manufactured locally and will not take up lots of space. Drillers can perform a bailer test after the rig has been moved off a recently drilled borehole and would not be delayed from moving on to a new site. Hydrogeologists can use the bailer test as part of their site reconnaissance to help inform the planning of a pumping test.

If the bailer test was applied consistently across Ireland on newly drilled boreholes, by collecting the basic data the bailer tests produce, improvements could be made to regional aquifer properties that can in turn help improve borehole prognosis by both drillers and hydrogeologists alike.

REFERENCES

Barker J.A., Macdonald D.M.J., (2000) ‘A manual for BGSPT: programs to simulate and analyse pumping tests in large-diameter wells.’ British Geological Survey Technical Report WC/00/17. Programme available online from: http://www.bgs.ac.uk/bgspt/; accessed January 2008.

Kruseman G.P., de Ridder N.A., (1990) ‘Analysis and evaluation of pumping test data.’ International ILRI Publication 47, Institute for Land Reclamation and Improvement, The Netherlands.

MacDonald A.M., Barker J.A., Davies J. (2008) ‘The bailer test: a simple effective pumping test for assessing borehole success.’ Hydrogeology Journal. 16: p. 1065-1075.

MacDonald A.M., Davies J., Calow R.C., Chilton P.J., (2005) ‘Developing groundwater: a guide to rural water supply.’ ITDG Publishing, Rugby, UK, p. 358

Papadopulos I.S., Cooper H.H., (1967) ‘Drawdown in a well of large diameter.’ Water Resources Research 3:241-244

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