INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS (IRISH GROUP)

presents

GROUNDWATER QUALITY: CURRENT ISSUES & CONCERNS

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FOREWORD

Groundwater Quality is an important asset to all sectors of Environmental Protection – drinking water, habitats, farming, due diligence, government, and the EU are examples. Many differing professions and sciences work in groundwater quality – laboratories, consultants, regulators, students, earth science agencies etc. There are old and new ways of treating contaminated groundwater – industry and their consultants are working more and more on passive monitored cleanup programmes such as Natural Attenuation and Permeable Reactive Barriers. Groundwater in Ireland is important resource and accounts for up to 15% of total drinking water supplied by local authorities. Groundwater Quality is now a key focus for national governments, regulators, local authorities and earth science agencies across Europe – The Water Framework Directive (2000/60/EEC) has made sure of this.

The objective of this seminar is to increase the awareness and understanding of Groundwater Quality – its issues and concerns. The seminar this year draws on thirteen papers. The papers contained within are comprehensive and deal with all aspects mentioned above. The Irish Group of the IAH welcome such high standards achieved this year and wish to express their gratitude to all the authors.

Malcolm Doak
Inspector
Hydrogeologist
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April 2002

1.

Water Framework Directive – An Overview of Groundwater Aspects

Donal Daly, Geological Survey of Ireland and WFD Groundwater Working Group.

WATER FRAMEWORK DIRECTIVE
AN OVERVIEW OF GROUNDWATER ASPECTS

Donal Daly

Geological Survey of Ireland and WFD Groundwater Working Group

ABSTRACT

The future is bright for groundwater in Ireland. The Water Framework Directive (WFD) establishes a requirement for river basin planning based on the maintenance, improvement, protection and sustainable use of the Community's water. Groundwater is at the heart of this Directive. It is seen as an integral element in the understanding and protection of our water resources and the ecosystems dependent on water. The WFD provides an opportunity and a challenge to the groundwater community. By 2005, groundwater throughout the country will be allocated to groundwater bodies, which are the groundwater management units of the WFD. The hydrogeology of each body will be evaluated and described to a level appropriate to the importance of the groundwater as a source of drinking water, and for the needs of surface water and terrestrial ecosystem, in combination with the threats posed by human activities. By 2007, an upgraded groundwater monitoring programme, both of water levels and quality, will be in place. River Basin Management Plans, which will integrate groundwater with the other parts of the hydrological cycle for the first time in Ireland, will be published in 2009. Measures designed to achieve the objectives of the Directive will be drawn up and ultimately our groundwaters must achieve 'good status' by 2015. But achieving these requirements and deadlines will not be easy: the timescale is short; a multi-disciplinary and multi-organisational approach is essential; seeing groundwater in terms of ecologically oriented objectives will be new to many hydrogeologists; and the precise technical requirements are not contained in the Directive. However, the WFD provides a vision and opportunity to inspire and sustain us for the future.

INTRODUCTION

WATER FRAMEWORK DIRECTIVE

The Water Framework Directive (WFD) (2000/60/EEC) is probably the most comprehensive piece of EC water legislation to date. It establishes a strategic framework for managing the water environment and sets out a common approach to protecting and setting environmental objectives for all groundwaters and surface waters within the European Community.

At the heart of the Directive is the requirement to produce a strategic management plan for each river basin, setting out how the objectives are to be achieved. The plans are based on the following:
♦ subdivision of the river basin into groundwater and surface water bodies;
♦ characterisation and risk assessment of the water bodies, including a description and an evaluation of the hydrogeology, a detailed analysis of the pressures on the water bodies within the river basin and an assessment of the impact of the pressures;
♦ monitoring, evaluation and presentation of the quantitative and qualitative status of groundwater and surface water bodies.

A comprehensive programme of measures can then be drawn up, tailored to the specific circumstances in each river basin, and in particular to target improvements and monitoring effort on those water bodies most **at risk** of failing to meet their environmental objectives. Characterisation and risk assessment must be completed by the end of 2004, which is a short time scale. The main emphasis subsequently from a hydrogeological perspective, is on monitoring. However, the tasks should be seen as following an iterative planning cycle (see Figure 1), and therefore improvement of the characterisation and risk assessment will continue after 2004.

River Basin Planning Cycle   *Main Tasks*
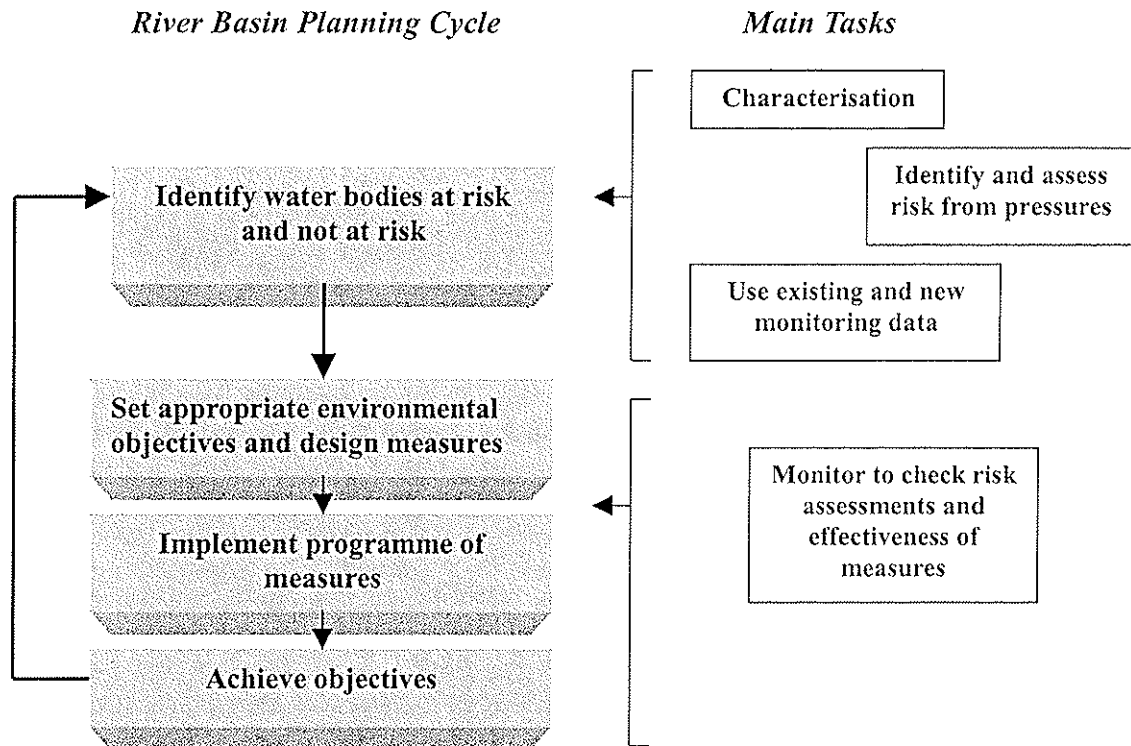
Figure 1   River basin planning cycle and main tasks

The WFD introduces new approaches to considering groundwater:

◆   It sees groundwater mainly in terms of whether it is of significance to surface water ecosystems, terrestrial ecosystems or water supply;

◆   It concentrates largely on groundwater in 'groundwater bodies';

◆   It introduces the concept of 'groundwater status';

◆   It requires greater integration of qualitative and quantitative aspects of both surface waters and groundwater, taking into account the natural flow conditions of water within the hydrological cycle.

The Directive is intended to provide a 'framework'; consequently it does not detail all the technical requirements. Also, it is a complex Directive, partly because it encompasses all waters in an integrated way; therefore interpreting the requirements on the groundwater aspects is not easy. In addition, there is continuing discussion at EU level on the precise requirements, particularly in the monitoring area. This paper summarises the main groundwater-related features of the Directive.

RIVER BASIN PROJECTS

In Ireland, practical implementation of the WFD will take place in the context of River Basin Management Projects. These will be established on the basis of a small number of areas to be known as River Basin Districts (RBDs). There will be four national RBDs (East, South-east, West and South) and three international (Shannon, North-east and North-west). Local authorities will have the primary role in promoting, establishing and implementing these projects, which are being funded by the Department of the Environment and Local Government (DoELG). Most of the work will be undertaken by consultants, with input from public bodies such as the Geological Survey of Ireland (GSI), Environmental Protection Agency (EPA), Teagasc, OPW, Met Eireann, etc. These projects will provide much of the basic data requirements and necessary analysis for characterisation of river basins, identification of pressures and impacts, mapping locations and boundaries of water bodies, establishing integrated water monitoring programmes and establishment of programmes of measures for the preparation of River Basin Management Plans.

GROUNDWATER WORKING GROUP

The Department of the Environment and Local Government has established a WFD Co-ordination Group to co-ordinate and promote, at national level, implementation of the Directive. Under the aegis of the WFD Co-ordination Group, a Groundwater Working Group has been established to assist in the technical interpretation of the groundwater aspects of the Directive, and to provide guidance for River Basin Projects in the delivery and implementation of groundwater work requirements. The Working Group on Groundwater is led by the Geological Survey of Ireland, and consists of the following members: GSI – Donal Daly (convenor), Vincent Fitzsimons, and Geoff Wright; EPA – Conor Clenaghan, Margaret Keegan and Micheál MacCarthaigh; DoELG – Pat Duggan; Dúchas – Jim Ryan; Local Authorities – Billy Moore and Ray O'Dwyer; Third Level Institutions – Paul Johnston; Environment and Heritage Service (NI) and Geological Survey of Northern Ireland (GSNI) – Peter McConvey; SE RBD – Colin Byrne. This paper is largely a summary of a report of the Working Group (2001)[1].

GROUNDWATER BODIES – A NEW CONCEPT FOR HYDROGEOLOGISTS

The 'Groundwater Bodies' Concept

Virtually all of the requirements of the WFD concerning groundwater relate specifically to 'groundwater bodies' rather than to 'groundwater' or 'aquifers'. As it is a term that is not normally used by groundwater specialists, a description and understanding of the concept is crucial to enable the requirements of the WFD to be outlined and undertaken.

There are 3 relevant definitions in the WFD:

"Groundwater" means all water which is below the surface of the ground in the saturation zone and in direct contact with the ground or subsoil.

"Aquifer" means a subsurface layer or layers of rock or other geological strata of sufficient porosity and permeability to allow either a significant flow of groundwater or the abstraction of significant quantities of groundwater.

"Body of groundwater" means a distinct volume of groundwater within an aquifer or aquifers.

The concept of 'Groundwater Bodies' embraces:
♦ groundwater that can provide for the abstraction of significant quantities of water and should be managed to ensure sustainable, balanced and equitable water use, and groundwater of satisfactory quality.
♦ groundwater that is in continuity with surface ecosystems, can place them at risk, and should be managed to prevent environmental impacts on them arising from human activity, either through the transmission of pollutants or by unsustainable abstraction.

The **Groundwater Body is consequently the groundwater management unit under the WFD** that is necessary for the subdivision of large geographical areas of aquifer in order for them to be effectively managed. They are used for characterisation, monitoring and reporting, and where they are in close proximity and have similar hydrogeological properties, may be grouped together for these purposes.

Groundwater Bodies must be divided into those that:
♦ are **not** considered to be 'at risk' and subsequently do not require *further characterisation* or a high degree of surveillance (for the purposes of the Directive) but are still management units and will require operational controls;

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[1] Groundwater Working Group, 2001. WFD River Basin Management Systems : Technical Requirements for Groundwater and Related Aspects. Interim Report. 31pp. Can be obtained from Groundwater Section, GSI and DoELG. Also on website: http://193.178.1.105/euwfd.htm

♦ are considered to be 'at risk' and therefore require *further characterisation* and should be monitored more intensively, subjected to trend analysis, management measures and reporting, etc., as relevant.

Therefore, the WFD is risk-based and receptor oriented. Traditionally, hydrogeologists see groundwater largely in terms of 'aquifers' and as a receptor in its own right, as well as being a source of water for abstraction. While the WFD approach still places emphasis on groundwater as a source of water for abstraction, it requires that groundwater be seen not in terms of 'aquifers' but in terms of 'bodies' and in terms of groundwater contributing to and linked to surface water and terrestrial ecosystem receptors. Consequently, a new and broader approach to evaluating and conceptualising groundwater will result from the WFD.

GROUNDWATER OBJECTIVES OF WFD

The groundwater objectives of the WFD can be examined in terms of 'protected areas', 'status' and 'reversal of significant and sustained upward trends'. These are summarised below:
♦ Protected areas include drinking water protected areas, which are groundwater bodies that are used to provide or are intended to provide more than 10 m^3/d or serving more than 50 persons. In practice, this covers the total land surface of the Republic of Ireland. The WFD objective is that the groundwater bodies must be protected '*with the aim of avoiding deterioration in their quality in order to reduce the level of purification treatment required in the production of drinking water*'.
♦ '*Deterioration of the status*' of groundwater bodies must be prevented, by implementing '*measures to prevent or limit the input of pollutants into groundwater*' and by '*ensuring a balance between abstraction and recharge of groundwater*'. Where the existing status is 'poor', restoration is required, such that 'good' status is achieved by 2015 (see Figure 2).
♦ If there is a significant sustained upward trend in the concentration of any pollutant that could threaten good groundwater status or protected areas, this must be reversed.

GROUNDWATER BODY STATUS

The 'good' and 'poor' status classes for groundwater bodies are considered (a) in terms of the impacts of the groundwater on surface water and terrestrial ecosystems to which it is linked and is contributing water, and (b) based on compliance with standards and objectives relating to 'protected areas'. The status depends on the flow (quantity) and quality of the water, since the two main potential detrimental impacts are from over-abstraction and pollutants. Therefore, the WFD sub-divides status into 'Quantitative Status' and 'Chemical Status'. In the context of the Directive, microbial pollutants are not a factor in consideration of groundwater status. Similar chemical concentrations or level of abstraction relative to recharge can result in either 'good' or 'poor' status in different groundwater bodies. This is because the status depends partly on the needs of the surface water receptors, and these can vary depending on the needs of a particular receptor. This illustrates the integrated approach and ecological orientation of the WFD.

QUANTITATIVE STATUS

For good quantitative status to be maintained or achieved, there must be a correct balance between abstraction, recharge and the water requirements of dependent surface water and terrestrial ecosystems of national and international importance for nature conservation. Poor quantitative status arises where over-abstraction decreases water levels (in the long term) and in the process impacts detrimentally on surface water receptors by decreasing flow and/or causing intrusion of saline or polluted water. The WFD places reliance on water levels as the means of determining quantitative status. However, as has been pointed out by the UK Groundwater Technical Advisory Group, this is not feasible without a comprehensive monitoring network with long runs of data and, in any case, it shows up problems when ecological damage has occurred, which is against the spirit of the Directive. Therefore, while monitoring has an important role, a key element in determining the status is groundwater body characterisation (see Figure 3).

CHEMICAL STATUS

For good chemical status to be maintained or achieved, the concentrations of pollutants must not cause:

1. A failure to achieve one of the environmental objectives for surface water objectives, such as deterioration in status.
2. Significant diminution in the chemical or ecological quality of a surface water body.
3. Significant damage to terrestrial ecosystems of national or international importance for nature conservation which depend on the groundwater body.
4. A failure to achieve one of the groundwater protected area objectives, such as protection of drinking water.

Currently a Daughter Directive on groundwater is being prepared, which will elaborate on the measures required to achieve good chemical status.
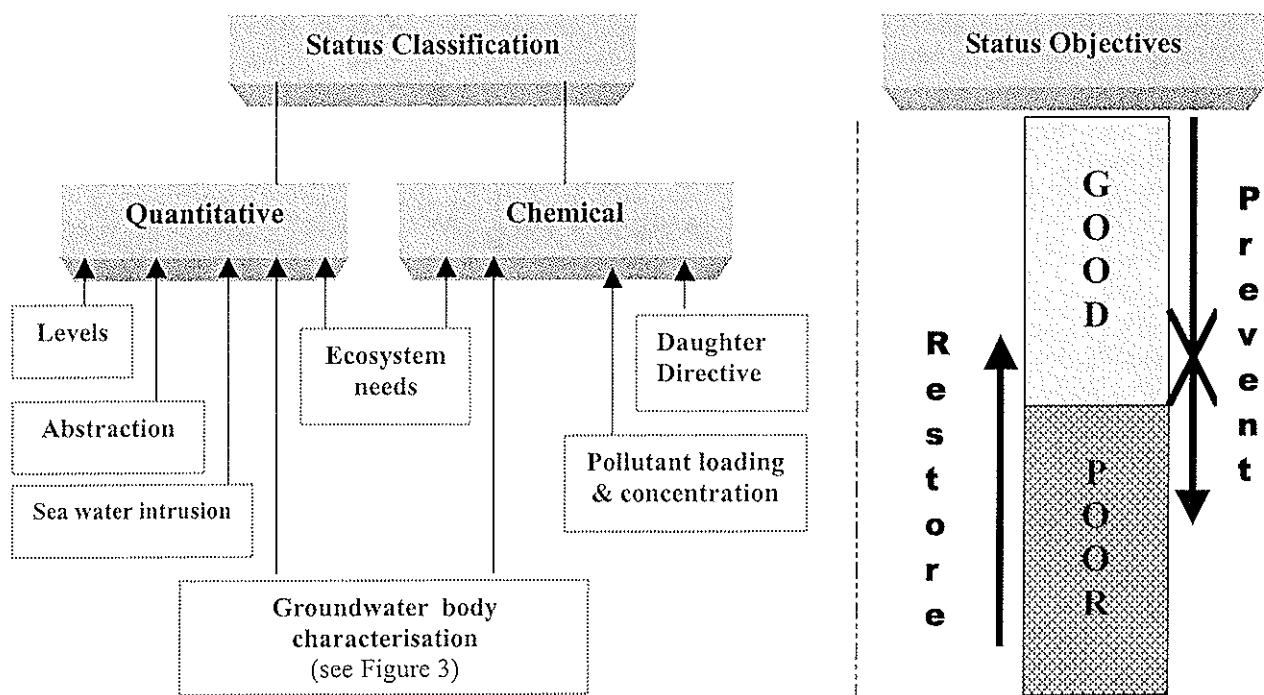
Figure 2   **Groundwater status classification and status objectives**

STEPS IN ACHIEVING THE GROUNDWATER OBJECTIVES OF THE WFD

The objectives are achieved by:

1. Characterising the groundwater bodies, including delineation of groundwater bodies and hydrogeological descriptions and understanding of groundwater flow, chemistry and interactions with surface water and terrestrial ecosystems.
2. Assessing pressures and impacts.
3. Environmental risk assessments to identify groundwater bodies 'at risk' and not 'at risk' of failing to meet the objectives of the WFD.
4. Monitoring programmes to i) assist with characterisation, ii) help (together with characterisation) determine groundwater status and iii) assess whether the objectives are being achieved.
5. Establishing and presenting the groundwater status.
6. Implementing measures to maintain the objectives (where the situation is satisfactory) and restore good status (where the situation is unsatisfactory).

The remainder of this paper will concentrate on the delineation, characterisation, monitoring and status evaluation of groundwater bodies.

CHARACTERISATION OF RIVER BASIN DISTRICTS (RBDs)

INTRODUCTION

A major element of groundwater characterisation is the mapping and description of 'groundwater bodies'. Characterisation is sub-divided into two stages – initial characterisation and further characterisation, where further characterisation is required for groundwater bodies deemed to be 'at risk'.

INITIAL CHARACTERISATION

Requirements of Directive for Initial Characterisation

According to the Directive (Annex II, section 2.1):

Member States shall carry out an initial characterisation of all groundwater bodies to assess their uses and the degree to which they are at risk of failing to meet the objectives for each groundwater body under Article 4. Member States may group groundwater bodies together for the purpose of this initial characterisation. This analysis may employ existing hydrological, geological, pedological, land use, discharge, abstraction and other data but shall identify:

- *the location and boundaries of the groundwater body or bodies*
- *the pressures to which the groundwater body or bodies are liable to be subject including:*
  - *diffuse sources of pollution*
  - *point sources of pollution*
  - *abstraction*
  - *artificial recharge,*
- *the general character of the overlying strata in the catchment area from which the groundwater body receives its recharge,*
- *those groundwater bodies for which there are directly dependent surface water ecosystems or terrestrial ecosystems.*

The process involved in undertaking the initial characterisation is summarised below and in Figure 3. A more comprehensive description is given in the Groundwater Working Group report (2001).

- ♦ The definition of 'groundwater body' means that all groundwater bodies will constitute part of an aquifer or aquifers. Therefore an aquifer map, using the aquifer categories defined in *Groundwater Protection Schemes* (DoELG/EPA/GSI, 1999) must first be completed to enable groundwater bodies to be delineated.
- ♦ The aquifers will then be grouped as follows:
  - ➢ Regionally Important Karstic (Rk) aquifers;
  - ➢ Gravel (Rg and Lg) aquifers;
  - ➢ Productive fractured bedrock (Rf and Lm) aquifers;
  - ➢ Poorly productive bedrock (Ll, Pl and Pu) aquifers.

  This grouping is based on similarities in a) hydrogeological properties; b) resource value; c) likely monitoring approaches; and d) likely influence on surface water characterisation. They can be considered as general groundwater body types.
- ♦ These groundwater body types are further sub-divided by catchment/surface water body boundaries to give a preliminary map of groundwater bodies. This will act as a framework for further sub-division based on areas that are considered to be 'at risk', and the nature of that risk.
- ♦ The simple conceptual model for each groundwater body obtained by this approach will then be evaluated and tested by various conventional hydrogeological approaches, which in practice will have been used in aquifer delineation. However, in addition, surface-groundwater interactions, recharge and groundwater chemistry will be considered.
- ♦ Each groundwater body will be defined in three dimensions.
- ♦ Groundwater-dependent conservation-designated ecosystems will be located and the relationship with the groundwater bodies described.
- ♦ A risk assessment will be undertaken. This will evaluate: groundwater body hydrogeology; water quality; presence of extremely vulnerable areas; the requirements of groundwater-dependent
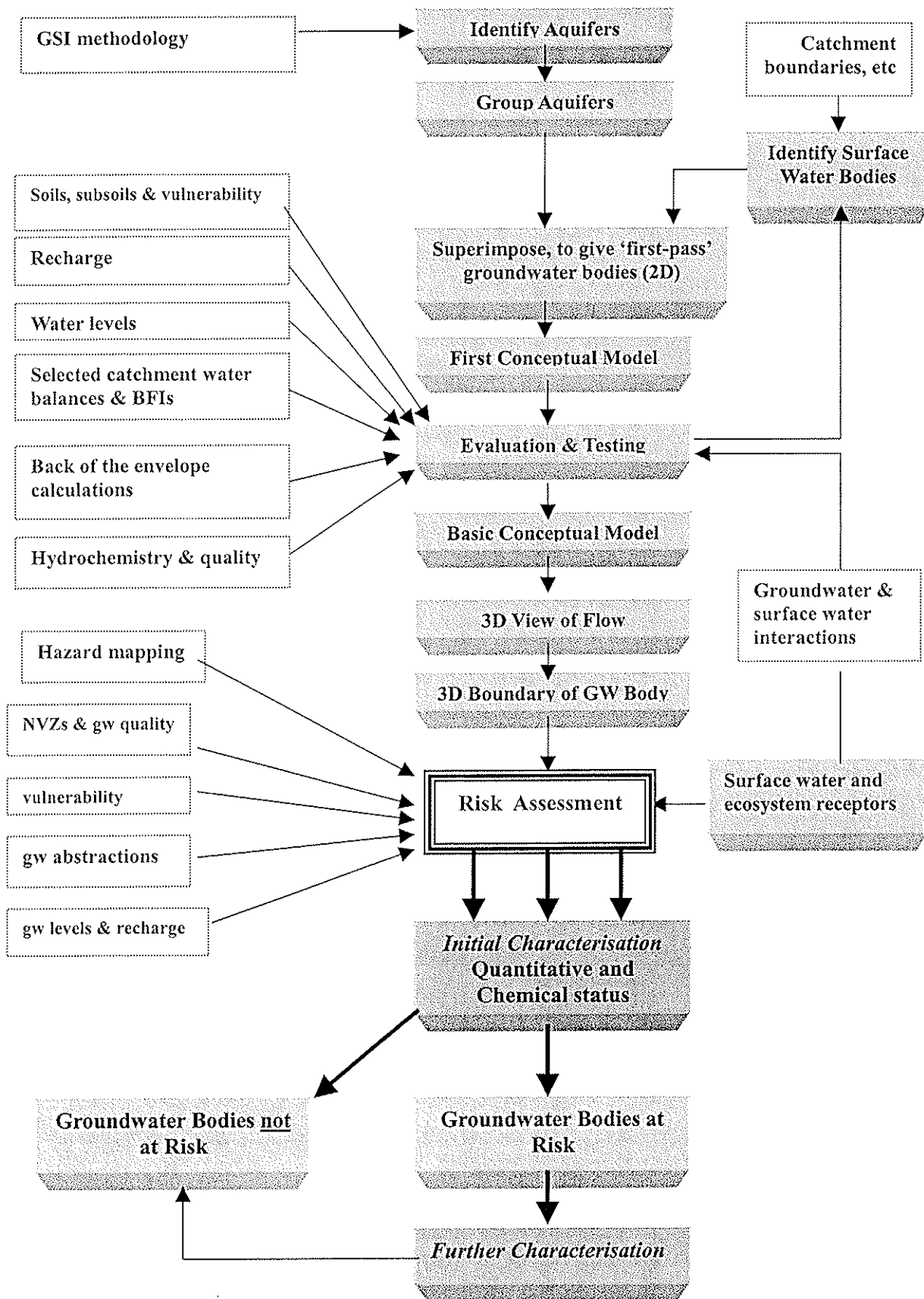
Figure 3 Summary of approach leading to the delineation of groundwater bodies

ecosystems and surface water bodies; recharge, abstraction and water levels; and pressures on the groundwater body, including both point and diffuse sources of pollution. In essence, it integrates the nature, sensitivity and requirements of the receptors with the hydrogeological characteristics of the groundwater body and the degree of pressure posed by human activities.

The outcome of this process will be a subdivision of groundwater bodies into those that are considered to either 'at risk' or **not** 'at risk' of meeting the environmental objectives of the Directive. Those bodies deemed to be 'at risk' require further characterisation.

FURTHER CHARACTERISATION OF 'AT RISK' GROUNDWATER BODIES

Further characterisation requires more detail on the following:
♦ the pressures, including mapping of hazards;
♦ the requirements of relevant receptors;
♦ the hydrogeology, including a more precise delineation of extremely vulnerable areas;
♦ collection, compilation and assessment of water level and abstraction data for groundwater bodies under threat of over-abstraction;
♦ evaluation of water quality data.

For some groundwater bodies thought to be at risk, it may turn out during further characterisation that the risk, which was assumed from the initial characterisation, is negligible. Once this has become clear, continued characterisation is no longer necessary.

The outcome of 'further characterisation' will be a comprehensive assessment of the natural character (hydrogeological, hydrochemical, ecological, hydrological, as appropriate) of the groundwater body at risk, which will be linked with an assessment of pressures and impacts. This will be followed by the identification of appropriate measures to achieve the environmental objectives of the Directive.

GENERAL POINTS

Groundwater Body Boundaries

Groundwater bodies will have hydraulic boundaries; preferably based on changes in geology, groundwater divides and, only when there is no alternative, on flow lines. Finalising the boundaries will be an iterative process, depending on the interaction with the delineation of surface water boundaries and the updating of hydrogeological information.

Grouping of Bodies

In describing groundwater bodies, adjacent bodies within an RBD will be grouped, if practicable, where there are similarities in hydrogeology, pressures and impacts.

Quantitative Status

Unlike most other EU countries, given our relatively low levels of abstraction and generally high rainfall, it is unlikely that there will be many groundwater bodies where the quantitative status is at risk.

Role of GSI and Teagasc

The GSI will be undertaking the aquifer delineation and contributing to the delineation of groundwater bodies. Teagasc will undertake soils and subsoils mapping.

The WFD and Groundwater Protection Schemes

Where groundwater protection schemes (GWPSs) have been completed (50% of the country by 2004), they will provide most of the mapping requirements for characterisation, although the earlier schemes will need to be updated by the GSI. For the remaining areas, the WFD work will not involve production of groundwater protection schemes; they will provide the aquifer component but not full vulnerability mapping (mapping of 'extremely' vulnerable areas only will be undertaken on groundwater bodies considered to be at risk, i.e. those areas with less than 3 m soil and subsoil). GSI plans to continue this work after 2004, as GWPS will still be recommended for land-use planning,

updating of characterisation and, together with the GWPS Responses, as a component of the groundwater-related measures required by the WFD to protect groundwater.

MONITORING OF GROUNDWATER STATUS

Monitoring is a critical element of the Directive. Monitoring programmes must be established by the end of 2006, with the aim of:

♦ assisting groundwater body characterisation;
♦ assisting and validating risk assessments as part of characterisation;
♦ helping to establish the groundwater status of groundwater bodies or groups of bodies;
♦ checking for trends; and
♦ checking the effectiveness of the programme of measures designed to achieve the environmental objectives.

Networks are required for monitoring the water levels and chemical status of groundwater bodies. The requirements and links are summarised in Figure 4. There is a danger that too much emphasis could be placed on monitoring. It is essential that the monitoring networks are based on a proper understanding of the hydrogeology of groundwater bodies and are focused on the objectives of the Directive, and that characterisation is seen as a continuing process after 2004.
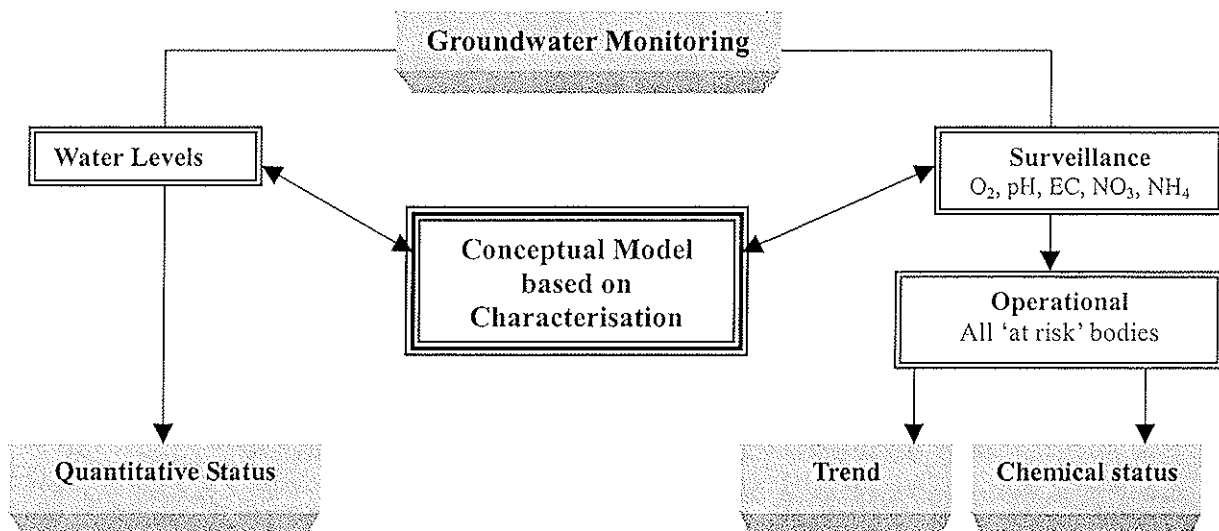
Figure 4   WFD Groundwater Monitoring

In Ireland some locally important aquifers and virtually all poor aquifers, due to their hydrogeological conditions, have short flow paths and the zones of contribution associated with monitoring sites are small. Therefore the usefulness of monitoring sites in terms of being representative of the waters within a groundwater body must always be assessed. Natural springs may assist in providing representative monitoring sites. Also, sampling baseflow in dry weather may assist.

QUANTITATIVE STATUS

Water level is given as the key parameter. The monitoring network must be designed to provide an early indication of any negative changes in the quantitative status of each groundwater body or representative groundwater body caused by abstraction. The network shall encompass both groundwater bodies 'at risk' and those 'not at risk', although monitoring will focus in particular on those bodies 'at risk'.

The distribution, density and frequency of monitoring sites must ensure that the spatial and temporal variability of the groundwater surface can be sufficiently well recorded within a groundwater body to enable status to be assessed and, in 'at risk' bodies', the impacts to be evaluated. It will depend on the hydrogeological conditions (i.e. the conceptual model of the groundwater body) and the degree to which the body is deemed to be 'at risk'.

CHEMICAL STATUS

Two types of monitoring of chemical status and trends are required – surveillance and operational.

Surveillance Monitoring

Surveillance monitoring is used to supplement and validate the procedure for the assessment of pressures and impacts, and provide information for use in the assessment of long term trends both as a result of changes in natural conditions and through human activity. The density of monitoring points will be determined by the hydrogeological regime in the groundwater body under review and the assessment of pressures and impacts on the groundwater body. A core group of parameters (O_2, pH, EC, NO_3, and NH_4) must be analysed in all cases. Other parameters, that are significant in the catchment area of the monitoring point and could constitute a negative influence on the groundwater quality, will be included as appropriate.

Operational Monitoring

Following the characterisation and risk assessment process and the evaluation of the surveillance monitoring programme, an operational monitoring programme, focused on water bodies at risk of failing to meet the Directive's environmental objectives, will be set up. The parameters to be measured for 'operational monitoring' will generally include those required for 'surveillance monitoring' but will be extended as necessary to include those additional parameters that are indicative of the identified risks. The frequency of monitoring will be (i) at a minimum of once per year and (ii) between the sampling dates of the surveillance monitoring programme. The frequency will depend on the hydrogeological conditions and on the vulnerability and flow regime of the groundwater body.

The data from the surveillance and operational monitoring is used to assess pollutant trends for the groundwater body (or group) as a whole. Any trend that could threaten good groundwater status or a protected area, such as for drinking water, must be reversed.

WFD AND RBD PROJECTS : AN OPPORTUNITY!

The requirements of the WFD provide a unique opportunity to:
1) advance our understanding of groundwater in Ireland;
2) improve our datasets;
3) have groundwater considered and managed as a vital part of the hydrological cycle;
4) improve our existing water level and water quality monitoring networks, not only in terms of the number of monitoring stations, but also in having each station based on sound and relevant hydrogeological information; and
5) enable our groundwater to be protected by providing an allied mechanism to Groundwater Protection Schemes.

ACKNOWLEDGEMENTS

2.

The Water Quality of Groundwater – *Chapter 5* **Water Quality in Ireland 1998-2000, EPA**

Margaret Keegan, Environmental Protection Agency, Wexford; Becci Cantrell Environmental Protection Agency, Wexford; M. MacCartaigh, Environmental Protection Agency, Dublin; Paul Toner, Environmental Protection Agency, Dublin.

Margaret.

Levels 350 stations
All aquifers covered
in monthly readings

Quality - 380 stations
All aquifers
~ twice yearly.

[handwritten annotations]

Chapter Five

THE WATER QUALITY OF GROUNDWATER

INTRODUCTION

Groundwater is an important water resource in the State and accounts for up to 15 per cent of total water supplied by local authorities (Daly, D., 1993). One quarter of the water abstracted for public and private drinking water supply is from groundwater with the proportion rising to 86 per cent in rural areas. A very large number of groundwater supply sources exist, e.g. Wright (1999) estimates that there are at least 200,000 wells in the country. However, only a small proportion of the available groundwater resource is currently being used.

In general, in Ireland the majority of private groundwater supplies are untreated. This heightens the need for aquifer and source protection and the treatment of groundwater to ensure that the quality of drinking water conforms to the requirements of the Drinking Water Regulations (SI No. 81 of 1988). Additionally, as groundwater may ultimately discharge from an aquifer as base or springflow to rivers, wetlands, estuaries, quarries or springs, the latter may be affected adversely if such discharge is polluted.

The Geological Survey of Ireland (GSI) has completed, or is nearing the completion of, a number of groundwater protection schemes for various local authorities and is planning to have developed such schemes for much of the country by 2010[1] (Daly, D. 1999). A groundwater protection scheme takes account of the nature of the hazard to groundwater (the potentially polluting activity), the pathway for contaminant migration to the aquifer (the groundwater vulnerability) and the value of the aquifer, spring or well at risk . The GSI, in combination with the DELG and the EPA, has published a methodology for the development groundwater protection schemes for Ireland that incorporates these elements of risk assessment (DELG, 1999).

The Department of the Environment and Local Government announced in July, 2000 that 13 groundwaters in counties Carlow, Cork, Kerry, Louth and Waterford had been identified by a Panel of Experts as being polluted or susceptible to pollution, by nitrates from agricultural sources. In line with the requirements of the EU directive on the prevention of nitrate pollution from agriculture (CEC, 1991), work is underway in delineating the catchments of these affected waters, and the land areas within the catchments contributing to nitrogen pollution. Nitrate Vulnerable Zone designations may be finalised for these waters before the end of 2001 with action plans completed by the end of 2002. It is envisaged that additional affected waters will be identified on the basis of a further review currently being undertaken (EPA, 2001a).

The EU adopted a new Drinking Water Directive (98/83/EC) in November 1998 and this was transposed into Irish law on the 18th December, 2000 by the European Communities (Drinking Water) Regulations, 2000 (S.I. No. 439 of 2000). This set of drinking water regulations is radically different from its predecessor and will entail very significant changes in virtually all aspects of implementation, including sample number, parameters, parameter classes and extent of coverage. However, it is important to note that the commencement date for the new Regulations is 1st January 2004 (O'Leary *et al.*, 2001)

In December 2000 the Water Framework Directive (2000/60/EEC) came into force and it establishes a strategic framework for managing the water environment and sets out a common approach to protecting and setting environmental objectives for all groundwaters and surface waters within the European Community. Specifically for groundwater the Directive aims to protect, enhance and restore all bodies of groundwater, which *inter alia* includes the maintenance and/or attainment of 'good chemical status'. However, the exact requirements of the WFD are not yet fully determined at this stage and, in particular, the European Parliament and the Council have yet to adopt specific measures to prevent and control groundwater pollution (Article 17) which, *inter alia*, shall include:

- Criteria for assessing good groundwater chemical status, in accordance with Annex II 2.2 and Annex V 2.3.2 and 2.4.5;

- Criteria for the identification of significant and sustained upward trends and for the definition of starting points for trend reversals to be used in accordance with Annex V 2.4.4.

In the absence of criteria adopted at Community level, Member States are required to establish appropriate criteria at the latest five years after the date of entry into force of the Directive. The EPA is

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[1] The same deadline as proposed by the EU in the Water Framework Directive for achieving a good status for all groundwater and surface water.

currently in the process of developing guideline values for the assessment of groundwater quality in Ireland (EPA, 2001b). The draft document sets out the Agency's proposed approach and application of guideline values for the protection of groundwater in Ireland. It has been proposed, therefore, that, on an interim basis and pending further elaboration of groundwater protection measures at national and Community level, these draft parameters and guideline values be used for the monitoring and characterisation of groundwater bodies for the purpose of river basin projects. Monitoring data collected during the course of these projects will further assist in the elaboration of national groundwater standards.

GROUNDWATER QUALITY IN IRELAND

Background

Groundwater quality is a function of natural processes as well as anthropogenic activities. Natural groundwater quality is generally good, although harmful concentrations of certain ions, e.g. iron, manganese, sulphate, hydrogen sulphide and, near coasts, sodium and chloride, can occur naturally and lead to problems (Daly, D. 1994). In Ireland, limestone bedrock and limestone dominated subsoils are common and consequently groundwater is often hard, containing high concentrations of calcium, magnesium and bicarbonate. However, in areas where volcanic rock or sandstones are present, softer water is normal (Daly, D. 2000).

It is important, therefore, that natural hydrochemical variations should be taken into account in establishing any baseline quality criteria, and in interpreting the results of groundwater monitoring programmes.

The concentrations of any contaminants detected in a groundwater monitoring programme will be influenced by source characteristics and proximity, the nature of the contaminant and the geological and hydrogeological influences, including, for example:

- the type of contaminant source (point source or diffuse);

- how far the source is located from the well or spring;

- the characteristics of the contaminant (e.g. solubility and mobility);

- the characteristics of the aquifer (primary or secondary permeability, presence of karst);

- and the aquifer vulnerability (e.g. the presence or absence of a protective layer of thick, low permeability subsoil above the aquifer).

Vulnerability is 'a term used to represent the intrinsic geological and hydrogeological characteristics that determine the ease with which groundwater may be contaminated by human activities' (Daly, D. and Warren, W.P. 1998).

The monitoring results will also be influenced by the monitoring regime itself, including:

- the type of groundwater sampling point (well or spring);

- the design of the well (e.g. for a discrete sampling depth or open hole completion);

- the construction of the wellhead (whether the surface casing has been properly sealed);

- the abstraction rate and hence zone of contribution (ZOC) to the well;

- the depth of sampling, and the method of sampling (pumped or bailed);

- the time of year;

- whether analyses are carried out on site or in the laboratory, sample storage procedures and related considerations, (see, for example, Hayes, T. 1997).

Groundwater Monitoring Programme

In the EPA programme, monitoring has been classified in three categories:

1. Representative or basic monitoring;

2. User-related monitoring; and

3. Pollutant-related monitoring.

The representative or basic monitoring is operated on a national basis by the EPA to define the state of groundwater quality, to detect trends in groundwater quality and to determine the causes of any changes in quality that are identified. Monitoring stations within this basic network have been selected taking into account hydrogeological conditions and groundwater use. Where drinking water abstractions are used as part of this network, samples are taken at a point antecedent to any treatment process.

User-related monitoring mainly consists of monitoring of those drinking waters originating as groundwater, as required under the European Communities (Quality of Water intended for Human Consumption) Regulations, 1988. Under these regulations, all waters used for human consumption as well as water used in the food industry, regardless of origin is covered. Monitoring of the water quality is required at the point where it is made available to the consumer.

Pollutant-related monitoring is intended to detect possible pollutant emissions from landfill sites, septic tank clusters, factories and other waste sources and includes the identification and mapping of potential sources of pollution.

The information set out below is based primarily on the analysis of samples taken by the EPA at monitoring stations in the representative network as part of the EPA's National Groundwater Quality Monitoring Programme. This programme commenced in 1995 and monitoring is carried out twice a year, to coincide with groundwater levels being at their lowest and highest levels.

The report contains the results of the analyses of groundwater samples taken in the period 1998-2000 in relation to a number of important parameters and indicates whether they meet the standards set in the Drinking Water Regulations for these parameters. This approach is taken in the absence of the finalisation of the proposed guideline values for the protection of groundwaters (EPA, 2001b). It is considered appropriate in the light of the fact that many groundwaters are put into supply systems with only minimal treatment, if any.

Sampling Frequency and Number of Monitoring Stations in the Period 1998-2000

The EPA commenced its groundwater monitoring programme in 1995 and results for the 1995-1997 period have been reported (Lucey et al. 1999). In the period 1998-2000, two sampling runs were carried out each year, giving a maximum of six samples per station.

As six samples were not taken at all locations in the period 1998 – 2000, the results, for the major parameters, from four sampling runs were taken as the minimum for the purposes of obtaining representative water quality data for use in this report. Thus, while the total number of individual samples taken on average was about 1350 at some 380 locations, approximately 200 sampling locations only had more than four sampling runs.

The counties shown below did not have the minimum number of sampling runs available for particular parameters and therefore have not been represented in the mean concentration figures or maps for those parameters. Details are given below.

Parameters	Counties not represented
Iron	Carlow, Tipperary (N+S), Waterford, Wexford
Manganese	Carlow, Tipperary (N+S), Waterford, Wexford
Chloride	Wexford
O-phosphate	Wexford
Ammonia	Wexford
Nitrate	Wexford

Faecal Coliforms Clare, Cork (N+S), Dublin, Kerry, Kildare, Limerick, Offaly, Wexford and Wicklow.

Since only three sampling runs for the selected parameters were carried out in Co. Wexford, this area did not have the required number of sampling runs to be included in the mean concentration calculations.

In addition, the number of monitoring stations has varied over the period of this review as sites were either found to be unsuitable for monitoring purposes or sources were no longer in use as the abstraction of water had ceased.

Results Of Analysis Of Groundwater Quality Samples Taken In The Period 1998-2000
Presentation of Data

To ensure uniformity in reporting throughout this report, the comments below are made on water quality results in terms of the mean of the sample results at each monitoring station, i.e., only samples from the approx. 200 locations with four or more sample runs are presented below. However, where relevant, individual sample results (from the approx. 200 locations) are also commented upon. The locations of the sampling points and data for selected parameters at each monitoring station are given in Maps 1-7 in Appendix VI.

