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DEPARTMENT OF CIVIL, STRUCTURAL AND ENVIRONMENTAL ENGINEERING

An investigation into the effects of the density of on-site wastewater treatment systems in Ireland

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DECLARATION

I, Patrick Morrissey, declare that:

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SUMMARY

Groundwater is an important resource in Ireland and given that over 25% of all water supplies are currently provided directly from groundwater abstractions, it is critical to protect this resource from contamination. Most streams and rivers in Ireland also receive baseflow from groundwater and the requirement under the European Water Framework Directive (WFD) is to achieve good status in these waters by 2015. There are nearly half a million on-site wastewater treatment systems across the country and given that the treated wastewater from these systems ultimately discharges to groundwater, any contaminants not treated or attenuated in the subsoil can migrate under natural gradients toward points of exposure for receptors of concern, e.g. humans and drinking water supplies or sensitive surface waters. Ireland's Environmental Protection Agency (EPA) has published recommendations aimed at defining subsoil conditions that will provide an acceptable level of treatment for on-site wastewater in order to protect such groundwater resources from contamination. However, there are currently no guidelines on what the maximum density of on-site wastewater treatment systems should be specific to the groundwater protection plans that are in place across the country. Given that these systems are usually constructed at rural developments that are clustered in nature this is an issue of some importance with respect to groundwater quality. The lack of guidance has led to an inconsistent planning policy with no national strategy in this area. The focus of this thesis is a 3 year study to quantify the effects of the density of on-site wastewater treatment systems on groundwater quality at different groundwater vulnerability areas across the country.

Four study areas were selected across the four different vulnerability ratings and monitoring boreholes were drilled upstream and downstream of the clustered developments. Water quality was monitored at these boreholes over an extended period for pollutants and indicators such as nitrates, phosphorus and bacteria. The results of this extended study were then used to make statistical comparisons between upstream and downstream water quality and therefore quantify the effects of the cluster of on-site wastewater treatment systems on groundwater quality. No statistically significant difference in downstream water quality was found when compared with upstream water quality at any of the study areas. The field study therefore indicated that at each of the study locations the current density is not having a cumulatively negative effect on groundwater quality. In addition a conduit karst site was monitored in order to investigate the impacts of density in a karst setting which is of great importance in Ireland particularly in the west of the country.

Numerical models were set up in a finite element modelling software, HYDRUS in order to simulate the attenuation and treatment of contaminants in the unsaturated zone (i.e. the subsoil

above the water table). These models were set up to represent the subsoil conditions at each of the study sites based on field tests that were carried out. The resulting simulations of concentrations and water fluxes at the water table were then used as input into MODFLOW MT3D models which were constructed for each of the study sites. The results of the MODFLOW models indicated that the cluster of on-site wastewater treatment systems at each of the sites were only contributing limited loading to groundwater even when inclusions were made for poorly functioning percolation areas. This agreed with the results of the field study which also indicated that the clusters of on-site wastewater treatment systems were not significantly impacting on groundwater in the study areas. Further simulations were then run using MODFLOW MT3D to access the impacts of increasing the density of treatment systems at each of the study locations. It was found that increasing the density of the treatment systems did not have a major impact at any of the study locations with the exception of the *EXTREME* vulnerability site where the models indicated that increasing the density past a threshold of 3 units/hectare did cause plumes of bacteria downstream albeit at low levels. Overall the study concluded that density of on-site wastewater treatment systems does not appear to be an issue for concern and a recommended maximum density of 6 units/hectare would appear to be appropriate based on meeting the accommodations of the EPA Code of Practice legislation.

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1 INTRODUCTION

1.1 Background

Groundwater is an increasingly important resource in the Republic of Ireland. It is estimated that up to 40% of total water supply in the Republic of Ireland is taken directly from groundwater sources (Daly, 1993; EPA, 1999). Given the current and projected reliance on groundwater as a source for drinking water in this country, it is therefore necessary to ensure that adequate protection is in place to ensure that groundwater sources are maintained to a high standard and that they are kept free from sources of pollution. From an environmental perspective groundwater quality must also be protected particularly given that baseflows in Irish rivers during dry weather periods are generally supported from groundwater sources. In this regard Ireland also has an obligation under the European Groundwater Directive to maintain a high quality standard for its groundwater resources. The main sources of contamination to groundwater resources in Ireland are from a large number of small point sources such as farmyards and single on-site wastewater treatment systems (EPA, 2003) or dispersed pollution from agricultural practices. The new Code of Practice, Wastewater Treatment Systems for Single Houses (EPA, 2009), defines subsoil conditions such as thickness and permeability that will provide an acceptable level of treatment for on-site wastewater effluent in order to protect groundwater resources from contamination. Recent research carried out as part of the preparation of this Code of Practice has identified how treated effluent from different on-site treatment systems disperses into the subsoil through the percolation area and how pollutants are attenuated (Gill et al., 2005; 2009). However, it is uncertain what the effects of higher hydraulic and contaminant loads generated by a cluster or grouped development would be when discharged in a concentrated area. This situation occurs in many areas across Ireland, particularly in a rural context whereby single dwellings incorporating on-site waste water treatment systems tend to be constructed in a clustered arrangement typically following a “ribbon” of dwellings along local roadways. Often these systems have been built to older standards and it is not known what the combined effects of groups of treatment systems in a relatively dense arrangement may have on groundwater quality, particularly in areas of varying groundwater vulnerability.

1.2 Research Context

In recent years a new document was published by the Environmental Protection Agency (EPA) in relation to on-site wastewater treatment systems and groundwater protection. The guidance manual Code of Practice: Wastewater Treatment Systems for Single Houses (EPA, 2009) provides guidelines for the selection, design, operation and maintenance of these systems to enable continued sustainable development in Ireland. In addition previous government documents including; the Groundwater Protection Schemes report (Department of the Environment and Local Government et al., 1999) aimed to maintain the quantity and quality of groundwater and in some cases improve it, by applying a risk assessment-based approach to groundwater protection and sustainable development. A new EPA strategy for the inspection of existing on-site wastewater systems has also been announced (EPA, 2013) with an accompanying risk guideline document. None of these documents or any other government agency document provide specific guidance to planning authorities on an acceptable density of these systems specific to the varying risks and geological conditions across the country.

During a review of current practices (see Chapter 2) it was found that local authorities, who determine whether to grant permission to construct these systems, do not have a common policy in this area and many adopt very different approaches and strategies when addressing the issue of density of clustered on-site wastewater treatment systems. In this regard it was identified that there was a requirement for guidance to be issued in this area and that that guidance needed to be based on appropriate field studies and numerical modelling. A conceptual section through a typical ribbon development incorporating a cluster of on-site wastewater treatment systems is given in Figure 1.1.