Ammonia Concentrations

Ammonia is generally present in natural waters, though in very small amounts, as a result of microbiological activity which causes the reduction of nitrogen-containing compounds (EPA, 2001c). It has a low mobility in soil and subsoil and its presence in groundwater much above 0.1 mg/l N may indicate direct sewage, industrial or agricultural contamination. From the view point of human health, significant concentrations of ammonia can indicate the possibility of sewage pollution and the consequent possible presence of pathogenic microorganisms (Lucey et al., 1999).

Under the EC (Quality of Water intended for Human Consumption) Regulations, 1988 (S.I. No. 81 of 1988), the maximum allowable concentration (MAC) of ammonia is 0.3 mg/l as NH_4^+ (ammonium) which is equivalent to 0.23 mg/l as N. Samples were analysed for total ammonia which consists mainly of ionised ammonia (ammonium, NH_4^+). For clarity, the parameter is referred to below as ammonia and the units used are **mg/l N**.

A total of 1166 individual samples were analysed for ammonia at 217 monitoring stations. The mean concentration results are summarised in Fig. 5.1 and are shown on Map 2 in Appendix VI for the individual locations. Mean ammonia levels greater than 0.23 mg/l N were recorded at 14 of the 217

Fig. 5.1 The number of sampling points with mean concentrations of Ammonia in the ranges indicated.

Fig. 5.2 Comparisons of the percentage of stations with mean concentrations of Ammonia in the ranges indicated between the 1995-1997 and the 1998-2000 period.

monitoring stations while 76 per cent of the stations had mean values less than 0.05 mg/l N. The majority (1096) of individual samples had ammonia concentrations less than the MAC value of 0.23 mg/l N. Of the 70 individual samples in which the measured concentrations were greater than or equal to the MAC value, 59 were taken at stations where the mean ammonia concentrations were also greater than 0.23 mg/l N. The highest individual ammonia concentration recorded was 2 mg/l N at a location in County Leitrim.

Mean ammonia concentrations greater than 0.23 mg/l N were found in samples from Cavan (1), Donegal (1), Kerry (1), Leitrim (3), Limerick (1), Meath (3), Monaghan (3) and Waterford (1). Of the 14 monitoring stations with mean ammonia concentrations above the MAC, five tested positive for total coliforms and two tested positive for faecal coliforms. However, two of the stations were not examined for bacteriological contamination. Any drinking water supply source with elevated ammonia levels should be regarded with suspicion pending the results of bacteriological examination.

Mean ammonia levels above 0.1 mg/l N were recorded at 31 monitoring stations in 13 counties. Ammonia concentrations greater than 0.1 mg/l N may indicate a nearby organic waste source. This demonstrates the need for disinfection of drinking water supplies but also the need for suitable borehole siting, construction and protection. All boreholes should be adequately grouted from the ground surface through the subsoils to prevent the ingress of surface contamination. Boreholes should also be located such that they are not too close to or downgradient of potential contamination sources such as septic tank percolation areas (DELG *et al*, 1999).

Comparison with the 1995-1997 Data The percentage of stations in each range for the period 1995-1997 and 1998-2000 are shown in Fig. 5.2. There was a slight decrease, from 95 to 93.5 per cent, of stations that had mean ammonia concentrations less than 0.23 mg/l N between the two sampling period. In addition, the number of stations with mean concentrations below 0.1 mg/l N declined from 89 per cent to 85.5 per cent.

Nitrate Concentrations

Nitrate contamination is generally observed in low yielding wells and in close proximity to potential point waste sources but may also arise from diffuse sources. It is of particular concern in some areas of Carlow, Cork, Kerry, Louth and Waterford where nitrate vulnerable zones are currently being delineated and other areas may still be identified. It may develop into a more widespread problem unless mitigation measures such as nutrient management planning are put in place.

Most nitrate found in natural waters is of anthropogenic origin, coming from organic and inorganic sources, the former including waste discharges and the latter comprising mainly artificial fertilisers. Nitrate is present naturally in water in low concentrations, typically in the range 5 - 9 mg/l NO_3.

There are health risks associated with excess nitrate consumption in the human diet. These include methamoglobinaemia in infants (blue baby syndrome) and possible carcinogenic hazards. The toxicity of nitrate to humans is thought to result solely from its reduction to nitrite. Nitrite is involved in the oxidation of normal haemoglobin to methaemoglobin which is unable to transport oxygen to the body's tissues (WHO, 1996). If water contains more than 450 mg/l NO_3 nitrate, it is unsuitable for livestock (Freeze and Cherry, 1979).

Under the EC (Quality of Water intended for Human Consumption) Regulations, 1988 (S.I. No. 81 of 1988), the MAC value for nitrate is 50 mg/l as NO_3 which is equivalent to 11.3 mg/l as N. A guide level of 25 mg/l NO_3 (or 5.65 mg/l N) was specified in the EU Directive 80/778/EEC and is recommended as an indication of contamination. The units used below for nitrate are **mg/l NO_3.**

A total of 1171 individual nitrate samples were recorded at 216 monitoring stations. The mean concentration results are summarised in Fig. 5.3 and are shown on Map 3 in Appendix VI for the individual locations. Mean concentrations of nitrate exceeded 25 mg/l NO_3 at 42 monitoring stations and exceeded the MAC of 50 mg/l NO_3 at 3 stations in Counties Carlow and Louth. The highest individual sample concentration was 98 mg/l NO_3 at a location in County Louth.

Mean concentrations of nitrate less than 25 mg/l NO_3 were recorded at 174 (80%) monitoring stations (Fig. 5.3). Concentrations greater than 25 mg/l NO_3 were recorded in 234 individual samples of which 32 exceeded 50 mg/l NO_3. Of the samples that exceeded the MAC, 11 were taken at stations where the mean nitrate concentration also exceeded the MAC. The remaining 21 samples were taken at 12 stations.

Elevated values (i.e. with a mean nitrate concentration >25 mg/l NO_3) were found in samples from Carlow (5), Cork N (2), Cork S (1), Kerry (3), Kildare (4), Kilkenny (1), Laois (5), Limerick (1), Louth (5), Offaly (2), Roscommon (1), Tipperary N (4), Tipperary S (1), Waterford (4), Westmeath (1) and Wicklow (2). The 42 monitoring stations at which elevated values (i.e. with mean concentrations >25 mg/l NO_3) occurred were not associated with raised concentrations of ammonia. Only two of the monitoring stations were
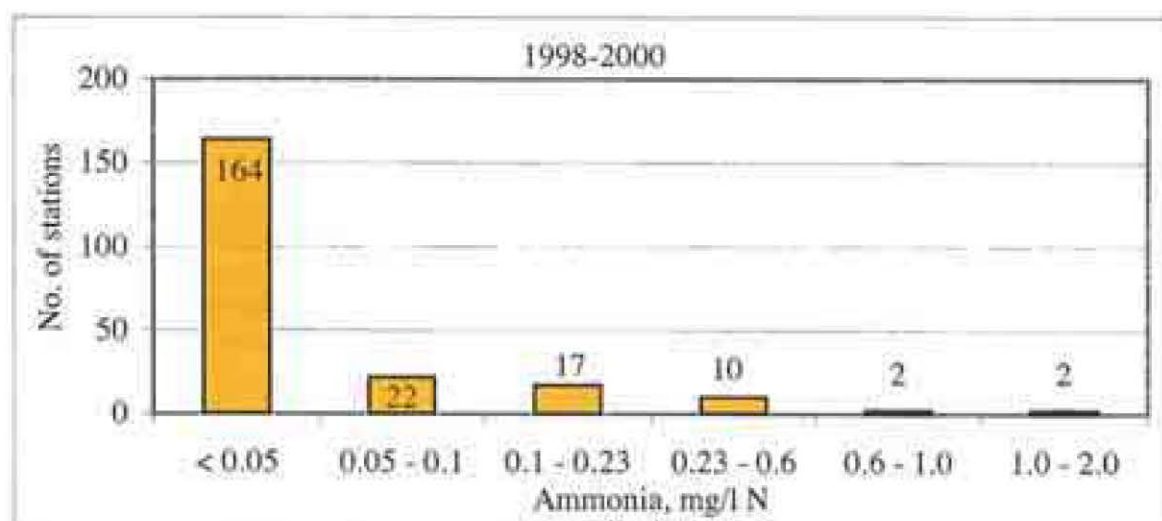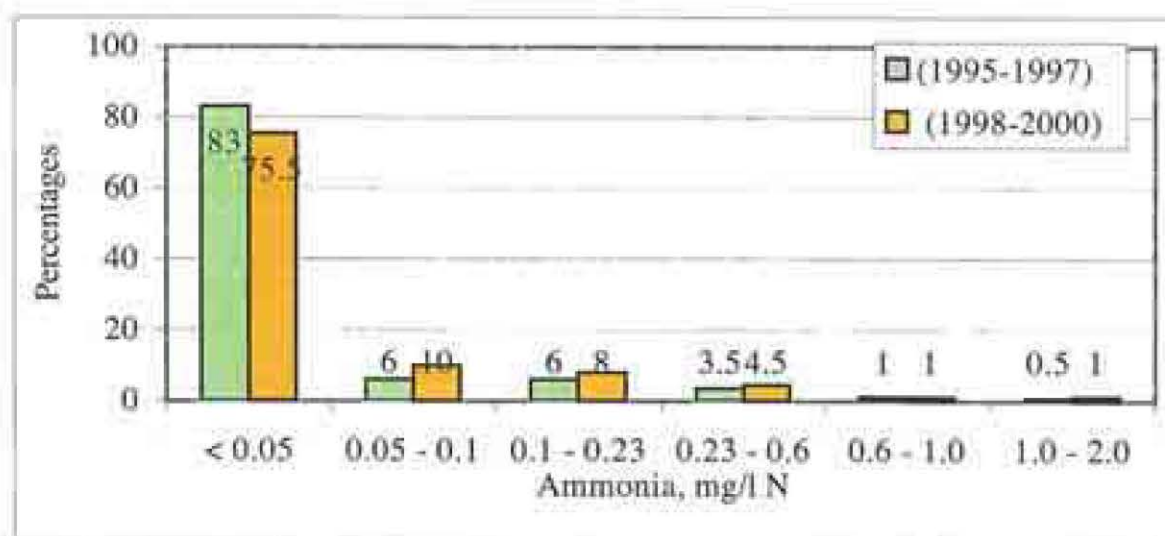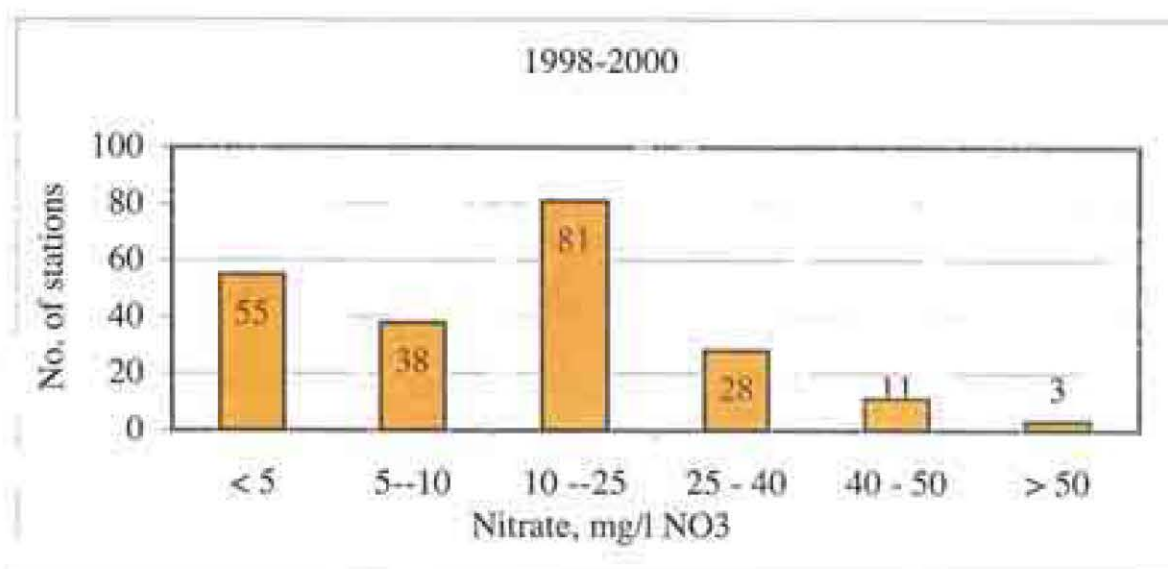
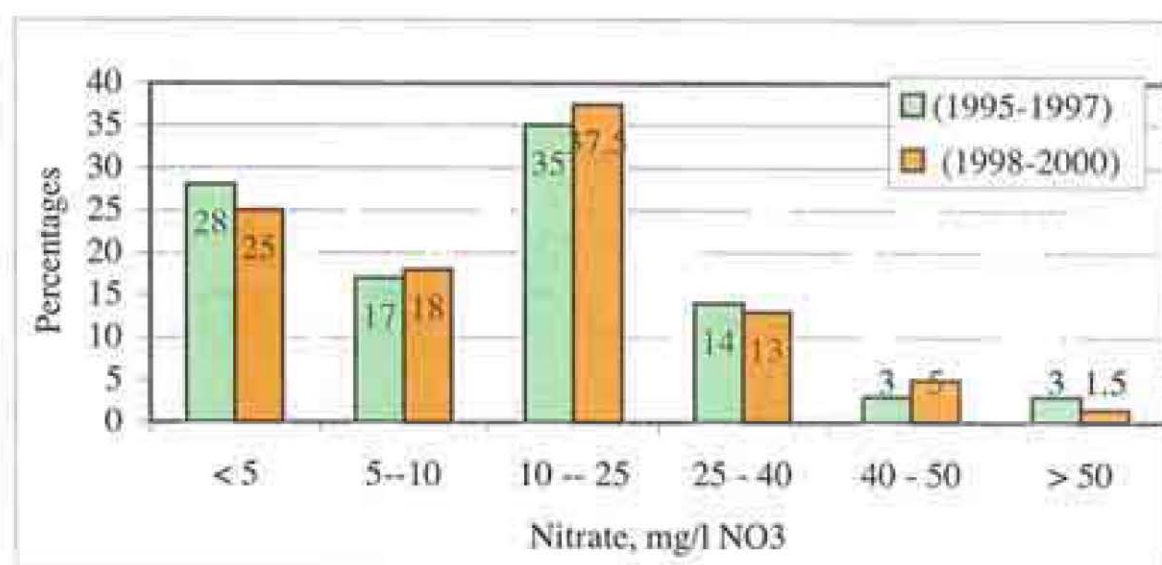Fig. 5.3 The number of sampling points with mean concentrations of Nitrate in the ranges indicated.

Fig. 5.4 Comparisons of the percentage of stations with mean concentrations of Nitrate in the ranges indicated between the 1995-1997 and the 1998-2000 period.

free of bacterial contamination and mean faecal coliform counts of zero were recorded at 17 of the stations.

Comparison with the 1995-1997 Data The percentage of stations in each range for the periods 1995-1997 and 1998-2000 are shown in Fig. 5.4. In the 1995-1997 sampling period 97 per cent of the stations had mean concentrations less than 50 mg/l NO$_3$. This figure increased slightly to 98.5 per cent for the 1998-2000 sampling period. The proportions (80% and 80.5%) of stations with mean concentrations less than the guide value of 25 mg/l NO$_3$ in the two periods were similar .

Chloride Concentrations
Chloride exists in all natural water, the concentration varying widely and reaching a maximum in sea water (average 20,000 mg/l Cl). Chloride does not pose a health hazard to humans and the principal consideration is in relation to palatability. Where there is a high chloride concentration there also may be an associated high sodium level. In freshwaters, chloride originates from both natural and anthropogenic sources, such as run-off containing de-icing salts, the use of inorganic fertilizers, landfill leachates, septic tank effluents, animal feeds, industrial effluents and seawater intrusion in coastal areas.

Because sewage is such a rich source of chloride, a high level or a significant increase of the ion may give rise to suspicions of pollution from sewage. However, in coastal areas elevated chloride values may be due to sea spray or salt water intrusion and not necessarily due to pollution (EPA, 2001c).

Under the EU (Quality of Water Intended for Human Consumption) Regulations, 1988 (SI No. 81 of 1988), the maximum admissible concentration (MAC) for chloride is 250 mg/l Cl; however, one would expect levels around 30mg/l in uncontaminated groundwater, except in coastal areas. Concentrations vary and what is important is not the absolute value but rather the relative level from one sampling period to the next.

A total of 1166 samples at 216 stations were analysed for chloride. The mean concentration results are summarised in Fig. 5.5 and results for the individual locations are shown on Map 4 in Appendix VI. The bulk (84%) of all samples taken from the 216 stations had a chloride concentration less than 30 mg/l. There were 181 stations where the mean result was less than 30 mg/l Cl while nine locations had a mean result greater than 50 mg/l. There were three samples from two different locations which exceeded the MAC of 250 mg/l Cl; two of these samples (Monaghan) also had high sodium and ammonia levels while the other one (Meath) was otherwise normal. High chloride levels

should be assessed in conjunction with ammonia, nitrite, bacteriological analyses and sodium to determine the possible source of contamination.

Comparison with 1995-1997 Data The percentage of stations in each mean concentration range for the period 1995-1997 and 1998-2000 are shown in Fig. 5.6. In the 1995-1997 sampling period 86 per cent of the stations had mean concentrations of Cl less than 30 mg/l, compared to 84 per cent 1998-2000. The data show a decrease in the percentage of stations with mean concentrations less than 20 mg/l and an increase in the percentage of stations in the 20-30 mg/l Cl range between the two periods.

Ortho-Phosphate Concentration
Phosphorus concentrations in natural water are normally no more than a few tenths of micrograms/litre (Hem, 1989). Phosphorus is used as an agricultural fertiliser and in household cleaning detergents as well as in industry. It is a major source of concern for surface waters because small amounts may lead to eutrophication of lakes and rivers. However, phosphorus is not a problem in groundwater; as it is not very mobile in soils, it is considered to be retained in the soil zone or sediments and is thus unlikely to penetrate to groundwaters. However, it may act as a pathway for P to receptors such as lakes, rivers and wetlands.

According to the Water Pollution Act, 1977 (Quality Standards for Phosphorus) Regulations, 1998 (SI No. 258 of 1998) in order to prevent eutrophication of surface waters, groundwater levels should not exceed 0.03 mg/l P as orthophosphate P when providing baseflow to a river or 0.02 mg/l total phosphorus if the groundwater feeds a lake.

A total of 1149 samples were analysed for orthophosphate at 215 stations. The mean concentration results are summarised in Fig. 5.7 and are shown on Map 5 in Appendix VI for the individual locations. There were 49 stations, where the mean result was greater to or equal to 0.03 mg/l P and of these 16 locations had mean concentrations greater than 0.05 mg/l P. The 49 locations are in the following counties: Carlow (4), Cavan (1), Clare (1), Cork N (1), Cork S (1), Galway (3), Kerry (3), Kildare (1), Laois (2), Leitrim (2), Limerick (3), Louth (2), Mayo (1), Meath (1), Monaghan (2), Offaly (1), Roscommon (5), Sligo (3), Tipperary N (4), Waterford (4), Westmeath (1), Wicklow (3). One sample taken in Co. Waterford in 1999 gave a value of 8.4 mg/l P, well in excess of the MAC of 2.2mg/l P for drinking water. This appeared to be an isolated contamination incident as the subsequent sampling

1998-2000

No. of stations

150

100

50

0

78

103

26

7

2

< 20   20 - 30   30 - 50   50 - 100   > 100

Chloride, mg/l Cl

Fig. 5.5 The number of sampling points with mean concentrations of Chloride in the ranges indicated.

Percentages

60

50

40

30

20

10

0

□ (1995-1997)
□ (1998-2000)

43

36

43

48

12   12

1   3

1   1

< 20   20 - 30   30 - 50   50 - 100   > 100

Chloride, mg/l Cl

Fig. 5.6 Comparisons of the percentage of stations with mean concentrations of Chloride in the ranges indicated between the 1995-1997 and the 1998-2000 period.

Fig. 5.7 The number of sampling points with mean concentrations of Ortho-phosphate in the ranges indicated.

Fig. 5.8 Comparisons of the percentage of stations with mean concentrations of Ortho-phosphate in the ranges indicated between the 1995-1997 and the 1998-2000 period.

in 2000 and 2001 respectively indicated levels of 0.031 and 0.009 mg/l P.

In general phosphate levels in groundwater are not a cause of concern in relation to drinking water supply. However, there are areas of the country where the levels of phosphate in groundwater may contribute to eutrophication of rivers and lakes particularly if they provide significant amounts of baseflow during the summer months. This interaction between groundwater and surface water is emphasised in the WFD and will be considered further in future monitoring programmes.

Comparison with the 1995-1997 Data The percentages of stations in each concentration range for the period 1995-1997 and 1998-2000 are shown in Fig. 5.8. In the 1995-1997 period, 89 per cent of the stations had mean values less than 0.04 mg/l P while in 1998-2000 the percentage decreased slightly to 87 per cent.

<u>Iron and Manganese Concentrations</u>

Iron is present in significant amounts in soils and rocks, principally in insoluble form. However, many complex reactions, which occur naturally in ground formations can give rise to more soluble forms of iron, which will therefore be present in water passing through such formations. Background levels vary considerably depending on the rock structure. Excessive concentrations of iron do not cause health problems but are of concern for aesthetic and taste reasons. Taste is not usually noticeable at iron concentrations below 0.3mg/l. Laundry and sanitary ware will stain at Fe concentrations above 0.3 mg/l. As a precaution against storage of excessive iron in the body the Joint FAO/WHO Expert Committee on Food additives established a provisional maximum tolerable daily intake of iron from all sources (PMTDI) in 1983. Allocation of 10 per cent of the PMTDI to drinking water corresponds to a concentration of 2 mg/l, which does not present a hazard to health. No health based guideline value for iron has been proposed by the WHO (WHO 1996).

Manganese is widely found in soils. Like iron, it may be present in solution in groundwaters due to reducing conditions and the excess metal will be deposited as the water is brought to the surface and re-aerated. The principal objection to the presence of relatively large concentrations of manganese in drinking waters is, again, aesthetic due to turbidity and taste, there being little significance for health. At concentrations exceeding 0.1 mg/l, the manganese ion imparts an undesirable taste to beverages and stains plumbing fixtures and laundry (Griffin A. E., 1960). The WHO has set a provisional health based guideline value of 0.5 mg/l

Mn, which should be adequate to protect public health (WHO, 1996).

Since organic pollution can lead to serious de-oxygenation of groundwater and provide reducing conditions to bring the two metals into solution, marked increase in levels of iron and manganese above background, can be considered indicative of such pollution.

Under the EU (Quality of Water Intended for Human Consumption) Regulations, 1988 (SI No. 81 of 1988), MACs of 200 μg/l and 50 μg/l have been set, respectively, for iron and manganese, equivalent to 0.2 mg/l Fe and 0.05 mg/l Mn.

Iron A total of 1026 samples were analysed for iron at 192 stations. The mean concentration results are summarised in Fig. 5.9 and are shown on Map 6 in Appendix VI for the individual locations. There were 34 stations where the mean concentration result exceeded the MAC of 0.2 mg/l Fe, of which eight had mean concentrations over 1 mg/l. Twenty individual samples from seven stations (Cavan (2), Kildare, Laois, Leitrim, Meath and Monaghan) had iron concentrations greater than 2 mg/l. The highest individual concentration was 19.4 mg/l for a location in Co. Leitrim.

Comparison with the 1995-1997 Data The percentages of stations in each concentration range for the period 1995-1997 and 1998-2000 are shown in Fig. 5.10. In the 1995-1997 sampling period 77 per cent of the stations had mean iron concentrations of less than 0.2mg/l compared to 83 per cent in 1998-2000. This represents a relative increase in the percentage of stations with mean concentrations greater than 0.2 mg/l and a relative decrease in the percentage of stations with values greater than 1.0 mg/l Fe in the 1998-2000 period.

Manganese A total of 1024 samples were analysed for manganese at 192 locations. The mean concentration results are summarised in Fig. 5.11 and shown for the individual locations in Map 7 in Appendix VI. There were 38 stations with mean concentrations greater than the 0.05 mg/l but only two stations with mean concentrations greater than 1 mg/l. Twenty seven individual samples from 10 locations (Dublin, Galway, Kerry, Laois, Limerick, Louth and Meath) had manganese concentrations greater that 0.5mg/l Mn. The highest individual sample concentration for manganese was 51.3 mg/l at a source in Limerick.

Comparison with the 1995-1997 Data The percentage of stations in each concentration range for the period 1995-1997 and 1998-2000 are shown in Fig. 5.12. The 1998-2000 sampling period showed and increase of 2 per cent in the number of sampling stations with mean concentrations of
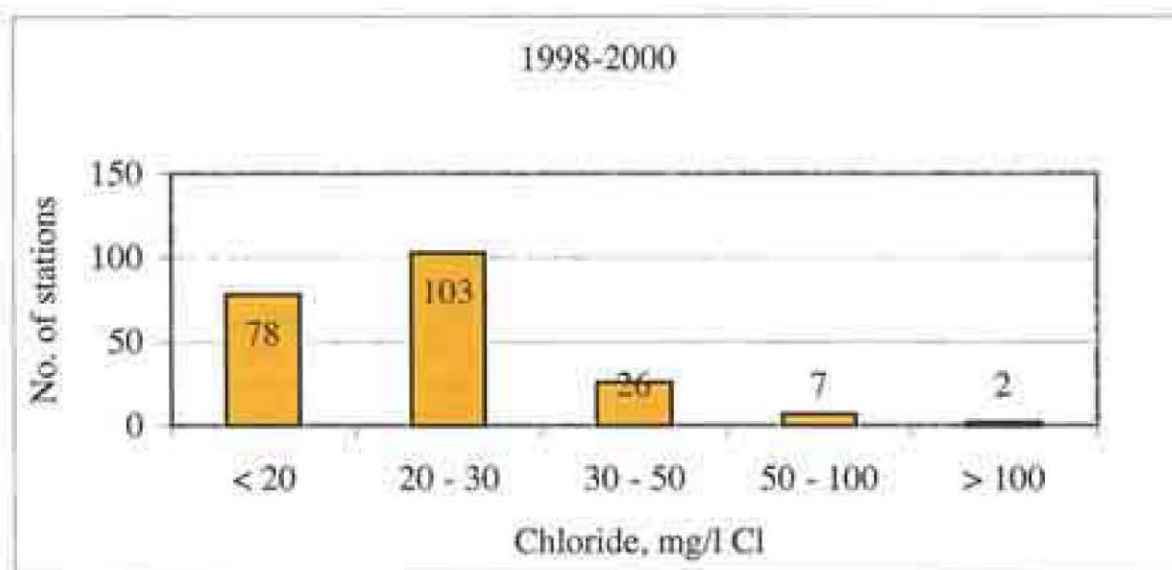
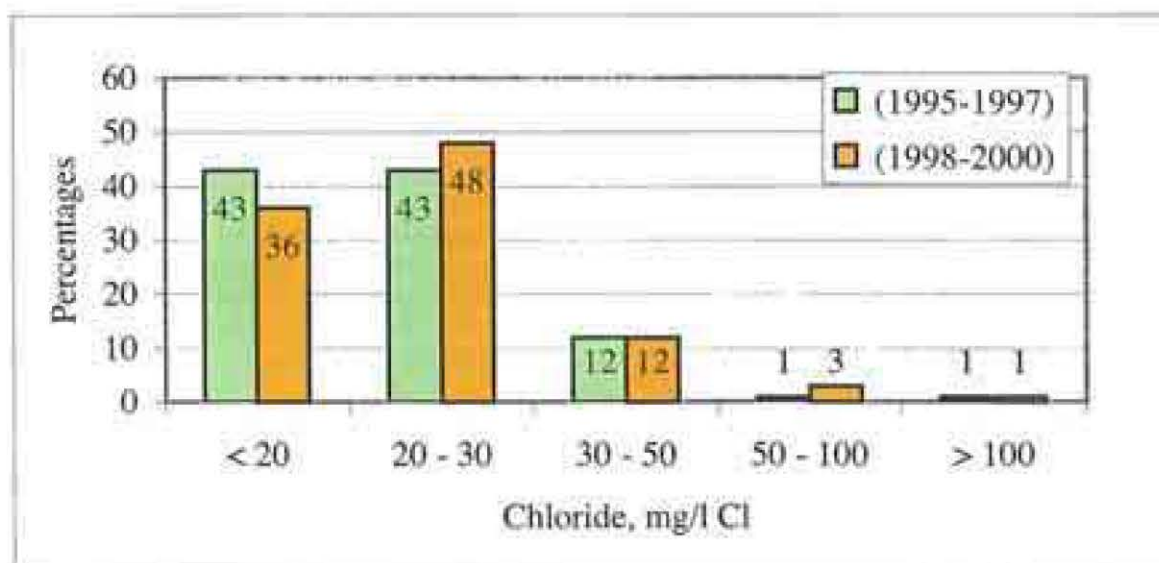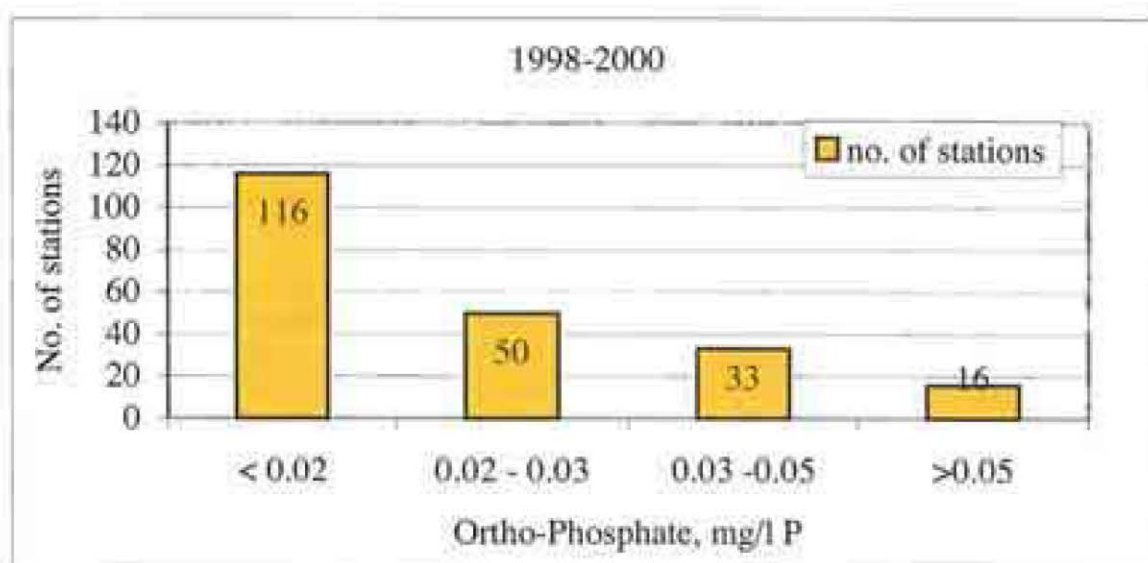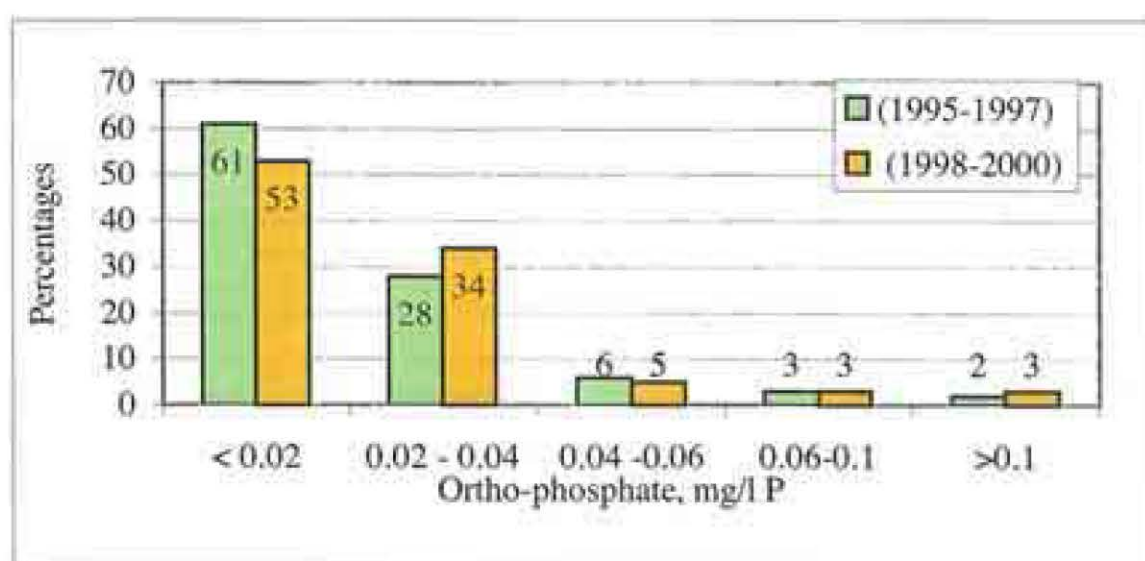Fig. 5.9 The number of sampling points with mean concentrations of Iron in the ranges indicated.

Fig. 5.10 Comparisons of the percentage of stations with mean concentrations of Iron in the ranges indicated between the 1995-1997 and the 1998-2000 period.
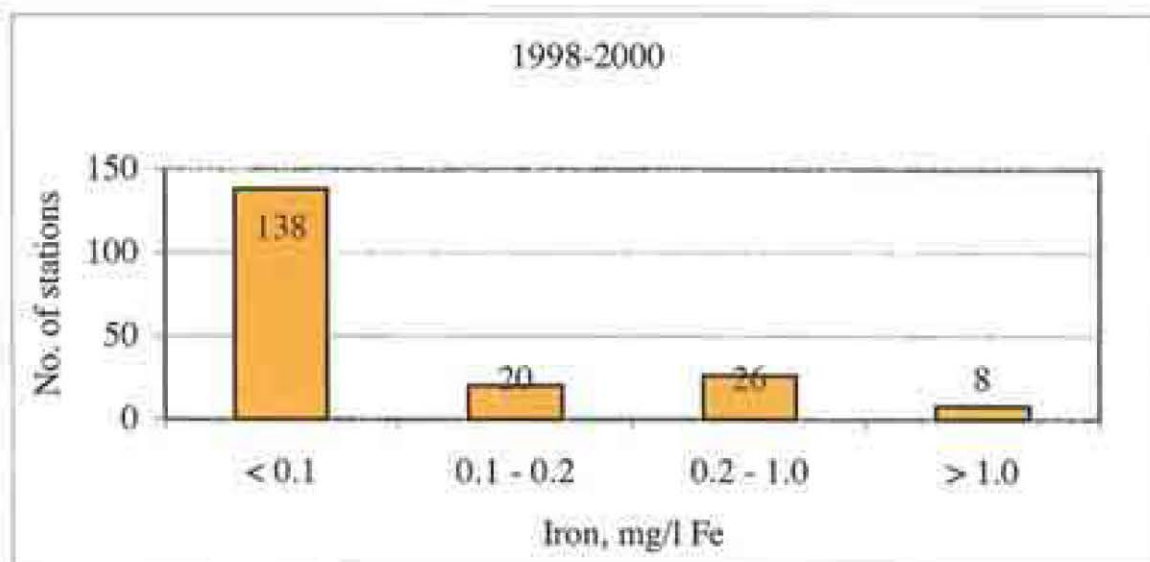
Fig. 5.11 The number of sampling points with mean concentrations of Manganese in the ranges indicated.
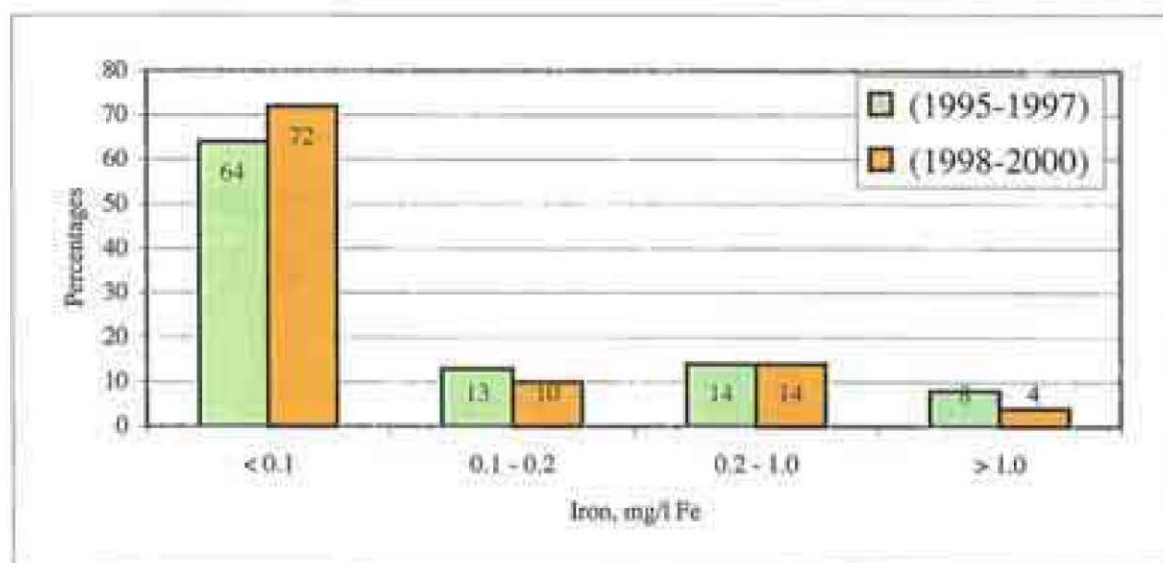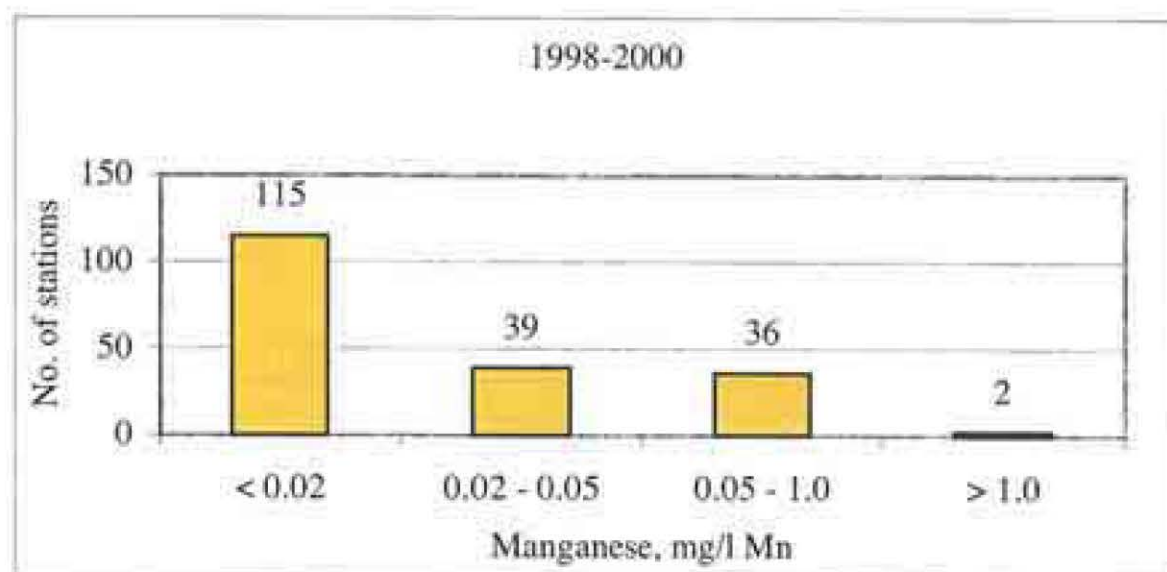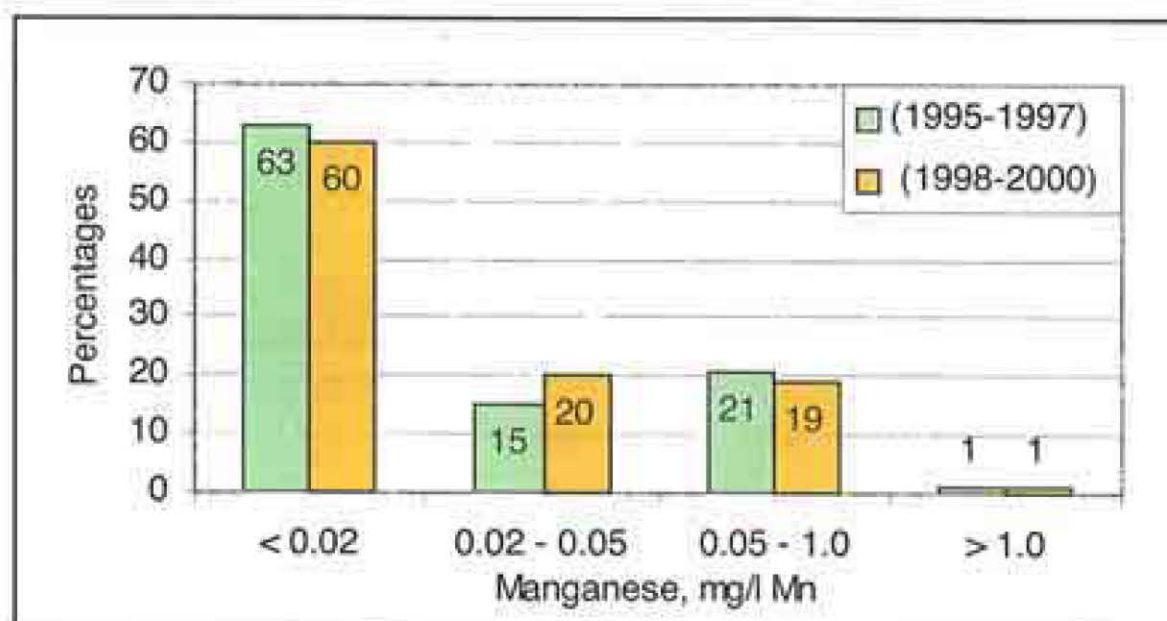

Figure 5.12:
Comparisons of the percentage of stations with mean concentrations of Manganese in the ranges indicated between the 1995-1997 and the 1998-2000 period.

manganese less than 0.05 mg/l Mn compared to the 1995-1997 period.

These results show that iron and manganese contamination of groundwaters is relatively common and likely to cause problems for consumers in certain geological areas unless appropriate pre-treatment is used. In general, samples with high concentrations of one metal usually also had elevated levels of the other metals indicating reducing conditions.

Bacteriological Examination of Samples
The 1995-1997 EPA groundwater surveys (Lucey et al. 1999) and earlier studies (e.g. Daly, D. 1994, Daly, E. and Woods, 1995) indicate that the main groundwater quality problems are associated with local microbiological contaminants rather than chemical contamination. In practice, faecal coliforms (e.g. *E. coli*) are the main microbiological contamination indicators analysed but other microbiological contaminants could be significant (e.g. viruses and the protozoan *Cryptosporidium*). The presence of *E. coli* is evidence that faecal contamination has occurred and is an indication that there is a potential presence of pathogenic micro-organisms i.e. those organisms capable of infecting or of transmitting disease. In general the majority of private groundwater supplies do not undergo any treatment prior to use.

Sources of *E. coli* include septic tank effluent, agricultural organic wastes and landfill sites. The natural environment, particularly soils and subsoils, can be effective in removing bacteria and viruses by predation, filtration and absorption. However not all areas are naturally well protected. High risk situations include karst areas, sands and gravels with a low clay content and a high watertable and extremely vulnerable fractured aquifers which allow the rapid movement of contaminants into groundwater with minimal attenuation. While the presence of clayey subsoils, tills and peat will, in many instances, retard the vertical migration of microbes, preferential secondary flow paths such as cracks in clay materials can allow the filtering effect of the subsoils to be bypassed.

From the perspective of human use and consumption of groundwaters, the most important consideration is the absence of pathogens. These organisms are not native to aquatic systems and usually require an animal host for growth and reproduction. They can survive and can be transported in natural water systems. The delineation of source protection areas (DELG *at al.*, 1999) is based on the premise that in some circumstances, bacteria and viruses can live longer than 50 days in groundwater. Pathogens include bacteria (e.g. faecal streptococci and *Salmonella*), viruses, protozoa and helminths (parasitic worms).

A total of 676 samples were examined for faecal coliforms at 134 monitoring stations. Positive counts were obtained in 257 (38%) of the individual samples, 49 of which exceeded 100/100ml (Fig. 5.13). Some of the stations at which nil counts were obtained had positive counts on other occasions. In 135 samples taken at 49 stations faecal coliform counts greater than 10/100 ml were recorded; this figure can be used as a threshold indicator of gross contamination. These samples were taken in Counties Cavan (2), Galway (12), Kilkenny (1), Laois (1), Leitrim (2), Longford (1), Louth (2), Mayo (6), Roscommon (12), Sligo (6), Tipperary S (1), Westmeath (3). This relatively widespread occurrence of coliform contamination emphasises the importance of disinfection of drinking water drawn from groundwater sources and demonstrates the importance of adequate borehole siting and construction.

Some coliforms grow naturally in soil and are not of faecal origin. A total of 666 samples at 132 monitoring stations were examined for total coliforms of which 395 showed positive counts. Faecal coliforms were not found in all samples with positive total coliform counts.Any indication of contamination must be regarded as a matter of gravity and the circumstances promptly investigated. This matter has been addressed in detail in the annual reports on drinking water quality published by the Agency (e.g. EPA, 1999).Comparison with the 1995-1997 Data The percentage of stations in each concentration range for the period 1995-1997 and 1998-2000 are shown in Fig. 5.14. The number of samples which tested positive for faecal coliforms increased from 34 to 38 per cent between the two sampling periods.
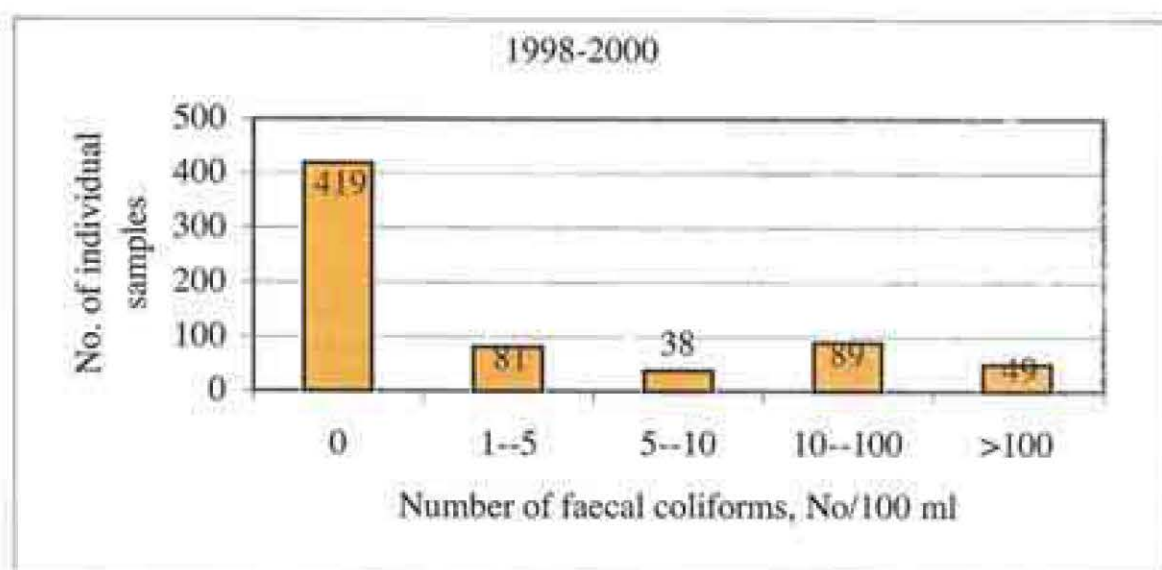
Fig. 5.13 The number of individual samples with mean counts of faecal coliforms in the ranges indicated.
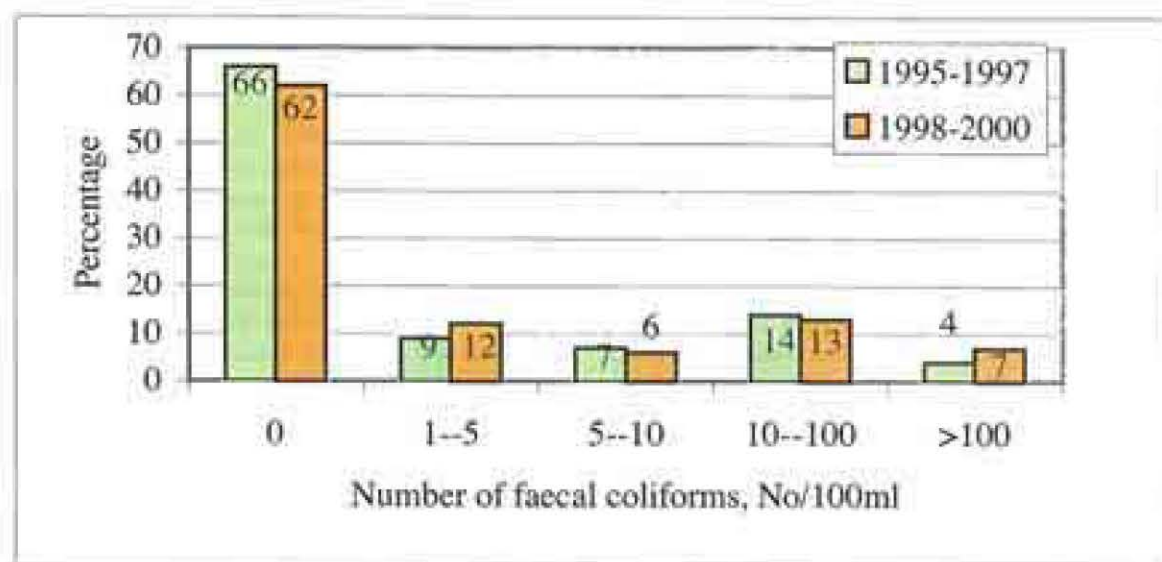
Fig. 5.14 Comparisons of the percentage of samples with mean counts of faecal coliforms in the ranges indicated between the 1995-1997 and the 1998-2000 period.

3.
Nitrate Vulnerable Zones – Latest Developments
Pat Duggan, Dept. of Environment and Local Government.

INFORMATION PAPER

ON

GOOD AGRICULTURAL PRACTICE

AND PROTECTION OF THE ENVIRONMENT

DEPARTMENT OF THE ENVIRONMENT
AND LOCAL GOVERNMENT

18 FEBRUARY 2002

DEPARTMENT OF THE ENVIRONMENT AND LOCAL GOVERNMENT

INFORMATION PAPER ON GOOD AGRICULTURAL PRACTICE AND PROTECTION OF THE ENVIRONMENT

Introduction

1. In the course of discussions on 22 January 2002 in relation to the strengthening of environmental protection in the context of agriculture, representatives of the Irish Farmers Association asked for a written statement as to the reasons in support of an "all areas" or "whole territory" approach to good agricultural practice and the Nitrates Directive. This paper was prepared in response to that request. In addition to the specific information requested, the paper sets out some general background information and some additional information in question-and-answer format.

Environmental Protection in Agriculture

2. There is a need for strengthened measures to protect the environment, particularly water quality, against pollution from agricultural activities and to give effect, insofar as agriculture is concerned, to certain EU Directives in relation to waste and water quality, including the Nitrates Directive. Agriculture is the last major sector remaining largely unregulated in terms of environmental protection. Industrial discharges to water, for example, have been subject to permits under the Local Government (Water Pollution) Acts since the 1970s. Controls on industrial activities were intensified by the introduction of IPC licensing under the Environmental Protection Agency Act 1992. Modern, statutory standards were applied to discharges from local authority sewage treatment plants in 1994 under Urban Waste Water Treatment Regulations and the EPA was assigned responsibility for supervising the activities of local authorities. A massive programme of investment is underway in relation to waste water treatment facilities under the National Development Plan 2000-2006. All waste management activities (other than recovery of agricultural waste) were made subject to licensing by the EPA (or by local authority permits) under the Waste Management Act 1996. The marketing of phosphate-based domestic laundry detergents has virtually ceased (90% phased-out) under a voluntary agreement made in 1999 with the detergents industry.

3. Numerous and significant measures for additional environmental protection have been applied in agriculture in recent years and significant investment has been made by farmers in waste storage

facilities and other infrastructure for pollution prevention. These measures are described more fully later. Farming remains nevertheless as a major source of environmental pollution, particularly in terms of water pollution and emissions to the atmosphere. The achievement of strengthened environmental protection in agriculture can best be done by Regulations which would provide a statutory basis to promote and support the application in all areas of established standards of good agricultural practice (GAP) for protection of the environment. Regulations for GAP would go most of the way towards implementing, for example, those provisions of the Framework Waste Directive and the Nitrates Directive which require the establishment of rules in relation to the management of certain agricultural wastes and other fertilisers. It would be possible, in the context of such Regulations, for the livestock manure limits specified in the Nitrates Directive to be applied to all areas or only to certain designated areas (i.e. "vulnerable zones"). The need to address these matters is given added urgency by the commitment given by Ireland, in the context of the CAP Rural Development Plan, to determine by end 2001 the areas to which the Nitrates Directive will be applied. Given that this deadline has now elapsed, there is a need for early decisions and action.

Agricultural Waste

4. Agriculture generates some 64 million tonnes per annum of waste or 80% of all waste (80 million tpa) generated in Ireland. (These data relate to agricultural waste requiring management and do not include the significant quantities of waste deposited on land directly by farm animals.) The organic load generated by agricultural livestock in Ireland is equivalent to that of a human population of 68 million persons. The vast bulk of agri-waste is in the form of animal manures and slurries which are applied to land. It is envisaged that landspreading will continue as the preferred treatment route for such wastes. This is an appropriate and sustainable approach provided that landspreading and related activities are carried on in accordance with good agricultural practice.

5. Unlike almost all other forms of waste, the management of agricultural waste is at present largely unregulated (i.e. it does not require a licence or permit and is not subject to detailed Regulations, with certain exceptions e.g. intensive pig/poultry units) even though –

- there is significant potential for water pollution and other environmental degradation arising from the management of such waste and related agricultural activities, and
- the application of certain controls on such activities is required by European legislation.

3

Agricultural Activities

6. The agricultural activities which give rise to water pollution are mainly :-

- inadequate farmyard management e.g. in relation to provision of waste storage facilities of adequate capacity, interception of soiled water, diversion of unsoiled surface water

- application of fertilisers (organic and chemical) to land in an inappropriate manner e.g. on wet or frozen land, too close to watercourses, in wet weather conditions, application of excessive quantities of liquid, on land sloping steeply towards watercourses, and

- over-application of fertilisers (organic or chemical) to land: increased levels of nutrients in the soil increase the incidence / risk of nutrients leaching from soil to water.

Codes of Practice

7. Guidelines for good agricultural practice have been established with the agreement of the main farming organisations. The established guidelines include –

- the Code of Good Agricultural Practice to Protect Waters from Pollution by Nitrates (CGAP): this code was issued in 1996 jointly by the Department of the Environment and Local Government (DELG) and the Department of Agriculture, Food and Rural Development (DAFRD), and

- the booklet on Good Farming Practice issued by the DAFRD in 2001 in the context of the CAP Rural Development Plan and Direct Aid Schemes.

8. A Protocol on Roles and Responsibilities of Fisheries Board Staff and Farmers was established in 2001.

9. The Code of Good Agricultural Practice was developed in consultation with, and with the agreement of, the main farming organisations, which also support and promote its implementation. These guidelines provide clear guidance as to the appropriate standards for good agricultural practice. Compliance with the guidelines would effectively prevent water pollution and other adverse effects on the environment. However, the guidelines are generally voluntary or apply only as administrative conditions in attached to agri-assistance schemes. Compliance is not legally enforceable. Non-compliance is not an offence. Compliance with the guidelines is uneven and, in the interests of promoting more

consistent performance by all farmers and strengthening measures for environmental protection, there is a need to convert the main provisions of the guidelines into enforceable Regulations.

Water Quality

EPA est 84% N less to agriculture. Coastal waters is due to agriculture.

10. Water quality in Ireland is generally good in comparison with that in most European countries but has deteriorated significantly from the position which prevailed when surveys commenced in 1971. In the intervening three decades, for example, the extent of river channel affected by pollution has significantly increased and some 30% of river channel is now affected by eutrophication, to a greater or lesser extent. Eutrophication of inland freshwaters has been identified by the Environmental Protection Agency (EPA) as "probably Ireland's most serious environmental pollution problem". In addition, in March 2001, the EPA assessed 17 estuarine waterbodies as eutrophic or potentially eutrophic. A recent report by the EPA on water quality in Ireland in the period 1998-2000 indicated a welcome halt to the continuing decline, and noted some improvement in river quality but pointed to the need for further major programmes to eliminate water pollution.

11. Eutrophication arises mainly from excess inputs of phosphorus from sources such as farming, sewage and industry. Excess inputs of nitrates also contribute to eutrophication, especially in saline waters i.e. estuaries, coastal and marine waters. The EPA estimates that agriculture is the source of -

   * 73 % of all inputs of phosphorus to waters, and
   * 82% of all inputs of nitrates to waters.

Nutrality most significant problem in Ireland

Eutrophication is the overenrichment of waters by nutrients causing excessive plant growth and consequential oxygen depletion in waters thereby reducing water quality and the capacity of the affected waters to sustain flora and fauna. Eutrophication occurs when sufficient quantities of phosphorus, nitrogen, carbon and trace elements are available in the water to generate and support excessive plant growth.

- high O₂ in day v. low in night

12. Pollution of water by nitrates and by total and faecal coliforms is also significant. The most recent EPA report on the quality of **drinking water** relates to the year 2000. This report indicates a compliance rate of just over 99% with the prescribed standard for nitrates (50 mg/l nitrates) in drinking water supplies (public and private), but also that breaches of the standard were recorded in ten, widely-spread counties (Carlow, Cavan, Cork, Galway, Kildare, Kilkenny, Louth, Mayo, Offaly, Waterford). It also indicates widespread contamination of group water schemes (i.e. a compliance rate of less than 80% for faecal coliforms) in 12 counties, mainly in western areas. In March 2001 the EPA completed an assessment of the trophic status of estuaries and bays, which identified

Medium river Quality doubled since 97

a total of seventeen estuarine waterbodies as eutrophic or potentially eutrophic due to excess inputs of phosphorus and/or nitrogen. The recent EPA report (issued January 2002) on **Water Quality in Ireland 1998-2000** indicates that, nationwide, 38% of groundwater samples showed faecal coliform contamination and high nitrate levels are reported in approximately 20% of well sampling stations. High ammonia values were found in groundwaters in 8 counties. All such contamination is likely to have organic waste of animal or human origin as source i.e. due mainly to poor control of animal wastes and poor siting or construction of septic tanks.

European Community Legislation

13. There is a need to give further effect to Directives of the European Community in relation to waste and water quality including –

   - **the Framework Waste Directive** (Directive 91/156/EEC of 18 March 1991)

   - **the Nitrates Directive** (Directive 91/676/EEC of 12 December 1991), and

   - **the Water Framework Directive** (2000/60/EC of 23 October 2000)

Waste Directive

14. The **Waste Directive** generally requires that a person carrying on an activity involving the recovery or disposal of waste must be licensed for that purpose and that the activity be carried on in accordance with the conditions specified in the licence. As an alternative to licensing, in the case of certain activities, waste recovery or disposal can be carried on in accordance with "general rules" by persons who are registered for the purpose. The Waste Directive has not yet been implemented in these respects in relation to organic agricultural waste (such waste being exempted from the licensing requirements of the Waste Management Act, 1996). The proposed Regulations for good agricultural practice are intended to rectify this deficiency and to provide for registration and general rules in relation to the recovery of agricultural waste as an alternative to the more onerous system of licensing of individual farmers. The proposed Regulations will relate to matters such as :

   - periods when the land application of certain types of fertiliser is not permitted
   - the quantity / type of fertilisers which may be applied to land
   - farmyard management.

The established standards of good agricultural practice will serve as the basis of the Regulations. It is envisaged that the Regulations will establish certain general requirements or principles but will allow for appropriate flexibility to reflect regional or local conditions arising from variations in climate, rainfall, soil conditions etc. It is envisaged, for example, that the Regulations should allow for different requirements in relation to waste storage capacity in different areas.

Nitrates Directive

15. The **Nitrates Directive** has the objectives of reducing water pollution induced by nitrates from agricultural sources and preventing further such pollution, with the primary emphasis being on the management of manures and other fertilisers. The Nitrates Directive requires Member States to –

- carry out monitoring of nitrate levels in waters
- establish a code of practice for farmers, and
- develop and implement action programmes to reduce and prevent pollution of waters by nitrates.

Action programmes must include rules (i.e. Regulations) in relation to matters such as those already mentioned for waste management. The Nitrates Directive and the Waste Directive, therefore, have essentially the same objectives in relation to agricultural organic waste.

16. The Nitrates Directive has generally been implemented in Ireland in terms of monitoring of waters, the establishment of a code of good agricultural practice (issued in 1996) and the implementation of a range of measures to protect water from pollution by agriculture. These consist of voluntary, obligatory and advisory measures. Voluntary measures include the REPS scheme, and the Farm Waste Management Scheme (previously the Control of Farmyard Pollution Scheme). Obligatory, statutory measures include local authority bye-laws and IPC licensing of intensive pig-rearing and poultry-rearing units. The DAFRD rules for Good Farming Practice are binding on farmers who are claiming aid under the direct payment schemes. Advisory measures include farm advisory services and nutrient management advice (Teagasc). There is a need, however, to formalise and extend these measures in the context of an action programme (or programmes) under the Nitrates Directive and to complement them with enforceable rules (i.e. Regulations). No such Regulations have yet been made. It is envisaged that the provisions of proposed Regulations on waste management (see paragraph 13 above), which will apply in all areas, will generally be adequate and appropriate to meet the requirements of rules the Nitrates Directive with the sole exception of the livestock manure limits under the Directive.

17. A particular requirement of the Nitrates Directive is that action programmes must impose limits on the amount of livestock manure that can be applied per hectare. The maximum amounts are 210 kgs of organic nitrogen per ha for the first four years and 170 kg of organic nitrogen per ha afterwards. A Member State may seek a derogation from these limits if such a derogation can be justified by reference to certain objective criteria. It is intended that the potential for seeking such a derogation will be considered and pursued in the light of successful implementation of the Directive. It is proposed, however, that a limit of 210 kg of organic nitrogen per ha should initially be applied i.e. for the first four years. The Nitrates Directive requires that action programmes be implemented in relation to the whole territory of a Member State or to areas designated by the Member State as vulnerable zones. A question arises, therefore, as to the area or areas to which the livestock manure limits should be applied i.e. whole territory or vulnerable zones.

18. Where a member State opts for the "vulnerable zones" approach, the zones must be designated by reference to –

    • waters which are, or may become, eutrophic due to enrichment by nitrogen compounds, or
    • waters which contain, or could contain, more than 50 mg/l nitrates unless appropriate action programmes are implemented.

    By reference to these criteria, there are extensive areas in the East, South East and South which would warrant designation as vulnerable zones (NVZs). It is considered, however, that there are compelling reasons for adopting a whole territory approach in preference to an NVZs approach (see paragraph 20 below).

Water Framework Directive

19. The **Water Framework Directive** requires that at least "good status" be achieved in relation to all waters by 2015 and that a programme of measures be established to achieve this objective. The programme must include measures to address all adverse impacts on water quality and quantity so as to achieve good water status by reference to biological criteria and physico-chemical criteria. "Good status" will be defined by reference to unpolluted conditions pertaining in the past or still prevailing at selected high-quality sites on rivers, lakes or estuaries or in groundwater bodies. Nitrates generally occur in unpolluted surface waters at background levels of between 5 and 10 mg / l nitrates. Any water quality standard to be established for protection of aquatic ecosystems against pollution by nitrates is likely to be considerably lower

than the level of 50 mg / l nitrates set for human health purposes in drinking water.

Advantages of Whole Territory Approach

20. The advantages of a whole territory approach include the following :-

advantages for farmers

- the designation of NVZs would discriminate between individual farmers who are carrying on exactly the same activities in different areas
- a whole territory approach provides a level playing-field, with all farmers subject to the same maximum application rates for livestock manure
- the livestock manure limits will impact only on those farmers operating at the highest levels of intensive farming e.g. above 2.5 dairy cows per hectare: while a majority of such farmers are likely to be located in areas which could be designated as NVZs, the whole territory approach ensures that all farmers operating at this level of intensity are subject to the same conditions
- a whole territory approach would help to ensure that any additional costs incurred by farmers would be properly distributed throughout the agricultural system
- the designation of NVZs may lead to (real or perceived) competitive variations e.g. in relation to farm production costs or agricultural land values
- a whole territory approach would help to preserve a clean, environmentally-friendly image for Irish agricultural produce and would avoid (real or perceived) "labelling" of particular areas as polluted or environmentally blighted
- the designation of NVZs involves the continuing uncertainty for farmers that additional areas might be designated in future years: advance planning is hindered by such uncertainty
- a whole territory approach is likely to provide the optimum basis for ensuring that appropriate derogations, consistent with environmental protection and sustainable development, can be achieved where warranted in relation to particular areas or crops or generally
- a whole territory approach would simplify the regulatory burden on farmers: it will enable several aspects of environmental protection to be addressed by one set of Regulations

advantages for the environment

- agricultural activities affect water quality in all areas: it is appropriate that similar standards / principles are applied in all areas

- pollution of waters by nitrates (i.e. at levels significantly exceeding natural background levels) is widespread
- an NVZ approach represents a fragmented, compartmentalised approach to environmental protection, addressing only one aspect (nitrates pollution) of one environmental medium (water)
- a whole territory approach is the most appropriate, holistic, comprehensive approach as it addresses a wide range of environmental impacts likely to arise from agricultural activities e.g.eutrophication of freshwaters or estuaries due to excessive inputs of phosphorus or nitrates, health-related risks due to elevated nitrates in drinking water sources, biological contamination of drinking water sources (including protection against possible contamination by E-coli 0157 which can cause serious illness): it would also support implementation of the National Climate Change Strategy (to reduce emissions of the greenhouse gas nitrous oxide), the Gothenberg Protocol and the National Emissions Ceilings Directive (2001/81/EC) (to limit emissions of ammonia)
- a whole territory approach will avoid creating pressures for export / relocation of livestock manure and pollution from one area to another
- a whole territory approach is more consistent with the requirements of the Water Framework Directive which requires that high standards be achieved in relation to all waters
- a whole territory approach is consistent with the actions being taken in all areas in relation to, for example, controls on industrial discharges and intensive piggery / poultry units, the massive programme of investment in waste water treatment facilities by the State and local authorities, supported by EU funds.

Example

21. An illustration of possible discrimination inherent in an NVZ approach is provided by comparing the operations of three farmers, all of whom are carrying on exactly the same activities of intensive farming and all of whom generate the same polluting outputs to the environment in terms of phosphorus and nitrates.

    A.   An intensive dairy farmer in one area might apply high levels of manure and fertiliser to land. Few if any of the neighbouring farmers operate at intensive levels. Local water pollution is slight.

    B.   An intensive dairy farmer in County Carlow carries on exactly the same activities. Many neighbouring farmers do likewise.

This causes significant pollution of waters by phosphorus and nitrates.

C.   An intensive dairy farmer in Limerick carries on exactly the same activities. Many neighbouring farmers do likewise. This causes significant pollution of waters by phosphorus but, instead of causing water pollution by nitrates, it contributes to Ireland's greenhouse gas emissions by the release of nitrous oxide into the atmosphere.

[Nitrous oxide is over **300 times** as potent a greenhouse gas as carbon dioxide and is released into the atmosphere when nitrogen is applied to wet / poorly drained land.]

In these examples, serious pollution of waters by nitrates occurs only in Example B i.e. the Carlow area. It would be basically unfair, however, to designate the Carlow area as an NVZ and apply certain conditions or restrictions on Carlow farmers but not the others given that all three farmers carry on exactly the same activities with the same environmental outputs, even though the effects of their activities are different in different areas.

GOOD AGRICULTURAL PRACTICE

SUPPLEMENTARY INFORMATION

What is the aim of the Department of the Environment and Local Government ?

To bring forward Regulations to provide for a higher level of protection for the environment against pollution arising from agricultural activities, particularly to protect water quality. The Regulations will aim mainly at giving a statutory basis to already well-established standards of good agricultural practice, and will form part of a programme of action to strengthen environmental protection in the context of agriculture. The detailed provisions of the Regulations remain to be determined in consultation with the main farming bodies and other interests.

Why is action required in relation to agriculture ?

Agriculture has been identified as the dominant source of water pollution by way of nutrient inputs to waters. Agriculture contributes 73% of all phosphorus inputs to water and 82% of nitrates inputs. These nutrients cause eutrophication (i.e. overenrichment) in surface waters. Eutrophication is probably Ireland's most serious environmental pollution problem. Excessive levels of nitrates also give rise to health risks for drinking water.

Why are Regulations needed ?

There is a need to strengthen the statutory basis for protection of water quality. The existing Codes of Good Agricultural Practice is voluntary. It is not possible to enforce compliance, or to prosecute non-compliance, in relation to a voluntary code of practice.

There is also a need to make Regulations to give further effect to a number of EU Directives on water quality and waste e.g. the Waste Directive 1991, the Nitrates Directive 1991 and the Water Framework Directive 2000

What aspects of agriculture will the Regulations address ?

The primary emphasis will be on promoting better management of manures and other fertilisers. The Regulations will address –

- Waste storage
- Farmyard management
- Nutrient management
- Landspreading of slurries e.g. manner, timing and rate of application

How were these aspects selected for attention ?

These aspects have been identified by many reports and studies as needing attention e.g. in the Lough Derg / Lough Ree Water Quality Monitoring and Management Project and in other catchment-based projects.

These are also the aspects which must be addressed in order to implement fully the Waste Directive and the Nitrates Directive

Will farmers be consulted in relation to the Regulations ?

Yes. It is intended that the proposed Regulations will be developed in consultation with the main farming organisations and other interests.

Will the Regulations apply nationwide ?

Yes, as is usual for national Regulations.

Will the Regulations impose the same requirements everywhere ?

No. The Regulations will establish certain basic requirements and principles which will apply nationwide. They will, however, allow for flexibility where this is appropriate to reflect local or regional variations in relation to soil conditions, climate etc. It is likely, for example, that different waste storage requirements should apply in different areas. Some basic principles for all areas might be that –

- the amount of slurry applied to land should not exceed the nutrient requirements of the soil, and
- adequate waste storage capacity will be provided to contain all waste generated for periods during which landspreading is not feasible.

How onerous will the Regulations be ?

An unduly heavy regulatory regime is not intended. The Regulations will aim mainly at giving a statutory basis to the well-established standards of good farming practice. These standards are already being exceeded by the 45,000 farmers participating in the REPS scheme.

Will a farmer have to get a permit to spread slurry on his own land ?

No. A licence or permit will not be required for the landspreading on a farm of organic waste generated on that farm. But all farmers will have to register with a local authority and provide some basic information on the location, nature and scale of their activities.Some additional controls may be required in relation to the acceptance and landspreading on a farm of waste from other farms or activities or the transfer of waste to other persons.

What will be the main impact on farmers arising from the proposed Regulations ?

The main financial impact for many farmers is likely to be a requirement to provide a certain minimum storage capacity for organic waste so as to promote storage, utilisation and distribution of such waste in a safe manner. There is known to be a significant deficit of waste storage capacity on Irish farms. The exact amount of required expenditure will depend in each case on the storage requirement established for each area and the storage actually available on the farm.

The provision of adequate storage capacity is an essential feature of sustainable farm management and good farming practice. It is essential for a farmer to have adequate storage to ensure that slurry does not need to be spread at inappropriate times due to lack of storage capacity.

Will financial assistance be available for farmers for this purpose ?

Grant support at a rate of 40% for such capital expenditure is available from DAFRD to farmers who meet certain qualifying criteria. Tax relief is also available to farmers for expenditure of a capital nature on such facilities.

Will farmers be compensated for extra operating costs arising from new Regulations ?

It is possible to provide assistance from State funds to support **capital** expenditure by farmers who may be required to provide additional waste storage capacity or other infrastructure. The EU rules on State aids to industry allow for such assistance up to certain levels of support.

It is not envisaged that compensation will be payable to farmers in respect of any additional **current or operational expenditure** (or loss of income) they may incur arising from compliance with legal requirements or the requirements of good agricultural practice. Any such compensation payments would be directly contrary to the polluter-pays principle.

Where a person is carrying on an activity which involves a risk of causing environmental pollution, the primary responsibility for ensuring that pollution does not occur rests with that person.

Agriculture is already supported to a significant extent from public funds. Some 56% of farming income (operating surplus) in Ireland is in the form of direct payments from public funds. In 1998, for example, the Department of Agriculture, Food and Rural Development spent about □2.4 billion in direct and indirect support to Irish farmers. This expenditure comprised over □1.27 billion in direct payments to farmers, some □635 million in market support measures (including export refunds and intervention) and □508 million in other measures and administration. Total EU financing for this expenditure amounted to over □1.9 billion.

How is it that payments can be made to farmers under REPS for environmental protection measures ?

The Rural Environment Protection Scheme (REPS) compensates and rewards farmers for additional expenditure incurred by them in carrying on a range of activities to a standard which **significantly exceeds** good agricultural practice or legal requirements.

How much time will farmers have to comply with new Regulations ?

The proposed Regulations will allow for appropriate phasing-in periods for different provisions. However, it is essential to achieve progress during the four-year period or it may become necessary to apply additional and reinforced requirements to comply with the Nitrates Directive.

Will REPS payments to farmers be affected ?

It is possible that payments to individual farmers under the Rural Environment Protection Scheme could be reduced by amounts of, perhaps, up to 5%. The exact impact, if any, on REPS payments will not be clear until the detailed provisions of Regulations are established. The operation of the REPS scheme will be kept under review by the the Department of Agriculture, Food and Rural Development in consultation with the European Commission.

Will the Regulations designate certain areas as Nitrate Vulnerable Zones?

The key provision of the Nitrates Directive in relation to NVZs is a limit on the amount of livestock manure that can be applied per hectare. The need for designation of NVZs will not arise if these limits are applied on a nationwide basis in conjunction with statutory standards of good agricultural practice.

What is the proposed limit for application of livestock manure ?

The Nitrates Directive allows a maximum amount of 210 kg of organic nitrogen to be applied per hectare during the first four years of operation. A limit of 170 kg organic nitrogen per ha applies thereafter, unless a derogation has been granted by the European Commission to the Member State in question. The potential for seeking such a derogation will be pursued in the light of the successful implementation of the Nitrates Directive.

How many farmers will be affected by this limit ?

The amount of 210 kg organic nitrogen is roughly equivalent to the amount of manure produced by 2.5 dairy cows. The average stocking rate on Irish farms is 1.4. livestock units (LU). (One dairy cow is generally equivalent to one LU.) The average stocking rate for intensive farms is 2.3 LU. The limit will therefore affect only those farmers who are operating **at the highest levels of intensive farming** i.e. some 4,000 farmers or 3% of all farmers. The limit of 170 kg organic nitrogen per ha could affect some 13,000 (10%) farmers. These are the most commercial, economically viable farming enterprises.

Farmers who operate at a high level of intensity should exercise a correspondingly high level of environmental responsibility. The 45,000 farmers participating in the REPS scheme are already operating within the 170 kg organic nitrogen per ha limit. Participation in REPS is expected to increase to some 70,000 farmers by end-2006. Compliance with the DAFRD rules of Good Farming Practice is already obligatory on farmers in the context of the CAP Rural Development Programme and Direct Aid Schemes.

Will an affected farmer be forced to reduce stocking levels ?

No. A farmer who does not hold adequate land to spread all waste generated on his / her own farm needs only to gain access to other spreadlands with surplus capacity for landspreading of livestock manure. This practice is already developing in relation to intensive pig-rearing facilities.

Are additional spreadlands available ?

It is understood (from Teagasc that adequate spreadlands are available locally (e.g. within the same DED) for organic waste from dairy farming in virtually all parts of Ireland. In the case of pig-rearing facilities, which are concentrated in North Cork and Cavan, longer travel distances will generally be involved.

Is EU-Funding of Agri-Assistace Under Threat ?

Payments under agri-assistance schemes are funded by the Irish Exchequer in the first instance and all such schemes will continue to operate.

In the context of EU support for Irish funding, however, Ireland must honour the commitment which it gave in the context of approval of the CAP Rural Development Plan 2000-2006 to give early effect to any aspects of the Nitrates Directive which remain to be implemented. A commitment was given in particular to identify by end 2001 the areas in which action programmes will be applied. It is likely that any revisions of agri-assistance schemes which might be sought by Ireland will not be considered by the EU Commission until this commitment is honoured. As recently as 12 February 2002, the EU Commission indicated that it would not approve any adjustments of the Compensatory Allowances Scheme (one of the four Measures under the CAP RDP) pending compliance with the undertakings for further implementation of the Nitrates Directive. The EU Commission also indicated that the continued funding of all four Measures under the CAP RDP would be contingent on compliance with the undertakings given in the Plan in respect of the Nitrates Directive.

Besides Regulations, what other measures will a programme of action contain to address pollution from agriculture ?

A wide range of actions is already in place aimed at providing for a higher level of protection for the environment against pollution arising from agricultural activities e.g. —

- the REPS scheme
- the Code of Good Farming Practice being applied by DAFRD under Agri-assistance schemes
- the grants available under the DAFRD Farm Waste Management Scheme
- Tax relief in relation to expenditure of a capital nature on certain farm infrastructure
- Teagasc advice booklets on nutrient management
- IPC licensing of intensive pig and poultry-rearing facilities
- farm inspections by local authorities
- implementation and enforcement actions by local authorities under the Water Pollution Acts and the Waste Management Acts
- measures taken in the context of local authority Catchment Based Water Quality Monitoring and Management Systems and River Basin Management Projects
- permission required under the Planning Acts.

Any new or formalised action programme will include the proposed Regulations together with measures of a type already in operation.

Is there a sound scientific basis to support the charges that agriculture is a major polluter ?

There is an abundance of scientific evidence to demonstrate that farming is a major source of water pollution. Research and surveys by the EPA and other bodies, such as Teagasc, over a long period of time clearly point to agriculture as the dominant source of water pollution. This conclusion is supported by detailed research carried out in particular areas in the context of catchment-based water management projects e.g. the Derg / Ree project. Similar studies in other countries confirm this situation. This is not surprising given the huge quantities of organic waste generated by agriculture and the acknowledged shortcomings in the management of such wastes e.g. inadequate storage capacity, landspreading at inappropriate times, over-application of slurries and fertilisers. Agriculture generates some 64 million tonnes per annum of waste or 80% of all waste, an organic load equivalent to that of a human population of some 68 million.

The most recent comprehensive data published in relation to water quality, and sources of water pollution, are contained in the EPA report on **Water Quality in Ireland 1998-2000** (January 2002). The data in the report are supported by some very detailed research and monitoring programmes (some in place for 30 years) particularly in the area of agriculture-related water pollution.   The conclusions in the report are drawn from –

1.   sampling, analysis and field surveys on many rivers, lakes, canals by a wide range of scientific personnel in many State and private laboratories employing a wide range of survey and analytical techniques e.g. ecological assessments at 3,200 sites, over 350,000 physico-chemical measurements on river water samples, chemical and bacteriological examination of groundwaters, surveys of toxic substances requiring advanced chemical analysis, remote sensing, sampling of cetacean tissue for microcontaminants

2.   detailed reports and research projects, which are cited in the text and can be referred to for clarification or additional information

3.   internationally peer-reviewed scientific papers.

Officials of the Department of the Environment and other appropriate bodies are meeting experts on behalf of farming interests to clarify any issues.

In addition to the abundance of scientific evidence outlined above, the **precautionary principle** which underlies environmental policy does not require that a causative link be proved between a suspected source of environmental damage and the damage caused. The European Treaty states,

for example, that Community policy on the environment shall be based on the precautionary principle and on the principles that preventive action should be taken, that environmental damage should as a priority be rectified at source and that the polluter should pay.

Who will enforce the Regulations ?

The Regulations will be enforced by local authorities in the context of their functions under the Waste Management Acts and the Local Government (Water Pollution) Acts. The proposed Regulations would strengthen, support and facilitate the existing enforcement actions of local authorities to protect water quality.

Most farmers comply with good farming practice. Why are new Regulations needed ?

There is a need for Regulations to enable action to be taken against the small minority of persons who persistently carry on activities which are contrary to good agricultural practice and which involve a high risk of causing water pollution. There is a need, for example, to prohibit the landspreading of slurry —

- at certain times e.g. the wettest months of the year, or
- at any time on wet or frozen ground, or
- in excessive quantities, or
- too close to wells, rivers or other watercourses.

Existing legislation is inadequate in that it places too high a burden of enforcement and proof on a local authority i.e. to prove that water pollution actually occurredd and was caused by the activities of a particular person at a particular place and time. Legislation should reflect more fully the basic principle that, where a person is carrying on an activity which involves a risk of causing water pollution, primary responsibility for **preventing** water pollution rests with that person.

Existing legislation prohibits a person from polluting waters. Is this not sufficient ?

Existing legislation of general application is not sufficiently explicit in terms of requirements for good agricultural practice. Farming practices are generally unregulated except, for example, in the case of –

- limited numbers of IPC-licensed intensive pig and poultry-rearing facilities

- farming in areas where bye-laws have been made by local authorities under the Water Pollution Acts (five local authorities have made such bye-laws in relation to limited areas)
- individual farmers on whom formal notices have been served by a local authority under the Water Pollution Acts (e.g. section 12 notices for pollution prevention, section 21A notices requiring preparation of nutrient management plans).

END

4.

The Drinking Water Regulations 2000 – Outline of Requirements
Richard Foley, Environment Unit, Enterprise Ireland.

The New Drinking Water Regulations

Introduction

Although a new directive on drinking water was ratified by the council of ministers in 1998, drinking water regulation in Ireland will continue to be governed by the 1988 regulations until January 1, 2004. This is despite the fact that new regulations, implementing the provisions of the new directive, were made by the Minister for The Environment and Local Government (DoELG) in December 2000.

The first directive (*Council Directive of 15 July 1980 Relating to the quality of water intended for human consumption (80/778/EEC)*) was ratified and published in 1980 and implementing regulations entitled *European Communities (Quality of Water Intended for Human Consumption) Regulations, 1988* were made by the Minister in Statutory Instrument 81 of 1988. This directive and implementing regulations will be effectively repealed in about two years time. The new directive is entitled *Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption*. The new regulations are entitled *European Communities (Drinking Water) Regulations, 2000* (Statutory Instrument 439 of 2000).

Why a new directive?

The preamble to the new directive lists the reasons why it was deemed necessary by the commission to replace 80/778/EEC. It is worth noting that originally consideration was given to amending 80/778/EEC, as happens to many directives (eg., natural mineral waters), but ultimately a complete new directive was the favoured proposal. There are of course some similarities between old and new, not least in that there are still quantitative standards for drinking water and monitoring requirements.

The reasons for making a new directive include:-
- Mandatory standards should be health based (risk assessment)
- The basis for standards should be founded on good science
- The precautionary principle should apply
- Application of the subsidiarity principle
- Scientific and technical progress (since 1980) should be recognised
- Assumption that 'Framework Water' directive will be adopted (it has been !)
- Action to be taken when standards not met; public information
- Reporting to commission

Exemptions

The 2000 regulations provides for an exemption from the regulations. An exempted supply is one which

(a) is provided from either an individual supply providing less than 10 m³/d on average or serving fewer than 50 persons and is not supplied as part of a commercial or public activity, or

(b) is used exclusively for purposes in respect of which the sanitary authority is satisfied that the quality of the water has no influence, either directly or indirectly, on the health of the consumers concerned.

Definitions

What is drinking water? A definition of drinking water is given in the 2000 regulations (Article 4(2)):-

Water shall be regarded as wholesome and clean if –
  (a) it is free from any micro-organisms and parasites and from any substance which in numbers or concentrations, constitutes a potential danger to public health, and
  (b) it meets the quality standards specified in Tables A and B in Part 1 of the Schedule.

A private water supply:-

means a water supply which is not in the charge or ownership of a sanitary authority

Application

The regulations apply to:-

  (a) all water, either in its original state or after treatment, intended for drinking, cooking, food preparation or other domestic purposes, regardless of its origin and whether it is supplied a distribution network or from a tanker,
  (b) all water used in any food production undertaking for the manufacture, processing, preservation or marketing of products or substances intended for human consumption unless the sanitary authority are satisfied that the quality of the water cannot affect the wholesomeness of the foodstuff in its finished form

Standards

Statutory Instrument 81 of 1988 divided the parameters into six groups:-

A Organoleptic Parameters
Colour, taste, odour and turbidity
B Physico-chemical Parameters (in relation to water's natural properties)
Temperature, pH conductivity, chloride, sulphate, sodium, potassium, aluminium, calcium, magnesium, dry residue
C Parameters Concerning Substances Undesirable in Excessive Concentration
Nitrate, nitrite, ammonium, kjeldahl nitrogen, hydrogen sulphide, hydrocarbons, phenols, boron, surfactants, THMs, iron, manganese, copper, zinc etc.
D Parameters Concerning Toxic Substances
Arsenic, cadmium, cyanide, chromium, mercury, nickel, lead, antimony, selenium, pesticides and related products, PAHs etc.
E Microbiological Parameters
Total coliforms, faecal coliforms, faecal streptococci, sulphite reducing clostridia, total bacterial counts etc.
F Softened Water
Total hardness, total alkalinity

The new regulations present the standards in a simpler form in that the 28 parametric values in Tables A and B are 'mandatory' and should not be exceeded. The values in Table C which are referred to as 'indicator parameters' and are linked to quality monitoring. The directive itself states *'as regards the parameters set out in ... the values need be fixed only for monitoring purposes and the fulfilment of the obligations imposed in Article 8 (Remedial Action)'.*

Schedule
Part 1 – Parameters and Parametric Values

Table A – Microbiological Parameters

Parameter	Parametric value
Escherichia coli (E.coli)	0 per 100 ml
Enterococci	0 per 100 ml

Table B – Chemical Parameters

Parameter	Parametric value	Unit	Notes
Acrylamide	0.1	µg/l	Note 1
Antimony	5.0	µg/l	
Arsenic	10	µg/l	
Benzene	1.0	µg/l	
Benzo(a)pyrene	0.010	µg/l	
Boron	1.0	mg/l	
Bromate	10	µg/l	Note 2
Cadmium	5.0	µg/l	
Chromium	50	µg/l	
Copper	2.0	mg/l	Note 3
Cyanide	50	µg/l	
1,2-dichloroethane	3.0	µg/l	
Epichlorohydrin	0.10	µg/l	Note 1
Fluoride	1.5	mg/l	Note 11
Lead	10	µg/l	Notes 3 & 4
Mercury	1.0	µg/l	
Nickel	20	µg/l	Note 3
Nitrate	50	mg/l	Note 5
Nitrite	0.50	mg/l	Note 5
Pesticides	0.10	µg/l	Notes 6 & 7
Pesticides – total	0.50	µg/l	Notes 6 & 8
Polycyclic aromatic Hydrocarbons	0.10	µg/l	Sum of concentrations of specified compounds; note 9
Selenium	10	µg/l	
Tetrachloroethene and trichloroethene	10	µg/l	Sum of concentrations of specified parameters
Trihalomethanes – total	100	µg/l	Sum of concentrations of specified compounds; note 10
Vinyl chloride	0.50	µg/l	Note 1

Note 1	The parametric value refers to the residual monomer concentration in the water as calculated according to specifications of the maximum release from the corresponding polymer in contact with the water.
Note 2	For the water referred to in sub-articles 6(a), (b) and (c), the parametric value to be met by January 1, 2004 is 25 μg/l. A value of 10 μg/l must be met by 25 December 2008.
Note 3	The value applies to a sample of water intended for human consumption obtained by an adequate sampling method* at the tap and taken so as to be representative of a weekly average value ingested by consumers and that takes account of the occurrence of peak levels that may cause adverse effects on human health. *The copper, lead and nickel parameters shall be monitored in such a manner as the Minister shall determine from time to time.
Note 4	For water referred to in sub-article 6(a), (b) and (c), the parametric value to be met by 1, January 2004 is 25 μg/l. A value of 10 μg/l must be met by 25 December 2013. All appropriate measures shall be taken to reduce the concentration of lead in water intended for human consumption as much as possible during the period needed to achieve compliance with the parametric value. When implementing the measures priority shall be progressively given to achieve compliance with that value where lead concentrations in water intended for human consumption are highest.
Note 5	Compliance must be ensured with the conditions that [nitrate]/50 + [nitrite]/3 $\leq$1, the square brackets signifying the concentration in mg/l for nitrate (NO_3) and nitrite (NO_2), is complied with and that the value 0.10 mg/l for nitrites ex water treatment works.
Note 6	Only those pesticides which are likely to be present in a given supply need be monitored. 'Pesticides' means: -- organic insecticides -- organic herbicides -- organic fungicides -- organic nematocides -- organic acaricides -- organic algicides -- organic rodenticides -- organic slimicides -- related products (*inter alia*, growth regulators) and their relevant metabolites, degradation and reaction products.
Note 7	The parametric value applies to each individual pesticide. In the case of aldrin, dieldrin, heptachlor and heptachlor epoxide the parametric value is 0.030 μg/l.
Note 8	'Pesticides — Total' means the sum of all individual pesticides detected and quantified in the course of the monitoring procedure.
Note 9	The specified compounds are: — benzo(b)fluoranthene — benzo(k)fluoranthene — benzo(ghi)perylene — indeno(1,2,3-cd)pyrene
Note 10	The specified compounds are: chloroform, bromoform, dibromochloromethane, bromodichloromethane. For the water referred to in sub-article 6(a), (b) and (c), the parametric value to be met by 1 January, 2004 is 150 μg/l. A value of 100 μg/l must be met by 25 December 2008. All appropriate measures must be taken to reduce the concentration of THMs in water intended for human consumption as much as possible during the period needed to achieve compliance with parametric value. When implementing the measures to achieve this value, priority must progressively be given to those areas where THM concentrations in water intended for human consumption are highest.
Note 11	The parametric value is 1.0 mg/l for fluoridated supplies. In the case of supplies with naturally occurring fluoride the parametric value is 1.5 mg/l.