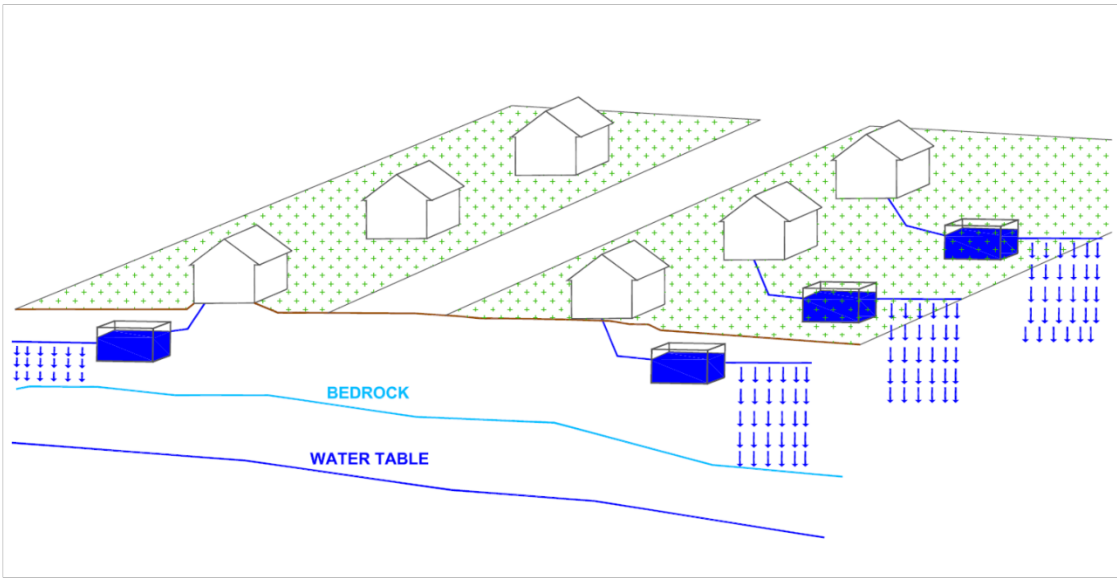


Figure 1.1 Cross-section through a typical ribbon development with on-site wastewater treatment systems

1.3 Aims and Objectives

The main aim of the project was to assess the impact of on-site wastewater effluent from different cluster systems on groundwater quality particularly with respect to density and groundwater vulnerability. This was achieved through both a detailed field study and numerical modelling. Given these requirements for both field based data collection together with desk based modelling, the project needed to be well structured and priority needed to be given early to identifying and selecting appropriate study areas in order to ensure completion within the time constraints involved. Listed below are the main objectives that were targeted in order to address the project's aim:

- Identify study areas with the required density of treatment systems in areas of Low, Moderate, High and Extreme groundwater vulnerabilities

- Complete an extended period of field monitoring for a range of water quality parameters with a desired duration of 24 months. This field monitoring would include any relevant field testing that would be required later in the study (i.e. to populate numerical models – see Figure 1.2 for conceptual field setup)

- Develop numerical models representing the unsaturated subsoil zone at each of the study sites to simulate the movement and attenuation of contaminants from the associated on-site wastewater treatment systems.

- Develop numerical models representing the saturated groundwater flow regime (i.e. the bedrock aquifers) to simulate the movement and attenuation of contaminants from the associated on-site wastewater treatment systems arising from the output of the unsaturated zone models as above
- Compare the field results with the numerical models
- Use the models to make further predictions for varying on-site treatment system densities

Following the completion of the tasks as outlined above, it is anticipated that recommendations will be made on what (if any) the impacts of density of on-site wastewater treatment systems are groundwater quality within the context of the varying geological and subsoil conditions that exist in Ireland.

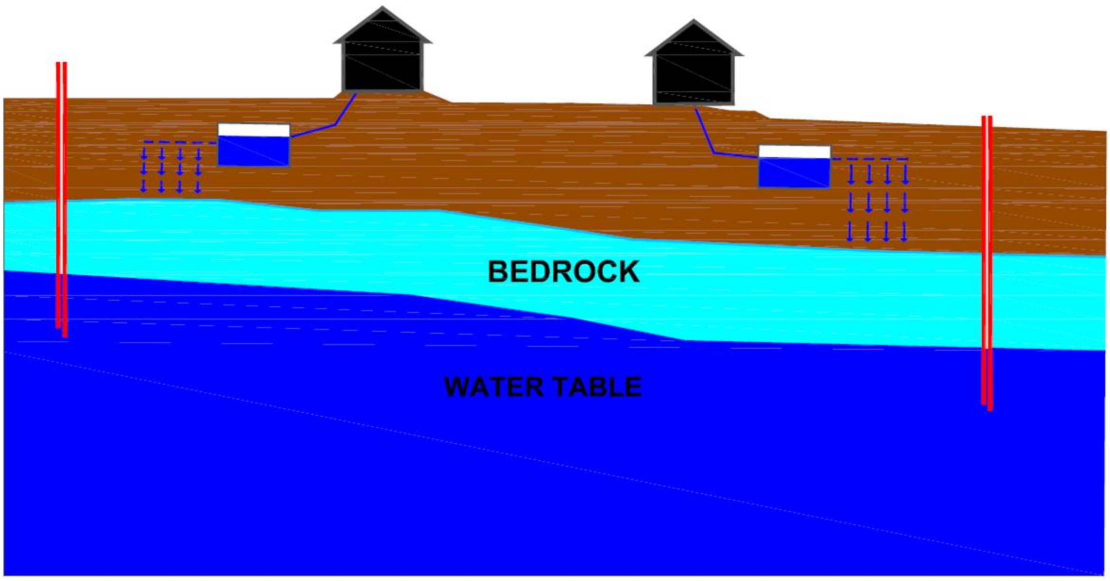


Figure 1.2 Arrangement of desired field monitoring setup for the project showing upstream and downstream boreholes (in red)

2 LITERATURE REVIEW

2.1 Overview of Water Supply and Wastewater Treatment in Ireland

2.1.1 Potable Water Supply

Potable drinking water in Ireland arises from a number of different public and private supplies. The majority of the population (~85%) receives drinking water through Public Water Supplies (PWS), which are operated by the Water Services Authority (EPA, 2010a). The Water Services Authority also supplies drinking water to “Public” Group Water Schemes (PuGWS); however the owners of the group schemes themselves are then responsible for distribution of the water supply. “Private” Group Water Schemes (PrGWS) are water supply schemes where the owners of the scheme itself will source and distribute their own water. “Public” and “Private” group water schemes account for approximately 12% of drinking water supply in Ireland (EPA, 2010a). The remainder of the population receive drinking water from Exempted Supplies and Small Private Supplies (SPS) – see Table 2.1. An exempted supply is one that serves less than 50 persons and does not supply water as part of public of commercial activity. Typically in Ireland an exempted supply is a private well serving an individual dwelling and these supplies serve approximately 13% of the population (EPA, 2010a). Small Private Supplies (SPS) generally consist of industrial water supplies and boreholes serving commercial and public buildings and comprises of 1,284 different supplies (EPA, 2010a).

Table 2.1 Water Supplies and Proportion of Population Served, 2010. (CSO, 2001; EPA, 2010a)

Type of supply	% of Population Served
Public Water Supply	75.6
Public Group Water Scheme	8.7
Private Group Water Scheme	2.8
Small Private Supply	9.8
Other Exempted Supplies	3.1

A large proportion (81.9%) of drinking water supply in Ireland (in particular for public water supplies) originates from surface water sources (EPA, 2010a). Approximately 10.3% of the drinking water supply for the country originates from groundwater with the remaining 7.8% originating from springs (EPA, 2010a).