" Values to be fixed only for monitoring purposes

Table C - Indicators Parameters

Parameter	Parametric value	Unit	Notes
Aluminium	200	μg/l	
Ammonium	0.30	mg/l	
Chloride	250	mg/l	Note 1
Clostridium perfringens (incl. Spores)	0	100 ml	Note 2
Colour	Acceptable to consumers and no abnormal change		
Conductivity	2500	μS/cm 20 °C	Note 1
pH	$\geq$6.5 & $\leq$9.5	pH units	Note 1
Iron	200	μg/l	
Manganese	50	μg/l	
Odour	Acceptable to consumers and no abnormal change		
Oxidisability	5	mg/l O_2	Note 3
Sulphate	250	mg/l	Note 1
Sodium	200	mg/l	
Taste	Acceptable to consumers and no abnormal change		
Colony count 22 °C	No abnormal change		
Coliform bacteria	0	per 100 ml	
Total organic carbon	No abnormal change		Note 4
Turbidity	Acceptable to consumers and no abnormal change; Note 5		

RADIOACTIVITY

Tritium	100	Bq/l	Notes 6 & 8
Total indicative dose	0.10	mSv/year	Notes 7 & 8

Note 1	The water should not be aggressive
Note 2	This parameter need not be measured unless the water originates from or is influenced by surface water. In the event of non-compliance with this parametric value, the supply shall be investigated to ensure that there is no potential danger to human health from the presence of pathogenic micro-organisms, e.g., cryptosporidium.
Note 3	This parameter need not be measured if the parameter TOC is analysed
Note 4	This parameter need not be measured for supplies of less than 10,000 m^3/d.
Note 5	In the case of surface water treatment, a parametric value not exceeding 1.0 NTU (nephelometric turbidity unit) in the water ex treatment works should be strived for.
Note 6	Monitoring frequencies to be set at a later date in Part 2 of the schedule.
Note 7	Excluding tritium, potassium-40, radon and radon decay products; monitoring frequencies, monitoring methods and the most relevant locations for monitoring points to be set at a later date in Part 2 of the schedule.
Note 8	A. The proposals required by Note 6 on monitoring frequencies, and Note 7 on monitoring frequencies and monitoring methods and the most relevant locations for monitoring points in Part 2 of the schedule shall be adopted in accordance with the Committee procedure laid down in Article 12 of Council Directive 98/83/EU. B. Drinking water need not be monitored for tritium or radioactivity to establish total indicative dose where, on the basis of other monitoring carried out, the levels of tritium of the calculated total indicative dose are well below the parametric value.

Article 6 defines the point of compliance for the parametric values in Part 1. For water supplied from a distribution network the point is the tap or taps at which the water is made available for human consumption.

Article 9(1) requires a sanitary authority to ensure that any failure to meet the parametric values specified in Part 1 of the schedule is immediately investigated so as to identify the cause of such failure. Article 9(2) requires a sanitary authority to take the necessary remedial action when monitoring shows that a parametric value is not being met. It is also obliged to use its enforcement action. In the case of a public water supply an action programme must be prepared and implemented not later that one year from the date of finalisation of the action programme if the non-compliance concerns Tables A and B in relation to a risk to public health. A period of two years for implementation is allowed for all of the parameters in Table B. In the case of non-compliance with the parametric values in Table C, the sanitary authority shall consider the risk to human health and if necessary take remedial action.

In the case of private water supplies where a non-compliance has been detected by a sanitary authority, the authority must serve a notice on the person or persons responsible for the supply to the effect that an action programme to remedy the non-compliance must be prepared. Measures to implement the action programme must be initiated within one year where the non-compliance concerns parameters in Tables A and B in relation to a risk to public health. A period of two years for implementation is allowed for all of the parameters in Table B.

A sanitary authority may apply to the EPA for a departure from the parametric values specified in Table B. The regulations prescribe the procedure and various time limits (Article 5).

Monitoring

Article 7 requires that regular monitoring be carried out by a sanitary authority and the frequencies are specified in Part 2 of the Schedule. There are two types of monitoring referred to as **check** monitoring and **audit** monitoring.

Part 2 – Monitoring

TABLE A – PARAMETERS TO BE ANALYSED

1.   CHECK MONITORING
The purpose of check monitoring is regularly to provide information on the organoleptic and microbiological quality of the water supplied for human consumption as well as information on the effectiveness of the drinking water treatment (particularly of disinfection) where it is used, in order to determine whether or not water intended for human consumption complies with the relevant parametric values laid down in Part 1 of this Schedule.

The following parameters must be subject to check monitoring:-
> Aluminium (Note 1)
> Ammonium
> Colour
> Conductivity

Clostridium perfringens (incl. Spores) (Note 2)
Escherichia coli (E. coli)
pH
Iron (Note 1)
Nitrite (Note 3)
Odour
Pseudomonas aeruginosa
Taste
Colony count 22 °C and 37 °C
Coliform bacteria
Turbidity

Note 1   Necessary only when used as a flocculant*
Note 2   Necessary only if the water originates from or is influenced by surface water*
Note 3   Necessary only when chloramination is used as a disinfectant*

*In all other cases, the parameters are in the list for audit monitoring.

2.   AUDIT MONITORING

The purpose of audit monitoring is to provide the information necessary to determine whether or not all the parametric values specified in Part 1 of this schedule are being complied with. All such parameters must be subject to audit monitoring unless it can be established by a sanitary authority, for a period of time to be determined by it, that a parameter is not likely to be present in a given supply in concentrations which could lead to the risk of a breach of the relevant parametric value. This paragraph does not apply to the parameters for radioactivity, which, subject to Notes 6, 7 and 8 in Table C in Part 1 of the Schedule will be monitored in accordance monitoring requirements adopted under the Committee procedure set out in Article 12 of Council Directive 98/83/EU.

TABLE B – Minimum frequency of sampling and analyses for water intended for human consumption supplied from a distribution network or from a tanker or used in a food-production undertaking.

Volume of water distributed or produced each day within a supply zone (Notes 1 & 2) m³		Check monitoring – number of samples per year (Notes 3,4 & 5)	Audit monitoring – number of samples per year (Notes 3 & 5)
≥10	≤100	2	Note 6
> 100	≤1000	4	1
> 1000	≤10000	4 + 3 for each 1000 m³/d and part thereof of the total volume	1 +1 for each 3300 m³/d and part thereof of the total volume
> 10000	≤100000		3 + 1 for each 10,000 m³/d and part thereof of the total volume
> 100 000			10 + 1 for each 25,000 m³/d and part thereof the total volume

Note1 A supply zone is a geographically defined area within which water intended for human consumption comes from one or more sources and water quality may be considered as being approximately uniform.

Note 2 The volumes are calculated as averages taken over a calendar year. The number of inhabitants in a supply zone may be used instead of the volume of water to determine the minimum frequency, assuming a water consumption of 200l/day/capita.

Note 3 In the event of intermittent short-term supply the monitoring frequency of water distributed by tankers is to be decided by the sanitary authority concerned.

Note 4 Where the values of results obtained from samples taken during the preceding two years are constant and are significantly better than the values specified in Part 1 of the Schedule, and no factor is likely to cause deterioration in the quality of the water, the number of samples specified in Table B of Part 2 of the Schedule and the reduction shall not (except in the case of a supply where the volume of water distributed or produced each day within a supply zone does not exceed 100 m³) be more than 50%.

Note 5 As far as possible, the number of samples should be distributed equally in time and location.

Note 6 To be determined by sanitary authority.

ARTICLE 9 – REMEDIAL ACTION (SUMMARY)

ANY FAILURE TO MEET STANDARDS IN PART 1 TO BE IMMEDIATELY INVESTIGATED BY SANITARY AUTHORITY TO IDENTIFY CAUSE OF FAILURE		
SANITARY AUTHORITY TO ENSURE THAT NECESSARY REMEDIAL ACTION TAKEN **ASAP** TO RESTORE QUALITY		
PUBLIC WATER SUPPLY		**PRIVATE WATER SUPPLY**
TABLES A & B NON-COMPLIANCE	1	**TABLES A & B NON-COMPLIANCE**
PREPARE ACTION PROGRAMME WITHIN 60 DAYS. ACTION PROGRAMME TO INCLUDE INTERIM MEASURES AS APPROPRIATE. CONSUMERS TO BE INFORMED UNLESS NON-COMPLIANCE IS TRIVIAL.	2	SANITARY AUTHORITY TO SERVE NOTICE ON PERSON/S RESPONSIBLE FOR SUPPLY WITHIN 14 DAYS.
IMPLEMENT ACTION PROGRAMME WITHIN I YEAR IN MATTERS WHICH PRESENT A RISK TO PUBLIC HEALTH.	3	PREPARE ACTION PROGRAMME WITH 60 DAYS. ACTION PROGRAMME TO INCLUDE INTERIM MEASURES AS APPROPRIATE. CONSUMERS TO BE INFORMED UNLESS NON-COMPLIANCE IS TRIVIAL.
WITHIN 2 YEARS FOR ALL STANDARDS IN TABLE B OTHER THAN THOSE REFERRED TO ABOVE (PREVIOUS BOX).	4	IMPLEMENT ACTION PROGRAMME WITHIN I YEAR IN MATTERS WHICH PRESENT A RISK TO PUBLIC HEALTH.
	5	WITHIN 2 YEARS FOR ALL STANDARDS IN TABLE B OTHER THAN THOSE REFERRED TO ABOVE (PRVIOUS BOX).
TABLE C NON-COMPLIANCE SANITARY AUTHORITY TO CONSIDER RISK TO PUBLIC HEALTH.	6	**TABLE C NON-COMPLIANCE** SANITARY AUTHORITY TO CONSIDER RISK POSED TO HUMAN HEALTH.
.WHERE RISK EXISTS TAKE REMEDIAL ACTION TO RESTORE QUALITY WHERE IT IS NECESSARY TO PROTECT PUBLIC HEALTH.	7	WHERE RISK EXISTS SANITARY AUTHORITY TO INITIATE 'NOTICE' 'ACTION PROGRAMME' & 'IMPLEMENTATION' PROCESS TO RESTORE QUALITY.
	8	RESPONSIBLE PERSON/S REQUIRED TO COMPLY.
	9	PENALTIES PRESCRIBED FOR FAILURE TO COMPLY WITH NOTICE

5.

Groundwater Quality – A Question of Perspective
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GROUNDWATER QUALITY

A QUESTION OF PERSPECTIVE.

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Abstract.

The primary concern of the engineer who commissions water supply schemes is the final treated water quality. The hydrogeologists and environmental scientists focus on the in-situ quality of the resource. A comparison of the bacteriological analyses of samples collected from surface water and groundwater show that raw groundwater quality is superior to that of surface waters, due to natural in-situ filtration of recharge. Statistics on poor drinking water quality are more a reflection of inadequate disinfection rather than environmental pollution. In areas of low to moderate vulnerability private well users are entitled to a water supply that is free from faecal contamination. In areas of extreme vulnerability groundwater quality becomes more similar to that of surface water. Effective disinfection can improve the quality of drinking water and protect the health of the consumer. Groundwater protection plans should increase their focus on the protection of the resource (aquifer) for the future, for the private well owner and for other receptors in the overall water environment. Groundwater supplies in areas of extreme vulnerability should be treated in the same manner as surface water. Nitrate contamination of Ireland's aquifers is not a problem at present. Protection measures must be implemented to prevent contamination by pesticides and nitrates, which are very costly to treat. The entire area of groundwater quality must be considered from both the environmental aspect and the public health aspect.

INTRODUCTION

Ireland as a country is blessed with an abundance of water in all of its many forms. Rain, rivers, lakes and aquifers. All these different phases of hydrological cycle interact. Water in the rivers and lakes constitute ecological habitats. Groundwater in the form of springs can feed ecologically important wetlands.

In addition to their in-situ environmental constitution, Irelands waters also form a resource that can be managed to provide public water supplies. Water is water regardless of whether it is collected rainwater, melted icebergs, surface water or groundwater. In Ireland water supplies are abstracted from both surface water sources and groundwater sources.

Current legislation and codes of practice consider water and water quality from two different aspects namely:

i)   The Environment.

ii)   Human Health.

The environmental aspect is best seen in the recently implemented Water Framework Directive which was conceived in order to get polluted waters clean again and to ensure clean waters are kept

clean. The Water Framework Directive has encouraged a more holistic approach to understanding the water environment. The directive encourages member states to consider water resources on a catchment basis as well as the interaction between the various phases. It is the water quality of the resource that is of concern.

Drinking water quality is approached more from the human health aspect. Steps must be taken to ensure that there is no adverse effect on the consumer. This risk management approach regards the consumer as the receptor.

DRINKING WATER SUPPLIES

The demand for potable drinking water is met by the public water supplies, group water schemes and private well supplies. In any water supply scheme there are three elements that must be considered jointly and separately.

i)	The Resource	-	The Aquifer, Lake or River
ii)	The Source	-	The Point of Abstraction from a Resource
iii)	The Consumer	-	The Supply at the Tap

Water quality is viewed differently and classified differently depending on which element of the supply system is being observed. In addition the quality is regarded differently if the resource or source is groundwater or surface water. Apart from being a potential source of water supply the resource is also integral part of the ecological environment and there are also environmental receptors that can be impacted by unnatural variations in quality.

The development of public water supplies is driven by the consumers within a community. The provision of public water supplies (and group water schemes) is managed by engineers.

The quantity required is calculated and a source is sought that will be capable of providing a sustainable supply of raw water. The raw water can be provided from surface water (river or lake abstractions) or from groundwater from wells, well fields or springs. The engineer is not overly concerned as to whether surface water or groundwater is used. He is concerned with sustainability, the raw water quality and the cost of provision.

Of primary concern is the protection of the health of the consumer. Treatment processes are incorporated into the system in order to ensure that the water received by the consumer meets the drinking water regulations.

BACTERIOLOGICAL QUALITY OF GROUNDWATER

Groundwater is the hidden or buried component of the hydrological cycle. It is seasonally recharged by rainfall and discharges to rivers as base flow and to springs. Vast volumes of groundwater are stored in the ground with the potential to be exploited as water supply. The overburden deposits that cover the bedrock in Ireland create a natural filtration system which polishes the quality of the infiltrating recharge removing bacteria and suspended solids. Consequently groundwater is characterised by its high clarity, low suspended solids and absence of harmful bacteria and would be regarded as having excellent quality.

Where the overburden cover is very thin or absent the natural filtration is not as effective, leading to

increased bacteria counts and problems with suspended solids after rainfall events. This would be the case in parts of the west of Ireland where there is an abundance of rock outcrop and thin soil cover. The presence of coliforms and faecal coliforms in samples collected from wells in these areas is very common. Whether their presence should be regarded as "natural" is debateable. Poorly constructed domestic wastewater treatment systems would be contributing to this statistic.

The rainfall that runs off to the rivers and streams does not undergo natural filtration and generally has much greater coliform counts. Similarly, problems with suspended solids are common in surface waters. In order to illustrate the effectiveness of natural filtration through the soil cover I have compared the bacteriological quality of water in its raw state both as groundwater and surface water.

Bacteriological Quality Surface Water vs Groundwater

In order to provide a comparison between the bacteriological quality of surface water and ground water, we looked at the bacteriological analyses carried out on groundwater samples collected by White Young Green in Ireland. All the samples were collected from wells that were constructed in connection with water supply.

The EPA have provided results of bacteriological monitoring of the raw water at 89 sampling points associated with surface water abstractions for public water supply in the south east region. A total of 577 samples were collected between 1998 and 2001.

The results of these surveys can be summarised as follows:

Surface Water

Total No. of Samples	577
Total No. of Sources	89
Samples that meet drinking water regulations in raw state	2
Samples with No Faecal Coliforms (but with total coliforms)	41
Samples with Faecal Coliform Counts between 1 and 100	208
Samples with Faecal Coliform Counts > 100 (of which 92 have > 1000)	326
Total	577

One random sample was then selected from each of the 89 surface water sources and the same analysis applied.

Samples that meet drinking water regulations in raw state	2 (2%)
Samples with No Faecal Coliforms (but with total coliforms)	7 (8%)
Samples with Faecal Coliform Counts between 1 and 100	35 (39%)
Samples with Faecal Coliform Counts greater than 100 (of which 16 (18%) are >1000)	45 (51%)

All of these public water supplies are chlorinated.

Groundwater

Total No. of Sources	72
Samples that meet drinking water regulations in raw state	58 (80.5%)
Samples with No Faecal Coliforms (but with total coliforms)	67 (93 %)
Samples with Faecal Coliform Counts greater than 100	1 (1.4%)
Samples with Faecal Coliform Counts greater than 1000	0

These analyses are taken from our in-house database and would cover the entire country although there is a greater representation of sources to the east and the midlands. These samples were collected at the end of 72 hour pumping tests on newly constructed wells. The wells would have been located some distance away from domestic wastewater systems.

In specific areas of high and extreme vulnerability the number of samples showing the presence of faecal coliforms would be expected to be much greater. In Roscommon, a survey of raw water quality in group water schemes showed the presence of presumptive faecal coliforms in 58% of the samples collected. It should be noted that the number of samples was considerably greater than the number of sources (i.e. wells were sampled frequently as part of a programme). It follows that even greater than 58% of the sources had presumptive faecal coliforms present at some time during the survey period.

The EPA survey of groundwater quality also shows a higher percent of faecal coliforms than the White Young Green database. Only 62 % of the samples had no faecal coliforms. The statistics from the EPA survey which is country wide may be showing a bleaker picture than is actually the case. The EPA admit that many of the sources in their sampling programme are domestic wells that could be impacted on by nearby poorly constructed wastewater treatment systems. Consequently the EPA survey reflects the quality of individual sources rather than the overall quality of the resource or aquifer. I believe that a monitoring well constructed within 200 metres of many of the wells showing the presence of faecal coliforms would be clear.

It is clear from this that terminology like "polluted" (used by hydrogeologists and environmental scientists) for groundwaters with faecal coliforms present can cause some confusion in the area of potable water supply. The engineer who is commissioning a water supply system would regard the "polluted" water as having excellent quality in comparison to most surface water sources.

BACTERIOLOGICAL TREATMENT

The disinfection of water is simple and relatively inexpensive. In the field of public water supply, risk management is concerned with the protection of human health. Effective disinfection can minimise this risk. The engineer regards in-situ groundwater source quality as raw water quality and treats accordingly. The degree of protection to the consumer will be same as if it was a surface water source. It is my opinion that all public water supplies (including group schemes) should be disinfected prior to distribution. The surveys of bacteriological quality of groundwater have shown that faecal contamination is very minor in comparison to the raw water abstracted from surface water sources. For the majority of public supply groundwater sources the "zap and a dash" disinfection (ultra violet treatment and a dash of chlorine to provide residual disinfection in the system) would be regarded as very effective. For sources with their inner protection areas classed as extremely vulnerable the raw water should be regarded in the same manner as a surface water supply.

The question of the effectiveness of chlorination against cryptosporidium and viruses applies equally to all pubic water supplies and those issues should be addressed in another forum. The concerns regarding the formation of trihalomethanes as a by-product of chlorination are generally unfounded in the context of the groundwater. The total organic carbon concentrations in groundwaters are well below the levels required to generate the problem. TOC concentrations in groundwater of greater than 2 mg/l are extremely rare.

It follows then that the occurrence of small quantities of faecal coliforms in the aquifer will not impact on the health of public supply consumers as long as that supply is effectively disinfected.

However, the environmental aspect which is concerned with minimising the presence of faecal coliforms in the natural groundwater (resource) takes into account the quality of supply that private domestic users can expect. Very few private wells are disinfected. A domestic well with 1 or 2 faecal coliforms and no treatment is unfit for human consumption under the drinking water regulations. I would suggest that aquifer protection plans should be more concerned with the protection of the resource (the aquifer). The strict guidelines for the design of small wastewater treatment systems should continue to be adhered to rigidly. The groundwater from private wells drilled in areas of low to moderate vulnerability can reasonably expect that their water supply be potable in its raw state. In areas where the overburden is thin or absent the groundwater may "naturally" contain some faecal coliforms. I would recommend that abstractions from private wells in such areas be disinfected. Ultra violet treatment would be considered appropriate and economic.

Statistics which show that an unacceptable percentage of water supplies don't meet the drinking water regulations can be reduced by introducing effective disinfection. Most of the problems with drinking water quality are a function of ineffective treatment rather than a widespread contamination of the groundwater resources. This does not reduce the onus on the environmental managers from introducing protective measures to protect the natural groundwater resource.

NITRATES IN IRISH GROUNDWATERS

The EPA have been monitoring nitrate concentrations in 216 sources throughout the country. In the period 1998 – 2000 only 1.5 % of these locations showed nitrate with concentrations above the drinking water regulations (SI 81 of 1988). 80% had concentrations less than the guide level of 25 mg/l as NO_3. There was no major change noted between this and the previous monitoring period. However, it was found that certain areas had higher levels than others.

The Dobris assessment (1995) quotes model concentrations of nitrate leaching from agricultural soils that indicate that 87% of the agricultural area in Europe has nitrate concentrations in the groundwater that are above the guide level value of 25 mg/l, and 22% that are above the 50 mg/l level. In many areas of Europe these levels are increasing with existing sources of drinking water having to be closed or being subject to expensive treatment processes at the expense of the consumer.

It is clear that in Ireland at present there is no widespread threat to groundwater supplies as result of nitrate contamination. However, it should be borne in mind that the nitrate directive is also concerned with environmental issues such as surface water quality and eutrophication.

All care must be taken to prevent nitrate concentrations in Irish groundwaters exceeding the MAC. Nitrate vulnerable zones will be mapped as part of the new water framework directive and measures will be undertaken in areas where the groundwater nitrate concentrations are approaching the MAC. The consequences of the nitrate concentration in groundwater exceeding the MAC in an area where an aquifer is being exploited for public water supply could be catastrophic.

The disposal of agricultural waste and applications of artificial fertilizers constitutes the greatest risk to the quality of our groundwaters yet it is the least controlled. The fact that the aquifers of this

country show lower nitrate concentrations than the rest of Europe is testimony to the natural attenuation of the overburden. In the absence of enforceable codes of practice it is only a matter of time before sources located in areas of extreme vulnerability will be threatened.

PESTICIDES

Very little is known about the occurrence of pesticide contamination in Irish groundwaters. There are limits for pesticides set out in the drinking water directives and associated regulations. Analysis for pesticides is rare. A comprehensive survey should be undertaken in order to establish whether a problem exists. If it is found that there is the potential for certain aquifers to be contaminated with pesticides then measures similar to the nitrate protection zones should be adopted. The new water framework directive could incorporate this evaluation and implement protection measures.

NATURAL IMPURITIES IN GROUNDWATER

While it has been shown that the quality of groundwater is generally vastly superior to surface water there are certain natural impurities that are associated with groundwater. The treatment required to render the raw water potable can be costly.

Iron and Manganese.

While groundwater quality is normally excellent there are certain impurities that occur naturally. The most common of these is elevated concentrations of iron and manganese that exceed the present regulations on the quality of water intended for human consumption (SI 81 of 1988). High iron and manganese concentrations are generally associated with groundwater abstractions from the dark limestone formations. The reason for the standard is the aesthetic problems associated with precipitation after the water is aerated on exposure to the atmosphere. The treatment is induced precipitation followed by filtration through a specific medium. In my experience it can be quite difficult to get the manganese to precipitate even with the addition of special catalysts. The question must be asked as to whether there is a problem if the manganese does not precipitate. I would suggest that if it can be proved that the manganese will stay in solution despite aeration and a rise in temperature (as in a washing machine) then it should not be required that the concentration be reduced. The new drinking water directive and regulations which are due to come into force next year do not have mandatory limits for iron and manganese concentrations. Iron and manganese are regarded as indicator parameters.

Ammonia and H_2S.

These can occur due to the anoxic conditions that prevail beneath the ground. Both of these parameters become reduced (particularly H_2S) when the supply becomes aerated. The cascading into a storage reservoir will remove the H_2S and to a lesser extent the ammonia. It is important to note that a break tank of some sort should be incorporated in the distribution system to ensure that the water becomes aerated before coming out of the consumers tap.

CONCLUSIONS

i)   The natural (raw water) quality of groundwater is generally vastly superior to that of surface water.

ii)   The risk management approach to public water supply is driven by the protection of the health of the consumer. The quality of the water coming out of the tap is of primary

concern. Whether the raw water is from surface water or groundwater is not a major issue.

iii)   Our investigation of surface water raw water source quality show that only 2.6 % of the sources examined meet the drinking water regulations in their raw state. The White Young Green database shows that 80% of groundwater sources have no coliforms or faecal coliforms. The EPA survey 1998-2000 gives this figure as 62%. It is felt that this figure may be skewed due to the possibility of domestic wells being contaminated by their own wastewater treatment systems. Prudent location of groundwater abstractions can reduce the possibility of contamination.

iv)   In areas where the protective overburden is thin or absent, the presence of some coliforms in the groundwater is not unusual. Routine disinfection of private well supplies using ultra violet radiation would provide an added protection to the health of private consumers.

v)   While contamination of groundwater by nitrates is not yet a serious problem in Ireland, protection schemes must be implemented. The cost of treating water supplies for nitrate removal is very high.

vi)   An assessment of background concentrations of pesticides in groundwater must be undertaken. Treatment is very costly.

vii)   The rate of travel of groundwater through the aquifer is much slower than surface water and therefore the consequences of a contamination event can be much longer lasting. Groundwater protection plans are used by planners to assess the risk of potential contamination by particular activities. Protection from potential contamination from hydrocarbons, leachates from landfills, and nitrates from landspreading can be provided by developing a clear understanding of the flow regime in the vicinity of abstraction wells.

viii)   An holistic approach that combines the protection of the resource, the source and the consumer must be developed in which similar methodologies are adopted regardless of whether the water is groundwater or surface water.

ix)   Enforceable controls on the disposal of agricultural wastes must be implemented as a protection against elevated nitrate concentrations and bacteriological contamination, which have both environmental and public health consequences.

6.

KEYNOTE: Current Knowledge on Groundwater Microbial Pathogens and Their Control
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Q. viruses

Cell culture methods ID few pathogens but show infectivity. PCR shows many pathogens but not infectivity.

Faecal bacteria data may not indicate previous occurrence. Faecal viruses more lethally in a city.

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potable

~55% GW supplies disinfect

~20% non-community "

Pathogens

Viruses: - viable for months
          - small enough to move long distances
            in all aquifers
          - very difficult to predict virus
            reduction from borehole to well

eg toraua

Bacteria: Can move in most aquifers
           Generally inactive for days
eg E. coli,   - Venge pathogens than viruses
salmonella,
cholera

Protozoa   Too large to move far in
eg Crypto unconsolidated.

Sources: Human sources dominate
          Urban - sewers. Rural - septic
          Baby mammals - bacteria protozoa farm.

Current knowledge on groundwater microbial pathogens and their control

Bruce A. Macler · Jon C. Merkle

Abstract Those who drink groundwater that has not been disinfected are at increased risk of infection and disease from pathogenic microorganisms. Recent studies have shown that up to half of all US drinking-water wells tested had evidence of fecal contamination. A significant fraction of all waterborne disease outbreaks is associated with groundwater. An estimated 750,000 to 5.9 million illnesses per year result from contaminated groundwaters in the US. Mortality from these illnesses may be 1400–9400 deaths per year. Control of these pathogens starts with source-water protection activities to prevent fecal contamination of aquifers and wells. These include assessment of wellhead vulnerability to fecal contamination and correction of identified deficiencies. Correction may include control of sources or rehabilitation of the well itself. Disinfection can serve as a useful barrier and is recommended as a prudent public-health policy for all groundwater systems.

Résumé Ceux qui boivent une eau souterraine non désinfectée présentent un risque accru d'infection et de maladie par des germes pathogènes. De récentes études ont montré que près de la moitié de tous les puits américains testés, captés pour l'eau potable, sont soumis à une contamination fécale. Une fraction significative de l'ensemble des premières manifestations de maladies liées à l'eau est associée aux eaux souterraines. On estime qu'entre 750 000 et 5,9 millions de personnes sont malades chaque année aux tats-Unis à cause d'eaux souterraines polluées. La mortalité parmi ces malades doit être de l'ordre de 1400 à 9400 décès par an. La protection contre ces germes pathogènes commence avec des mesures prises au niveau du captage pour empêcher la pollution des aquifères et des puits. Celles-ci comprennent une évaluation de la vul-nérabilité des têtes de puits à la pollution fécale et une correction des insuffisances mises en évidence. Cette correction peut comprendre une maîtrise des sources de pollution ou la réhabilitation du puits lui-même. La désinfection peut être une précaution utile et est recommandée comme une mesure prudente de santé publique pour toutes les nappes aquifères.

Resumen Beber agua subterránea no desinfectada supone un riesgo de infección por microorganismos patógenos. Estudios recientes muestran que la mitad de los pozos de abastecimiento analizados en los EEUU presentan evidencia de contaminación fecal. Un porcentaje significativo de la aparición de enfermedades transmitidas por el agua puede asociarse a las aguas subterráneas, estimándose que por contaminación de las mismas se registran, sólo en ese país, entre 750.000 y 5.9 millones de personas enfermas y entre 1400-9400 muertos por año. El control de estos patógenos empieza con la protección de la fuente para prevenir la contaminación fecal de pozos y acuíferos. Esto supone evaluar la vulnerabilidad y corregir las posibles deficiencias detectadas, lo que incluye controlar los trabajos de rehabilitación del propio pozo. La desinfección puede servir como una barrera a los microorganismos patógenos, por lo que se recomienda como una política prudente de salud pública en zonas abastecidas con aguas subterráneas.

Introduction

The majority of articles in this issue of *Hydrogeology Journal* discuss soil and groundwater microbiology with respect to the ability of various organisms to transform inorganic and organic materials in the subsurface, with potentially beneficial consequences. This article discusses quite a different topic, the potential pathogenicity, consequences, and control of microorganisms in groundwater.

Protection from waterborne microbial disease has been a US public-health goal for decades. A variety of control approaches are in place today, including pro-

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tection of the source waters from microbial contamination, treatment to remove or inactivate microbial pathogens, and development of operational criteria for drinking-water systems to prevent contamination at the wellhead or in distribution. However, as recently as 1990, the US Environmental Protection Agency's (USEPA) Science Advisory Board, an independent panel established by Congress, cited drinking-water contamination as one of the highest-ranking remaining environmental risks (USEPA 1990). The Science Advisory Board reported that microbiological contaminants (e.g., bacteria, protozoa, and viruses) are likely to be the greatest remaining health risk-management challenge for drinking-water suppliers. These risks are most likely associated with groundwater. Whereas stringent regulations to control microbial contaminants apply to drinking-water systems using surface water, only limited regulations apply to systems using groundwater.

This article reviews the current understanding of microbial contamination of groundwater and the implications for public health. It also discusses some approaches that have been successful in controlling pathogens in groundwater.

Organisms and Their Associated Diseases

A large number of microbial pathogens are known to contaminate or may plausibly contaminate groundwater (Bennett et al. 1987; Herwaldt et al. 1992; Moore et al. 1993; Benenson 1995; Kramer et al. 1996). Some of these are listed in *Table 1*. These include more than 100 viral and several bacterial pathogens. This list also includes protozoa, such as *Giardia* and *Cryptosporidium*, although contamination of groundwater with protozoa generally indicates surface-water influence (USEPA 1989a, 1994). Most of these organisms are of fecal origin and are transmissible via a fecal–oral route of exposure.

The possible microbial illnesses that result from infection vary with the organism and vary markedly in their severity (*Table 1*). The predominant recognized illness is generalized acute gastrointestinal illness (AGI), resulting in fever, nausea, diarrhea, and/or vomiting. Most cases of AGI are of short duration, self-resolving, and may not be of major consequence to otherwise healthy individuals. However, for others this may not hold true. AGI may be chronic, severe, or fatal to some people. These include the elderly, infants, pregnant women, and especially the immuno-suppressed and immuno-compromised (Kaplan et al. 1983; Modlin and Kinney 1987; Lew et al. 1991; Gerba et al. 1996). Studies reported by Sobsey et al. (1991) indicate that more sensitive subpopulations, including infants and those over 70 years old, have mortalities of 3–5% from diarrhea requiring hospitalization.

An additional concern is that some organisms produce other illnesses beyond AGI that may have more serious consequences. Coxsackie virus infection, for example, is associated with heart and circulatory disease (myocarditis, pericarditis), aseptic meningitis, and insulin-dependent diabetes melitis. Hepatitis A infection may result in fever, jaundice, or liver damage, or it may progress to death. The Center for Disease Control and Prevention (CDC) calculated death rates from hepatitis A illnesses in the US at 0.3% of those who are ill (Bennett et al. 1987). Pregnant women have a 10–20% mortality to hepatitis E infections (Craske 1990). The CDC presents data that indicate that overall death rates from waterborne illnesses from a variety of organisms approach 0.1% (Bennett et al. 1987). Estimated annual waterborne disease deaths in the US are 900–1800 (Bennett et al. 1987; Morris and Levin 1995).

Table 1 Pathogenic microorganisms of concern in groundwater

Organism	Associated health effects
Viruses	
Coxsackie	Fever, pharyngitis (sore throat), rash, respiratory disease, diarrhea, hemorrhagic conjunctivitis, myocarditis, pericarditis, aseptic meningitis, encephalitis, reactive insulin-dependent diabetes, hand, foot and mouth disease
Echo	Respiratory disease, aseptic meningitis, rash, fever
Norwalk	Gastroenteritis (fever, vomiting, diarrhea)
Hepatitis A	Fever, nausea, jaundice, liver failure
Hepatitis E	Fever, nausea, jaundice, death
Rota	Gastroenteritis (fever, vomiting, diarrhea)
Enteric adeno	Respiratory disease, hemorrhagic conjunctivitis, gastroenteritis
Calici	Gastroenteritis (diarrhea)
Astro	Gastroenteritis (diarrhea)
Bacteria	
Escherichia coli	Gastroenteritis (diarrhea)
Salmonella spp.	Enterocolitis (fever, diarrhea, vomiting), endocarditis, meningitis, pericarditis, reactive arthritis, pneumonia
Shigella spp.	Gastroenteritis (diarrhea, fever, vomiting), reactive arthritis
Campylobacter jejuni	Gastroenteritis (diarrhea, fever, vomiting), Guillain-Barre syndrome
Yersinia spp.	Diarrhea, reactive arthritis
Legionella spp.	Legionnaires' disease, Pontiac fever, death
Vibrio cholera	Diarrhea, vomiting, death
Protozoa	
Cryptosporidium parvum	Diarrhea
Giardia lamblia	Chronic diarrhea

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Occurrence of Microbial Pathogens in Groundwater

Contamination of groundwater with pathogenic microorganisms is generally believed to result from migration or introduction of fecal material into the subsurface. Primary sources of fecal contamination of health concern to humans include other humans and some animals. Whereas human diseases result largely from organisms specific to humans, the reoviruses, *Cryptosporidium, Giardia,* and several bacterial species pathogenic to humans occur in cattle, other mammals, and some birds.

Fecal contamination can reach groundwater from many routes. Of primary concern are concentrated point sources, such as failed septic systems, leaking sewer lines, and cesspools. Animal feedlots, dairy farms, and other intensive animal-husbandry operations may be significant sources in some settings but are far less common. Transport to groundwater is primarily a function of the hydrogeological setting and climatic conditions.

Although the authors are aware of data from more than two dozen studies of microbial contamination of groundwater, relatively few have been published (Hibler 1988; Bauder et al. 1991; Rose et al. 1991; Abbaszadegan et al. 1998, 1999; Hancock et al. 1998; USGS 1998). Pathogenic viruses and their indicators, indicators of fecal bacterial contamination, *Giardia,* and *Cryptosporidium* occur in significant fractions of tested wells, both public and private, throughout the US.

All public drinking-water systems are required to monitor for total and fecal coliform bacteria under the Total Coliform Rule (TCR) (USEPA 1989b). Analysis of EPA data on drinking-water enforcement shows that about 40,000 of the 156,000 groundwater systems have had TCR maximum contaminant level (MCL) violations, indicating coliform bacterial contamination of their groundwater, well, storage, or distribution system during the last 5 years. Because of limited monitoring requirements and state surveillance of non-community systems (systems that serve water to people on a less-than-full-time basis, such as schools or rest stops), these violations came primarily from the 44,000 community groundwater systems. For the larger, urban groundwater systems, violations were more often associated with distribution systems and may have been related to biofilm growth or from cross-connection events. For the smaller, rural groundwater systems, contamination is more often at the wellhead, indicating well- or source-water contamination. The majority of these violations involved detection of total coliform bacteria rather than fecal coliforms or *E. coli.* Because many coliform bacteria are not pathogenic, the implications of their occurrence for public health are not direct.

Some argue that protozoa such as *Cryptosporidium* or *Giardia* should not occur in true groundwaters, because their relatively large sizes make them more subject to natural filtration by soils than would be so for the smaller bacteria and viruses. One of the criteria to determine if a groundwater is under the direct influence of surface water is to consider the presence of protozoa. It appears, however, that some hydrogeological settings that would not indicate a surface-water influence do allow protozoan contamination. These include fractured rock and karst with limited unconsolidated soil overlayers. Hancock et al. (1998) report *Cryptosporidium* and/or *Giardia* detections in 12% of 199 groundwater sites across the country. Although the majority of detections were in springs, infiltration galleries, and horizontal wells, which are generally considered to be more subject to direct surface-water influence, *Cryptosporidium* was also detected in 5% of vertical wells, some of which could not be associated with surface sources. This result is consistent with the earlier findings of Hibler (1988) and Rose et al. (1991).

In terms of public health, most concern for groundwater has focused on pathogenic viruses resulting from fecal contamination. Their small sizes relative to the larger bacteria and protozoa may allow freer movement through unsaturated and saturated media. A major study of the occurrence of groundwater pathogens, supported by the American Water Works Association Research Foundation (AWWARF) and the USEPA, tested about 550 public water-supply wells (prior to any treatment) for various indicators of fecal contamination. These indicators included total coliform bacteria, *E. coli,* enterococci, somatic and male-specific coliphage (viruses infecting coliform bacteria), and human viruses detected by polymerase chain reaction (PCR) and cell-culture techniques. Wells were generally sampled only once. Results from the first phase of this work (244 samples) have been published (Abbaszadegan et al. 1998, 1999). About 50% of wells initially considered more vulnerable to contamination and 40% of wells initially considered less vulnerable were positive for one or more of these indicators. More specifically, 38% were positive for one or more pathogenic viruses using PCR, the most sensitive and encompassing of the currently available techniques. About 18% were positive for coliphage using assays not yet fully optimized. Human viruses were detected by cell culture in about 7% of samples, even though this method detects a distinctly limited subset of the human viruses of concern. About 10% were positive for coliform bacteria and 18% for enterococci. No correlations were seen between occurrence of fecal indicators and either soil type or distance from fecal sources.

An earlier USEPA/AWWARF study (Lieberman et al. 1995) examined 30 public water-supply wells judged to be vulnerable to fecal contamination. These were sampled monthly for 1 year. The authors report 24% positive for culturable viruses, more than 50% positive for one or more coliphage, 50% positive for *E. coli,* and 70% positive for enterococci. Their results

indicate that multiple samples of a source were necessary to determine contamination.

Other studies of groundwater microbial contamination are more limited in scope and/or unpublished at this time. Bauder et al. (1991) examined private wells in Montana and report about 40% contaminated with coliform bacteria. This level is nearly identical to that in an unpublished study by the CDC of more than 5500 private wells in the midwest US. The US Geological Survey (USGS 1998) reports about 8% of wells positive for culturable human viruses and 8% positive for coliphage in a study of 109 public wells in Missouri.

It is uncertain how representative these data are in characterizing the occurrence of fecal contamination in groundwaters in general across the US. Major factors affecting groundwater-system vulnerability include hydrogeological setting, well construction, well depth, and proximity to sources of fecal contamination (Wireman and Job 1997, 1998). The studies of Abbaszadegan et al. (1998, 1999) relied on volunteer utilities and predominantly examined larger urban groundwater systems that provided disinfection as a matter of best management practice. They selected wells that were generally deeper, better constructed, and better maintained in order to reduce vulnerability and wellhead-related effects. As a result, smaller systems, rural systems, and wells with poorer construction were less represented. Thus, their results probably under-represent national occurrence. At the other end of the spectrum, Lieberman et al. (1995) selected specifically for vulnerable wells. Their results were three- to five-fold higher for specific indicators, although the overall well-specific fecal contamination levels were similar. If these results are approximately representative of groundwaters used for drinking water, they are substantially higher than previously believed and challenge the idea that few groundwaters are contaminated with microbes.

Public Health Concerns Regarding Microbial Pathogens in Groundwater

Although data exist that some drinking-water wells are contaminated with fecal material, it is not known how many of the 110 million people served by groundwater-based public water systems and the 20 million or so on private wells become sick each year, the key question for protection of public health. Epidemiological studies to determine this directly have not been carried out. However, several lines of evidence give us reason to believe that a health problem exists.

Outbreaks of Waterborne Disease

Microbial pathogens in groundwater systems are known to have caused numerous disease outbreaks in the US. Craun and Calderon (1997) report 356 outbreaks from 1971 to 1994 caused by contaminated

groundwater systems, 58% of the total of all waterborne outbreaks. Community systems were involved in 32% of these outbreaks, non-community systems in 68%. Contamination of the groundwater source was considered responsible for 70% of these outbreaks; contamination of the distribution system for 30% (Craun 1991; Craun and Calderon 1997). For community groundwater systems, inadequate disinfection was cited in 23% of outbreaks, lack of disinfection in an additional 20%. For non-community groundwater systems, these numbers were respectively 28% and 53%.

Acute gastroenteric illness was the most common disease described in these outbreaks, accounting for 35% in community systems and 75% in non-community systems. Frequently, no causative agent was identified. When a disease agent has been identified with an outbreak, *Shigella,* hepatitis A virus, norwalk virus, *Giardia lamblia, Campylobacter jejuni,* and *Cryptosporidium parvum* have been implicated. However, the number of individuals reported ill from these outbreaks is generally understood to underestimate the actual levels of microbial disease associated with drinking water for the overall population (Frost et al. 1996). Two reasons are offered: (1) the nature of outbreaks themselves, where significant numbers of individuals get sick over a short time, does not describe endemic levels of disease; and (2) reporting of disease outbreaks in the US is poor.

The CDC maintains a database of information regarding outbreaks of waterborne diseases in the US. This database is based upon responses to a voluntary survey that is completed by state and local public-health officials. To be considered a waterborne outbreak, acute illness must affect at least two persons and be epidemiologically associated with the ingestion of water (Frost et al. 1996; Craun and Calderon 1997). Outbreaks are generally recognized when a significant fraction (1–2%) of a population gets sick within a few weeks, such that local physicians or laboratories recognize that something unusual is happening. As noted above, groundwater outbreaks are typically associated with some sort of contamination event or treatment failure, where unusually high numbers of organisms may occur in the water delivered to consumers.

The majority of disease outbreaks is not reported to the CDC and, therefore, they are not represented in this survey. The likelihood that individual cases of illness will be epidemiologically linked and associated with water varies considerably among locales and is dependent on factors such as public awareness, physician interest, availability of laboratory-testing facilities, and surveillance activities of state and local health agencies. Therefore, the states that report the most outbreaks may not be those in which the most outbreaks actually occur (Frost et al. 1996). This often additionally requires that the disease itself be noteworthy or reportable (cholera, giardiasis, cryptosporidiosis) and not simply "viral gastroenteritis" or

diarrhea. In large cities, small outbreaks may be obscured, because ill people may consult so many different physicians that nothing unusual is noticed. Outbreaks associated with community water systems are more likely to be recognized than those associated with non-community systems, because the latter generally serve transient populations. Outbreaks associated with private systems are believed most likely to be under-reported, because they generally involve few people and occur in a rural setting.

Endemic Illness

Endemic illness, not necessarily associated with a massive contamination event leading to a disease outbreak, occurs at levels below that necessary to detect an outbreak. Epidemiological studies have shown that the average individual has 1.5 cases of diarrhea over the course of a year from any of a variety of sources (Bennett et al. 1987; Payment et al. 1991). Data indicate that drinking water may contribute 10–30% of these cases (Payment et al. 1991). However, no epidemiological studies have looked directly for waterborne disease. Annual waterborne microbial illnesses in the US have been estimated at between 7 and more than 27 million cases (Haas 1993; Morris and Levin 1995). The majority of these is probably associated with groundwater systems, because all systems using surface water are currently required under the Surface Water Treatment Rule (USEPA 1989a) to have substantial disinfection in place and maintain a disinfectant residual in their distribution systems. Most surface-water systems provide substantial physical removal (coagulation, sedimentation, and filtration) as well. As described later in this article, substantially fewer groundwater systems disinfect to these standards. About 22 million Americans were estimated to drink water that was not disinfected from community groundwater systems and perhaps as many from private wells.

Because direct information on waterborne disease is not available, estimates based on occurrence data and the pathogenic properties of the organisms may be useful. Methods exist to develop quantitative risk assessments for groundwater systems (Regli et al. 1991; Haas et al. 1993; Gerba et al. 1996; Hurst et al. 1996; Crabtree et al. 1997). However, available data on both the occurrence and the organisms of interest limit this approach.

Interpretation of the significance to public health of the occurrence studies is difficult for several reasons. Whereas the presence of indicators of fecal contamination in drinking water would not be acceptable from the standpoint of public health risk management, fecal contamination itself does not mean that the water contains pathogens, only the potential for pathogens. Many pathogens with specific infectious properties could be and probably are present, but any one organism may not be. Depending on the size of the population providing the fecal source material, the prevalence of specific diseases in that population, and other environmental factors, specific pathogens may not occur at all or only during certain times of the year. As a result, fecal-occurrence data alone cannot easily be used for quantitative assessments of health risk.

If quantitative assessments of risk are based solely on data for human pathogens, such as virus PCR and cell culture data, other problems arise. For example, PCR detects viral pathogen nucleic acids but does not require intact virus particles; hence, positive results may be obtained from non-infective material. The PCR results presumably are associated only with the presence of fecal material; thus, the method is useful as an indicator, but again quantitative risk calculations cannot be done. Reliance on cell culture data for viruses is also problematical, but for another reason. Very few of the pathogenic viruses of health concern can be successfully cultured at this time. Therefore, although quantitative risk assessments can be done using these data, they would be expected to underrepresent true risks.

Complicating issues also exist with respect to the pathogenicity of the organisms of interest. Limited data are available on the pathogen doses necessary to yield infection. Acceptable data exist on only a few viruses, bacteria, and protozoa (Regli et al. 1991). Additionally, only a limited understanding of the relationship between infection and the various forms of illness observed exists. Perhaps only about 50% of infections result in illness (Gerba et al. 1996; Hurst et al. 1996). Immunity to these pathogens is poorly understood, much less quantifiable. In some cases it may involve true immunity to infection. In other cases, infection may occur, but it is asymptomatic or sufficiently mild that it is not recognized. This may explain "tourist's syndrome," where residents of a small community or homeowners using private wells do not report illnesses, but visitors do.

Groundwater Microbial Risk Assessments

If the uncertainties from these data limitations are accepted and plausible assumptions are used, assessments can be performed to estimate the magnitude of endemic waterborne illness from groundwater systems. Such assessments may at least provide a range of possible impacts useful for public policy and management discussions.

As an example, estimates were constructed of the annual number of people in the US that might become ill from consuming groundwater that has not been disinfected from community water systems and from all public water systems. These estimates were generally based on rotavirus, because data sets are most complete for this virus group. However, because

groundwater outbreaks have occurred where other pathogenic microorganisms have been involved, estimates based solely on rotavirus are likely to under-represent true levels of waterborne illness.

The occurrence-input data were developed as follows:

- The range in the percentage of wells with evidence of fecal contamination was taken as from 7% (based only on the presence of culturable human viruses) to 46% (based on all indicators).
- The data are assumed to be representative of wells across the US.
- A recovered most probable number (MPN) of 0.4/100 L and a recovery rate of 0.5, based on Abbaszadegan et al. (1999), were assumed for all data sets.
- Because the species composition of the cell-culture data is unknown, rotavirus occurrence is assumed for positives based on the PCR results of Abbaszadegan et al. (1999): 18% of all wells detected rotavirus of the 38% total wells with one or more fecal detections = 0.47.

The exposure assessment combines occurrence with populations and ingestion rates:

- Occurrence in wells that have not been disinfected was assumed to be the same as for disinfected wells.
- Disinfection, if in place, was assumed to be adequate to reduce the risk to zero.
- Twenty two million people were estimated to drink from community water systems (CWS) that had not been disinfected, for 350 days/year (from USEPA databases).
- Five million people were estimated to drink from non-transient, non-community (NTNC) water systems that had not been disinfected (schools, factories), for 250 days/year.
- The standard water consumption rate of 2 L/day was used for these populations. This value was converted to a "day-L" exposure: 15.4 billion day-L CWS exposures; 2.4 billion day-L NTNC exposures.
- Two hundred and fifty million people were estimated to drink from transient non-community (TNC) water systems that had not been disinfected, twice a year.
- A water-consumption rate of 1 L/day was used for this population and an equivalent day-L exposure was calculated: 500 million day-L TNC exposures.

Only dose-response data for rota- and echoviruses are available to represent human pathogens of concern.

- The dose response for rotavirus for annual risk of infection at 0.4/100 L MPN is 0.834 (Gerba et al. 1996).
- An illness rate of 50% infection rate was used (Haas et al. 1993).
- To estimate mortalities, the elderly were taken as the primary subpopulation of concern. A mortality rate for elderly populations of 0.01 the illness rate

was used and a population of elderly of 16% assumed (Gerba et al. 1996).

- A 25% secondary infection rate was assumed.

Based on the above data and assumptions, risks are as follows:

1. Waterborne illnesses in groundwater-based community water systems were estimated to be 750,000–5.0 million illnesses/year [22 million people exposed × 7–46% contamination rate × 0.47 rota contribution × 0.5 illness/infection/0.5 recovery × (834 rota dose-response @ 0.4/100 L MPN) × 1.25 secondary spread rate = 750,000–5.0 million illnesses/year].

2. Waterborne illnesses for all public supplies using groundwater were estimated to be 890,000–5.9 million illnesses/year [calculated by multiplying the above CWS illness estimate by a ratio of day-L exposures for (CWS+NTNC+TNC/CWS) = (15.4 + 2.4 + 0.5)/15.4 × (750,000 – 5.0 million) = 890,000 – 5.9 million illnesses/year].

3. Mortality from waterborne illness was estimated to be 1400 – 9400 deaths/year [calculated by multiplying the above illness estimates by deaths divided by illness and percent elderly 0.01 × 0.16 × (890,000 – 5.9 million) = 1400 – 9400 deaths/year].

These estimates are consistent with CDC mortality data (Bennett et al. 1987) and with independent assessments of waterborne illness and mortality (Haas 1993; Morris and Levin 1995; Hurst et al. 1996; USEPA 1998).

Approaches to Control Waterborne Microbial Pathogens

USEPA databases indicate that about 400,000 public drinking-water wells and many more private wells exist in the US. The public wells are spread between about 44,000 CWS, perhaps 19,000 NTNC water systems, and 93,000 TNC water systems (rest stops, campgrounds, restaurants, etc.). Almost all of these 156,000 systems are very small. Fewer than 400 community groundwater systems serve more than 50,000 people; another 4000 systems serve between 3300 and 50,000 people. About 40,000 community systems serve fewer than 3300. The NTNC and TNC systems generally serve relatively few people on a regular basis but may serve many people irregularly or infrequently.

The USEPA has articulated public-health goals for microbial contamination of drinking water from surface sources in the Surface Water Treatment Rule (SWTR) (USEPA 1989a) and Interim Enhanced SWTR (IESWTR) (USEPA 1998). Maximum contaminant level goals (MCLGs) have been established at zero for *Giardia, Legionella, Cryptosporidium*, total coliform bacteria, and viruses. MCLGs are not enforceable, but they provide the public-health directions for the enforceable National Primary Drinking Water Regulation (NPDWR) (Macler 1993). The cor-

responding NPDWR primary-treatment goal is to achieve no more than one *Giardia* (SWTR) or *Cryptosporidium* (IESWTR) infection per 10,000 exposed people per year. Prevention of outbreaks of waterborne-microbial diseases is another public-health goal of these regulations. The public-health goal for the TCR NPDWR (USEPA 1989b), which applies to both surface- and groundwater-based drinking-water systems, is to have no total or fecal coliform bacteria detected in the system. In public discussions for USEPA's Groundwater Rule, which will specifically address microbial contamination in groundwater-based public water systems, public-health goals to prevent outbreaks of waterborne diseases and to reduce levels of endemic waterborne disease were articulated. A regulatory goal to prevent fecal contamination from reaching the consumer by requiring a series of barriers was developed (Macler 1996; Macler and Pontius 1997).

Control of waterborne microbial pathogens thus centers on eliminating the route of exposure from fecal sources through groundwater to the consumer. This may be accomplished at every step in the process, for example, by eliminating sources of fecal contamination from the recharge zones or zones of influence around wells, siting wells to provide adequate natural attenuation of microorganisms, ensuring proper well construction to prevent surface contamination at the wellhead, controlling cross-connection contamination events in distribution systems, and providing disinfection treatment.

Best Management Practices and Supporting Regulations

These and other public-health practices are employed by drinking-water utilities, either voluntarily or in response to regulatory requirements. These "best management practices" (BMPs), listed in *Table 2*, generally have had long, successful track records in protecting public health. Many have served successfully as regulations, policies, or guidelines at the state level. Until passage of the federal Safe Drinking Water Act (SDWA) of 1974, utilities and states acted as mostly independent experimentation laboratories of drinking-water protection. After 1974, the subsequent NPDWRs placed a minimum foundation under these activities. However, experimentation did not cease, as individual states continued to tighten their programs in response to local needs. To get a better understanding of the utility and commonality of these practices, the USEPA collected groundwater-related statutes, regulations, guidance, and disinfection practices from all states (Merkle et al. 1996; Merkle and Reeverts 1997).

Source-Water Protection Barriers

Several BMPs act to control pathogen sources or their proximity to the wellhead. Source-water assessment and protection programs and wellhead protection programs aim to control or eliminate sources of contamination. Well-siting criteria that use hydrogeological information and appropriate setback distances from fecal sources can help ensure that contamination does not reach the well. Monitoring for microbial pathogens or surrogates (such as total or fecal coliform bacteria) provides direct information on contamination.

However, monitoring for pathogens at the wellhead, while desirable and useful, is inadequate by itself to ensure protection to the public. Monitoring results are useful only in a reactive mode, in that they are generally available only after exposure has occurred. Because infection may result from very limited exposure to pathogenic microorganisms, protective elements should focus on proactive measures. Monitoring results may be equivocal for other reasons: (1) if a system is positive for a pathogen or fecal indicator at a given time, uncertainties remain in the frequency and magnitude of this contamination, as well as in the types and health significance of other organisms that might co-occur; (2) if a system is found negative for indicators, it may be in fact contaminated, but the limitations of monitoring frequency, sample size, and level of quantitative analysis may not show this; (3) a system that is negative may be without contamination now but not in the future.

To supplement monitoring, assessments of source-water vulnerability may be useful. Such assessments

Table 2 Best management practices to control microbial pathogens in groundwater

Source-water protection barriers	Approved source-water protection or wellhead-protection program Minimum setback distance(s) specified from microbial contamination to wells Hydrogeological criteria used for well siting Wellhead-monitoring data required
Well and water-system integrity barriers	Sanitary survey and corrections required Well-construction codes
Operations and system-maintenance barriers	Well and pump disinfection Periodic flushing of distribution system Disinfection of new/repaired water mains Cross-connection control programs Requirements for certification of operators
Disinfection requirements	Specified disinfection $C \times T$ values Microbial kill/reduction values Specified minimum disinfectant or chlorine residual in distribution system

generally consider land uses and sources of contamination, combined with determinations of hydrogeological sensitivity (Wireman and Job 1997, 1998). Common vulnerability factors include time of travel, presence of confining layers, soil type, depth of the unsaturated zone, nitrate levels, presence and location of contamination sources, and monitoring data. However, substantial uncertainty may exist in estimating vulnerability.

Hydrogeological criteria may indicate that a well is vulnerable to contamination, but the water may still be safe if adequate natural filtration of microbial pathogens has occurred in the subsurface. The traditional setback distance between source and well is an application, based on best professional judgment, of this concept. This is an extension of the assumption that protozoa do not contaminate true groundwaters, because their large size results in their being filtered out by the soil within a short distance from their source. Evidence exists that bacteria and viruses are similarly filtered out or absorbed by some soils under some conditions (McDowell-Boyer et al. 1986; Bales et al. 1989; Gerba et al. 1991). Hydrogeological conditions of productive aquifers and well sites vary widely across the US. Significant disagreement exists among experts in this field concerning whether this filtration can be adequately predicted or quantified to be of use for predicting risks and establishing public-health guidance on either a site-specific or national basis.

Similarly, pathogenic microorganisms may be inactivated in the subsurface over time such that, even if physically detected in groundwater, they are incapable of causing infection. The rate of this inactivation is organism-specific and highly influenced by subsurface physical and chemical conditions (Hurst et al. 1980; Hurst and Goyke 1986; Kutz and Gerba 1988; Hurst et al. 1989). As with physical removal, substantial disagreement exists about whether and how data on inactivation can be used to predict vulnerability. Setback distances and time-of-travel requirements from fecal sources used by states and communities to site wells are seldom if ever field validated.

Source Water Assessment and Protection Programs

The 1996 Amendments to the SDWA established USEPA programs designed to support source-water protection in two ways. The Amendments mandate that states develop and implement statewide source-water assessment programs (SWAPs). Some funding for this can be obtained from set-asides to State Drinking Water Revolving Funds. States are encouraged to develop and implement voluntary source-water protection programs (SWPPs) as well.

SWAPs are to be implemented by the states for all sources of drinking water on a one-time basis over the next few years. They are to provide a consistent baseline of information on the vulnerability of sources to problematic contamination. A SWAP for groundwater sources includes three components: (1) a delineation of zones of influence around production wells; (2) an inventory of sources of contamination within these zones of influence; and (3) an assessment of well "susceptibility" (vulnerability) to contamination based on the contaminant occurrence and characteristics and hydrogeological information (USEPA 1997). States may be able to use SWAP products in their sensitivity determinations under the USEPA's proposed Groundwater Rule. Information from SWAPs can be used by local communities to create a SWPP to address current problems and prevent future threats to drinking-water quality.

SWPPs are community-based approaches to protecting sources of drinking water from contamination. With respect to providing enhanced microbial protection, SWPPs can be designed to promote a barrier to fecal contamination reaching the wellhead recharge area or zone of influence. The goal would be to eliminate or control fecal sources or ensure that adequate natural disinfection occurs in the unsaturated zone or groundwater prior to reaching the well. Management of sources is optional under the 1996 Amendments, although program guidance can provide approaches to manage existing and potential contamination problems. This approach supports collective efforts among local governments, farm and business interests, and citizen groups at the community, regional, and watershed levels. The success of these efforts depends on the involvement of all stakeholders in the implementation process. Because SWPPs are based on community and site-specific activities, the extent of the program's effectiveness cannot easily be measured and compared from one locality to the next.

SWPP activities include state and local wellhead-protection programs, state and local groundwater-protection programs and Underground Injection Control Class V programs. Local watershed-protection activities that target groundwaters or groundwater recharge areas for protection may also add to the source-water assessment and protection barrier.

Wellhead Protection Programs

Wellhead protection is the protection of the area surrounding a well from significant potential sources of anthropogenic contamination. The USEPA has approved wellhead protection programs for 44 states and 3 territories. Some states have adopted mandatory requirements, whereas others use approaches based on voluntary activities.

The functioning unit of this program is the wellhead protection area (WHPA), which is defined as the surface and subsurface area surrounding a well or well field that supplies a public water system. WHPA boundaries are determined based on factors such as well pumping rates, time of travel of groundwater flowing to the well, aquifer boundaries, and degree of

confinement. These hydrogeologic characteristics have a direct effect on the likelihood and extent of contamination.

A WHPA can be established for any type of aquifer. The extent of the areas within WHPAs varies depending upon the program goals of individual states and municipalities and the hydrogeologic settings. These programs involve forming a local team, delineating a protection area, identifying potential sources of contamination within that area, and managing these sources to protect the wellhead. By defining a WHPA and conducting an inventory of potential contaminant sources, a water supplier can gain valuable insight into the potential threats that exist to the water supply. Where possible, a public water system may help ensure the protection of existing wells from contamination through the implementation of effective management controls on these sources.

Possible Source Controls

Source-control activities may be generalized to the area, such as through zoning requirements, health regulations, land acquisition, and conservation easements. They may also be site- or activity-specific, such as requirements for septic systems, sewer lines, discharges to groundwater, or feedlots. *Table 3* lists examples of source-specific control measures.

Examples of zoning controls include prohibition of certain sources, such as cesspools; limits on density of sources, such as septic tanks, through large-lot zoning; performance standards; and special permitting. Because zoning typically applies to future development and often exempts existing activities and systems, it is best used during planning.

Health regulations could include prohibition of microbial sources within a specified distance from the wellhead (e.g., setbacks). They could include performance standards for particular sources, such as design, operation, maintenance, and inspection requirements for septic tanks, sewer lines, or privately owned small sewage-treatment plants. In some cases, existing setbacks may not be large enough to protect the wellhead from microbial contaminants. The effectiveness of performance standards in preventing contamination of the wellhead by microbial contaminants may not be known.

Control by establishing requirements for land acquisition and conservation easements includes buying land in a source-water protection area and prohibition of potential sources of contamination. It includes buying or mandating conservation easements that restrict all or a portion of the property to open space or limited development. Success of this tool may depend on the public water system's or community's commitment to promote it, the presence of willing sellers, real-estate values, and the resources available to buy land or easements. Under the SDWA Amendments of 1996, states may now set aside up to 10% of

the Drinking Water State Revolving Fund for loans to public water systems for purchase of land or easements.

It has proven difficult to assess the efficacy of wellhead protection programs and few such assessments have been attempted. Merkle and Reeverts (1997) report that states with approved wellhead programs had an average total coliform maximum contaminant level (MCL) violation rate of 27% (FY1991–1995) for community systems compared to 26% for states without approved programs (total coliform MCL violations indicate the presence of coliform bacteria in the drinking water system). Many assumptions were involved in this assessment For example, in this comparison they did not attempt to separate the older, established programs from the younger programs that may have had little time to make any impact upon TCR violations. However, the study does not suggest that this approach is positively affecting microbial contamination of groundwater systems in general.

Well-Siting Criteria

The majority of states have well-siting requirements based either on hydrogeological criteria or on setback distances from sources of fecal contamination to the wellhead (Merkle et al. 1996; Merkle and Reeverts 1997). Twenty-four states always and five states some-

Table 3 Examples of source-specific control measures

Septic systems	Meets design standards
	Installation provides adequate separation above groundwater
	Adequate inspection and pumping requirements
	Density restrictions, possibly based on nitrate loading analysis
	Discharge is treated to kill pathogenic microbes
	Additional requirements for new systems or systems needing repair
Sewer lines	Stricter standards for sewer line construction, testing, and manhole installation
	Leak-detection system
	Plan for corrective action if leak detected
Wastewater discharges to ground/injected wastewaters	Disinfect wastes before discharging
	Alternative treatment that results in wastewater that meets a nitrate standard or other limit set by a state
	Regular monitoring
	Inject wastes only below aquifer used for drinking water
Feedlots	Divert runoff from feedlot area
	Minimize runoff by reshaping area
	Collect and treat runoff
	Require lined manure pits
	Collect and treat pit effluent
	Establish a size threshold above which disinfection of waste materials is required

times consider local hydrogeological criteria in the approval of the siting of a well. How carefully or with how much detail these criteria are applied in state practice is not known, nor is it known how frequently wells approved on erroneous information are required to be refitted or replaced. Without further research, the most that can be said about these programs is that they appear or do not appear in the regulations of a specific state. Most of these states give the regulatory agencies general authority to consider local hydrogeological characteristics or place the requirement to consider this feature upon the driller. Highly technical analyses, such as time-of-travel determinations or sub-surface modeling, are not explicitly required. Hydrogeology is examined most commonly to determine whether to apply standardized setback distances, and to determine the depth of the well, the length of the casing, and the extent of the grouting, especially when confining layers are present. Some states, such as Louisiana, require consideration of general protective measures: "The earth formations above the water-bearing stratum shall be of such character and depth as to exclude contamination of the source of supply by seepage from the surface of the ground" (Louisiana State 1994).

The presence or absence of hydrogeological requirements can be compared with the statewide TCR MCL violation rates to determine whether these programs are associated with reduced violations at community water systems. States that do not employ hydrogeological criteria have a mean TCR MCL violation rate of 33%; states that employ such criteria have a mean TCR MCL violation rate of 23%, which represents a 30% reduction in violations. Put another way, among the 18 most successful states (i.e., those states with TCR MCL violation rates less than or equal to 20%), 72% of them use hydrogeological criteria. Of the 18 least successful states (i.e., those states with TCR MCL violation rates greater than or equal to 29%), only 28% use these criteria. Thus, the use of hydrogeological criteria in well siting appears to be associated with fewer TCR violations across a broad range of state groupings.

Well and System Integrity Barriers

Other important means to control microbial contamination in well water focus on ensuring the integrity of wells and distribution systems. These include proper design and construction of wells, distribution lines, and storage systems according to applicable state criteria, codes, or regulations. They also include proper operations and maintenance activities.

Important examples include periodic inspections (sanitary surveys) of sources, well and system hardware, distribution lines, and storage, followed by the appropriate correction of deficiencies. Provision for a state-certified operator and implementation of an emergency response plan covering major equipment failure (e.g., well pump, transmission mains, etc.) or natural disasters help ensure successful operations.

Protection of the distribution system from fecal contamination entering via cross-connection events or siphonage is critical. A substantial portion of water-borne-disease outbreaks is associated with failures of distribution systems (Craun 1991). In urban settings, sewer lines and water mains may occupy the same trench or lack sufficient setback distances. Because sewer lines leak, pathogens can be expected external to water mains. Provision of a cross-connection control and backflow prevention program is desirable. Maintenance of an acceptable distribution system pressure at all times, water-main flushing programs, and maintenance of a disinfectant residual in the distribution system have all proved successful.

Disinfection Barriers

Disinfection can provide a barrier at any or all points in the system, and it can provide protection to almost all of the source and system deficiencies possible in groundwater systems. At a minimum, all groundwater systems with known fecal contamination should have to disinfect, unless they immediately correct outstanding problems by some other means.

Technologies for inactivating microorganisms in groundwater are well understood, practical, and relatively inexpensive. These include the traditional use of chlorine, as gas, hypochlorite, or chlorine dioxide. They also include ultraviolet (UV) light, ozone, and ultrafiltration. The USEPA has assembled documentation on available UV, ozone, membrane filtration, chlorine, and other technologies suitable for small systems (USEPA 1996).

The degree of necessary disinfection is relevant. The SWTR (USEPA 1989a) specifies that systems must achieve a 99.99% inactivation of viruses at the first customer. This level of disinfection is considered to also ensure protection from pathogenic bacteria. This treatment level appears most appropriate to address situations with known microbial contamination. The disinfection approaches currently available are all capable of achieving this level of treatment. Another approach, frequently used for small systems, is merely to require a measurable (chlorine) disinfectant residual in the distribution system. This appears technically and economically feasible in almost all applications, yet still yields substantial disinfection credit.

Of all the groundwater protective practices studied in this work, none showed a greater state-to-state variation than disinfection. Disinfection of groundwater, defined here as the application of at least a detectable chlorine residual or its equivalent, has been addressed in some fashion by all but one state (Connecticut). Nationwide, about 55% of community water systems, 28% of non-transient non-community water systems, and 17% of transient non-community groundwater

systems are disinfected. Individual state disinfection rates in these three categories range from 7%, 1%, and 1%, respectively, in Rhode Island, to 100% in these three categories in Florida, Kansas, Kentucky, and Texas.

The strikingly heterogeneous disinfection practices of the 50 states presented an opportunity to measure the relationship between statewide disinfection and statewide TCR MCL compliance rates. "Success" in this context was defined as having low TCR MCL violations. Results show that high disinfection rates are strongly associated with success. The ten highest disinfecting states have a mean TCR MCL violation rate of 18% over the 5-year period; the ten lowest disinfecting states have a mean TCR MCL violation rate of 38%. This represents a decrease in violations of more than 50% from the lowest disinfecting states to the highest. When the disinfection rates of the ten most successful states are compared with the disinfection rates of the ten least successful states, a similar result in favor of the disinfecting states is observed: the most successful states have an average disinfection rate of 72%, whereas the least successful states have an average disinfection rate of 29%. Disinfection, even at the low level of residual maintenance, is clearly associated with contaminant reduction.

Conclusions and Recommendations

A variety of studies indicates that significant fecal contamination of groundwater wells occurs in the US. Data on waterborne-disease outbreaks in groundwater systems suggest a range of problems leading to these events. Given the apparent risks associated with groundwater systems that have not been disinfected, disinfection of all groundwater systems is a prudent public-health policy. Additionally, protection of sources from fecal contamination and maintenance of well and system integrity are likely to act as substantial barriers to contamination reaching consumers.

There remains a need for additional research on microbial contamination in groundwater and its impacts: (1) data are required to better define the public health problem. These include information on known and estimated public health risks (outbreak and endemic-disease information, pathogenicity of contaminant organisms, and microbial risk assessments) and microbial occurrence in groundwaters and distribution systems. Answers to these will help further define the nature and scope of any public health problem; (2) a better understanding is needed of the factors affecting and limiting microbial contamination of groundwater sources. Outstanding issues are the site-specific hydrogeological properties affecting vulnerability to contamination and the physical and chemical properties governing fate and transport of microbials in the subsurface.

Disclaimer

The views expressed are those of the authors and do not necessarily represent those of the USEPA.

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7.
Group Water Scheme Bacteria Problems – A Local Authority Perspective
Ray Spain, Offaly Co. Co.

GROUP WATER SCHEME BACTERIA PROBLEMS – A LOCAL AUTHORITY PERSPECTIVE

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ABSTRACT

The EPA Quality of Drinking Water Report in Ireland for 2000, for the first time has separately assessed public drinking water supplies and group water scheme supplies for water quality. This was done in response to the poor performance of Group Water Schemes, which was highlighted in previous reports, and the lack of progress in tackling quality deficiencies. The principle water quality parameter of concern is bacterial and the Group Water Scheme sector face a number of hurdles in its efforts to overcome this problem and achieve the required results. Training programs for operatives and a quality assurance scheme are being developed. Increased funding is being made available through improvements in grant aid and annual operational subsidies. This drive to improve water quality in Group Water Schemes is a major element of the Governments Rural Water Program and the ongoing protection of our groundwater's is an integral part of this process.

BACKGROUND

In the 1950's public water supplies were confined to the larger urban areas. A need for water schemes to serve the rural areas of Ireland was recognised and a unique solution to this problem was brought into being. In 1952 the Minister for Local Government introduced a system of grants for the provision of private water supplies. This evolved in due course into the Group Water Scheme (GWS) sector as we know it today, with the first GWS being constructed in County Wicklow in 1958. An evaluation of the success of the programme was carried out in 1989 and concluded that the programme was an effective and efficient way of providing water supplies in rural areas. It also recommended greater involvement by Local Authorities, to help overcome problems being experienced by GWS and in 1997 responsibility for the management of the programme, as part of an integrated strategy for Rural Water Supplies was assigned to Local Authorities. This was a positive development placing the decision-making and prioritisation of resources as close as possible to the communities affected with Local Authorities and Group Schemes operating in partnership.

PRESENT POSITION

There are over 14,000 GWS in the country ranging in size from those serving just 2 houses to those serving over 2,000 houses. These schemes supply 150,000 households or 25% of the households in Ireland, with groundwater being the predominant source of water. The main aim when setting up the schemes was to provide water to each household, with schemes being constructed on a tight budget and often to a poor standard as a result. Sources were regularly selected because of their convenience rather than their suitability as a sustainable source of good quality water. GWS in essences are local do-it-yourself schemes which are managed on a day to day basis by voluntary committees who give of there time freely.

Up until 1997, many Local Authorities gave little consideration towards Group water schemes in their areas. They looked on them as inferior, poorly operated schemes with a substainal percentage requiring "take-over" due to poor construction standards and the excessive demands placed on GWS organisers. (Offaly County Council has taken over 12 private Group water schemes). Local Authority

100% GWS in Offaly[1] are groundwater

resources, which were already over stretch, were further stressed to cater for these schemes, leading to negative feelings by Local Authorities towards GWS. This was counter balanced by GWS, at times not unreal, perspective's of Local Authorities as uncooperative, dictatorial, slow moving, and bureaucratic organisations whose aims were not seen as compatible with their own.

Among the GWS sector there is a strong resistance to the use of chemicals in the treatment of water, often to the extent that where a water supply was shown to be of poor bacterial quality, schemes refused to provide disinfection facilities. This resistance to the use of chemicals in water treatment stems from not only a lack of understanding, but also of a legitimate fear with respect to the health implications, which the use of some of these chemicals may incur. This negativity towards traditional treatment systems by GWS has had a positive effect with the introduction into Ireland of "chemless" treatment systems such as membrane filter technologies and Ultra Violet systems. Of course water quality is not merely a matter of treatment, but of better protection of water resources in the environment, and of a co-ordinated groundwater and catchment management planning.

Although 85% funding is available from central government towards various improvement works on schemes, local funding may not be available or there is no will on the part of the already hard working scheme organisers to take on further projects. This results in a lot of schemes being in poor condition due to the lack of maintenance, which is exasperated, by the lack of funds.

WATER QUALITY

A recently completed water quality study by the National Federation of Group Water Schemes on raw water quality in the sources of 734 Group Water Schemes in the midlands showed that only 1 scheme was 100% compliant. Samples were taken on the schemes monthly for 12 months and were analysed for bacterial contamination including a range of physio-chemical parameters. This study was based on a pilot project in County Roscommon which sampled 65 sources of Group Water Schemes over a 12 month period from March 1999 and showed 91 (14%) out of 640 samples free from presumptive total coliforms. When looked at by source however, only 2 were clear of presumptive coliforms.

Group schemes in general operated independently until the introduction of the drinking water regulations in 1988, at which point Local Authorities began sampling of Group schemes for water quality. Sampling began at a low intensity, low frequency rate increasing annually up to the present monitoring regime as reported in the annual Drinking Water Quality Report published by the Environmental Protection Agency. Through this closer interaction, it became obvious that many Group Schemes were unaware of their responsibilities with respect to water quality. They did not understand the implications of the various water quality parameters and for the most part were willing to leave this up to the Local Authorities.

GROUP WATER SCHEMES IN COUNTY OFFALY

In County Offaly there are 23 private GWS serving 13,000 people with a further 18,000 on there own private source of water supply , and all of which use groundwater as their source of water. They range in size from those serving 12 houses up to the largest, which serves over 1,000 houses. Since the early 1980's Offaly County Council has been carrying out water quality monitoring on GWS in line with its own programme for public schemes and in excess of that which is required by the drinking water regulations. Over the years a good relationship has been built up between Offaly County Council and the GWS in its area, with co-operation and the sharing of resources been the norm. Nineteen of the GWS have disinfection facilities, 18 using sodium hypochlorite systems and 1 with a U.V. system. Of the 4 schemes without treatment, 2 have appointed contractors, to install U.V. systems in conjunction with sodium hypochlorite to provide a residual disinfection presence in the

34% schemes show water bacti contan after treatment.

network. One scheme is considering installation of treatment and the last is encountering problems with respect to right of access to their source.

Bacteriological quality of schemes when compared with the trend nationally for Group Schemes is good (see fig. 1.) however, is still far short of what is being achieved by public schemes. This difference is in part due to the number of schemes without treatment but probably more significantly as a result of inefficiently or incorrectly operated treatment systems. The levels of bacteriological contamination are shown in fig. 2 and fig. 3. The water quality project carried out by the National Federation of Group Water Schemes on raw water quality included 19 Offaly schemes, all which showed varying levels of bacteriological contamination and therefore with out adequate treatment, the poor performance of GWS with respect to water quality is set to continue.

THE ROSCOMMON PROJECT (RWMP, 2000)

In 1999 the Government approved a special water quality initiative for the group water scheme sector which would
- examine new approaches and new technologies for treating and disinfecting small scale rural water supply schemes
- closely monitor the quality of water in the source and group scheme distribution systems to determine the effectiveness of existing group water scheme treatment and disinfection system
- pilot test recommended technologies across a range of group schemes

A program was developed by Roscommon County Council in association with the National Rural Water Monitoring Committee and the Water Services National Training Group. The following is a concise summary of there findings and recommendations.
- Quality control and quality assurance procedures should be implemented.
- A Geographical Information System with mapping of all the assets of each scheme should be prepared.
- Monitoring of all Group Schemes to be increased in line with that carried out by the project.
- Many schemes are vulnerable to contamination from runoff or percolation. Source protection should be improved and protection plans prepared for each source.
- Operators of Group Schemes should be given adequate training.

An interesting point to note from the intensive monitoring program was the relationship between residual chlorine and observed presumptive total coliform levels (see fig. 4).

THE MAIN PROBLEMS AFFECTING GROUP WATER SCHEME WATER QUALITY

LOCATION OF SOURCES

The groundwater sources for Group Water Schemes can be divided into 2 categories. Boreholes normally 150 to 200mm in diameter, which have been located at a convenient location, usually close to the centre of population which the scheme will serve and springs/shallow wells located at a natural groundwater outfall, usually in a reasonably remote area. Of the 23 G.W.S. in County Offaly, 9 are served by boreholes and 14 by springs/shallow wells. The standard of wellhead protection on these sources varies greatly and the close proximity of some to pollution sources leaves a lot to be desired. A number of examples given below, probably bests illustrates the problems being experienced by GWS in this regard.

A GWS, which serves a population of 150 in north Offaly is served by a single borehole, located on the hard margin of a national secondary road. A heavy-duty unsealed manhole cover covers the borehole chamber. During periods of heavy rainfall, surface water seeps into the borehole chamber. This chamber is without adequate drainage outlets, which results in surface water entering the borehole.

A borehole serving a scheme of 15 houses is located in the garden, beside the house of the GWS operator. This is one of 11 houses within 170m of the borehole, an area of ribbon development, all of which are on septic tank systems.

A spring source serving a GWS of over 200 households is located on the flood plain of a river. The land in the immediate vicinity of the spring is of poor agricultural quality and is mainly covered by gorse interspersed by small pockets of grass. The spring's only means of protection is a 900mm concrete pipe and cover, which is sunk 600mm into the soil surface. Water is collected by a 150mm concrete pipe, and feeds a pump sump, with the water surface in the spring at approx. 300mm below ground level. Cattle are over wintered outdoors, which leads to considerable cutting up of the soil surface around the spring and animals regularly gather in the vicinity of the spring.

Concrete pipes and cover act as a pump sump again, to protect a GWS which serves 350 houses from a spring located adjacent to a surface water drain. There is a stock proof fence, which only extends to a distance of 1.0m around the spring. During periods of overflow in the spring, the outfall enters the drain, however during periods of heavy rainfall, water levels in the drain rise and back feed into the spring.

A series of springs, which feed a large GWS, are located in an area of poor peaty ground with low vulnerability, as defined by a groundwater protection scheme. The outer protection zone of the spring has a large area of extreme vulnerability and a sinking stream. Located within this area of extreme vulnerability and close to the sinking stream are a number of poorly maintained large dairy enterprises. A number of gross contamination events have occurred on this scheme and on each occasion incidents, which may have led to the contamination, have been investigated on the dairy enterprises. Section 12's have been served by the Local Authority on these enterprises requiring remedial woks to be carried out.

PROTECTION OF SOURCES

Offaly County Council incorporates all GWS sources in its Groundwater Protection Scheme, which was set-up in 1986 and consisted of a series of concentric circles around each source. Individual Groundwater Protection Schemes have been prepared for 4 Group Water Scheme sources and 10 sources for Local Authority schemes under the methodology prepared by the Geological Survey of Ireland, the Department of Environment and Local Government and the Environmental Protection Agency which, was published in 1999. It is also the policy of Offaly County Council, where a GWS is undertaking an upgrade that a protection scheme be developed for its groundwater sources.

TREATMENT

Nineteen of the 23 GWS in Offaly have installed disinfection facilities, however most are poorly operated. Sodium hypochlorite systems are the preferred option with only one U.V. system. At present only 2 of the GWS have standby units in the event of a breakdown in the primary system.

The large scheme mentioned above, first installed a disinfection system following a contamination incident where boil water notices were in place for 2 months. Following installation and a number of teething problems the GWS received the all clear with respect to water quality. The

following month, samples again showed the presents of coliforms and the GWS where requested to issue boil water notices and check on the disinfection system, to insure that it was operating satisfactorily. Follow up samples a number of days later again received the all clear, but the next months samples again showed the presence of coliforms. In liaison with the Environmental Health Officer an investigation was carried out on the Group scheme, identifying sample locations, possible dead ends in the pipe network with stagnant water, areas where re-infection may occur, and the checking of chlorine residual levels. Investigations continued for 2 days without identifying a reason for the contamination. Samples again showed the water to be free of coliforms and the all clear was given. During this time the Local Authority was receiving negative comment in the local press and radio, and from the Group Scheme. Daily visits and sampling on the scheme were commenced by Local Authority personnel and it was found that following the "all clear" the operator of the scheme was switching of the disinfection system, as he understood the water was now safe and therefore didn't require disinfection.

Complaints were received from a member of the public, who had just moved into the area which was served by a small Group Water Scheme, of a very noticeable swimming pool smell every Saturday in the water supply. Following a visit to the Group Scheme it was discovered that the operator was adding approximately 1 gallon of sodium hypochlorite directly into the well each Saturday morning. This was being done, explained the operator, as a result of a number of bad samples a few years ago. He was advised by the Local Authority to install a disinfection system and disinfect the well. They had installed the chlorination system but he had read in an article somewhere that for private water supplies it was good practice to disinfect a well on a regular base, this he had been doing every Saturday for a number of years.

A GWS in a neighbouring county has been on a boil water notice for the last 2 years and still refuses to install a disinfection system. This same County indicates that approximately 50% of its 80 private GWS are without disinfection facilities.

SCHEME MANAGEMENT

GWS are largely run by voluntary committees in there spare time and therefore may not be in a position to devote the time required or the expertise to operate the schemes satisfactorily. They are also untrained with respect to operating and managing a water scheme. Many of these people however, show great enthusiasm and personal interest in their schemes and should not be under estimated. This has been recognised by the National Federation of GWS and in association with the Water Services National Training Group, a training program has been put in place. A quality assurance scheme is also being piloted at present, based on the HACCP system (Hazard Analysis Critical Control Points) used in food safety, with the aim to improve scheme management and provide standard operating and recording procedures) (Byrne, 2001). The present focus of many schemes also needs to be changed, with the primary emphasis being placed on providing a good quality product to consumers. The present focus of many schemes is to provide cheap water and indeed many consumers believe that management are doing a good job once they provide water in the tap at low cost, with little regard to quality until something goes wrong.

Each year, when invoices are sent to GWS in Offaly to cover the cost of the annual water quality monitoring programme for their scheme, inquiries are received as to the extent of this monitoring and its necessity. Comments are generally of the nature, "it is very expensive and our water is okay anyway, sure we don't really need it".

There are a number of misconceptions of the source of groundwater by not only GWS but also my many members of the public. These range from the idea that there is a large reservoir/aquifer of water underneath the ground which is all interconnected throughout the entire country and the opposite to this in that each well taps into its own separate source, i.e. the next door neighbours water is different from mine. This was highlighted recently when carrying out a site visit to a Group Scheme in County Offaly who were contemplating an upgrade of the scheme and who had a history of

poor bacterial water quality. It was suggested to them, that this would be a good time to also have a groundwater protection scheme developed for the source, a shallow spring, and that this would be an aid in the prevention of any further deterioration in water quality. The operator relayed the story of an "expert" having checked out the source a number of years previously and identifying the source of the water as an underground river, which had as its source a sinking stream originating in Co. Cavan.

RESOURCES

Many GWS lack the resources to cope with everyday management of schemes or to deal with emergencies. Prior to the introduction of the annual subsidy grant in 1997, GWS lacked the funds to carry out required maintenance of schemes. Maintenance was restricted to that of an essential nature to maintain water in the tap. This mainly took the form of repairing leaks if pressures dropped and the replacement of pumps, which broke down. This meant that treatment facilities, where install, were not maintained and source protection was allowed to deteriorate. On a recent visit to a small GWS it was noticed that the collection chamber and its cover for a series of springs had disintegrated to the extent that it was now open to the elements and animals. . The "chamber" was now cover with a number of white-thorn bushes to prevent the ingress of large animals. This lack of resources is now being counteracted by the recent improvements in the annual subsides and the encouragement of GWS to take out annual operation/maintenance contracts with water treatment companies, This is being further enhanced by GWS coming together, with the aim of a combined contract for upgrading of schemes under Design Build & Operate approach.

CONCLUSIONS

The poor water quality and condition of many Group Water Schemes reflects the historical lack of investment in the sector as the 75% maximum grant was not an incentive for groups to provide treatment facilities and improve distribution systems. In the year 2000, a 100% grant was introduced for the provision of treatment facilities and an 85% grant for other improvement works. Also the annual operational subsidy was increased for schemes with treatment, all of which should now allow Group Schemes to initiate improvement works. Schemes who wish to employ full time personnel to operate treatment facilities are now in a position to do so and/or enter into operational contracts with treatment companies or provide treatment facilities under a "Design, Build and Operate" basis.

Hand in hand with the provision of adequate treatment facilities is the proper management of these facilities and this can only be achieved by the training of operators and management. This is being provided for by a series of training centres which have been set-up by the Water Services National Training Group around the country and the introduction of a quality management system.

The proverb "Prevention is better than the cure", is one which should be applied to water quality. This is best achieved by the development and implementation of river basin management and groundwater protection plans. River basin management projects have been set-up for most river catchments with groundwater being included as an integral part of these projects.