2.1.2 Groundwater Sources

One of the most important natural resources in Ireland is groundwater. It originates from rain that soaks into the ground and is stored and flows through either fractures in the bedrock or the pore spaces of sand and gravel deposits. An aquifer is defined as "any stratum or combination of strata that stores or transmits groundwater" (Water Pollution Act, 1990). However, a geological deposit is commonly referred to as an aquifer if it can yield enough water for a significant water supply (DELG/EPA/GSI, 1999). Groundwater provides approximately 20% - 25% of drinking water supplies, however in some counties such as Roscommon groundwater accounts for nearly 90% of the drinking water supply (DELG/EPA/GSI, 1999); in many rural areas of the country it is the only source of supply. A large proportion of the productive aquifers in Ireland are karstified limestone which can transmit large volumes of water through conduit, fissure and cave systems that have developed underground in the past (GSI, 2000). Given the current and projected usage of groundwater as a source of drinking water supply and Ireland's obligations under European Legislation, groundwater quality must be protected. Discharges to groundwater are regulated and authorised by local authorities and the Environmental Protection Agency (EPA) under relevant European Legislation including The Water Framework Directive (2000/60/EC) (WFD) and the Groundwater Directive (2006/118/EC) (GWD). Groundwater quality is at risk from agricultural, industrial and other human activities such as private on-site wastewater treatment systems (DELG/EPA/GSI, 1999).

The EPA has identified parameters that are indicators of anthropogenic pollution to groundwater, these are summarised below (EPA, 2010b):

- Ammonium
- Nitrate
- Phosphate
- Faecal Coliforms
- Pesticides
- Chemical Organic Parameters

After an initial monitoring period, the EPA decided that the instances of groundwater being contaminated with pesticides or chemical organic parameters were proportionately low enough to initiate a less intensive monitoring programme. Table 2.2 gives guideline threshold values for the remaining parameters listed above as outlined by the EPA. Values observed above these thresholds are usually indicative of a nearby source of organic pollution (EPA, 2010b), however the values given are usually much lower than those given in the European Drinking Water Regulations

Table 2.2 Threshold Values for groundwater quality usually indicative of anthropogenic activities (EPA, 2010b)

Indicator Parameter	Guideline Threshold Value
Ammonium (NH_4)	0.15 mg/l
Nitrate(NO_3)	10* mg/l
Ortho-Phosphate (PO_4)	0.035 mg/l P
Faecal Coliforms	100 cfu/100ml

** Schedule 5 of SI No. 9 of 2010 sets a threshold of 35mg/l however EPA, 2010b suggests values greater than 10mg/l indicates anthropogenic organic or inorganic inputs*

It should be noted that the presence of even one faecal coliform in a drinking water supply is considered a breach of the Drinking Water Regulations (S.I. No. 278 of 2007). A new strengthened regime for the protection of groundwater has been established under The European Communities Environmental Objectives (Groundwater) Regulations, 2010 (S.I. 9 of 2010) which is in line with the requirements of the Water Framework Directive (2000/60/EC) and the Groundwater Directive (2006/118/EC). The EPA has been identified as the responsible body for:

- Establishing and maintaining a list of Threshold Values (TVs) for pollutants in groundwater
- Assessing the chemical and quantitative status of groundwater bodies
- Undertaking pollutant trend and trend reversal assessments

The EPA have recently published guidance documents in the area of groundwater quality and these documents (EPA, 2010c & d) together with the existing Code of Practice on

Wastewater Treatment for Single Houses (EPA, 2010e) and the Nitrates Directive ((91/676/EEC) form the current best practice and guidance for mitigating the risks to groundwater from anthropogenic activities. Given that a large proportion of groundwater fed drinking water supplies are private wells, the proportionate level of monitoring in quality is far lower than that of public water supplies and this presents an even higher risk to public health if contamination does occur.

2.1.3 Surface Water Sources

The majority of public water supply in Ireland originates from surface water sources. The greater Dublin region receives almost all of its water supply demand from a number of dams on the River Liffey with reservoirs at Poulaphuca in Wicklow (shown in Figure 2.1) and Leixlip in Co.Kildare. Another older reservoir at Roundwood in Co. Wicklow which abstracts water from the River Vartry, also provides water to the city and surrounding areas.

Figure 2.1 The Poulaphuca Reservoir in Co. Wicklow (Water Supply Project - Dublin Region, 2010)

Water is treated at a number of Water Treatment Plants (WTP) located in counties on the periphery of Dublin close to the associated reservoirs. The largest of these WTP's is at Ballymore Eustace in Co. Kildare with an output of 318 million litres per day (Mld) in 2010 (Water Supply Project - Dublin Region, 2010).

Water quality in Ireland is monitored by the Water Services Authority in conjunction with the EPA. There are 48 parameters to be monitored under the 2007 Drinking Water Regulations (EPA, 2010a). Compliance is assessed by comparing the results of the analysis of samples taken from supplies with the required standard as set out in the Drinking Water Regulations. Of the 48 parameters required to be monitored under the regulations, seven key parameters have been identified by the EPA which are:

- E. coli
- Enterococci
- Lead
- Nitrate
- Trihalomethanes,
- Aluminium
- Turbidity (at the water treatment plant)

The EPA published a summary of results of sample analysis for Irish drinking water in 2010 (EPA, 2010a) and for the majority of parameters the compliance rate was between 97 – 100%. However a lower compliance rate of approximately 94% for both pH and coliform bacteria was achieved – these results are biased by a number of smaller supplies which frequently exceed the threshold values and inflate the national statistics. The report concluded that these supplies would have to be focused on in order to achieve a higher compliance rate particularly for coliform bacteria.

Over the past 10 – 15 years the Dublin region has grown at a rapid rate and consequently the public water demand has grown significantly also. Water demand in to the future is expected to be met through extractions from the River Shannon at Lough Derg thus ensuring Ireland's continued reliance on surface water as our main source of potable water supply.

2.1.4 Wastewater Disposal

2.1.4.1 On-site Single House Disposal

On-site wastewater treatment systems provide a means for the disposal of foul effluent from a single house and are contained within the boundary of the dwelling. There are two main types of systems available for on-site treatment systems, conventional septic tanks or on-site proprietary treatment systems (secondary treatment). The population of Ireland tends to be quite spatially distributed outside of the main urban areas and therefore the use of on-site single house disposal systems is quite prevalent. The different types of treatment systems available and the associated implications for groundwater quality will be discussed in detail in Section 2.3.