The European Union have issued warnings to Ireland with respect to drinking Water quality, however this should not be the only driving force behind providing a potable water supply. It is incumbent on everyone involved in the water industry, be they GWS, Local Authorities or professionals in the area to provide a quality service and a quality product. Structures are now in place for the GWS sector to allow this to happen and we will see major developments in the sector over the next number of years.
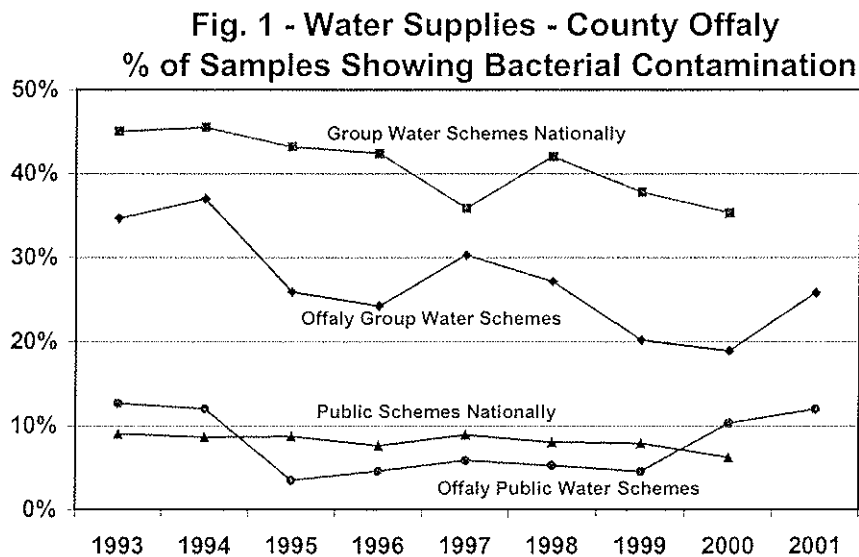
ACKNOWLEDGEMENTS

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Byrne,D. (2001). Introduction to Quality Assurance Scheme. National Federation of Group Water Schemes – GWS NEWS, Vol. 3, Issue 3, Dec 2001.
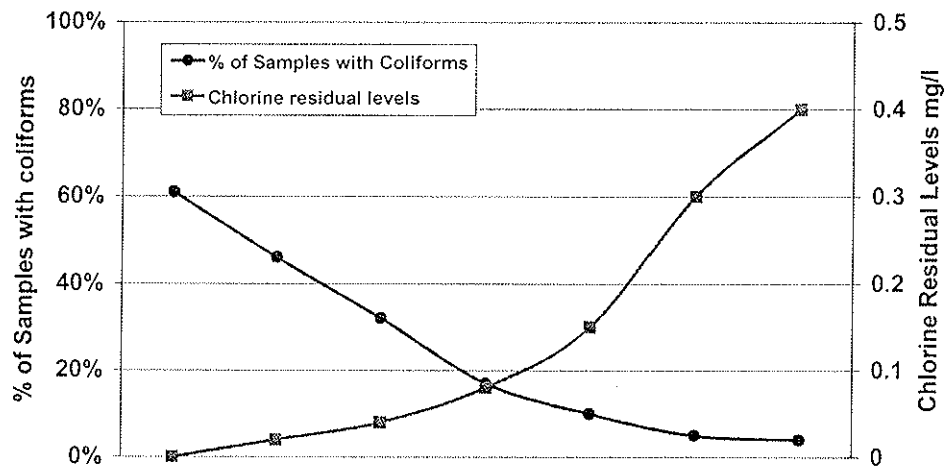
Fig. 1 - Water Supplies - County Offaly
% of Samples Showing Bacterial Contamination

Fig. 2 - Offaly Group Water Schemes
Faecal Coliforms Exceedance's

No. / 100ml

7
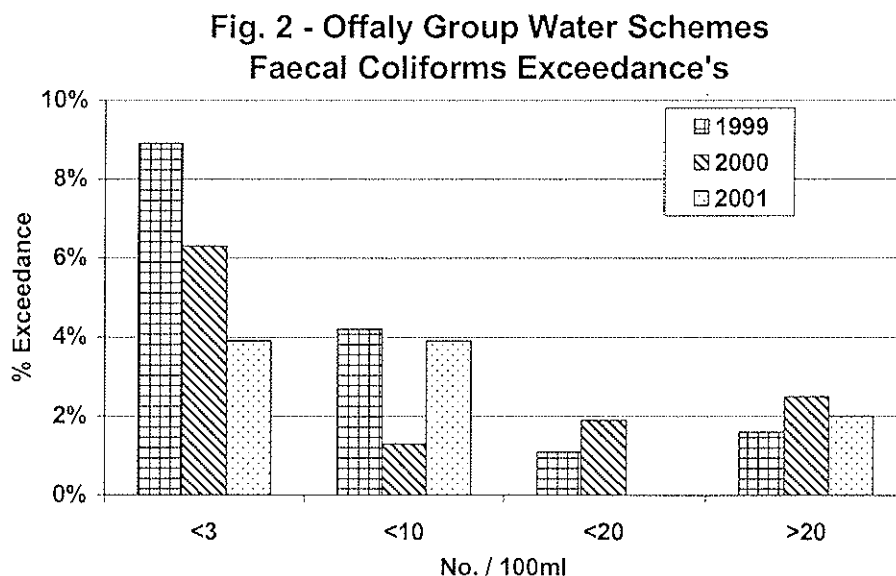
Fig. 3 - Offaly Group Water Schemes
Total Coliforms Exceedance's
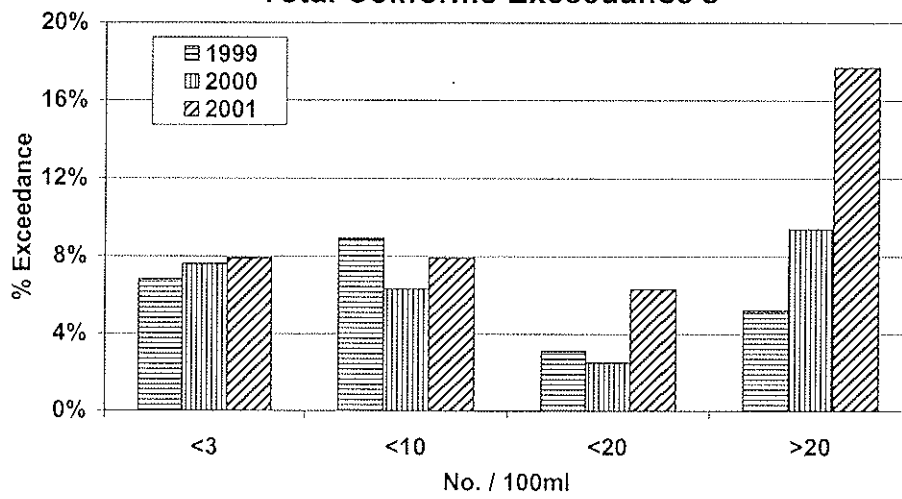
Fig. 4 - % of Samples with Coliforms Compared
with Chlorine Residuals

8.

Groundwater Quality – Sampling and Analysis
Peter Webster, Environmental Protection Agency, Cork.

GROUNDWATER QUALITY : SAMPLING & ANALYSIS

Peter Webster. Regional Chemist, Environmental Protection Agency

Abstract

Consideration of the design, implementation and quality assurance of groundwater sampling and analysis programs is essential to adequately characterising the chemical and microbiological composition of such waters. All too often however such programs are initiated without adequate consideration and definition of key variables resulting in the generation of analytical data that is, potentially, neither reliable, representative nor cost-effective. Since groundwater quality is influenced by wide regional variations in geology and land usage it is perhaps not surprising that few definitive protocols exist for the sampling and analysis of such waters. A central tenet of environmental programs should be that at every step in the process procedures should be identified to reduce the potential for error. Data quality objectives should be formally specified and techniques applied which ensure that data is fit for purpose. This paper does not purport to address all of the elements necessary to ensure that the most reliable data is obtained in every situation but aims to provide the reader with an insight into the range of variables, both practical and analytical, which should be considered if a satisfactory groundwater monitoring program, and the consequential environmental decisions and outcomes from it, are to be realised. The paper will concentrate on aspects of sampling, sample handling and analysis.

DEFINING THE SAMPLING PLAN

The adage "Garbage In = Garbage Out" applies equally well in the arena of groundwater monitoring as it does in Information Technology. There can be no doubt that the quality of data obtained on any sample is only as good as the sample itself. All too often however the attention given to sample program design falls short of that required to ensure reliable data. In part this is often influenced by the location of existing boreholes, knowledge of the aquifer history or land use, and economic considerations.

SAMPLING DESIGN

In developing a pilot study with no prior knowledge of aquifer composition or hydrogeology it is clear that there must be some element of "suck it and see" in initial design. Data obtained will require the program to be reassessed but this latter component is the first major hurdle. The choice of sampling location will depend largely on the objectives of the program but may require to consider the following aspects of sample design:

- **Haphazard**: Not recommended for groundwater flow since underlying assumptions are difficult to verify.
- **Judgemental**: The target population needs to be clearly defined, homogeneous and completely assessable so that sample bias is not a problem or specific environmental samples selected for their unique value rather than for purposes of making inference on the underlying population.
- **Simple Random sampling**: Not to be confused with haphazard sampling. This is useful for homogeneous populations. Other methods will give more accurate estimates of means if the population contains trends or patterns.
- **Stratified Random sampling**: Useful when a population can be broken down into parts that are internally homogeneous.

- **Cluster sampling**: Useful when population units cluster together and every unit in each randomly selected cluster can be measured.
- **Systematic sampling**: Usually the method of choice when estimating patterns or trends over space or time. Also useful in estimating the mean when trends and patterns are not present or when strictly random methods are impractical
- **Random sampling within Blocks**: Combines the benefits of Stratified Random and Systematic sampling
- **Double sampling**: Useful when there is a strong linear relationship between the variable of interest and a less expensive or more easily measured variable.
- **Search sampling**: Useful when historical information, site knowledge, or prior samples indicate where the object of the search may be found.

HOW MANY SAMPLES

The number off samples required to achieve a pre specified margin of error is defined by

$$N = \left(u * \frac{k}{p} \right)^2$$

where u = the standard Normal deviate
k = coefficient of variation (s.d. expressed as proportion of the mean)
p = desired precision (expressed as proportion of underlying mean)

The following table shows the approximate sample numbers required to estimate the mean (at 90% confidence) to various levels of precision D (expressed in standard deviation units)

Desired Precision D (at 90% confidence) as multiples of σ	0.1	0.15	0.2	0.25	0.3	0.4	0.5	0.75	1.0
No. samples	271	120	68	44	30	17	11	5	3

Thus if we wished to estimate the mean nitrate concentration to ± 3 mg/l for a data set with an underlying population mean of 12 mg/l we would need to take at least 44 samples. If instead we were satisfied with a wider precision say ± 6 mg/l we would only require 11 samples thus illustrating that if the precision is relaxed by a factor of two then *four* times fewer samples are required.

Similarly we can determine the precision achieved by a given sampling frequency by the relationship

$$P = t(\alpha) * \frac{k}{\sqrt{n}}$$

where n = number of samples
k and p are as defined above
t = Students "t" statistic for a given level of confidence α

The following table shows the precision (D) as a multiple of standard deviation with which the mean is estimated (at 90% confidence) for various sample sizes

No. of samples	6	12	18	24	36	52	104	365
Precision achieved (D) at 90% confidence as multiple of s	0.83	0.52	0.41	0.35	0.28	0.23	0.16	0.09

For example a mean nitrate concentration of 50mg/l calculated from 6 samples and having an s.d. of 10mg/l would have a precision of 8.3 mg/l, i.e. the 90% confidence interval will be 41.7 - 58.3 mg/l.

The large numbers of samples required to meet such statistically defined objectives will, in most cases, simply prove unrealistic or unaffordable. Little surprise then that programmes, including those designed to meet the EPA's own landfill monitoring recommendations are more likely to be based on a pragmatic compromise of monthly, quarterly etc. i.e. Systematic sampling. One of the outcomes of this approach is that it can take many years to build up a reliable estimate of underlying population trends, patterns or variability and thus early conclusions as to the quality of underlying populations should be avoided at all costs.

WHERE TO SAMPLE

This will often be determined by the objectives of the program and the history of aquifer / land use but consideration will need to be given to e.g. up-gradient, down-gradient, depth profiling and radial sampling to define aquifer boundaries and variability in composition.

SAMPLING PROTOCOLS

There is **NO** universally acceptable technique for groundwater sampling. Unlike surface waters they do not lend themselves to time weighted or flow proportioned automatic sampling devices. All groundwater sampling devices are intrusive and have drawbacks though some are better than others.

The most common approaches to sampling are set out in Table 1 below:

Equipment	Main Advantages	Main Disadvantages
Bailers	Low cost, simple to use, portable, dedicated or disposable options, suitable for VOCs	OK for sampling in water column but labour intensive if used for purging. Sample transfer causes aeration.
Inertial Pumps (e.g. Waterra)	Modest cost but portable and can be used as dedicated pump for purging and sampling. Operable up to ca. 60m depth. Dedicated tubing left in well	Causes significant mixing of water column and problems of sediment trapping.
Suction lift pumps (e.g. peristaltic)	Relatively inexpensive & portable. Tubing may be left in well	Only useful for shallow boreholes to ca. 10m. Can cause losses of VOCs and dissolved gases
Bladder pumps / Gas driven pumps	Easy to operate, reliable, portable and easy cleaning. Suitable for low flow sampling and organics.	Expensive and low discharge rate. Gas source required. Time consuming for sampling of deep wells
Submersible pumps	Variable flow rates suited to both purging and sampling. Easy to operate and suited to deep wells	Expensive, power source needed, requires decontamination between uses, very susceptible to damage from grit / sediment

Several references are available giving guidance on approaches to sampling of groundwater and these are listed at the end of this paper. In essence the objective is to obtain as representative sample as is practicable. This clearly infers some element of compromise particularly for parameters such as organics where the distribution of non-aqueous phase components (LNAPLs, DNAPLs, VOCs) may be distributed by the sampling technique. In such cases consideration should be given to a multi-tasking approach which may involve discrete sampling of the static water column or selected phases before, during and post well purging.

With respect to the degree of purging opinions vary on this however a figure of at least 3 well volumes is generally regarded as adequate for most circumstances. Bear in mid however that for a 50m deep, 50 mm well casing this involves the purging of at least 300 litres of water.

SAMPLING PREPARATION

It is highly desirable that all aspects of the sampling are agreed between the requestor, sampler and the receiving laboratory and these should include aspects such as site maps, contact names, site safety information, sampling sequences (clean wells first), sampling protocols, cleaning / decontamination procedures for submersible pumps etc., purging protocols, QA/QC procedures (see later), field measurements, list of parameters for analysis, sample bottles and preservation procedures, storage and transport criteria and chain of custody documentation.

SAMPLE HANDLING

While the physical act of sampling has a marked influence on the sample quality it is also important to recognise that not all parameters are stable once out of the ground. Exposure to air can result in changes in solubility of gases affecting e.g. pH, Alkalinity or result in precipitation of metals from samples taken from anoxic wells. Some protocols may insist on field measurements of key parameters such as pH and on immediate sample pre-treatment in the field for parameters such as metals.

While some field measurements can be made with a degree of accuracy approaching that of laboratory measurements they are generally somewhat poorer. In addition while from a comfortable office it may seem both practical and desirable for example to filter samples for metals analysis in-situ the reality of doing this on a cold wet and blustery February morning in Ireland is somewhat less ideal.

It is important therefore that how the sample is to be handled is discussed before commencing any sampling. Similarly preservation measures needed for analyses such as Cyanides, Ammonia, Metals, Organics and the sample bottles required should be fully documented. In the case of some samples such as those for microbiological assay there will be a need to transport them to the receiving laboratory within a fixed time period.

Some widely respected references stipulate cooling samples to the mystical 4°C for transport however while this is achievable using refrigerated vehicles it is seldom, if ever, achieved in common practice. All samples, particularly those for microbiological analysis, should be kept in a cooled environment and it is more common for samples to be transported in picnic style cool boxes (with ice packs) or held in portable 12v refrigerators during transportation to the lab. This is an important aspect of the sampling process and one which should not be overlooked particularly where subsequent data may be the subject of legal proceedings and where inappropriate sample handling has the potential to invalidate any case no matter how good the analysis.

SPECIFYING THE ANALYSIS

The variability of groundwater and the reasons for its monitoring mean that specifying the analysis required is often site-specific. Nonetheless there are a number of key parameters that will allow the quality of groundwater to be characterized. While most of these bulk chemistry parameters are easily defined and well understood there are some that require a little more consideration. While much of the focus of groundwater monitoring is generally directed towards characterizing its principal inorganic parameters and physical characteristics e.g. pH, nutrients, anions, cations, soluble metals etc. the analytical suite may include empirical parameters such as COD, BOD, and other parameters defined by the methodology used e.g. Phenol Index, TOC / NPOC, SVOCs depending on the intended use.

Where this is the case it is <u>important</u> that both the requestor, the laboratory, and any regulatory body are fully agreed on analytical protocols to be used and the performance to be achieved.

Many conventional laboratory analytical techniques have the capability for sub-mg/l measurement (equivalent to 1 minute in 2 years) and routine trace analysis is capable of sub μg/l concentrations (equivalent to 1 second in 32 years). Most analysis at mg/l concentrations will present few serious challenges for the analyst however and there are circumstances where use of portable field test kits for in-situ use can be of enormous value as a screening tool particularly e.g. in the assessment of contaminated waters. Their principal disadvantage is that they are generally less sensitive than laboratory techniques and more prone to operator error. On the other hand analysis to ppb concentrations represents, in many cases, considerably greater analytical challenges and with it and attendant increases in cost.

A common area of confusion is metals analysis where the failure to specify whether or not samples should be filtered can give rise to serious problems. Filtration is usually taken as meaning the sample is passed through a 0.45 μm filter and the filtrate (soluble metals) portion analysed after acidification. It may be the case that the sedimentary fraction is also of interest but release of metals from this (solid) material is dependent on the approach used. For groundwater evaluation is is recommended that samples be filtered but this should preferably be stated in the accompanying test report.

Considerable attention requires to be given to complex but often poorly defined analysis such as trace organics. The proliferation of acronyms is common in analytical chemistry e.g. VOCs, SVOCs, DNAPLs, PAHs, PCBs, O-CLs, TPHs. In many cases these represent families of compounds that are defined by the methodology used for their determination. There are several international reference procedures which are widely used, e.g. US EPA Method 524.2 (Purge &Trap GCMS for Volatile Organic Compounds), however it is important to realise that these generally cover only a family of substances which are quantifiable under specific conditions and despite the broad range of substances reported (52 in the above case) they may <u>not</u> include what is in your groundwater.

For this reason it may be useful to regard such techniques as a screening tool that will give an indication of problems but to examine samples in some greater detail using techniques optimised for specific classes of substance e.g. Phenols, PAHs, Pesticides etc. Even then there is no guarantee that we will have categorised everything in the groundwater as all analytical techniques have some limitations.

QUALITY CONTROL OF DATA

You've gone to the hard work, effort, and expense of obtaining your sample and deciding what you want done on it ... now all that's left is to get it analysed. Unfortunately this is an area where all the preceding effort can be undone if the receiving laboratory does not operate an adequate Quality Assurance (QA) and Analytical Quality Control (AQC) program

In relation to the quality of data produced there are several factors that should be considered before embarking on the choice of laboratory:

- Does the laboratory have experience of this type of analysis
- Are its staff trained and competent in this work
- What equipment will it use for the analysis
- Is this subject to regular maintenance and calibration
- Does the laboratory have documented SOPs
- Has the performance of test methods been validated and are performance criteria available
- What are the Practical Reporting Limits
- How much will it cost

Although cost is at the bottom of this quality list it is invariably going to be at the top of someone else's list and thereby presents a potential problem for the operation of the monitoring program.

There are many in-house, company, and commercial laboratories capable of undertaking groundwater analysis who will, as a matter of good practice, have in place a program of measures to ensure that the quality of their data is reliable, however there also many who don't. It is imperative that you are satisfied that the accuracy, precision and reliability of the resulting data is fit for its intended purpose.

There are many aspects to a good QA / QC program including all of the above points. I would like to consider a few of the key areas further.

CHOICE OF TEST METHOD

There is generally no shortage of analytical methods at the analysts' disposal and in some cases the choice is almost too wide. European (BS / ISO / CEN / DIN) reference methods tend to be written in broad terms. By contrast many US EPA methods tend to incorporate strict requirements designed to meet their Federal legislation. It is not surprising then that most laboratories will opt to use documented in-house procedures which are based on these international or recognised standard test methods. Even if the test method source is a compendium series such as AWWA *"Standard Methods"* or the UK Standing Committee of Analysis "MEWAM" series this will invariably be the case.

What is critical however is not whether the lab can follow the recipe book but that it is able to adequately define how the method performs using their procedures, equipment, and staff in their laboratory. In this respect factors such as Accuracy, Precision, % Recovery, Linearity, Limits of Detection, Practical Reporting limits should be known. These factors can readily be determined by analysing blanks and samples of varying concentration in replicate over several batches to obtain an estimate of procedural variability. Statistical assessment of the variations will yield information on the robustness of the technique and allow for an assessment of the lowest value reportable with a known statistical confidence i.e. the Practical Reporting Limit ... often misquoted as the Limit of Detection.

Routine performance can be assessed by means of Statistical Control Charts (Shewart Charts), use of Certified Reference Materials (CRMs), replicate analyses, and participation in inter-laboratory quality assurance schemes such as WRc *"Aquacheck"*.

MEASUREMENT UNCERTAINTY

Measurement uncertainty in an engineering or metrology context is often controlled and calculable to the smallest margin so it comes as something of a shock to find that, at best, estimates of the uncertainty of chemical measurements tend to be in the range 2 – 20% depending on the concentration of the parameter and the complexity of analysis.

While metrology defines the uncertainty by means of systematic identification and evaluation of contributory factors such approaches are only really applicable to simple chemical scenarios. For most routine chemical analysis measurement uncertainty is more pragmatically defined by consideration of random variations over time using replicate analysis.

Statements of measurement uncertainty are seldom, if ever, quoted or requested on test reports however their significance can be important in determining compliance with legislative criteria such as Maximum Allowable Concentrations (MACs) as prescribed in, e.g. standards for water abstraction, EPA licences etc. and they must be given consideration if the data is to be used in legal context.

DATA INTERPRETATION

LIMIT OF DETECTION

Of all the common terminology used in analytical chemistry the term Limit of Detection must surely be the most misused and misunderstood. It is important, particularly when comparing performance quoted by different laboratories, e.g. commercial contract laboratories, that the requestor satisfies himself or herself as to the manner in which this parameter is being defined.

There are various statistical approaches can be employed to derive the lowest concentration which can be differentiated from background noise at a given level of confidence. These mathematically derived values are then usually subject to some precautionary factoring by some laboratories to yield a higher Practical Reporting Limit (PRL). The result of this is that the uninformed observer may wrongly conclude that one laboratory is "better" than the other simply because one uses the initial calculated LoD and the other a factored PRL. Of the two the PRL approach is the more desirable since it provides and additional margin of safety in declaring that a parameter has been quantitatively determined.

Watch out for this pitfall and if you are unsure ask your laboratory for further details ... if they can't provide you with a suitable explain as to why they are reporting what they do then don't use them!.

REPORTING NOTHING

While for the bulk parameters in groundwater the question of how to report nothing found is hardly a common occurrence it is almost the norm for most trace measurements especially organics. Some of the approaches that have been observed include:

- "–"   Tells you absolutely nothing at all about the measurement
- 0   Is it really zero or is your measurement system not sensitive enough ?
- NA   "Not analysed" / "Not available" or what ?
- ND   "Not Detected" ... only really helpful if the PRL is also quoted
- < 10   Probably the most common approach but does not confirm the measured value for statistical evaluation.
- < 10 (6.3)   Identifies both the statistically valid measurement and the actual measured value. Good for stats purposes but runs the risk that 6.3 will be cited rather than < 10

LABORATORY ACCREDITATION

It is worth concluding this paper with a cautionary note on laboratory accreditation. The process of laboratories being accredited by their national competent bodies (e.g. NAB , UKAS) has grown in recent years as commercial competitive pressures have increased. Many public services and regulatory authorities both in Ireland and the UK have followed suit to ensure that the quality of their data is robust. The result is often that this "added value" is reflected in the costs of analysis.

Achieving and maintaining accreditation is both a costly and time consuming process for laboratories. It requires control over training, analytical procedures, documentation and auditing of laboratory practice, procedures for redress, criteria for service delivery, contracts and much more. This does not mean to say that non-accredited laboratories are not capable of producing data of comparable quality however accreditation requires much more than being able to produce good results and the requestor should always satisfy themselves, before any regulator does, that data quality is of an adequate standard.

While it is not a prerequisite to use accredited laboratories even for EPA monitoring returns, there appears to be a growing and welcome trend to do so particularly for specialist organics analyses. One important point to bear in mind however is that laboratories are accredited for <u>specific</u> analysis only and this is defined in their Scope of Accreditation. You may find that while you think the lab is accredited for organics it might only have accreditation for pH so do check its Scope before commissioning any analyses.

REFERENCES

The following are a list of general references on this overall subject area which may be of interest to the reader.

SAMPLING

- BS ISO 5667 / BS 6068 series "Water Quality"
- UK Standing Committee of Analysts "MEWAM" series *"Principles of Sampling and Accuracy of Analytical Results 1980"*, HMSO, UK. ISBN 0 11 751491 8
- NS 29 *"Handbook on the design and Interpretation of Monitoring Programmes"*, Water Research Centre, UK. ISBN 0 902156 72 1
- *"Monitoring of Landfill sites - Guidance on Monitoring of landfill leachate, groundwater and surface water"* Environment Agency, UK. ISBN
- *"Low flow (minimal drawdown) groundwater sampling procedures"* US EPA Report No. EPA/504/S-95/504

ANALYSIS

- *"Standard Methods for the Examination of Water and Wastewater 20th Ed"*, AWWA/ APHA ISBN 0-87553-235-7.
- UK Standing Committee of Analysts "MEWAM" Blue Book series *"Method for the Examination of Waters and Associated Materials*, HMSO, UK.
- BS / ISO / CEN / DIN International standard test methods

QUALITY CONTROL

- ENV ISO 13530:1999 *"Water Quality - Guide to Analytical quality control for water analysis"*
- NS 30 "A manual on Analytical Quality Control for the Water Industry" Water Research Centre, UK, ISBN 0 902156 85 3
- Report No. CO 4239 "Quality Control Charts in Routine Analysis", Water Research Centre, UK.
- ISO 7870 :1993 "Control Charts - General Guide and Interpretation"
- ISO 8258: 1991 "Shewart Control Charts"
- ISO 17025 *"General Requirements for the competence of testing and calibration laboratories"*, Available from NSAI
- *"Quantiflyng Uncertainty in Analytical Measurement"* Eurachem/CITAC

GENERAL

- EPA Landfill Monitoring Manual 1995 (under revision)
- *"Practical Handbook of Groundwater Monitoring"* D.M. Neilsen , ISBN 0-87371-124-6
- *"Statistical Procedures for analysis of Environmental Monitoring data & Risk assessment"* E.A.McBean , F.A. Rovers, ISBN 0-13-675018-4
- *"Principles of Environmental Sampling"* L.H. Keith, ISBN 0-841203152-4

9.
In-Situ Bioremediation of High Nitrate Chalk Aquifers
Paul Godbold, WRc PLC, UK

Ion exchange — resin.
But produces sludge
- not common UK treatment

Bio reactors
bacteria grown in reactor
problem - NO_x residual
+ requirement to clean
up bio-fouling

IN-SITU BIOREMEDIATION OF HIGH NITRATE CHALK AQUIFERS

Paul Godbold, MSc, FGS, WRc plc

High nitrate concentrations have a major impact on the availability of groundwater for potable supplies. The costs of conventional surface treatment methods can be high and where possible, blending high and low sources is a preferred supply option in the UK. With increasing nitrate concentrations in groundwater, alternative, cost-effective and sustainable treatment technologies will be required to combat the problem in the foreseeable future. Enhancing the intrinsic denitrifying capacity of aquifer systems to reduce nitrate could therefore offer major technical, economic and environmental advantages over conventional treatment solutions or blending. This technology offers a sustainable approach to nitrate removal without the generation of waste streams. Successful laboratory-scale studies (undertaken by WRc plc and Imperial College London) have used flow through microcosms to develop data sets for the parameterisation of a biochemical transport model. The results of this study have been encouraging, and have successfully facilitated quantitative modelling of the denitrification process for the first time using British aquifer material. This paper presents a brief overview of the denitrification process (stimulated by addition of organic carbon), and outlines the transition from laboratory studies to the implementation of a pilot in-situ field trial currently being managed by WRc. The field trial, initiated in March 2002 aims to examine process optimization, and to assess the technical feasibility of in-situ nitrate bioremediation in Chalk groundwaters. As yet there are insufficient data to enable detailed interpretation, but initial indications are encouraging.

INTRODUCTION

Enhancing the intrinsic bioremediation potential of aquifer systems to reduce nitrate (NO_3^-) concentrations in potable water to comply with drinking water standards could offer major technical, economic and environmental advantages to conventional engineering solutions to water treatment or blending.

Nitrate movement into groundwater is principally from diffuse sources and is related primarily to land use practices and intensive agriculture that depends heavily on nitrate for crop production. Although measures to reduce agricultural inputs of nitrate have been implemented in Nitrate Vulnerable Zones (NVZs), nitrate leaching will continue to present a problem and is difficult to control in practice because of the complex interactions between soil management, fertiliser use and environmental factors. Significant areas of our major aquifers are now impacted to such an extent that the nitrate concentrations in the water abstracted from many UK potable supply boreholes are approaching or exceeding the maximum admissible concentration (MAC) for nitrate in drinking water of 50 mg NO_3^- l^{-1} (11.3 mg NO_3-N l^{-1}) (CEC 1998).

The present methods of treatment available to industry are mixing with low-nitrate water or removal by on site treatment. European opinions are moving against such on-site and end of pipe treatment and towards pollution prevention and aquifer clean-up, i.e. *in situ* treatment (CEC 1996).

Cost-effective and sustainable treatment technologies will therefore be required in the foreseeable future to remove nitrate from water supplies as resources of low nitrate water for blending become fully exploited. A treatment technique, which meets such a requirement, is to harness natural biochemical processes in aquifers to destroy nitrate by *in situ* denitrification. The principal by product of this process is gaseous and the technique would provide a completely new efficient groundwater treatment option that is in line with the current environmental expectations of the European Community.

Used in Austra, France, Netherlands, US, Israel

S+G to chalk.

0·5 to 5 Ml/day

Denitrification is the reduction of nitrate to gaseous forms of nitrogen (primarily dinitrogen [N_2] and some nitrous oxide [N_2O]); and occurs naturally in the ground and in groundwater under anaerobic conditions.

Denitrification is mediated predominantly by indigenous bacteria and can be summarised by:

$$2NO_3 + C_2H_5OH \rightarrow N_2 + 3H_2O + 2CO_2$$

nitrate   glucose?
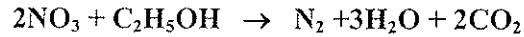
The feasibility of enhancing natural biochemical processes in aquifers as a treatment process to remove groundwater nitrate by *in situ* denitrification has been the subject of recent study by WRc and others. The result is a strategy and design for the future development and implementation of *in situ* bioremediation of nitrate in groundwater by denitrification to full operational scale.

Effective strategies for restoring groundwater quality, however, require realistic predictions of the effects of different management options as a result of the difficulties in observing and characterising the subsurface. Therefore, a linked experimental and modelling approach was developed to facilitate the cost-effective and objective evaluation of the technical feasibility and specification of *in situ* bioremediation systems for nitrate removal.

The overall objectives of this project were:
- To develop a biochemical transport model of enhanced denitrification in groundwater and assess the feasibility of this approach for potable water treatment of nitrate;
- To assess biofouling potential and optimum denitrification management strategies and to provide model parameterisation data using a laboratory experimental approach;
- To critically assess recent international and UK literature on denitrification in aquifers and identify and compare possible treatment design options for *in situ* bioremediation of nitrate;
- To assess the significance of biofilm-water flow interactions in fractured aquifer matrices from a critical review of scientific literature and recommend an experimental and modelling framework to account for these processes during *in situ* bioremediation;
- To undertake an economic appraisal of in situ bioremediation by denitrification relative to conventional surface treatment methods of nitrate removal from groundwater;
- To optimise the configuration and determine operational boundaries of *in situ* bioremediation systems for nitrate; and
- To design an *in situ* bioremediation scheme and pilot scale system for nitrate destruction in a sandstone aquifer.
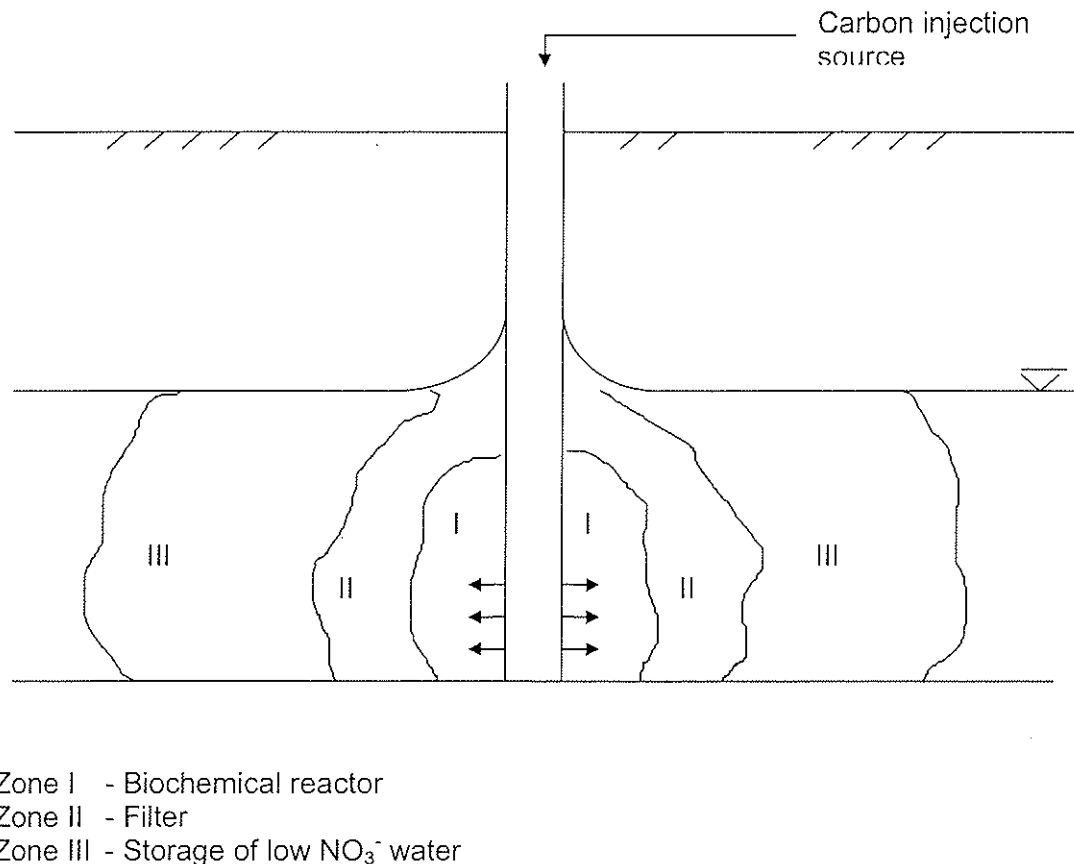
A detailed and comprehensive technical and economic assessment of the feasibility of *in situ* bioremediation of high nitrate groundwater by enhanced denitrification is outside the scope of this paper. In depth information is available in other publications (Cartmell *et al.* 2002).

ENHANCEMENT OF NATURAL DENITRIFICATION

Denitrification is a naturally occurring process, the enhancement of which can be achieved by producing favourable conditions for the development of heterotrophic populations of denitrifying bacteria. This is achieved through the injection of an organic carbon food source into the aquifer creating a natural biologically anoxic active zone (Zone I in Figure 1) in the vicinity of the well, where nitrate can be reduced to nitrogen gas. Zone II serves as a filter zone in which the turbidity and suspended solids are removed. Zone III will subsequently contain groundwater containing reduced concentrations of nitrate. Successful *in situ* microbial denitrification depends mainly upon the prevention of well/aquifer clogging and on process control.

The controlled introduction of an organic carbon food source, for example glucose, stimulates microbial growth. Once the microbial population is increased the practice of glucose limitation by

regulated injections encourages the increased bacterial population to utilise nitrate as an alternative source, facilitating denitrification.

Carbon injection source

Zone I   - Biochemical reactor
Zone II - Filter
Zone III - Storage of low NO$_3^-$ water

Figure 1   Schematic Description of *in-Situ* Denitrification

LABORATORY DENITRIFICATION STUDIES

Effective strategies for restoring groundwater quality require realistic predictions of the effects of different management options as a result of the difficulties in observing and characterising the subsurface. This requires a thorough understanding of biologically-mediated transport processes in porous media (Clement *et al.* 1997). Therefore, a linked experimental and modelling approach was developed in this project to facilitate the cost-effective and objective evaluation of the technical feasibility and specification of *in situ* bioremediation systems for nitrate removal. The experimental programme of laboratory trials concerned biofouling and evaluation of the carbon supply regime.

The objective of the experimental work was to assess biofouling potential and optimum denitrification management strategies and to provide model parameterisation data using a laboratory experimental approach.

The principal problem in assessing the feasibility of subsurface denitrification is the lack of technical understanding of the complex chemical and biological processes controlling the activity of denitrifying bioremediation. Thus, controlled laboratory experiments are critical in understanding biochemical process responses particularly in relation to controlling biofouling episodes in bioremediation. The laboratory work facilitated parameterisation of the biochemical model particularly in relation to biofouling potential and temperature using a flow-through microcosm apparatus. Methods to optimise bioremediation through recirculation or pulse flows were also examined. Specific experimental objectives were:

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- Determination of the commencement of denitrification under anaerobic aquifer conditions using excess carbon to generate denitrification conditions. Further optimisation of the minimum carbon concentration required;

- Determination of the potential for biofouling of substrate-injection borehole facilities and surrounding aquifer material. The significance and extent of biofouling was measured by correlating potential decreases in permeability and porosity of the sandstone media with carbon input, flow rate and microbial biomass Adenosine TriPhosphate (ATP) concentrations;

- Determination of the effectiveness of recirculation of nutrient solution using injection boreholes to enhance nitrate removal and/or improve water quality;

- Assessment with regard to mode of operation of substrates, e.g. continuous or pulse-flow to optimise bioremediation and reduce biofouling; and

- Determination of the quality of the treated water in relation to residual organics, nitrite, oxygen concentrations or bacterial contamination.

The microcosms used were constructed of glass and divided into sections so as to obtain biomass measurements throughout the length of the sandstone core. The design incorporated entry of the nutrient solution at the base of the microcosm controlling the upward vertical flow of the solution to minimise the possibility of preferential flow mechanisms. The microcosms were supplied with nutrient solutions under specified flow regimes at rates determined from a radial flow model simulating aquifer abstraction and injection boreholes set at a distance of 15m apart.

The experimental work has shown that the introduction of a glucose carbon source into sandstone aquifer material can stimulate denitrification activity within several hours reducing nitrate concentrations to below the EC limit of 11.3 mg NO_3-N l^{-1} within 10 days at groundwater temperatures of 12 °C and in the presence of a continuous oxygen supply of 8 mg O_2 l^{-1}. The supply of carbon can therefore be used as the mechanism for precise control and management of enhanced *in situ* denitrification to remove groundwater nitrate concentrations. A carbon concentration in the region of 40 mg C l^{-1} was required to reduce 25 mg NO_3-N l^{-1} to below 10 mg NO_3-N l^{-1}. Residual carbon concentrations were in the region 2.0 mg C l^{-1}.

Biofouling due to excess biomass growth was not observed to be significant in either the continuous or the pulse flow microcosms, probably as a result of the limiting carbon concentrations applied.

Whilst the information provided by laboratory experimentation does not necessarily relate directly to denitrification in the field, this approach allows for the close control of experimental variables governing denitrification, producing data upon which the representative description of the true system can be based. The data can also be applied further in the development of models of denitrification at the aquifer scale.

COMPUTER MODELLING

Mathematical models are playing an increasingly important role in the development of groundwater protection and bioremediation strategies. The attainment of predictive capabilities for denitrification is best achieved through the use of mathematical models that examine the transport of nitrate and its full biological transformations in an aquifer. The combination of a physically based transport model simulating solute movement and a biochemical model describing the interactions of solutes and biomass offers the potential to test, optimise and evaluate the feasibility of nitrate removal systems and configurations.

In this study a biochemical model has been integrated into a three dimensional model of a groundwater abstraction/injection system to develop a biochemical transport model to assess *in situ* bioremediation treatment systems. The biochemical model applied is based on a denitrification numerical model (Kinzelbach *et al.* 1991) for biological transformations in unidirectional flow. The

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three dimensional model simulates the same processes but through an aquifer in response to abstraction and injection patterns. In Kinzelbach's model the nitrate and carbon sources are mixed in the input water to ensure the best conditions for denitrification. This cannot be achieved in the real world, but is approximately reproduced in a daisy wheel abstraction/injection configuration. The flow in this case is predominantly radially convergent, and interaction between the native, nitrate-rich water and the carbon source is governed by the geometry of the daisy wheel system and by the natural aquifer dispersivity. The efficacy of the process depends on the degree of interaction which can be achieved, and a conceptually realistic model was required to provide an understanding of, and prototype design for, the *in situ* denitrification treatment system.

Simulations were undertaken using parameter values considered to be appropriate to a field scale design. The basic output of the model provided concentrations of carbon and nitrate in water discharged at the abstraction well.

Extensive model runs have been completed and five particular issues have been studied, with the results summarised in the accompanying figures:

- the relationship between the number of wells in the daisy wheel and system efficacy;

- the relationship between the radial position of the injection wells and system efficacy;

- the effect of dispersivity on the predictions;

- the effect of bypass flow on the predictions;

- the effect of short term fluctuations in the background nitrate concentration on the output from the abstraction well.

DENITRIFICATION TREATMENT SYSTEM DESIGN

The experimental and modelling work has demonstrated that *in situ* bioremediation of nitrate in groundwater is a technically feasible proposition. Moreover, the understanding and modelling capability of the biochemical and hydrogeological processes in sandstone aquifer material has developed to the extent that a full pilot scale treatment plant can be designed for the treatment of nitrate in sandstone aquifers. Although the original laboratory studies and subsequent modelling have been conducted using sandstone cores the technology is equally applicable to Chalk aquifers.

The basic design of the denitrification treatment system is based on the premise that a groundwater resource, exploited by a public supply borehole, is polluted by nitrate to unacceptable levels (>50 mg NO_2 l^{-1}). The borehole will have to be abandoned or the groundwater will require treatment if it is to be used as a potable supply. The treatment system comprises a ring of boreholes around a public supply borehole, through which a nutrient solution is injected into the groundwater to supply the energy to enhance the natural bioremediation processes of denitrification. The groundwater flow in the aquifer supplying the groundwater, in the vicinity of the public supply borehole, is assumed to be radial towards the borehole. The ring of injection boreholes is commonly known as a 'daisy wheel' and the overall surface design of an example daisy wheel is shown in Figure 2.

5
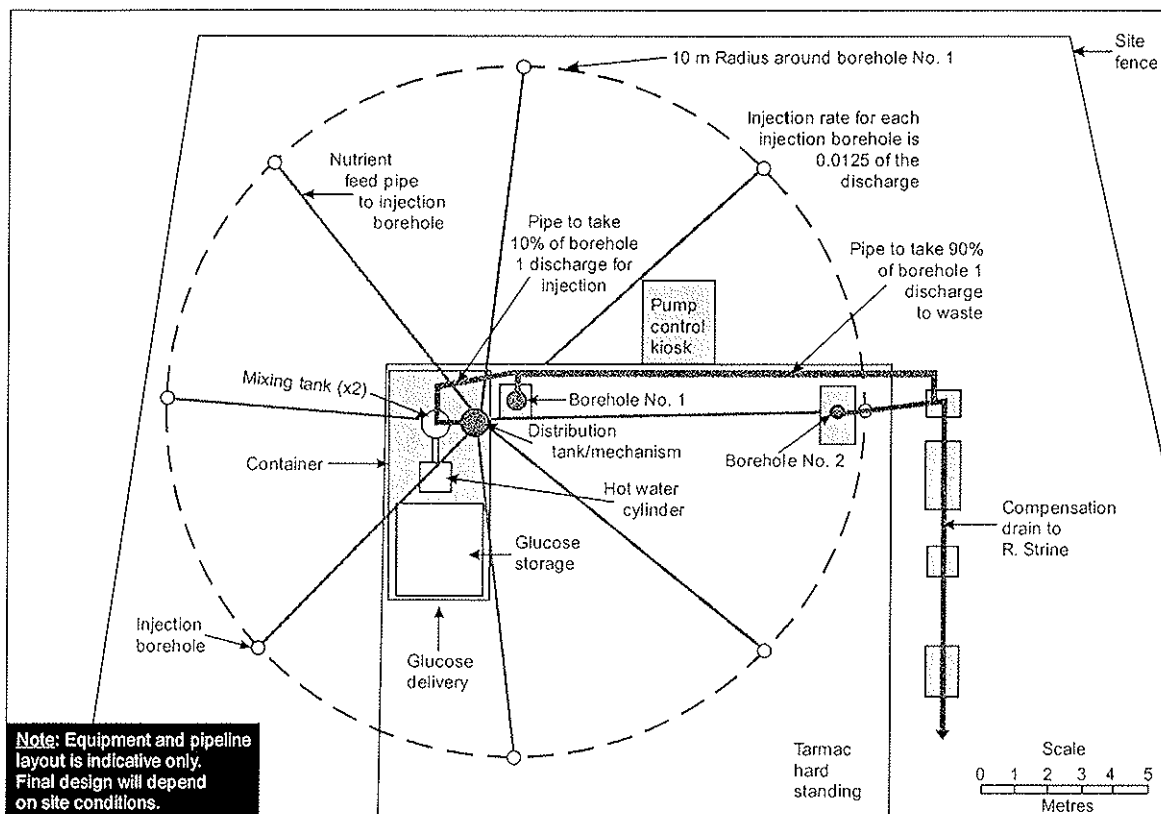
Figure 2:   Diagrammatic Plan of a Daisy Wheel Denitrification Plant

The main features of a full scale daisy wheel configuration are shown in Figure 2 but the final design at any site will differ from this slightly, depending on site characteristics. The factors underpinning the design of a 'daisy wheel' are:

- radius of daisy wheel;
- spacing of injection wells around daisy wheel;
- depth of all boreholes;
- length of screen in all boreholes;
- diameter of all boreholes;
- discharge rate (size of pump) in discharge borehole;
- recharge rate in all injection wells.

The success of the denitrification treatment design and operation will be measured in the observed reduction of nitrate in the discharge from the central borehole under the different test scenario conditions. The success of the system relies on groundwater movement from each of the injection boreholes towards the central abstraction point, with the volume of treated water between each borehole being a function of the dispersivity within the aquifer.

CASE STUDY: PILOT SCALE IN-SITU FIELD TRIAL

Following considerable interest from the UK Water Utility companies an operational pumping station has been selected for the installation of the first UK based pilot scale denitrification scheme. The station draws water from the Lower Chalk aquifer and has been taken out of supply for the duration of the trial.

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As the field trial is a pilot study the full scale geometric daisy wheel configuration was reduced to comprise three injection boreholes effectively positioned in an arc covering approximately one fifth of the radial area surrounding the abstraction borehole.

In order to update the biochemical model and provide sufficient understanding of the site hydraulics it was necessary to advance an observation borehole to undertake pumping tests, a tracer test and a geophysical logging survey. The results of this exercise provided groundwater travel times, an indication of travel pathways (a combination of fissure and matrix flow), Chalk porosity ranges and an evaluation of the effective depth of the aquifer contributing to the abstracted flow.

Using this site specific information the biochemical model was updated, and used to determine an abstraction rate to provide sufficient 'residence time' for groundwater passing through the system to be subjected to denitrification. In addition, the radial distance of the injection boreholes was optimised and the injection concentration for the glucose food source was determined (Godbold 2001).

Following the finalisation of the design, three injection boreholes were advanced at 80m from the abstraction borehole. In addition to the original observation borehole a second borehole was placed between the injection and abstraction boreholes to provide two locations at which groundwater could be sampled prior to reaching the abstraction point. By this means WRc would be able to evaluate the progress of the treatment process.

SYSTEM OPERATION

The system configuration (an adaptation of the example shown in Figure 2) was installed and commissioned in February 2002. The pump control kiosk draws approximately 10% of the abstracted groundwater to an above ground mixing tank to which a controlled quantity of glucose is added using an automated dosing system. The mix tank is continually agitated to ensure that full mixing of the glucose is achieved. The prepared glucose / water mix is then automatically injected into the three boreholes at the chosen flow rate via dedicated pipes which descend below the water table within each borehole. At all times during the operation a minimal volume of background groundwater is also injected into the same boreholes to encourage mixing and facilitate movement of the injected glucose mix away from the borehole and into the aquifer.

During the course of the trials system variables, such as injection concentration, injection frequency and duration, and abstraction rate will be altered to determine the degree of influence each has on the optimization of the denitrification process. The injection regime was initiated on 4th March 2002 based on the most efficient criteria determined from the laboratory experiments, and will be adjusted in an iterative fashion based on the results achieved.

The key parameters to indicate change within the system are as follows:
- Glucose: high levels at observation boreholes will indicate that sufficient stimulation of microbial activity has not occurred to date.
- Plate Counts: Increased values will indicate stimulated microbial activity.
- Nitrate / nitrite: Changes will indicate denitrification activity.
- Total Organic Carbon: Monitoring substrate consumption and microbial activity.
- Turbidity: Increases could indicate increased biological activity.
- Dissolved Oxygen: Decreased levels will indicate microbial activity stimulated and increased likelihood of denitrification.

At the time of writing the trial is entering only the fourth week of operation, so as yet there are insufficient data available to enable significant trend analysis and interpretation. However, with early indications of denitrifying activity already present (declining nitrate, increased bacterial counts, increased nitrite and declining dissolved oxygen) the outlook for future weeks is extremely encouraging.

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10.
Problems with Iron and Manganese
Jer Keohane, Geotechnical and Environmental Services, Carlow.

PROBLEMS WITH IRON AND MANGANESE

Jer Keohane, Geotechnical and Environmental Services, Ground and Water Environment Consultants

ABSTRACT

Iron and Manganese are not generally harmful to health, but above certain concentrations can be an aesthetic problem or can impair the operation of a water supply system. The source of the iron and manganese can be direct from the groundwater or appear indirectly as a result of certain circumstances. It is important to understand the problem before attempting to manage or treat it and care must be exercised in the application and interpretation of results of analyses since the actual numerical result is not as important as with other parameters. The easiest way to avoid the problem is to site a borehole in a geological strata that is not problematic, but this may not always be possible. At low concentrations the problem can be managed by employing some basic approaches. There are a number of treatment options available

1.   INTRODUCTION

Iron and Manganese in a groundwater supply can become a water quality problem for reasons as diverse as aesthetics and operational difficulties.

It is a common problem, but a problem that can be complex and often poorly understood.

As with any problem, it is first important to attempt to assess and understand the problem and then look at ways to manage or treat it.

This paper aims to discuss some of these issues.

2.   BACKGROUND CHEMISTRY

IRON

Iron (Fe) comprises approximately 5% of the earths crust mainly as iron compounds. Iron can occur in stable or unstable valencies as Ferrous (Fe $^{2+}$) or Ferric (Fe $^{3+}$) compounds.

Iron typically occurs in the form of oxides, hydroxides, carbonates and sulphides.

The form and solubility of iron in natural water are strongly dependent upon the pH and the oxidation-reduction potential of the water.

MANGANESE

Manganese (Mn) occurs naturally in the earths crust, but is much less abundant than iron.

Manganese can occur as in the divalent (Mn^{2+}) and the quadrivalent (Mn^{4+}) forms.

Solutions of manganese compounds are more stable and therefore more difficult to treat than ferrous solutions.

3. HOW does Iron or Manganese become a problem ?

In the ground, after time the water reaches equilibrium with the surrounding environment, and iron and manganese stay in solution.

When we drill and we encounter groundwater, and then pump the water to the surface, we change the conditions, and the iron and manganese can precipitate.

The general rule of thumb is that oxygenated water will have only low concentrations of iron and manganese. The reason is that both iron and manganese react with oxygen to form compounds that do not stay dissolved in water, and this can happen within the household plumbing or within a distribution network, creating the perceived problem.

Oxygen therefore acts both in creating the problem or as the key to treating the problem.

4. WHAT are the effects ?

Elevated iron and manganese concentrations, in general have no harmful effects on human health.

The effects caused by elevated concentrations are generally aesthetic in nature (not understating the problem to the consumer), or can impair the operation of the water supply system.

Water with high iron concentrations can impart a reddish brown colour on laundry or sanitary fittings.

Manganese stains laundry, sanitary fittings and can form a black coating in pipes.

High concentrations of both elements may impart a bitter taste to the water.

Iron can also enhance undesirable bacterial effects in water distribution systems. Most iron bacteria get their energy from the oxidation of ferrous iron into ferric iron. Iron is either obtained from the pipe itself or from the water inside the pipe. The process causes deposits in pipes and slimy coatings. The deposits cause a reduction in the carrying capacity of the pipework and this in turn can exacerbate the problem. The organisms can impart a bad colour, taste and odour to the water.

Once the effects are noticed it is generally recommended that elevated concentrations are reduced. However the onset of effects does not occur at a specific concentration.

For instance the relative direct usage of water in a household may not create the conditions whereby the iron or manganese comes out of solution. Whereas the same concentration in a group or public scheme, with a longer residence time in the pipe network, may mean that the iron and manganese comes out of solution, thus creating a problem.

5. WHERE does the iron come from ?

The appearance of iron in a water supply can have a number of origins.

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In some cases the problem may only become apparent some years into the life of the borehole. Some of the possible causes are:

Corrosion

Corrosion of the rising main can be caused as a result of a galvanic cell forming between two metals in the rising main and pump, leading to enhanced corrosion of (in particular) lower quality steel.

Low pH and Redox potential (Eh) can provide the potential for direct solution of steel.

Reduction in borehole yield

If the saturated thickness of the aquifer is reduced for some reason (for instance a dry summer), the borehole yield may reduce, the well can then be over-stressed, causing greater drawdowns and introducing oxygen to deeper levels in the borehole.

Iron biofouling (iron slime caused by iron bacteria) or iron precipitation or a combination of both may then occur in and around the slotted liner causing reduction in well efficiency, and instigating a vicious circle that can lead to abandonment of the borehole within a few years of the problem first appearing

Biofouling

The appearance of biofouling deposits is highly variable and it is this that has helped to perpetuate the belief that such deposits are chemical rather than biochemical or microbial in origin. Such deposits found coating and clogging well screens, pumps rising mains and distribution systems range in nature from hard brittle powdery sludge, to soft and slimy and in colour from white buff to orange red brown, olive brown to black.

Iron precipitation and biofouling in rock fractures may occur due to the introduction of oxygen during pumping.

Again the problem may become worse with time.

Groundwater Pollution

The introduction of organic matter into the ground can create conditions whereby iron and manganese are mobilised. The solubilised ions can then migrate through the aquifer causing the groundwater to become enriched. This can often create conditions whereby iron and manganese then become a problem, possibly in a source that previously did not have a problem.

In this case it is worth looking at the full chemistry to identify other problem parameters, such as ammonia and chloride.

In other cases Iron and Manganese may be a problem from the start, and may have its origins in the following:

The strata

If the well is constructed in rock types or strata in which the groundwater has the potential to have high iron or manganese concentrations, then the problem will become apparent quite soon after drilling.

Wells that are inefficient and constructed in Strata as above

Precipitation of iron at and above the water table may occur during pumping and this iron is then available at the start of the next pumping. This can occur when the well efficiency is low either (i) where the slotted liner does not have a sufficient open area in the vicinity of the water bearing rock fractures to maintain low flow velocities thus causing large drawdowns due to well loss or (ii) alternatively in open holes where flow through fissures is turbulent and drawdown is large.

Bogs

If the well receives recharge from a bog, iron may often be elevated in a well, even if the well is not in or adjacent to the bog. The chemical conditions in the bog may create conditions whereby iron appears in the water.

Mining

Groundwater emerging from or in contact with old adits or mineworkings can be enriched and this water may be providing recharge to the borehole.

Artifical Recharge

Problems have been experienced when water has been recharged through one strata for storage in an upper aquifer. For instance problems arose in London, when the Greensand was used as a medium to artificially recharge the Chalk aquifer.

6. QUANTIFYING the problem

The need to test for iron and manganese in the water is not as critical as it is for other types of contaminants that can cause potential health problems.

Iron and Manganese are not a problem in household water until they become detectable by the senses. Consequently, elaborate laboratory analyses are not required to determine if iron or manganese are a problem.

Laboratory analyses for iron and manganese are therefore generally only needed to quantify the problem and to provide information for the development of a management or treatment strategy.

For this reason, it is important when taking a sample to record the status of the water at the wellhead when the sample is taken, because the water may look very different when it arrives at the laboratory.

Exposure of the sample to air will cause precipitation of iron and manganese. To get an estimate of the amount of iron and manganese originally dissolved in the well water, precipitation must be prevented or the precipitated material must be redissolved. Before sampling for iron and manganese a certified laboratory should be consulted. They will recommend a sampling procedure that will provide an accurate estimate of dissolved iron and manganese in the source water.

The information recorded at the wellhead may be critical in the selection of an appropriate treatment system.

Samples taken immediately after drilling may not be representative of the long term chemistry, because during drilling air is blown into the hole, there is a turbulent environment, and peculiar conditions may arise.

Making a decision on the basis of one singular sample may be problematic. The problem may be intermittent, may occur for instance only after heavy rain, or out of the blue for no specific reason.

Taking more than one sample is recommended before a treatment or management strategy is decided upon.

For normal water supply purposes the results obtained by colourimetric determination can be sufficient. The concentration is directly related to the colour developed when certain reagents are added to the sample.

The most usual method used by contract laboratories is based on atomic absorption.

7.   HOW much is too much ?

To answer this question we can simply consult the Drinking water guidelines.

0.20 mg/l for Iron
0.05 mg/l for Manganese

However using these as the universal standard can be problematic.

For instance iron at a particular concentration in a domestic supply, may not present any problems, whereas the same concentration in a group scheme or public supply may create an operational headache. T he reason may simply be related to the longer residence time for oxygenated water in the distribution system.

There may be an argument that the standard should not apply to households since iron and manganese are not health problems and only become problems, when the effects are noted, and the effects may not necessarily become apparent at or above the guide level.

In fact Iron and Manganese may exceed the guide levels, and treatment may be recommended, but other parameters may be just below their threshold level, and present much more of a risk, or be more indicative of poor quality water.

Furthermore it is not uncommon for consumers to reject water with elevated concentrations of these metals for mainly aesthetic reasons in favour of less safe water that has a more acceptable taste.

8. AVOIDING the problem.

The problem is not confined to any particular geographical area. It is probably safe to state that the problem can be found somewhere in every county in Ireland.

Reference to the bedrock Geology is probably the best indicator of where the problem is likely to occur, although all that is needed is a single layer within an otherwise "non-problematic formation" to create the problem.

In general the cleaner Limestones do not present the problem, except where the recharge area may be a source of iron or manganese.

Shales, mudstones, many sandstones, muddy limestones, and metamorphic rocks and derived from these, are the rock types generally associated with the problem. Similarly subsoils derived from all of these rocks can be problematic.

On the stratigraphic column the problem can range from some Precambrian rocks, through Ordovician Volcanics, Devonian Sandstones and Shales, into Carboniferous Limestones, Upper Carboniferous shales, mudstones, sandstones. Permo-Triassic rock into Quaternary Sands and Gravels particularly those derived from rocks that are problematic.

One of the most consistently common Formations presenting the problem is the Calp Limestone.

Outside of the stratigraphic column, sources close to bogs also appear with high frequency on the problem list.

In the simplest of situations it may be possible to avoid problematic rock types by locating a source in a less problematic rock type. There is no definitive map of the country showing areas to be avoided, although a number of aquifers where the problem was identified is part of a study undertaken by the Geological Survey of Ireland in a review of groundwater vulnerability and quality undertaken in 1983.

In other cases local knowledge provided by drillers, or members of the public may be of assistance in avoiding the problem.

In the worst case, with such a widespread geographical distribution there may not be a choice to avoid areas where iron and manganese are endemic.

9. UNDERSTANDING the problem

Once the problem is identified and quantified, it is important to attempt to identify the source of the problem.

In the domestic case, the well is generally drilled and paid for before the problem is identified, and this can cause logistical difficulties in the installation of a treatment system.

It possibly would be a good idea if householders were alerted or prompted to potential problems, by their engineer, well driller, or local authority, at an early stage in the development of a house either at the planning stage or when the well is drilled.

In the case of group or public supplies, a hydrogeologist may be involved and they may carry out an investigation involving trial drilling. Some of the key elements of this investigation may comprise :

- A good well log to identify potential source horizons of iron or manganese. In some cases these could then be cased out, when the production well is designed

- Representative water samples to quantify and provide information on the variability of the problem. This should include a full suite of parameters to assess other issues such as aggressiveness, contamination from other sources.

- Pumping tests, particularly, step tests enable the well efficiency to be calculated. Repeating these tests at a later stage can help to identify the onset of the problem.

- A conceptual hydrogeological model to assess where the recharge is coming from and to assess the variability in this recharge. Some hydrogeological conditions such as confined aquifers and reducing conditions in the aquifer may contribute to the problem and need to be identified.

- Development of a source protection plan to assess the groundwater vulnerability and to develop protection protocol.

10. MANAGING the problem

The following are some general recommendations that may assist in managing the problem.

- Disinfection of the well rig before drilling may prevent the introduction of iron bacteria ✳ from another source.

- Disinfection of the well after drilling and at regular intervals thereafter may limit the growth of iron bacteria.

- Use of corrosion resistant casing may limit the appearance of iron, with aggressive ✳ waters.

- Installation of a system for the recording of water levels during pumping to allow measurement of general variations in water level and an assessment of changes in the efficiency of the well.

✳ = measures suitable for domestic well operators.

- Regular measurement of dissolved iron and manganese, pH, Eh, temperature and electrical conductivity at the site can provide information on changes in the characteristics of the water.

- Implementation of a good pumping regime whereby the wells are operated for longer periods or continuously at an appropriate lower pumping rate.

- Where possible, the pumping regime should ensure that pumped water levels do not fall below the top of the screen.

- The pump should not be installed in the slotted liner/screen section, to limit turbulence and well velocities.

- If iron biofouling and/or precipitation have reduced the well yield, the following options can be considered.

  Air surging and airlift pumping
  Chlorination
  Acidisation
  Drilling a new well

- Blending water from a surface water source or source with lower concentrations may be an option for Public Supplies.

9.   HOW can the problem be treated

Because iron and manganese occur in various chemical forms, it is impossible to offer a universal solution to the problem.

The most common approach from hydrogeologists or engineers is to contact a firm who specialise in the supply of treatment systems. There are firms who will set about analysing the problem and tailoring a solution or there are firms who will provide an off-the-shelf system.

Nowadays there is a trend from consumers to request treatment technologies that do not involve the addition of chemicals.

The most widely used treatment technologies involve some form of media. At low concentrations a media filter on its own may provide the solution. In more severe situations there may need to be some form of pre-treatment to condition the water before it passes onto the filter.

Upto recently the most common filter media was a compound known as BIRM. Use of this material in certain conditions, such as low pH and in the presence of H_2S is problematic.

FILOX ™ is a new media manufactured in America, that is an alternative to the previously preferred BIRM ™. FILOX needs no regenerative solution, and has a lifespan of 4-7 years. The media is effective at Iron and Manganese removal. The media is heavy however and needs strong backwashing to clean it. It is therefore important to install a pump that can deliver the necessary capacity to effectively backwash the filter.

Another system called Crystal Right ™, which works on an ion-exchange basis, requires the use of a salt such as sodium chloride or potassium chloride.

8

In some cases such as very high concentrations, particular pH conditions, it may be necessary to condition the water, before filtering, by pH correction or aeration using a cascade or ozone.

More sophisticated treatment may be required for public supplies or for industrial applications, where the treated water has to be to a high specification.

It has been shown that a big problem with treatment is not the lack of technical quality in the design, but rather shortcomings in operation and maintenance. It is important even in the domestic situation that some basic training on the operation and maintenance of the filter is provided.

Costings for a domestic situation may vary from € 1000 for a basic filter not requiring a regenerant, to € 1400 for a more sophisticated system.

If pre-treatment is required such as Ph correction, or aeration you could add an extra €800.

In other cases alternative treatment approaches have been used.

Constructed wetlands to treat iron rich groundwater from mine

Biological systems, using bacteria isolated from an infected borehole, may be placed in a bio-reactor to interact with the water, and remove iron and manganese. This approach is still in the early stages of development, but offers good potential in the future.

Re-injection of oxygenated water may be used to condition groundwater prior to treatment.

Acknowledgements

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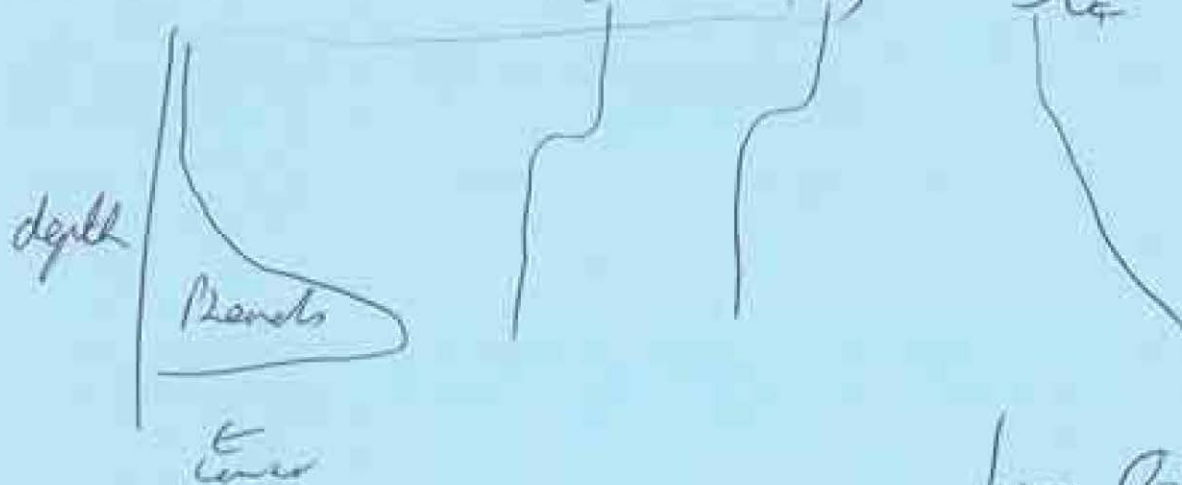*KEYNOTE:* **The Use of Monitored Natural Attenuation as a Cost-effective Technique for Groundwater Restoration**

David Lerner, Steven Thornton, Ruth Davison, Groundwater Protection and Restoration Group, University of Sheffield, UK.

Methods

3 pieces evidence needed

1) evidence mass reduction in field
2) Geochem evidence biodeg (eg $O_2 \downarrow$ point ($\bar{F} \uparrow$ daughter))
3) Microcosm evidence from lab in field.

Case study — higher cores planet demonstrate
because had conductor, higher in the past

O_2 NO_3 SO_4

depth

Phenols

t
time

and Geochem evidence from $O_2 + NO_3$
⇒ electron acceptors drop at plume
⇒ active degradation. SO_4 not so clear
Isotope studies showed there had been
partial SO_4 reduction.

Acetic acid very high cores. Unusual —
fermentation by-product

The use of monitored natural attenuation as a cost-effective technique for groundwater restoration

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ABSTRACT: There are many groundwater pollution problems caused by the spillage of fuels and industrial chemicals throughout the world. There has often seemed to be between strategies of ignoring the problem or going for a full scale and expensive cleanup. These extremes are now seen as simplistic, and there has been a growing acceptance of a risk-based approach to evaluating individual sites and designing appropriate strategies. This allows account to be taken of the natural processes which reduce concentrations along the pathway from source to receptor, particularly biodegradation. Taking formal account of these processes can often reduce or eliminate an engineered restoration, replacing it with a requirement that the plume is monitored in future. Protocols are available to assist with the assessment of such natural attenuation. They are especially useful for straightforward hydrocarbon plumes. Studies of two more complex cases, involving phenolic contaminants, are presented to show that natural attenuation can still as assessed on sites which do not fit within the protocols. In one case, natural attenuation has been successful. In the other, biodegradation is so slow that an alternative risk management strategy will be required.

1 GROUNDWATER POLLUTION PROBLEMS

Contaminated land and groundwater remain a large liability on the balance sheets of many companies, local authorities and nations. Groundwater pollution is particularly difficult to cleanup, and researchers have often argued that no severely polluted site has been successfully restored by engineered intervention (National Research Council, 1994). The underlying problem is that of inaccessibility of the pollutants. Heterogeneity causing greatly variable flow rates, diffusion of pollutants into low permeability materials and long term sorption, all mean that the time scales for cleanup are controlled by diffusion, a very slow process. This has been recognised in the USA, where it is now possible to obtain a "Technical Infeasibility Waiver" for some sites. Pump-and-treat, which is the standard approach when active restoration is undertaken, is best seen as a containment approach.

Until now, most "owners" of polluted groundwater in the UK have only dealt with the problems when they wish to sell land, or they have had such a serious incident that the Environment Agency becomes involved. This is going to change. Recent and imminent legislation will create a new contaminated land regime. This will require every contaminated site to be identified and evaluated for the risk it may cause to controlled waters. The Environment Agency is now empowered to require cleanup where groundwater is polluted but not linked to any contaminated land. The new IPPC legislation will require landholders to put baseline surveys of the environmental condition of their sites on the public register. Then there will be public information about which sites are polluted and powers to require cleanup – ignorance and inaction will no longer be options.

There is a potential conflict between the increasing pressure on landholders to act on groundwater pollution, and the acknowledged difficulties of engineered cleanup. Natural attenuation is a potential strategy to resolve this conflict for many sites – if natural processes can be demonstrated to be reducing risks to acceptable levels, then active cleanup will be unnecessary.

2 NATURAL ATTENUATION CONCEPTS

Natural attenuation of polluted groundwater refers to the suite of "....physical, chemical or biological processes that act without human intervention to reduce the mass, toxicity, mobility, volume, or concentration of contaminants in soil and groundwater......" (U.S. EPA, 1998). These in-situ processes include biodegradation, dispersion, dilution, sorption, volatilisation, and chemical or biological stabilisation,

transformation, or destruction (U.S. EPA, 1999). It is also called *intrinsic (bio)remediation*, and the term *monitored natural attenuation* (MNA) is becoming standard as it emphasises that monitoring must be undertaken if MNA is to be accepted as a strategy for any site.
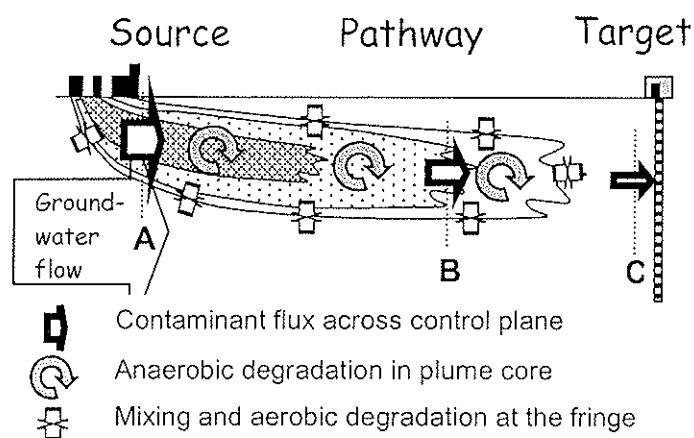
Source   Pathway   Target

Figure 1 Concept sketch for natural attenuation of biodegradable contaminants in groundwater, in which fluxes are reduced during flow from source to receptor.

The processes of natural attenuation are shown schematically in Figure 1. With time, polluted groundwater moves from the source at A, through intermediate points in the plume (B), to the receptor at C. MNA will be successful if the flux and concentration of pollutants is reduced enough to make the risks to the receptor acceptably low. For organic pollutants, biodegradation is usually the most important process. Anaerobic degradation will occur inside the plume, but is slow and limited for many situations. Usually aerobic and nitrate reducing degradation are the most important reactions, but they can only take place at the fringe where background waters mix slowly into the plume. The plume will grow until the influx of pollutants at A is balanced by the rate of destruction of the combined fringe mixing-controlled and the slower internal reactions.

3   WHEN IS MNA APPROPRIATE?

The use of MNA has increased substantially over the last decade with, for example this approach implemented in 23% of US cases in 1997 compared with only 5% in 1991 (Kremer, 1998a). MNA was the preferred choice for the remediation of 46% of petroleum hydrocarbon-contaminated sites, and was also widely used for inorganic contaminants and chlorinated solvents (35%). The exploitation of MNA as a remediation technology is supported by the U.S. EPA "within the context of a carefully controlled and monitored site cleanup approach that will reduce contaminant concentrations to levels that are

protective to human health and the environment within a reasonable time frame" (U.S. EPA, 1999). It requires sound technical justification within a site-specific decision process, that allows for contingency measures in the event that its predicted performance fails to meet regulatory cleanup criteria (Wilson, 1998). This implies a sophisticated site assessment.

MNA has many potential advantages, but also some possible limitations, when compared with conventional engineered approaches. It is not for those in a hurry, or for landholders who want to install equipment for PR reasons. Nor is it for all compounds. For example, chlorinated solvents only degrade anaerobically and incomplete degradation produces vinyl chloride which is more toxic than the parent compounds.

MNA is a viable option for compounds which do attenuate, and for sites with enough travel distance and time to allow the processes to operate. If these hurdles are passed, non-technical factors are considered, such as the institutional controls over long periods, the availability of adequate funding for reliable monitoring and performance evaluation, and public acceptance of the extended time scale required for remediation.

4   HOW TO DEMONSTRATE MNA

Natural attenuation is evaluated using multiple lines of evidence (Wiedemeier, et al., 1999). These distinct but converging lines of evidence are based on hydrochemical and microbiological data that document the occurrence and extent of in-situ contaminant removal processes. The nature of the information required to demonstrate MNA of organic contaminants includes:

– *Primary lines of evidence* such as historical trends in contaminant data showing plume stabilisation and/or contaminant mass loss over time;

– *Secondary lines of evidence* showing that *in-situ* hydrogeochemical conditions are suitable for biodegradation and that active biodegradation has occurred. This is based on geochemical indicators of naturally occurring contaminant biodegradation, such as the depletion of electron acceptors and donors, increased metabolic by-product concentrations, decreasing parent compound concentrations and increased daughter compound concentrations;

– *Tertiary or optional lines of evidence* that prove the processes and estimate rates.

Primary and secondary lines of evidence are obtained by sampling groundwater monitoring boreholes along the plume flowpath, including the source area, and uncontaminated zones of the aquifer. They are used to show that a plume is shrinking, stable or only growing at a rate slower than that predicted by

conservative groundwater flow calculations. They also demonstrate that this behaviour is consistent with the hydrochemical environment. Tertiary lines of evidence include data from microcosm studies and are used to support an MNA assessment, when the other evidence is inconclusive, or when information is required on a specific degradation mechanism or on environmental factors that may limit biodegradation processes.

5 DEVELOPMENT OF PROTOCOLS FOR MNA

The emergence of MNA as a remediation technology has resulted in the development (which is ongoing) of technical protocols for its application. These are guidance documents which present good practice in site investigation and analysis of data required to implement and monitor MNA. The development of these protocols has been driven by American experiences, producing guidance for petroleum hydrocarbons (Wiedemeier, et al., 1995) and chlorinated organics (Wiedemeier, et al., 1997). Protocols have also been developed by the American Society for Testing and Materials (ASTM, 1998), and private sector companies in response to specific contaminant problems (e.g. Buscheck and O'Reilly, 1995). Significantly, there is little technical guidance for the application of MNA in many European countries, although this is presently under consideration in Denmark and in the Netherlands. A guidance document, commissioned by the Environment Agency, for the assessment of MNA in the UK is currently under development (Carey *et al.*, 2000).

The American protocols have proved very useful. They have introduced a common language and philosophy to MNA assessments. They have made studies more cost effective and enabled MNA to be widely accepted, reducing the costs of contaminated groundwater to society. However other countries are not the same as the USA. They have different legal and administrative arrangements, and different pressures on national resources. Often their hydrogeology is different. National protocols can be quickly developed from existing documents to take account of these factors. We strongly recommend that any protocol includes a requirement for peer review. More than anything, this will raise standards of investigation and interpretation, and share experience amongst the community. Without such review and transparency there is a serious risk that the public will see MNA as a do-nothing, money saving option which is justified in a secret report. MNA is more than this, it is a valuable addition to a repertoire of solutions to the massive problems of contaminated land and groundwater.

6 SITE INVESTIGATION AND MONITORING STRATEGIES

The site investigation programme needed to evaluate the performance of NA technology incorporates initial site characterisation and long-term monitoring (Kremer, 1998b). Site characterisation for NA is fundamentally different and more comprehensive and than that required for an active remediation scheme, because it requires greater understanding of processes affecting the contaminant plume and there is greater emphasis on data collected from within the plume. This initial monitoring phase should identify the location and extent of contaminant source area(s), the spatial distribution and concentration of contaminants, heterogeneity in the aquifer geological and hydrogeological characteristics and variations in the groundwater hydrochemistry. The objective of the initial site characterisation for NA assessment is to define the baseline conditions. This is necessary to ascertain whether NA is likely to be a viable remediation option and, if so, to provide a reference state from which its performance can be monitored over time. A typical network of observation boreholes used to evaluate NA of contaminant plumes is shown in Figure 2. At a *minimum* the borehole network needs to include wells that assess unimpacted background groundwater quality upstream of the plume (well A), the contaminant composition in the source area (well B) and groundwater quality along the plume flow path (wells C to G). To correctly delineate the plume, boreholes should also
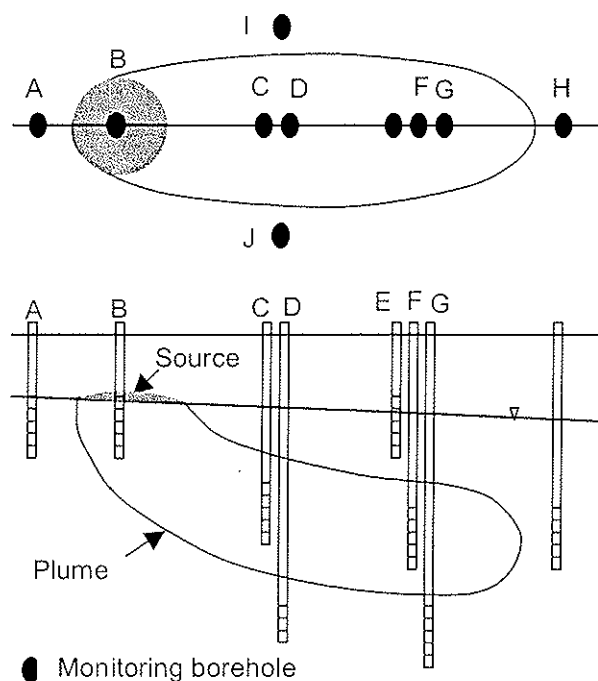

Figure 2 Network of observation boreholes used for assessment of natural attenuation (modified from Wiedemeier and Haas, 1999).

be positioned in the flow path ahead of the plume to define the downgradient extent of contamination (well H) and transverse to the plume to define the lateral extent of contaminant migration (wells I and J). The monitoring borehole network (e.g. wells A to J) is usually installed in phases since the extent of plume migration is unknown until these wells have been drilled.

Long-term monitoring (LTM) is required to assess the behaviour of the contaminant plume over time and to confirm that NA is occurring at rates which are protective of downgradient receptors. Additional wells are likely to be needed, their number, location, screened intervals and frequency of sampling being dependent on contaminant distributions, site stratigraphy, plume velocity, groundwater geochemistry and travel time to sensitive receptors (Barcelona, 1994; Wiedemeier and Haas, 1999).

7 CASE STUDIES

Of course real life is often not as simple as the conceptual models which are used to explain ideas, or as the protocols would sometimes have us believe. Two case studies below illustrate how natural attenuation can be assessed outside the protocols. In the first case it is probable that the risks have been reduced sufficiently, while the other shows such slow degradation that MNA is a not sufficient management strategy.

7.1 Rexco plant, Mansfield, UK

The site produced smokeless fuel, by the Rexco process, between 1935 and 1970 using coal from the on-site colliery. The site is on the Permo-Triassic Sherwood Sandstone, the second most important aquifer in the UK. There is no superficial protective cover to the aquifer, the water table is some 16-20 m below ground at the site, and groundwater velocity is about 60 m y^{-1}. Both ammonium liquor (quench water) and coal tar have contaminated the unsaturated zone but only the ammonium liquor has affected the groundwater. Ammonium liquor from a modern plant using a similar process was analysed and found to contain high concentrations of ammonium (~12800 mg l^{-1}), phenol (~7700 mg l^{-1}), chloride and sulfate. Extensive fieldwork (multilevel boreholes creating 109 monitoring points at 39 locations; Jones et al., 1999) discovered a plume of ammonium (up to 400 mg l^{-1}) down gradient from Rexco but no organics were observed. A second plume was inferred, originating from a lagoon some 800 m east of the plant; this is the subject of continuing investigations (Davison and Lerner, this volume).

Phenol is expected to degrade in groundwater, although there are known to be inhibitory effects at high concentrations. An electron balance suggested that sufficient electron acceptors have mixed into the plume to degrade all the organics (Thornton et al.,
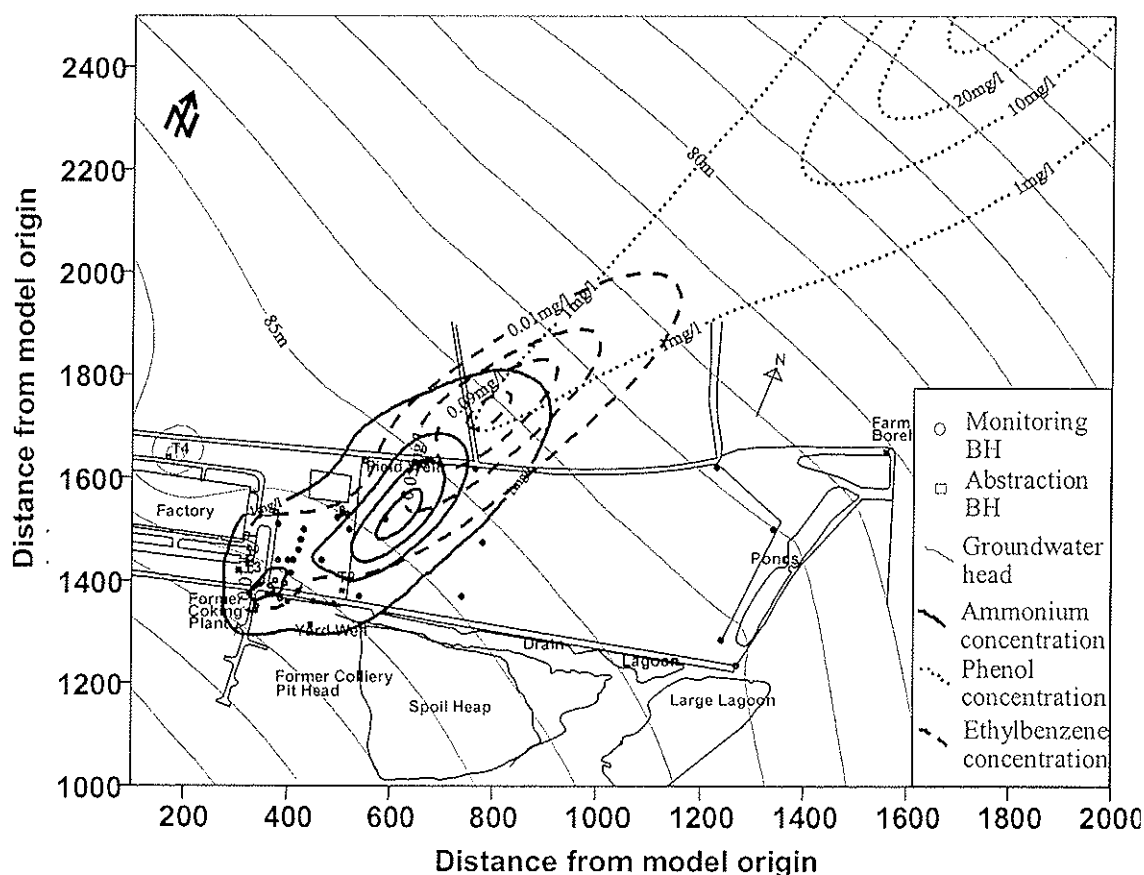
Figure 3: Simulated positions of phenol, ammonium and ethylbenzene plumes at Rexco in 1996, assuming no biodegradation.

1998). However there were no field data to confirm that complete degradation had occurred. The reactions would have been completed some time ago, allowing the geochemical and microbiological signatures in the aquifer to return to a more normal condition before the investigation took place. In addition, the still present ammonium undergoes similar oxidation reactions and so will confuse the evidence provided by field data. Another approach was required to evaluate whether NA had been successful for the phenol.

A groundwater flow and solute transport model was constructed of the site, and calibrated against the ammonium data (Davison and Lerner, in press). Figure 3 shows the model predictions for ammonium and phenol in the absence of biodegradation, and reveals that the phenol was likely to have moved off-site. It is probably beyond the locations of the observation boreholes, which had been drilled before the model was constructed. Thus the absence of phenol in the observation boreholes does **not** prove NA by itself, as it can be explained by the chromatographic separation of the pollutants during transport since the input stopped at the ground surface around 1970.

To differentiate between these explanations an organic compound that has a similar affinity to degrade as phenol and a similar retardation to ammonium was simulated. Ethylbenzene has such properties, and was known to be present in the source. When modelled with just advection, dispersion and sorption processes active, ethylbenzene was predicted to remain around the field site (Figure 3). In reality it is absent and so must have degraded, indicating that degradation is active at the site; NA has probably been successful at this site (but see Davison and Lerner, this volume).

7.2 4A tar distillery, Wolverhampton

The 4A plant has manufactured specialty organic chemicals since the site was opened in 1950. Originally it was a coal tar distillery, and it now uses feedstocks brought in from other chemical plants. As at Rexco, the plant overlies the important Permo-Triassic Sherwood Sandstone. Groundwater is about 4 m below ground, and the aquifer is 250 m thick in the vicinity of the site. Flow is towards the west, at about 10 m/y. In 1987, a pollution plume of tar acids and neutral aromatic compounds was found under the site. It extends about 500 m westward, and to a depth of 60 m when defined by the 1 mg l^{-1} phenol contour. The total concentration of organic compounds in the source area is currently 24800 mg l^{-1}. This includes about 12500 mg l^{-1} of phenol, substantial concentrations of cresols and xylenols, and lower concentrations of mono-aromatics and other compounds.

Consultants have provided two contrasting interpretations of the site data, one arguing that NA has been insignificant and that a substantial, engineered cleanup is required. This is estimated to cost £20 000 000. The other consultant has argued that substantial biodegradation has taken place and that no engineered clean up is needed. The difference between these views arises out of interpretation of the source term. As spillages have occurred over 50 years or so, and records are negligible, the amount spilt is almost impossible to estimate from site records.

Our approach was to investigate the plume in more detail with two multilevel samplers which provided sampling ports at 1 m intervals over 35 and 42 m deep boreholes (Lerner et al., in press). Anions (F, Br, Cl, NO$_2$, NO$_3$, SO$_4$, PO$_4$), cations (Ca, Mg, K, Na, Fe, Mn, Al, Cd, Ni, Zn, NH$_4$), Si, S^{2-} total P, total S, organics (phenol, o-cresol, m/p-cresol, 2,3-xylenol, 2,4/2,5-xylenol, 2,6-xylenol, 3,5-xylenol, 3,6-xylenol, 4-hydroxybenzoic acid (4-HBA) and 4-
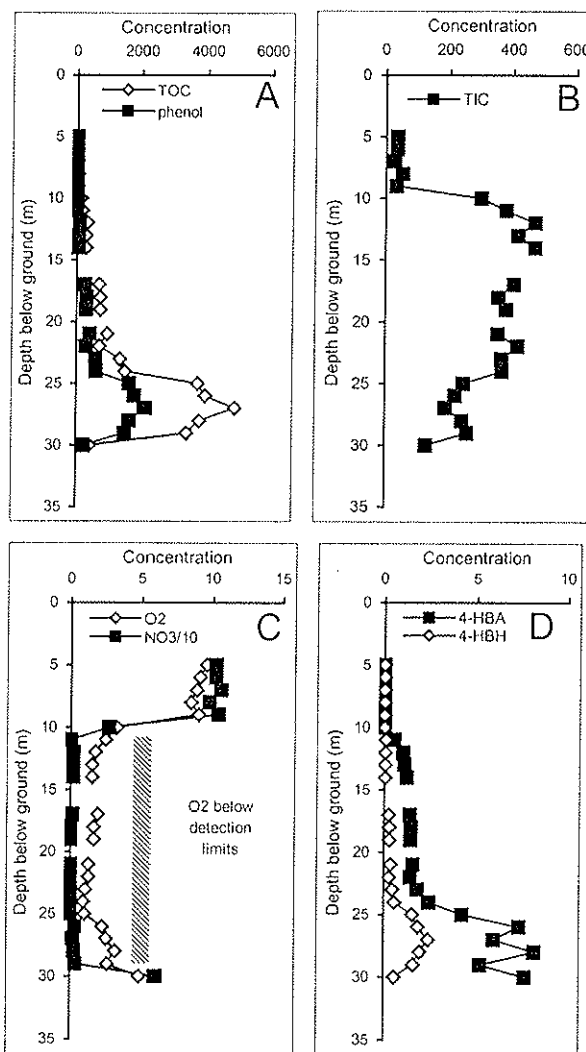

Figure 4. Selected vertical profiles from BH59 at 4A, showing (a) distribution of major phenolic pollutants, (b) concentrations of total inorganic carbon (TIC), (c) O2 and NO3 profiles, and (d) two metabolites of anaerobic biodegradation of phenol.