2.1.4.2 Decentralised Systems

In addition to on-site disposal for single houses, wastewater can also be disposed of in small decentralised systems which are sometimes referred to as cluster systems. Decentralised systems usually involve grouping a number of single dwellings into a localised central treatment system and can contain anything from 2 to 100 houses or a Population Equivalent (PE) of 10 to 500 (EPA, 1999). The types of treatment that are available for decentralised systems tend to be similar to those for centralised systems just at a smaller scale. Decentralised systems are not very prevalent in Ireland and tend to only be utilised in isolated instances usually by a Local Authority whereby the on-going maintenance is assured, however in the last number of years rural housing developments have begun to incorporate decentralised systems to maximise on the demand for housing outside of urban areas in commuter belts. Local Authority policy has also promoted this trend in many areas of the country, with rural clusters featuring in many county development plans leading to ‘ribbon’ style developments which can be very isolated from services and amenities.

2.1.4.3 Centralised Systems

Centralised wastewater management is comprised of wastewater collection, treatment and reuse or disposal of effluent and sludge and is the most practical and cost-effective method of wastewater disposal in urbanised areas. Urban wastewater treatment comprises of up to four generic stages:

- **Preliminary treatment (pre-treatment)** involves the screening of large debris and the removal of grit, fat and grease. Flow into the Wastewater Treatment Works (WWTW) is also regulated during pre-treatment

- **Primary treatment** involves settlement or sedimentation of suspended solids with removal of the resulting sludge

- **Secondary treatment** aims to substantially degrade the biological content of the sewage usually using aerobic biological processes

- **Tertiary treatment and Nutrient removal** in the final stage at a WWTW and aims to further improve the effluent quality before it is discharged to the receiving environment (sea/river/lake) and can be achieved by filtration, lagooning and internal mixed liquor recycle pumps

The Urban Waste Water Treatment Regulations 2001-2010 and the 1991 Urban Waste Water Treatment Directive (UWWTD) set requirements on; the provision of waste water collection systems and treatment plants, provide for the monitoring of waste water discharges and also specify limits for certain parameters in the discharges (EPA, 2011). A summary of the provision of wastewater in Ireland in 2012 is given in Table 2.3 below

Table 2.3 Provision of waste water treatment 2001-2009 (based on PE) (EPA, 2011)

Year	No Treatment/ Preliminary Treatment	Primary Treatment %	Secondary Treatment %	Secondary Treatment with Nutrient Reductions %
2009	6	1	78	15
2007	9	1	75	15
2001	30	41	21	8

The EPA (2011) report on urban wastewater discharges in Ireland highlighted eleven large urban areas (≥2,000 PE) that did not meet the UWWTD requirement for secondary treatment and eight urban areas with a PE greater than 10,000 that did not meet the UWWTD requirement to provide nutrient reduction in addition to secondary treatment for discharges to sensitive areas. In 2011 discharges from 57 wastewater treatment works were directly impacting negatively on rivers or bathing waters in Ireland. Under Ireland's requirements as set out in the UWWTD, Ireland must achieve compliance with the required level of treatment at these WWTW's by 2015. It is hoped that these improvements will also assist in meeting targets under the Water Framework Directive (WFD) whereby Ireland must achieve at least 'good" status in many of the country's major rivers, lakes, estuaries and coastal waters (EPA, 2011).

2.2 Groundwater in Ireland

2.2.1 Bedrock Aquifers

An aquifer is defined as a geological deposit that can yield enough water for a significant water supply (DELG/EPA/GSI, 1999). Groundwater flows through fissures, fractures and faults in most Irish bedrock aquifers with the proportion of gravel or similar aquifers limited (GSI, 2007). Consequently the amount of water that can flow through the associated aquifer unit tends to be limited by the number, size and connectivity of fissures, with more groundwater flow in rock that has many large and well connected fissures and less groundwater flow occurring in rocks that have only few small fissures that are poorly connected (Bear, 1979; Daly et al., 1980). Aquifers in Ireland have been classified by the GSI as part of the development of the Groundwater Protection Schemes document (DELG/EPA/GSI, 1999). This classification reflects the groundwater flow regime (i.e. fissure/karst/gravel) and the resource potential of the aquifer (i.e. regional/local/poor importance). Aquifers have been classified into eleven categories as listed below. An accompanying aquifer map of Ireland is given in Figure 2.2

Regionally Important (R) Aquifers

- Karstified bedrock **(Rk)**
  - **Rkc** – dominated by conduit flow
  - **Rkd** – dominated by diffuse flow.
- Fissured bedrock **(Rf)**
- Extensive sand & gravel **(Rg)**

Locally Important (L) Aquifers

- Bedrock which is Generally Moderately Productive **(Lm)**
- Bedrock which is Moderately Productive only in Local Zones **(Ll)**
- Sand & gravel **(Lg)**
- Locally important karstified bedrock **(Lk)**

Poor (P) Aquifers

- Bedrock which is Generally Unproductive except for Local Zones **(Pl)**

- Bedrock which is Generally Unproductive **(Pu)**

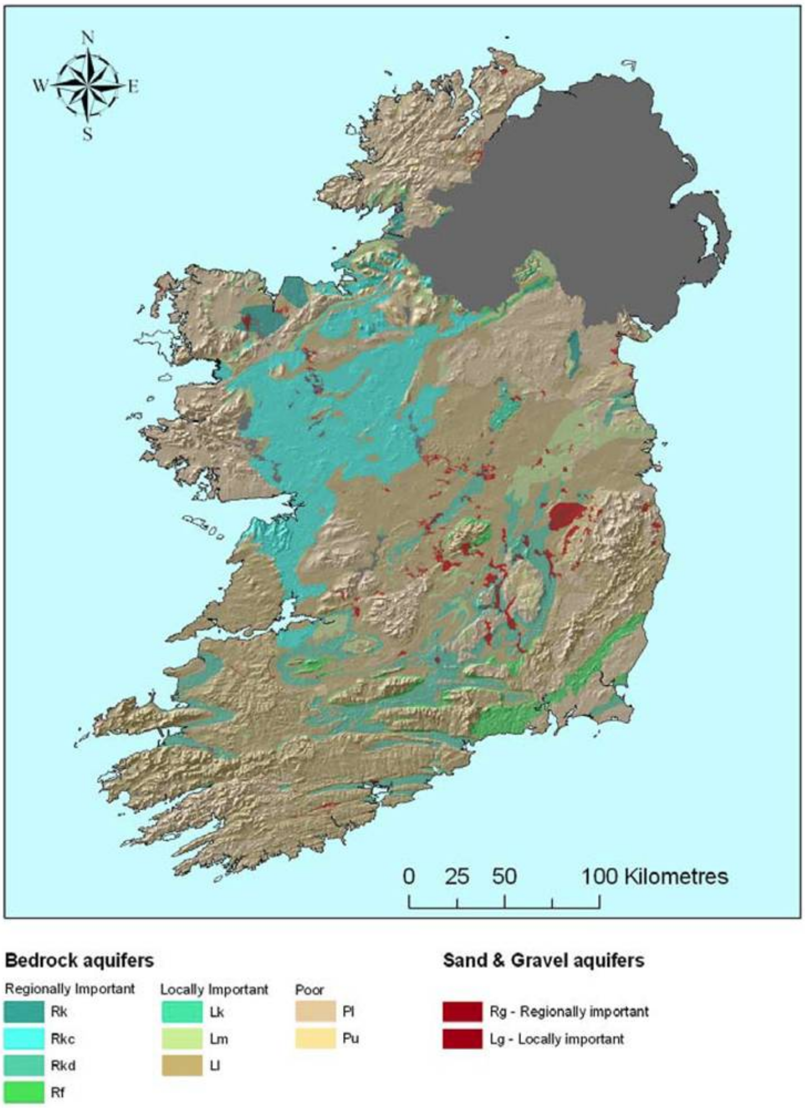


Figure 2.2 Aquifer map of Ireland

The extent to which the fissures and faults through which groundwater flow occurs will develop depends on a number of factors including; the rock type (i.e. limestone, sandstone, granite), its original structure (i.e. bedded, unbedded, cooling joints, etc.) and the type and quantity of deformation that the rock has been subjected to (e.g., folding and faulting) (Todd and Mays, 2005). Much of Ireland is underlain by limestone bedrock (GSI, 2000) and consequently the purity of these limestone units indicates their susceptibility to dissolution (karstification) and the presence of bedding influences the likely presence or absence of jointing. The proportion of deformation that rocks have undergone will influence the quantity, size and connectivity of fissuring in them with rocks that have undergone little or no deformation tending to have very little fissuring and rocks that have undergone higher proportions of deformation likely to have more abundant fissuring present (GSI, 2007).

Moe et al. (2007) estimated that groundwater abstractions supply up to 200 million m^3 of water annually for both domestic and industrial use accounting for almost 30% of national usage. Groundwater quantity and quality can be generally classed as good with only 1% of groundwater bodies nationally estimated to be at risk of over-abstraction (EPA/RBD Co-ordinating Authorities, 2005). There are however areas of the country that are subject to repeated cases of microbiological contamination and elevated levels of nutrients specifically nitrates (EPA, 2006). It is estimated that there are over 100,000 groundwater abstractions in Ireland through both wells and springs (EPA, 2010a) and bedrock aquifers are therefore a hugely important national resource. In addition to providing water supply resources for human use and consumption, bedrock aquifers also provide base flow for many of the country's main river systems and many other ecological systems are either partially or wholly dependent on groundwater to sustain them such as fens and peat bogs (Otte, 2003).

2.2.2 Subsoils and Bedrock

Given that bedrock aquifers in Ireland are such an important national resource, it is important to understand the relationship between the overlying subsoils with the bedrock aquifers beneath. The nature and extent of subsoils influences groundwater quantity by limiting or allowing recharge and also affects quality by providing protection to the aquifer from contamination either from human or animal activities (Todd and Mays, 2005). Subsoils are the unlithified looser sediments that are located beneath the topmost layer of the ground surface (topsoil) and the substratum beneath – typically bedrock. The topsoil layer which is subjected to both biological and weathering processes is usually less than one metre thick, but this varies across the country. In Ireland, the majority of these subsoils consist of glacial deposits or 'tills'. The method used to describe subsoils in Ireland is BS5930 (British Standards Institute, 1981) and contains textural descriptions of fine-grained materials based on behavioural characteristics, such as plasticity and dilatancy. It also includes a description of the colour, density/compactness, bedding and the presence of discontinuities, where appropriate (Swartz et al., 2003). The vulnerability (described in more detail later) of an aquifer is defined by the permeability and thickness of the overlying subsoils, however other material factors such as particle size distribution, plasticity, dilatancy and mass factors such as density/compactness, bedding and discontinuities can influence the vulnerability at specific sites (O'Luanaigh, 2009).

Subsoil Description and Factors Affecting Permeability

The particle size distribution of a subsoil will influence its permeability as subsoils containing a greater proportion of fine grained materials will have fewer pore spaces for water to travel through and will have a low permeability; conversely coarser grained subsoils will have more well connected pore space and can therefore transmit water faster and more freely and will be high permeability (Stephens, 1996). The plasticity of a soil is the ability of a soil material to deform, or change shape, without breaking when subject to an external force or pressure. Higher plasticity subsoils tend to be less permeable due to their tendency to include more clay particles. When describing soils, dilatancy describes the expansion in volume of a subsoil sample on shearing. Typically sands and some silts with high permeability tend to be 'dilatant' whilst lower permeability clays tend not to be.

The colour of a subsoil can indicate its drainage as higher permeability subsoils will give rise to well aerated conditions and tend to have a red/brown colour caused by the oxidation of iron to its highest state (Fe^{+3}). Subsoils that have a low permeability tend to be more saturated and consequently anaerobic conditions are prevalent with these soils tending to have a grey colour owing to the reduction of iron by bacteria to Fe^{2+}, a non-coloured state. Mottling can occur in soils that have repeated seasonal variations in their degree of saturation and this gives rise to a distinctive blotchy colour pattern of mixed rust colours and grey. Subsoils containing peat or organic matter usually have a dark brown or blackish colour (Swartz, 1999).

In addition to the above there a number of other methods of estimating the likely permeability of subsoil including:

- Density – the more dense the subsoil is the more compressed the pore spaces are, and the lower the relative permeability.
- Discontinuities – the presence or absence of preferential flow pathways in the subsoil
- Bedding – the degree of heterogeneity of the subsoil i.e. the non-uniformity of the sediment and the presence of beds, laminations or lenses of different sediments within the subsoil

As mentioned earlier, most Irish subsoils are derived from glacial drift or 'till' and are relatively immature being derived from the geological era known as the Quaternary period – the shortest and most recent geological period covering the last 1.6 million years of the Earth's history (GSI, 1999). The dominant force that shaped the landscape and derived most Irish subsoils during this period was the last Ice age which ended c.12, 000 years ago. The parent materials of nearly all mineral soils in Ireland arise from erosion and deposition caused by ice sheets that crossed the land during this period. The way in which these glacial ‘till’ materials were deposited across the country in a non-uniform manner gives rise to a highly heterogeneous subsoil spatial distribution. The main subsoil groups present in Ireland are summarised below.

Till

Till, otherwise known as boulder clay, is the most common and widespread Quaternary subsoil type. It is often tightly packed, unsorted, unbedded, possessing many different particle and clast sizes and types which are often angular and subangular.

Glaciofluvial Sands and Gravels

The highest permeability Quaternary deposits found in Ireland are typically glaciofluvial sands and gravels or esker sands and gravels which were deposited by running water as the ice sheets began to melt and decay. In certain areas of the country these sands and gravels are present in sufficient thickness and purity to give rise to high yielding and important aquifers. Where these sands and gravels are present also gives rise to areas of high groundwater recharge due to their associated high permeability (Daly, 1985).

Glaciolacustrine deposits

Glaciolacustrine deposits consist of sorted sediments that were deposited in ancient lakes or stream beds and include gravel, sand, silt and clays. They are normally located in wide flat plains or in small depressions in the landscape given that they were deposited by melting glaciers (GSI, 1999). These can be of extremely low permeability, for example the Macamore marls in County Wexford.

Alluvium

Alluvium is a product of river flow and flooding and is usually of sand/silt grade but can sometimes contain some gravels and generally occurs in river floodplains.