Table 1 Comparison of NA case studies at Rexco and 4A

Nature and name of site	Rexco - coal carbonisation	4A coal tar distillery
Source of pollution	Aqueous ammonium liquor containing ~ 13 g phenol l⁻¹.	Non-aqueous phenolics mixture, leading to current in situ concentrations of 13 g phenol l⁻¹.
	Total spillage ~ 400 t of phenol, also NH₄, Cl, SO₄ and other organics in source.	Total spillage ~ 1000 t of phenol, also a substantial load of other polar organics.
	Active at surface 1955-1970	Active at surface 1950-1990
Unsaturated zone	20 m	3 m
Groundwater velocity	60 m y⁻¹	10 m y⁻¹
Changes in flow direction	Major changes up to 90° due to changing local abstraction on several occasions	Minimal
Current status	All organics believed to be degraded.	<10% of organic load degraded
	Substantial NH₄ present in aquifer	
Investigation and assessment	Large field investigation inconclusive. Solute transport modelling required to assess behaviour	Initial large field investigation inconclusive. Detailed interpretation of multilevel profiles, using isotope and geochemistry required
Natural attenuation outcome	NA has eliminated risk	NA will not eliminate risk

hydroxybenzaldehyde (4-HBH)), TOC, TIC, sulphur isotope ratios and wellhead parameters were measured. A small subset of the results are shown in Figure 4 for one multilevel sampler, BH59 which is 150 m from the source area.

The principal electron acceptors (O_2, NO_3) in the uncontaminated groundwater decrease from background to undetectable in less than 2 m (Fig 4c), mirroring an equally rapid decrease in TOC from the plume to background. These observations indicate relatively rapid aerobic and NO_3-reducing activity in the plume fringe. There are three primary lines of chemical evidence for active anaerobic degradation in the plume. Firstly, two metabolites (4-HBA and 4-HBH) characteristic of anaerobic degradation of phenolic compounds are observed throughout the profile (Fig 4d). Total inorganic carbon (TIC) is also higher than background throughout the plume (Fig 4b). Dissolved Fe and Mn are present above background levels across the profile, as are small amounts of methane (0.02-12 mg l⁻¹) (data not shown).

Comparing TIC with TOC concentrations suggests that a maximum of 1% of the TOC has degraded in the 15 y travel time from source to BH59. At a plume scale, the residual TOC is about 2000 t. Both an electron balance and a carbon balance suggest that the residual carbon is about 95% of the original spillage. At both local (BH59) and plume scales, NA has been slow and will not reduce concentrations sufficiently to protect the receptors at risk (Thornton et al., 2000).

7.3 Comparison of the case studies

Both Rexco and 4A concern the same organic pollutants (phenolics) in the same aquifer (Sherwood Sandstone) and yet the outcomes are remarkably different. The similarities and differences are high-

lighted in Table 1. A number of factors came together at Rexco to help NA, such as the thickness of the unsaturated zone and greater mixing caused by higher and changing velocities. None of these factors were beneficial at 4A, with the consequence that NA has not been successful.

8 CONCLUSIONS

The severity of groundwater pollution problems is well known, at least to hydrogeologists. Similarly, the costs and difficulties of engineering clean-ups are well known, with the consequence that society cannot afford to take engineered action to restore many sites, and they have been left alone with little or no formal assessment. Rather than ignoring such sites, a more sophisticated approach is developing in which full account is taken of the natural processes which reduce pollutant fluxes and concentrations. This natural attenuation can sometimes reduce risks from the pollution sufficiently to allow a formal and well-argued decision that active restoration is unnecessary.

Assessing natural attenuation can be difficult, because of the complexity of the biogeochemical reactions involved. Protocols have been developed to help in such assessments, and they work well for straightforward cases, particularly of the common hydrocarbon fuel and chlorinated solvents spillages. Two more complex case studies have been used to show that NA can be assessed even when the protocols cannot be applied.

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12.

Case Studies in Monitored Natural Attenuation
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CASE STUDIES IN MONITORED NATURAL ATTENUATION

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ABSTRACT

Monitored natural attenuation of contaminated groundwater is increasingly being considered as a possible remedial solution, particularly since the alternative remedial options can be very expensive. In the UK, the increasing use of natural attenuation has resulted in the Environment Agency publishing a guidance document, which defines a staged process of investigations and analysis that needs to be followed before a natural attenuation strategy is considered viable. This paper presents three different case studies, focussing on natural attenuation as a remediation option. The first study shows how natural attenuation was demonstrated to be sufficient to protect the environment, while the second study illustrates a situation where natural attenuation is not feasible. The third study shows how a combined positive and passive remedial solution was accepted as the best practicable option.

1.0 INTRODUCTION

Natural attenuation of contaminants in groundwater is defined as the combination of naturally occurring physical, chemical and biological processes which reduces the mass and concentration of polluting substances in groundwater. Monitored natural attenuation (MNA) is the remediation term used to describe the groundwater monitoring regime required to confirm that the natural attenuation processes are sufficient to achieve the remedial objectives. In the UK, MNA is increasingly being considered as a possible remedial solution for contaminated groundwater. However MNA is only an applicable under certain favourable circumstances. Natural attenuation processes need to take place at a rate that protects the wider environment within an acceptable timeframe and without resulting in a significant increase in the volume of polluted groundwater, before achieving the remedial objectives.

Various guidance documents and procedures defining the assessment of the natural attenuation processes to assess whether MNA is a feasible remedial strategy have been produced in the US (AFCEE, 1995, ASTM, 1998 and USEPA, 2001). In the UK the Environment Agency has developed its own MNA guidance document to fit in with the UK's regulatory regime (Carey et al, 2000), which ensures that acceptance of MNA strategies are based on scientific evidence that MNA will achieve the remedial objectives and thus will protect the environment.

Over the last three years URS has worked on a number of remedial assessments throughout the UK, where a component of the studies has been to assess whether MNA is a feasible remediation strategy, using the Agency's guidance. This paper describes three of these studies, each presenting different aspects of MNA in relation to the Agency's guidance to demonstrate both the advantages and disadvantages of MNA.

2.0 SUMMARY OF THE UK REGULATORY GUIDANCE ON MNA

Before the Environment Agency produced guidance on MNA in 2000, approaches to demonstrate the effectiveness of natural processes in remediating groundwater varied widely. The Agency's method for assessing the appropriateness of MNA is based on a four-stage process in which different levels of data collection and analysis complexity are applied through each stage of the process to filter out circumstances in which MNA is not appropriate. The four stages are as follows:

- **Stage 1: Initial Screening**, where an initial assessment of the viability of the approach is conducted from a limited data set. This screening needs to take account of the technical reliability of the natural attenuation mechanisms, the timeframe of natural attenuation and the economics of natural attenuation over other remediation options.
- **Stage 2 Demonstration of Natural Attenuation**, where scientifically based evidence is developed to demonstrate the natural attenuating mechanisms: dilution, dispersion and chemical and biological degradation. Three lines of evidence are available:
  - **primary** lines of evidence, where historical hydrochemical data are used to demonstrate a declining trend in contaminant concentrations and pollutant mass
  - **secondary** lines of evidence where changes in chemical data (contaminant and electron acceptors) are used to demonstrate a loss in contaminant mass,
  - **tertiary** lines of evidence, where laboratory microbiological testing is used to demonstrate that the indigenous microbiota is capable of degrading the contaminants.
- **Stage 3 Assessment of Natural Attenuation,** to establish whether natural attenuation is an appropriate long term mitigation measure through modelling and/or analytical predictions from existing field evidence.
- **Stage 4 Implementation of MNA**, through the development of a monitoring and data review plan, used to demonstrate that the remedial objectives are being achieved according to the predictions in Stage 3.

Within the overall decision framework, there are a number of important factors that need to be considered before the process of demonstrating that MNA will achieve the overall remedial objectives. First and foremost, MNA is only acceptable for remediating residual contamination. That is, the source of the contamination must be removed or disconnected from the groundwater. Secondly, it may be necessary to define a contingency plan which could be implemented should the attenuation processes not take place as predicted. Thirdly, MNA is not considered an acceptable remedial strategy if the natural attenuation processes do not reduce contaminant concentrations everywhere in the aquifer within 30 years (i.e. a single generation). Finally, although the overall costs of an MNA strategy are likely to be lower than active remedial solutions, the initial costs associated with demonstrating and assessing the natural attenuation processes could be significant.

3.0   MNA CASE STUDIES

3.1   General

It should be standard practice to consider MNA at an early stage in the remedial review process where the contaminant concentrations that exist in groundwater are proven to be a significant risk according to contaminated land and water legislation. MNA may be the primary component of the remediation strategy, although more frequently, MNA is carried out following the implementation of an active remedial scheme. In such circumstances the overall cost of the remediation scheme may be significantly reduced by the requirement to achieve higher "risk based clean up levels" by the active scheme. These are then subject to further reductions through the natural attenuation process.

TABLE 1: DETAILS OF MNA SITES				
Site	Aquifer	Contaminants of Concern	Status of the Source of Contamination	Current status of MNA strategy
Former Oil Storage Depot, Oxfordshire	Whitchurch Sand	Benzene and Toluene	Contaminated soils removed	Demonstrated, assessed and currently being implemented
Helpston Landfills, Cambridgeshire	Jurassic Limestone	Mecoprop (pesticide)	To be contained at the landfills	Screening & demonstration. MNA insufficient due to potential time constraints
Chemical Works, Staffordshire	Triassic Sandstone	Phenols, creosols	Industry controls in place to minimise spillages/leakages of chemicals	Demonstrated and assessed – only applicable for low level concentrations at edge of the plume

The following case studies outline how the Agency guidance has been adopted by URS at three sites, to demonstrate different components of MNA, the impact of MNA on the overall remedial costs and to illustrate the key factors that have been used to establish whether MNA is suitable.

Details of the three sites are shown in Table 1.

3.2 Former Oil Storage Depot. Oxfordshire

A quantity of benzole, a blend of benzene and toluene, was accidentally released from an oil storage depot in the 1960's. Site investigations in the 1990's indicated that it was present in soil and groundwater. The geology comprises around 1m of Gault Clay over approximately 18m of Whitchurch Sand. In the eastern half of the site, the Gault Clay is absent. Groundwater is present at a depth of between 2.75m and 3.5m below ground, and flows in a south westerly direction, as shown in Figure 1. Various groundwater investigation boreholes were installed to delineate the horizontal extent and concentration variation in the plume. The maximum concentration of benzene was found to be 74 mg/l.
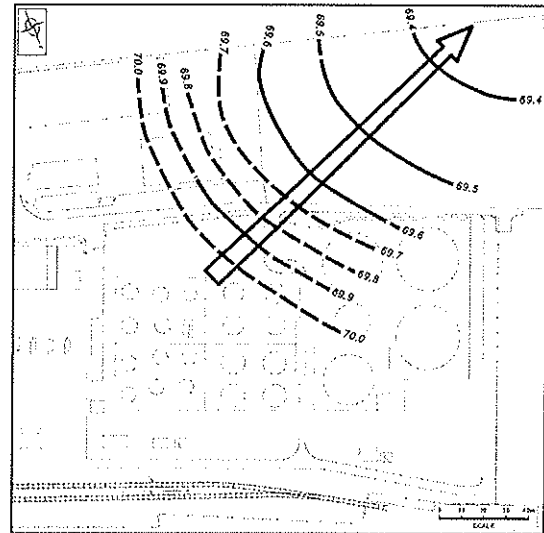

Figure 1 Groundwater Flow

Once it had been established that an MNA approach was feasible (i.e. Stage 1 screening), groundwater monitoring continued at six-monthly intervals, in which contaminant concentrations, electron acceptor concentrations and microbe bacterial counts were measured. These data were used to demonstrate and assess that MNA was a feasible remedial strategy for the contaminated groundwater.

Primary Lines of Evidence
Based on the dimensions and behaviour of the plume, it was shown that natural attenuation of the plume was occurring (Figure 2). In particular, the down-gradient edge of the plume did not migrate any distance between 1996 and 1999, suggesting a plume being actively degraded.
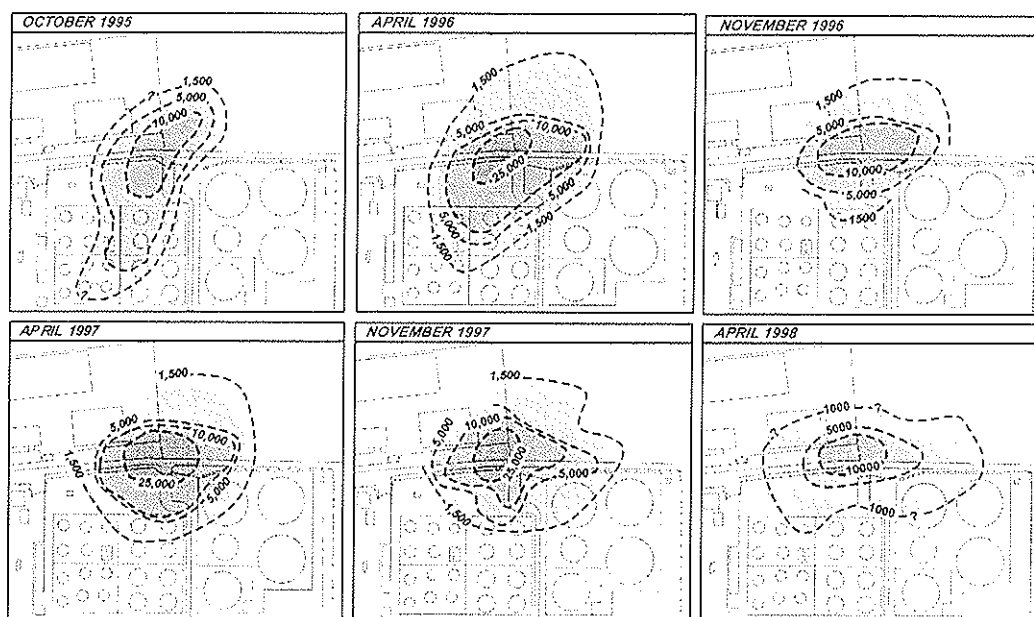

Figure 2 Attenuation of Benzene 1996 to 1998

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Secondary Lines of Evidence

Analysis of the electron acceptor concentrations (oxygen, nitrate, sulphate, manganese and iron), demonstrated that sulphate was the most significant electron acceptor for benzene oxidation. The concentration of sulphate in groundwater up-gradient of the plume was measured at around 220 mg/l and at 7mg/l within the area of maximum benzene contamination. That is a negative correlation between sulphate and benzene exists unless we actually calculated correlative coefficient should leave out. A similar relationship was observed for iron which was manifested by an increase in soluble ferrous ion. However the distribution of reduced iron so formed is complicated by sulphate reduction, as the hydrogen sulphide generated reacts with the iron in solution to yield iron sulphide precipitate.

Tertiary Lines of Evidence

Further evidence of the natural attenuation processes was provided through the use of immunoassay test kits to identify the presence of enzymatic activities of sulphate reducing bacteria (SRB). SRB activity is detected in 75% of groundwater samples, providing evidence of sulphate reducing conditions.
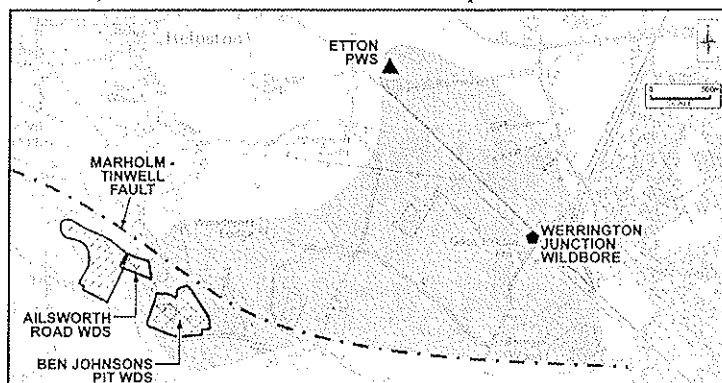
Further Assessment

In addition to field data, the natural attenuation processes were modelled using BIOPLUME III (USEPA, 1998). Although the model proved useful in demonstrating the likely timescale of full attenuation of the benzene plume, confidence in the model was insufficient to accurately predict the benzene and electron acceptor decay rates. In particular, the relative magnitude of attenuation as a result of dispersion and attenuation resulting from biological decay could not be established with confidence.

However, there was sufficient evidence to prove that an MNA was an acceptable remediation strategy. This is currently being implemented for the site.

3.3   Helpston Waste Disposal Sites, Cambridgeshire

Low levels of the herbicide, mecoprop, were first detected at a public water supply borehole (Etton) in 1988. It was established that contamination source was two former, unlined waste disposal sites (WDS), Ailsworth Road WDS and Ben Johnson's Pit WDS, 3km to the south west of Etton. Up to 40 tonnes of mecoprop were deposited within the WDS's between the mid 1970's and 1990. The WDS's, are located within former quarries in the Lincolnshire Limestone, a highly permeable fractured aquifer. Immediately to the east of the WDS's lies a major geological fault, the Marholm-Tinwell fault, which at one time was considered to be a barrier to groundwater flow. To the east of the fault, the limestone is overlain by relatively impermeable formations, resulting in confined conditions and artesian flows.

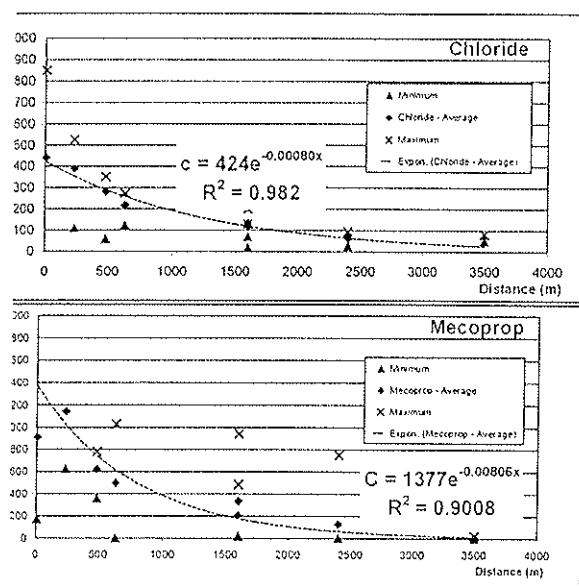Figure 3 Contaminated Extent of Lincolnshire Limestone

Since the contamination was first detected, the mecoprop plume has been monitored on a monthly basis. In addition a three dimensional contaminant transport model (Dames & Moore, 1998), coupled to a groundwater flow model of the Lincolnshire Limestone (Birmingham University, 1994) has been developed to predict the behaviour of the plume in the future, under various remedial strategies both at the WDS's (source removal and containment) and within the confined aquifer system (incorporating groundwater abstractions and MNA). In this paper, only the confined aquifer is discussed, as it has been established that a containment scheme will be developed in the immediate vicinity of the WDS's to ensure that no

further mass crosses the fault to the confined aquifer. The question therefore is whether the residual contamination within the confined aquifer will attenuate within an acceptable timescale, or whether groundwater abstractions are required to accelerate the process. A full MNA strategy, incorporating the WDS area as well as the confined aquifer, has been discounted because it has been estimated that of the 40 tonnes deposited, between 250 and 500kg is estimated to have entered the confined aquifer between 1984 and 2000 (i.e. only 0.04% of the deposited mass per annum). Consequently a full MNA strategy, without source removal, would take over 1000 years to complete.

Since there is an ongoing source of contamination of the confined aquifer, it is not possible to establish if there is any no primary evidence of biodegradation. Three different methods have been adopted to establish wither an MNA strategy for the confined aquifer is acceptable: an assessment of the hydrochemical data; modelling and laboratory investigations into the biodegradation of mecoprop.

Hydrochemical Data
Plots of concentrations of chloride and mecoprop along the centre line of the plume from the WDS's, a distance of 3km, indicated that there was very little additional attenuation of mecoprop compared in addition to dilution and dispersion of chloride. The first order decline in concentrations along this pathway is shown in Figure 4. This indicates that the average first order decay rate with distance is 0.00086 m^{-1} for mecoprop compared with 0.00080 m^{-1} for chloride, suggesting additional biological attenuating mechanisms are limited.

Laboratory Experimentation
BGS (2000, 2002) are currently conducting various laboratory experiments on the mecoprop originating from the Helpston WDS's. Mecoprop exists in two mirror image forms (known as the R and S forms) and, as such is known as a chiral compound. BGS has been studying both the biodegradation potential of each form and whether there is interconversion between forms, which could influence biodegradation. Based of various on-going laboratory experiments, BGS have concluded that:

Figure 4 Attenuation of Chloride and Mecoprop with Distance

- under anaerobic conditions, in the immediate vicinity of the WDS's, only the R-form degrades, with nitrate as the electron acceptor;
- under aerobic conditions, the S-form degrades at a faster rate than the S-form;
- there is no evidence of inversion of R-form to S-form under aerobic conditions.

Within the confined aquifer, the enantiomeric ratio (defined as the ratio of R-form to total) increases with distance from the WDS's, indicating that either the S-form is degrading preferentially or S-form is inverting to the R-form. Given that aerobic conditions do not exist over most of the confined aquifer, it is likely that any biodegradation is limited, a view that is supported by the results of monthly monitoring data.

Consequently, natural attenuation processes within the confined aquifer are considered to be limited to dilution and dispersion.

Groundwater Modelling
Using the contaminant transport model, different remedial options for the main aquifer were assessed. These include MNA and the abandonment of abstractions from Etton, continued abstraction from Etton and the installation of a pump and treat system. The simulated concentrations 3km to the east of the WDS's is shown in Figure 5, to demonstrate the differences.

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With no groundwater abstractions at Etton, the model indicated that that there is potential for the existing plume to migrate further to the north, towards other licensed abstraction boreholes. In addition, the model indicated that it will take over 30 years before the plume has fully attenuated below current drinking water standards. However, if Etton continues to abstract at between 5000 and 6000 m³/d, the plume is contained and attenuates within 20 years. With additional abstractions from a new borehole in the centre line of the plume, this attenuation duration reduces to 11 years. The choice between Etton and a new remediation borehole is, therefore, economic based.
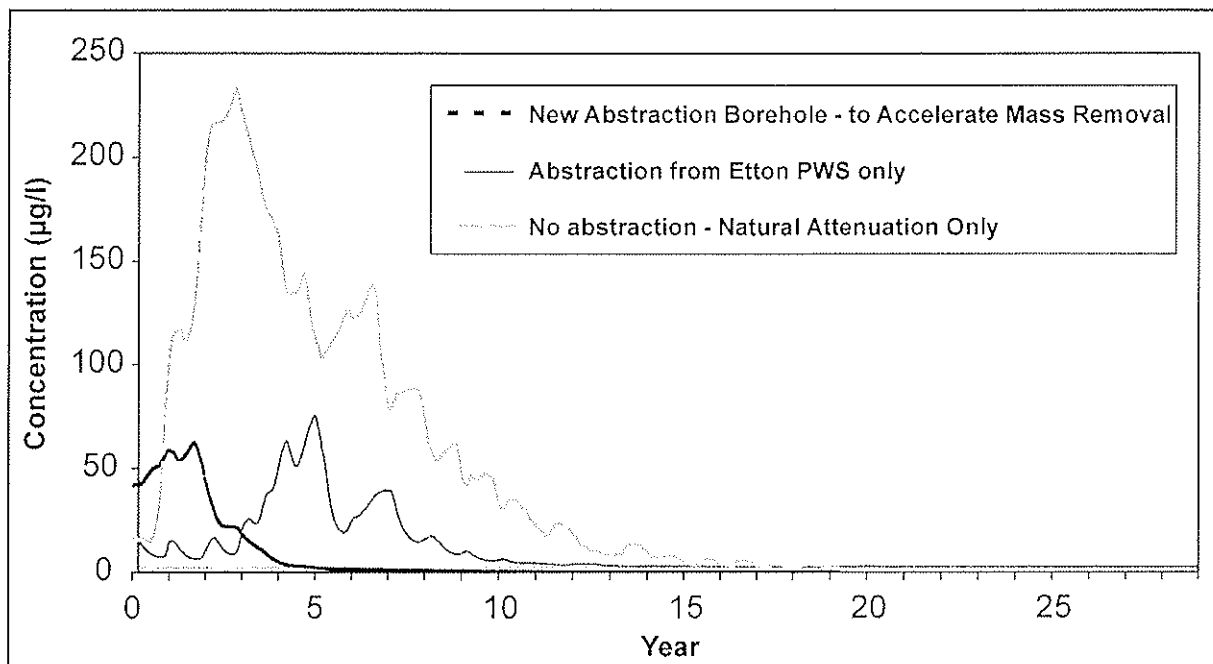

Figure 5 Simulated Recovery in Groundwater Quality

Consequently, through a combination of hydrogeological and chemical studies, laboratory experiments and groundwater modelling it has been shown that a MNA strategy alone is not feasible for the residual contamination within the confined, limestone aquifer.

3.4 Chemical Works, Staffordshire

This factory has manufactured organic chemicals since 1950. It was originally a coal tar distillery but now uses feedstock from other chemical plants. The site overlies the Permo-Triassic Sherwood Sandstone, a major aquifer, with groundwater encountered at approximately 4m below ground level. Groundwater flows to the west towards a public water supply (PWS) borehole and a river at a rate varying from 4 to 11 m/year. The PWS is situated approximately 1.8 km down-gradient of the site.

An increasingly complex sequence of groundwater sampling and analysis was undertaken at the site over the past 15 years. This has identified four principal plumes of phenolic contamination apparently emanating from the former tank farms and process areas. A concentration of total phenols of up to 25,000 mg/l has been measured from long-screened, observation boreholes. Data from multi-level samplers have indicated concentrations in the core of the plume greater than 50,000 mg/l.

The site has been the subject of significant research into the biodegradation of the phenolic compounds in recent years (e.g. Lerner et al, 2000), which has assisted URS in establishing the optimum remediation strategy for the site. Based on the results of the various research components, a full MNA strategy to remediate the different plumes was considered infeasible. Lerner indicated that degradation within the core of the plume is limited, primarily as a result of the chemical toxicity of the contaminants, although at the fringe of the plume, where concentrations are lower, biodegradation of

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the contaminants will be enhanced. Consequently a pump-and-treat remedial scheme for the plume core combined with MNA was considered the most cost effective strategy. The present value cost of a pump and treat scheme which remediated the whole of the contaminated groundwater to drinking water standards was estimated to be around £20million, whereas the combined strategy cost is estimated to be between £6 and £8 Million.

Whilst there is no primary evidence of natural attenuation at the site, both secondary and tertiary evidence of natural attenuation mostly in the more dilute areas of the plume has been identified, this being:

- Depletion of the electron acceptors oxygen, nitrate, and sulphate in the plume fringe, an increase in dissolved species of iron and manganese arising from utilisation of iron (III) manganese (IV), together with methanogenesis within the anaerobic body of the plume;
- Enrichment of inorganic carbon measured as total inorganic carbon (TIC) in the plume compared to background, to the extent that this balances the consumption of oxidants;
- The occurrence of anaerobic phenol degradation products 4 hydroxybenzoic acid, (4-HBA) and 4 hydroxybenzaldehde (4-HBH)
- The demonstration through an electron and carbon mass balance model, simulating phenolic oxidation for each of the different electron acceptors, that there has been a net reduction in mass of carbon based on an assumption of mass entering the aquifer since the 1950's;
- Tertiary evidence including the presence of a viable microflora with a demonstrable metabolic capability for utilisation of phenol and related aromatic substances, microcosm-based studies indicating phenol degradation in contaminant-exposed sediments and finally enrichment of the less preferentially metabolised sulphur isotope ^{34}S, in sulphate, and depletion of ^{34}S in sulphide, at the margin of the plume:

An MNA strategy has not been provisionally developed for this site, and accepted by the Environment Agency. The strategy comprises the following key components:

- establishment of a monitoring well network, which will characterise up-gradient, side-gradient, down-gradient and plume concentrations, allowing the fate and transport of the plume and associated chemicals to be analysed to demonstrate that the plume is attenuating as anticipated;
- a defined monitoring frequency and analytical schedule to ensure seasonal and long-term trends in groundwater levels, flows and quality are evaluated.
- the establishment of a GIS/modelling system to process, analyse and store the MNA data.

The cost of the MNA strategy, to last over 15 years, has been estimated to be around £0.5 million.

4.0   CONCLUSIONS

Whilst the development of a conceptual site model is essential for evaluating the suitability of any remedial strategy, a much greater level of understanding is required before MNA can be accepted as a viable approach. The degree of information gathering required to provide supportive evidence for occurrence of NA should therefore not be underestimated, neither should MNA be viewed as an 'easy' option, given the onus to demonstrate that it will indeed be sufficiently protective of the receptors in question.

At the same time, the potential cost benefits, particularly in the long-term from the application of MNA either as a stand-alone approach or as part of an overall remedial strategy can be enormous, as has been demonstrated in the above examples. In a wider aspect, the integration of MNA into 'active' remedial approaches potentially resulting in less onerous RBCLs could favour the application of in situ approaches such as bioremediation both in the saturated and unsaturated zones, which may

otherwise have difficulty in meeting particularly low target concentrations. The application of such technologies may have environmental as well as cost benefits over alternative approaches.

The challenge for research is to be able to provide long term reliable data on the bioattenuation of organic chemicals, so that better informed predictions of the rate of biodegradation in the long term can be made according to environmental circumstances.

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Permeable Reactive Barriers
13.
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PERMEABLE REACTIVE BARRIERS

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Abstract

Permeable Reactive Barriers are a novel means of in-situ groundwater plume management that incorporate the use of semi permeable reactive media to transform or immobilise contaminants. Their application is consistent with the risk based management of contaminated land in circumstances when source treatment is neither technically or economically feasible. Permeable reactive barriers can potentially incorporate a wide range of active materials that can address both inorganic and organic contaminants and a number of alternative techniques are available for construction. The results from the evaluation of systems installed to date are encouraging and the further use & development of these technologies seems likely.

INTRODUCTION

Over the last 5 years Permeable Reactive Barriers (PRBs) have emerged as a viable alternative to conventional groundwater treatment systems and are currently the subject of considerable academic and commercial interest. The objective of this paper is to provide a brief overview of the origins, design and implementation of PRB systems drawing on experience both in the USA and Europe.

PRB DEVELOPMENT

The development of PRBs appears to have been stimulated by two distinct drivers:

- Recognition of the limitations of 'pump and treat' as a remedial technology; and
- A desire to enhance the performance of existing passive in-ground containment systems.

The limitations of pump and treat

The abstraction of groundwater as a means of hydraulic containment coupled with above ground treatment is commonly referred to as 'pump and treat' and is frequently selected as a remedial alternative. When combined with re-infiltration of groundwater and other amendments then pump and treat systems can form the basis for in-situ treatment of source zones.

Despite the common use of pump and treat in soil and groundwater remediation its effectiveness in restoring groundwater quality has been questioned (1) with the chief technical difficulties reported as:

- Physical heterogeneity of the subsurface and difficulties in its characterisation;
- The presence of non-aqueous phase liquids (NAPLs);
- Diffusion of contaminants into inaccessible regions such as low permeability clays; and
- Sorption of contaminants to the subsurface matrix.

The presence of such difficulties typically resulted in considerably extended periods of operation with associated high maintenance and operational costs.

Passive Containment

Traditional, passive containment does not eliminate contamination or treat contamination but restricts the pathways by which the target may be exposed to mobile contaminants. This form of containment is a common response to contaminated sites where treatment of the source of contamination is not technically feasible or economic (2). Indeed, across Europe, containment is only second to landfill disposal as a means of site remediation (3).

Currently, containment is achieved by using varied combinations of in ground barriers using materials of low permeability, such as a cement–bentonite slurry wall, to physically restrict groundwater flow rather than intentionally alter or eliminate dissolved or suspended contaminants as they migrate through the wall.

In ground barriers are therefore relatively crude but effective mechanisms for controlling contaminant migration but can significantly alter local or regional hydrogeological conditions and additional hydraulic controls may be required within the contained site.

PRBs

Of the above drivers, probably the key to early development was the recognition of the limits of pump and treat. Development of PRBs was stimulated with the publication of a number of key papers in the USA as early as 1985 (4), but more importantly in the period 1992 to 1994 when the concept of permeable reactive barriers was more clearly defined (5,6), together with proposals for a funnel and gate type configuration (7) and the emergence of a treatment technology (zero valent iron) (8) that was ideally suited to configuration in PRBs.

In essence, the opportunities envisaged through the application of PRBs related to:

- The creation of smaller more clearly defined treatment zones in which subsurface treatment can be optimised;
- Contamination would be directed through these treatment zones by natural groundwater flow and would therefore be less energy intensive than pump or treat;
- The development of treatment technologies that would require little if any energy or maintenance; and,
- The creation of an effective long-term containment mechanism for sites where the source cannot be treated.

A PRB has been defined as "an emplacement of reactive materials in the subsurface designed to intercept a contaminant plume, provide a preferential flow path through the reactive media, and transform the contaminant(s) into environmentally acceptable forms to attain remediation concentration goals at the discharge of the barrier." (9). This definition considers PRBs distinctly from technologies where 'barriers' are formed through the creation of ill defined treatment zones, and emphasises the physical placement of a reactive material. The requirement to meet remediation goals at the outlet of the reactor is also onerous and there is scope for the use of PRBs as a cost effective means of reducing concentrations in a particular contaminant plume so allowing the operation of other mechanisms between the barrier and receptor.

The configuration of an idealised PRB system is illustrated in Figure 1.

Figure 1: A Permeable Reactive Barrier
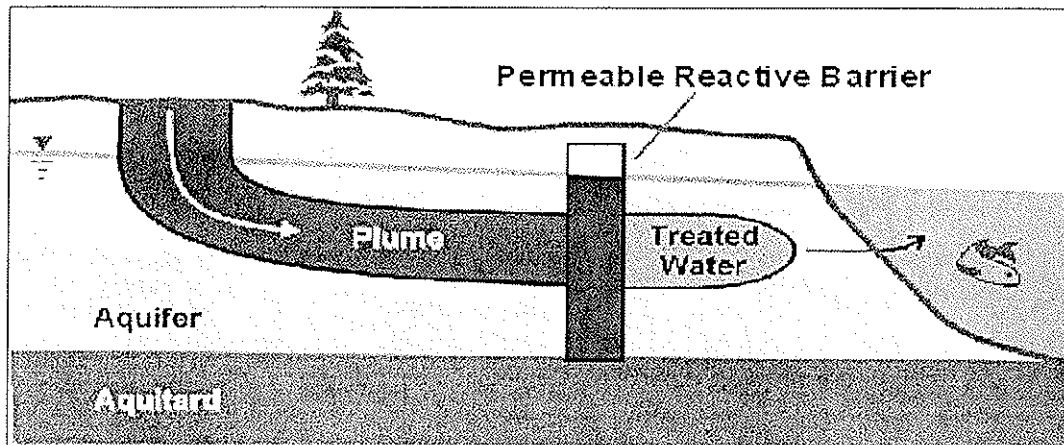
PRB DESIGN

Fundamentals

PRBs utilise semi permeable media that cause chemical or biochemical reactions to transform or immobilise contaminants as they migrate through the barrier. PRBs are passive in operation and are typically constructed perpendicular to a contaminant plume. For a given location, two of the key design related parameters are the groundwater flow velocity and mass loading:

- The groundwater flow velocity determines the potential residence time within a barrier of a given thickness, and comparison of the residence time with the predicted reaction rates for the contaminants of concern determines the extent to which treatment can be expected; and
- The groundwater flow velocity and concentration of contaminants within the plume at the point of construction determine the mass loading of contaminants at the barrier, this can be an important determinant of the longevity of the barrier.

For barrier emplacement a number of configurations may be considered including:

- The use of a continuous trench of reactive material;
- The use of a 'funnel & gate' configuration whereby the contaminated plume is directed by impermeable treatment walls to a treatment gate in which the reactive treatment is located; and
- The use of multiple treatment gates within an impermeable barrier.

Each of the above scenarios has a number of advantages and disadvantages. For example, continuous trenches are relatively easy to construct, though ensuring adequate thickness of the barrier along the whole length of the barrier does require adequate quality control. Monitoring of system performance also requires careful consideration and placement of multiple points to ensure adequate coverage. Funnel and gate systems focus the groundwater treatment process in a single area and so maximise the ability to monitor both up and down gradient concentrations and allow easy access to the treatment medium should it require maintenance or renewal. Funnel and gate systems do however have a significant impact on groundwater flow hydraulics and can lead to groundwater mounding or by pass of the plume.

The design and installation of a PRB therefore requires a broader understanding of the site characteristics. Typical requirements and methodologies have been documented (10, 11) and are summarised in Table 1.

Final design of the barrier is often undertaken with the aid of groundwater modelling that enables the design team to integrate the physical and chemical characteristics of the groundwater plume in relation to different design scenarios ensuring that the groundwater plume is captured and that the treatment process has sufficient residence time with which to meet the objectives.

Table.1   Site Characterisation Requirements for Permeable Reactive Barriers

Aquifer Characteristics	Chemistry
Depth and width of the plume/aquifer	Types and concentrations of contaminants of concern
The geotechnical characteristics of the aquifer materials	Types and concentrations of non-target contaminants
Aquifer Permeability	Existing groundwater redox status
Groundwater velocity	The inorganic composition of the groundwater
Heterogeneities	
The presence and depth of confining layer	
Seasonal Variations in Groundwater flow direction	

Active Mechanisms

Ideally, the active mechanism used in a PRB system should be:

- Effective in treating the contaminants of concern;
- Physically stable and durable for the design life of the barrier;
- Available in a form that permits placement and provides the conditions for the relevant physical and chemical processes to occur; and
- Economic to use, maintain and if necessary replace.

These requirements are reflected to a greater or lesser degree in the variety of active mechanisms that have been used or are under development for use in PRBs. These operate using physical, chemical or biologically mediated reactions in which contaminants may be immobilised, degraded of transferred to another phase. In a recent review of PRB technologies (12) the primary removal processes used in PRB construction were indicated as being:

- Sorption and precipitation;
- Chemical reaction; and
- Biologically mediated reactions.

Examples of reactive materials used in each of these categories is summarised in Table 2.

Table.2   Examples of reactive materials considered for use in reactive barriers

Material	Mechanism	Contaminants Treated
Ferric oxyhydroxide	Sorption	Uranium, Molybdenum, Chromium
Peat moss	Sorption	Heavy metals, organics
Spodic material	Sorption	Cadmium
Sawdust, Lignite, coal	Sorption	Heavy metals, organics
Ferric Chloride	Sorption	Mercury, cadmium, lead
Limestone	Precipitation	Heavy Metals
Hydrated lime, fly ash	Precipitation	Uranium

Material	Mechanism	Contaminants Treated
Zero Valent Iron	Chemical Reduction	Chlorinated aliphatics, some pesticides, chromium VI
Bi metallic iron	Chemical Reduction	Chlorinated aliphatics
Oxygen Releasing Compounds	Microbial Oxidation	BTEX, TPH
Hydrogen Releasing Compounds	Microbial Reduction	Chlorinated aliphatics
Organic Materials	Microbial Sulphate Reduction and Precipitation	Acid mine Drainage
Organic Material	Microbial nitrate reduction	Nitrate

It should be noted that many of the above have been demonstrated at field scale or at a limited number of commercial sites and the long term performance for many of the technologies is limited. The most successful commercial application of PRBs to date has been in the use of zero valent iron (ZVI). The reductive properties of ZVI were first identified in the early 1990s (8) and have since been the subject of intense research (13). ZVI has been shown to be particularly effective for the degradation of chlorinated aliphatics and ZVI barriers have been constructed at over 60 locations (14). The first application in Europe was installed at the Nortel Networks site in Belfast, N.Ireland in 1995 (15). The latter has been the subject of detailed study and a review of the design, installation and performance of the barrier was recently published by CL:AIRE (16).

Construction & Installation

A number of different methods have been used to construct PRBs and can be broadly defined according to those that use trench based system and those that are emplaced using point based systems. Each have particular characteristics and a summary of the main techniques is presented in Table 3.

Table 3: *Methodologies used for construction of PRBs*

Trench Based Systems	Point Based systems
Backhoe Trenching	Augured overlapping holes
Biopolymer Trenching	Hydro Fracturing
Continuous Trenching	Pneumatic Fracturing
Cofferdam	Jetting
Caisson Method	Soil Mixing

Trench based systems can be used to create a well defined treatment zone though are limited by the maximum depth of construction. Bioploymer trenching in which a degradable biopolymer is used to hold open the trench while the iron is emplaced can extended up to 18m in depth and is currently under development (17). With point based systems emplacement of the iron is rather less precise and point based systems often require a series of overlapping treatments to ensure adequate residence time for the contaminants of concern. An advantage of these systems is the greater depth that can be achieved with hydro fracturing reported to depths of 36m (14).

In funnel and gate systems impermeable sections have been created using sheet piles or cement bentonite slurry walls. The reactive gates have been formed either by an enlarged cofferdam or caisson technique or by using reactor vessels. A number of varying designs have been developed for such reactors using either horizontal or vertical flow, and located within or external to the barrier itself (16,17,18 & 19). An example of a reactor system is illustrated in Figure 2.

Figure 2: Example of Reactor Vessel Construction
(Suthersan 1997)

Cross section of treatment gate
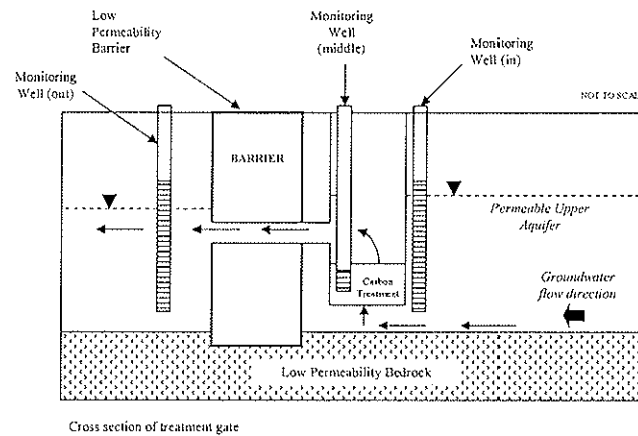
COMMERCIAL APPLICATIONS OF PRB TECHNOLOGY

As indicated above the most successful commercial application of PRB technology to date has been in the use of Zero Valent Iron for the treatment of chlorinated aliphatics where over 80 full scale projects have been or are in construction. This is an excellent example of where an innovative treatment technique (ZVI) has been combined with means of implementation (PRB) to fill a market niche (the long term treatment of chlorinated solvent plumes - where source treatment is difficult and plumes may persist for many years). The capital costs of construction of ZVI barriers have been documented for a number of case studies and an example illustrating three different techniques and barrier dimensions is illustrated in Table 4.

Table 4: Construction Costs for Three ZVI barriers using differing construction techniques

	Construction	Iron	Total
Backhoe Construction, OH '99			
• 8 ppm TCE			
• 6 m deep, 60 m long	$36,000	$28,000	$64,000
BioPolymer Trench, NH '99			
• 10 ppm cDCE; 5 ppm TCE; 1 ppm VC			
• 10 m deep, 46 m long	$200,000	$130,000	$330,000
Trench Box, WY '99			
• 21 ppm TCE; <1000's ppb cDCE, VC			
• 7 m deep, 172 m long	$255,000	$745,000	$1,000,000

Average Cost (10 Sites) = $425,000

Source: EnviroMetal Technologies Inc.

Economic analysis of both capital and operational costs of ZVI PRB barriers has confirmed their cost effectiveness when compared to traditional hydraulic containment (16) with one source claiming that in a potential US market of 500 barriers over 10 years then cost savings over traditional techniques could collectively amount to between $500million and$1billion (20). The success of the ZVI technology does in many ways contrast with the other available treatment technologies where the advantages over existing techniques are less apparent and the long term performance of the reaction mechanism is less certain. However there may be significant opportunities for barrier type technologies in the shorter term either as supplements to source treatment or in combination with strategies such as natural attenuation.

CONCLUSIONS
Permeable Reactive Barriers are an innovative technology that can offer a cost effective means of groundwater plume management. The successful application of PRBs requires an appropriate degree of site characterisation and the integration of chemistry, geology and hydrogeology in order to develop a satisfactory design. To date experience of PRBs is greatest using ZVI as a reactive medium for the degradation of chlorinated aliphatics and the reduction of chromium. Potential concerns with the technology have always focussed on loss of performance due to fouling or precipitation. Current field data suggests that these will not be major issues (at least at the sites studied). The technology is still in its infancy and the long term performance of PRBs will require on going evaluation.

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Useful Web Links for PRBs:

www.prb.net
www.rtdf.org/public/permbarr/default
www.inquip.com/prb
www.bps-zwickau.de/files/wasserb/prb
www.environment-agency.gov.uk/gwcl