Peat

Peat developed in post glacial times (following the most recent ice age) usually in lake basins and consists mainly of dead organic material which only partially decomposed and reflects excessively moist conditions. Extensive peat deposits in Ireland usually overlie badly drained glaciolacustrine silts and clays which enabled the extended saturated conditions required for their development.

The soils of Ireland have been mapped by Teagasc (formerly known as The Agricultural Institute / An Foras Talúntais) during the National Soil Survey (NSS) and have been described in detail in the accompanying explanatory bulletin (Gardiner and Radford, 1980). There are also a number of maps and explanatory documents for individual counties that were also produced by An Foras Talúntais. To understand how groundwater flow occurs it must first be determined how groundwater is replenished and sustained in the hydrological cycle as illustrated in Figure 2.3 below.

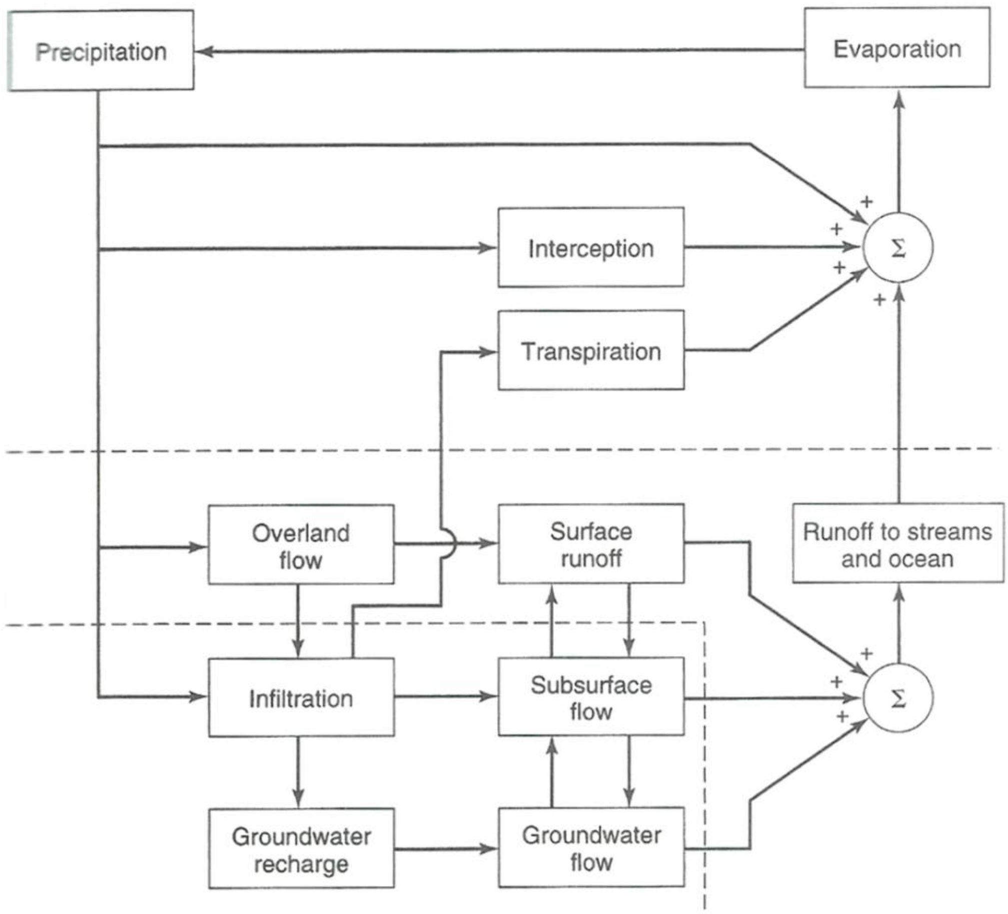


Figure 2.3 Block-diagram representation of the hydrological system (Todd and Mays, 2005)

Groundwater recharge is water that enters the groundwater system usually as infiltrating water from the unsaturated or vadose zone above. Typically groundwater recharge occurs naturally through infiltrating participation and stream, river and lake inflows; however recharge can occur through artificial means for example excess irrigation, seepage from man-made water courses and artificial recharge through boreholes (Todd and Mays, 2005). Misstear et al. (2009) produced a methodology aimed at making estimates of groundwater recharge based on the thickness and permeability of the subsoil overlying the bedrock aquifer. It is on this basis that recharge to groundwater is estimated in Ireland and given this, a national recharge map has been produced for the country by the Eastern River Basin District (ERBD) together with geologically-based recharge coefficients for the country (ERBD, 2007; Working group on groundwater, 2008). However given that 70% of the country is underlain by 'poorly productive' bedrock aquifers with low storage capacity and with many of these aquifers underlying the areas of the country that receive the greatest volumes of effective rainfall, much of the effect rainfall cannot be accepted by these aquifers and ends up in surface water bodies through runoff. In contrast areas of the country that are underlain by the highly productive aquifers tend to be dominated by low permeability subsoils and even though the bedrock aquifer has the ability to receive the recharge, it is limited by the poor infiltration properties of the soil (GSI, 2007). The nature of bedrock aquifer response to inputs and outputs in Ireland has been investigated by Tedd et al. (2011) and indicates that groundwater level minima or maxima timing estimates that had been used previously as a basis for developing understanding of recharge in Ireland were generally correct. The movement of groundwater in Ireland tends to be dominated by faulting in the region and therefore a general understanding of the geology of an area is needed to make estimates of groundwater flow and direction in bedrock aquifers (Daly et al. 1980).

2.2.3 Groundwater Protection

Research by Daly and Warren (1998) provided the basis for the development of a groundwater vulnerability protection scheme for the country which defines a vulnerability rating for an area based on the thickness and permeability of the overlying subsoil. The vulnerability classification for Irish hydrogeological conditions is given in Table 2.4 below.

Table 2.4 Groundwater Vulnerability Classification (DELG/EPA/GSI, 1999)

	Hydrogeological Conditions
	Subsoil Permeability and Thickness

Vulnerability Rating	High Permeability	Moderate Permeability	Low Permeability	Unsaturated Zone (Sand and Gravel Aquifers Only)	Karst Features (<30 m Radius)
Extreme	0 – 3.0 m	0 – 3.0 m	0 – 3.0 m	0 – 3.0 m	Yes
High	>3.0 m	3.0 – 10.0 m	3.0 – 5.0 m	>3.0 m	N/A
Moderate	N/A	>10. m	5.0 – 10 m	N/A	N/A
Low	N/A	N/A	>10.0 m	N/A	N/A

The basis for the vulnerability classifications given in Table 2.4 above were further investigated by Swartz et al. (2003) and were found to be satisfactory.

Given that groundwater is an important resource and must be protected from potential contamination, it was necessary for a further layer of protection in addition to the vulnerability classifications as outlined above. A groundwater protection response matrix was therefore developed which gives guidance as to whether a potentially polluting activity (namely the installation of an on-site wastewater treatment system) should be permitted based on the vulnerability classification combined with the importance of the aquifer as a resource (DELG/EPA/GSI, 1999). It must be noted that groundwater abstractions in Ireland such as wells or boreholes are allocated Source Protection areas that provide protection to that individual supply and these Source Protection areas are delineated based on a number of criteria as outlined in the Groundwater Protection Scheme document (DELG/EPA/GSI, 1999). The groundwater response matrix is a key element in the new EPA CoP for single houses (EPA, 2009) and the current groundwater response matrix is given in Table 2.5 below with the associated responses listed beneath.

Table 2.5 Response Matrix for on-site wastewater treatment systems (EPA, 2009)

Vulnerability Rating	Source protection area[a]	Resource protection area Aquifer category		
		Regionally important	Locally important	Poor aquifers

	Inner (SI)	Outer (SO)	Rk	Rf/Rg	Lk	Lm/Lg	Ll	Pl	Pu
Extreme (X and E)	R3²	R3¹	R2²	R2²	R2²	R2¹	R2¹	R2¹	R2¹
High (H)	R2⁴	R2³	R2¹	R1	R2¹	R1	R1	R1	R1
Moderate (M)	R2⁴	R2³	R1	R1	R1	R1	R1	R1	R1
Low (L)	R2⁴	R1	R1	R1	R1	R1	R1	R1	R1

R1 Acceptable subject to normal good practice (i.e. system selection, construction, operation and maintenance in accordance with the EPA CoP, 2009).

R2[1] Acceptable subject to normal good practice. Where domestic water supplies are located nearby, particular attention should be given to the depth of subsoil over bedrock such that the minimum depths required in Section 6 are met and that the likelihood of microbial pollution is minimised.

R2[2] Acceptable subject to normal good practice and the following additional condition:

1. There is a minimum thickness of 2 m unsaturated soil/subsoil beneath the invert of the percolation trench of a septic tank system

or

1. A secondary treatment system as described in Sections 8 and 9 is installed, with a minimum thickness of 0.3 m unsaturated soil/subsoil with P/Tvalues from 3 to 75 (in addition to the polishing filter which should be a minimum depth of 0.9 m), beneath the invert of the polishing filter (i.e. 1.2 m in total for a soil polishing filter).

R2[3] Acceptable subject to normal good practice, Condition 1 above and the following additional condition:

2. The authority should be satisfied that, on the evidence of the groundwater quality of the source and the number of existing houses, the accumulation of significant nitrate and/or microbiological contamination is unlikely.

R2[4] Acceptable subject to normal good practice, Conditions 1 and 2 above and the following additional condition:

3. No on-site treatment system should be located within 60 m of a public, group scheme or industrial water supply source.

R3[1] Not generally acceptable, unless: A septic tank system as described in Section 7 is installed with a minimum thickness of 2 m unsaturated soil/subsoil beneath the invert of the percolation trench (i.e. an increase of 0.8 m from the requirements in Section 6)

or

A secondary treatment system, as described in Sections 8 and 9, is installed, with a minimum thickness of 0.3 m unsaturated soil/subsoil with P/T-values from 3 to 75 (in addition to the polishing filter which should be a minimum depth of 0.9 m), beneath the invert of the polishing filter (i.e. 1.2 m in total for a soil polishing filter) and subject to the following conditions:

1. The authority should be satisfied that, on the evidence of the groundwater quality of the source and the number of existing houses, the accumulation of significant nitrate and/or microbiological contamination is unlikely
2. No on-site treatment system should be located within 60 m of a public, group scheme or industrial water supply source
3. A management and maintenance agreement is completed with the systems supplier.

R3[2] Not generally acceptable unless:

A secondary treatment system is installed, with a minimum thickness of 0.9 m unsaturated soil/subsoil with P/T-values from 3 to 75 (in addition to the polishing filter which should be a minimum depth of 0.9 m), beneath the invert of the polishing filter (i.e. 1.8 m in total for a soil polishing filter) and subject to the following conditions:
1. The authority should be satisfied that, on the evidence of the groundwater quality of the source and the number of existing houses, the accumulation of significant nitrate and/or microbiological contamination is unlikely
2. No on-site treatment system should be located within 60 m of a public, group scheme or industrial water supply source
3. A management and maintenance agreement is completed with the systems supplier.

The above groundwater response matrix provides protection for new developments where an OSWTS is proposed with the OSWTS being the risk source, however in Ireland many areas are at risk from intensive and poor agricultural practises and in that regard the response matrix does not protect against this potential source of contamination. The Nitrates Directive (91/676/EEC) was issued by the European Council concerning the protection of waters against pollution by nitrates from agricultural sources and was implemented in Irish law in 2009 and amended in 2010 (DELG, 2009; DELG, 2010). The Statutory Instrument (SI No. 610 of 2010) provides for the following in order to protect both groundwater and surface water (DELG, 2010):

- A site-specific, risk-based approach for setback distances from drinking water abstraction points;
- A prohibition on the application of chemical fertiliser within 2 metres of a watercourse;
- New controls on storage of baled silage;
- Amendments to the maximum nitrogen and phosphorus fertilisation rates for cereal crops including a measure to address the issue of low protein levels in malting barley;
- Time-limited extension for transitional arrangements covering the use of pig and poultry manure and spent mushroom compost;
- Revision of certain dates where the establishment of green cover is required.

The implementation of Nitrates Directive in Ireland is carried out by the Department of Agriculture, Fisheries and Food (DAFF) under the Farm Waste Management Scheme. In

2010, the DAFF carried out a certain amount of on-farm inspection on behalf of the Local Authorities in every county.

From an agricultural perspective, research has been carried out by Premrov et al. (2012) to investigate the effects of over-winter green cover with a view to providing protection to groundwater beneath tillage land. Holman et al. (2010) provided an assessment of the risk to surface water ecology from groundwater systems that contain elevated levels of phosphorus and found that surface water in Ireland has shown many cases of eutrophication caused almost completed by elevated phosphorus levels originating from groundwater sources. Recent research carried out by Hynds et al. (2012) demonstrated that groundwater abstractions such as residential wells and springs are at a much higher risk in low vulnerability areas. This is due to a combination of low permeability subsoils resulting in agricultural contamination at the ground surface and poor protection being provided to wells and springs (i.e. wellheads not being sealed with concrete and grout). This illustrates that even though the groundwater protection matrix and source protection areas may seem to provide protection, contaminates can enter groundwater in ways that may not be expected. Finally, Wilson and Rocha (2012) demonstrated that the number of aquifers in Ireland that discharge to the sea (submarine discharges) are much higher than previously thought and the coast must therefore also be thought of as a potential sensitive receptor.

2.2.3.1 Aquifers in Karst Environments

Limestone bedrock dominates Ireland and in many areas the presences of purer limestone has led to karstification with many areas scattered with karst features such as sinkholes, caves, and underground drainage systems (GSI, 2000). Karst conditions occur through the dissolution of a layer or layers of soluble bedrock, usually carbonate rock such as limestone. Aquifers in karst environments can be at a very high risk of contamination due to the presence of preferential flow paths and the very fast travel times that can exist unusually in conduit flow (GSI, 2000) as illustrated in Figure 2.4.

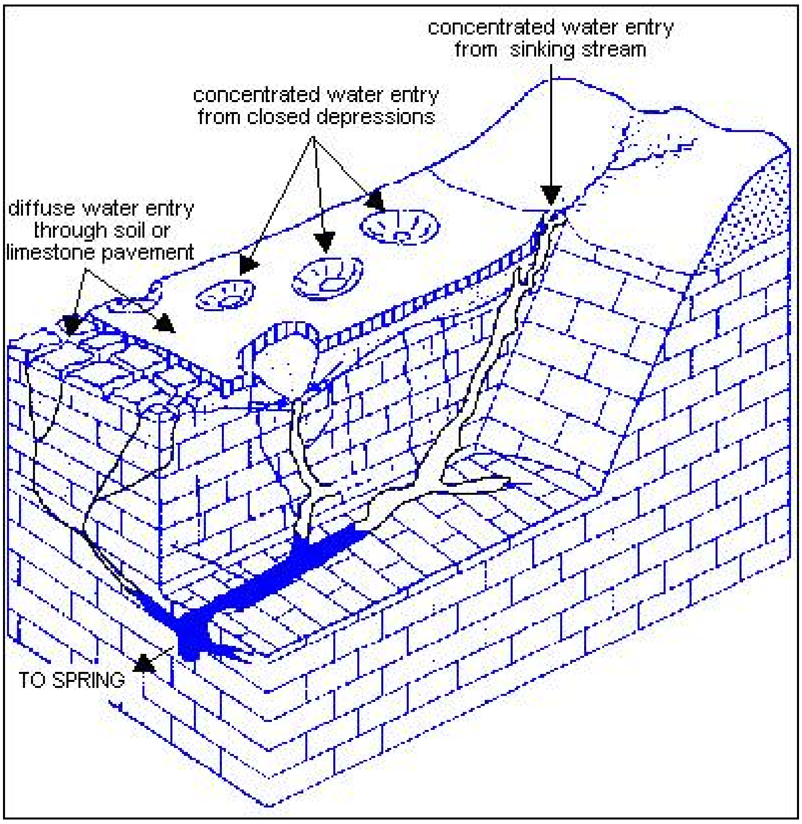


Figure 2.4 Possible entry routes for contaminants to enter karst groundwater (GSI, 2000)

In Ireland groundwater flow in karst environments is likely to occur in three main hydrogeological regimes (Deakin, 2000):

(1) an upper, shallow, highly karstified weathered zone, known as the epikarst, in which groundwater moves quickly, through solutionally enlarged conduits, in rapid response to recharge;
(2) a deeper zone, where groundwater flows through interconnected, solutionally enlarged conduits and cave systems which are controlled by structural deformation and bedrock lithologies. Groundwater flows along the less permeable, cherty units until it intersects a vertical fissure; and
(3) a more dispersed slow groundwater flow component in smaller fractures and joints outside the main conduit systems.

All three of these groundwater flow regimes will be hydraulically connected in places with the degree of interconnection depending on the presence of less permeable bedrock units and the faults and joints associated with the structural deformation (Deakin and Daly, 1999; Ford and Williams, 2007).

Drew (2008) also noted that the majority of lowland areas underlain by limestone are the principal sources of groundwater abstraction in Ireland and these areas also coincide with the most economically developed and intensively farmed areas of the country. Given the ability of nutrients and microbiological contaminants to travel very quickly within a karst bedrock aquifer these areas are therefore highly vulnerable to anthropogenic influences. A study by Kilroy and Coxon (2005) observed highly temporal variation in observed phosphorus concentrations in karst springs located in south-west Ireland over an extended monitoring period with spikes corresponding to high rainfall events illustrating the low travel times that can occur. The difficulty in identifying the full extent of a source protection area in a karst limestone bedrock aquifer is illustrated by Deakin et al. (2000) whilst developing the groundwater protection scheme for the Drumcliff Springs water supply in Ennis, Co. Clare. Research carried out by Landig et al. (2011) investigated the nitrate discharge loading from a dairy farm in southern Ireland to a fractured limestone aquifer via karstic springs to a local watercourse. Landig et al. (2011) developed a stream tube model in order to quantify the nitrate loading from the springs. The study found that 18 tons of nitrate per year were being discharged from groundwater to the nearby river, which equated to 54% of the current total agronomic nitrate load on the farm. Drew (2008) estimated that at least 50% of Ireland is underlain by limestone bedrock that are sufficiently pure to be karstified. Katz et al. (2010) investigated the fate of OSWTS contaminants in an area overlying a karts aquifer in Florida, USA. A number of chemical indicators arising from OSWTSs in the area were detected in the karst aquifer, however the study indicated that whilst movement of contaminants from the OSWTS percolation fields to groundwater there concentration and extent was highly related to water usage, subsoil lithology and meteorology at each site (Katz et al. 2010).

2.3 On-site Wastewater Treatment Systems

An On-site Wastewater Treatment System (OSWTS) provides treatment for individual dwellings typically in the form of a primary treatment tank with or without secondary treatment with discharge of the treated wastewater to the subsoil and unusually is contained within the boundaries of the development lot or site (Radcliffe and Šimůnek, 2010).

2.3.1 Septic Tank Treatment Systems

A septic tank acts as a primary settlement chamber removing the majority of settleable solids as well as grease and other floatable solids, usually resulting in the formation of a scum layer at the top of the liquid in the tank and a sludge layer at the bottom of the tank which builds up and must eventually be removed (Viraraghavan, 1976) as shown in Figure 2.5 below.. The main treatment process that occurs in a septic tank other than settlement of solids is anaerobic digestion whereby nutrients are converted from one form to another e.g. some of the organic nitrogen (organic-N) is converted to ammonium and phosphorus (P) is converted to ortho-P (phosphate). Microbiological contaminants are also reduced in this anaerobic environment. However, given the amount of treatment that occurs in a septic tank is limited and does not generally result in an overall reduction of nutrient loading (Gill et al. 2009), the majority of treatment that occurs in a septic tank treatment system happens in the subsoil or soil treatment unit (STU) beneath the percolation area or drain field (Siegrist and Boyle, 1987).

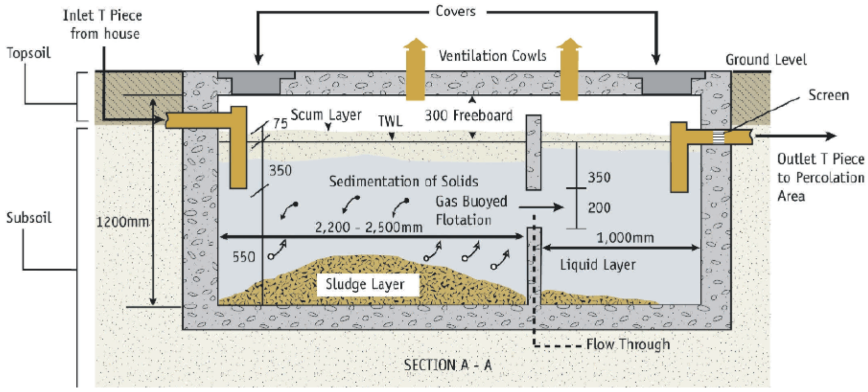


Figure 2.5 Cross-section through a typical septic tank (EPA, 2009)

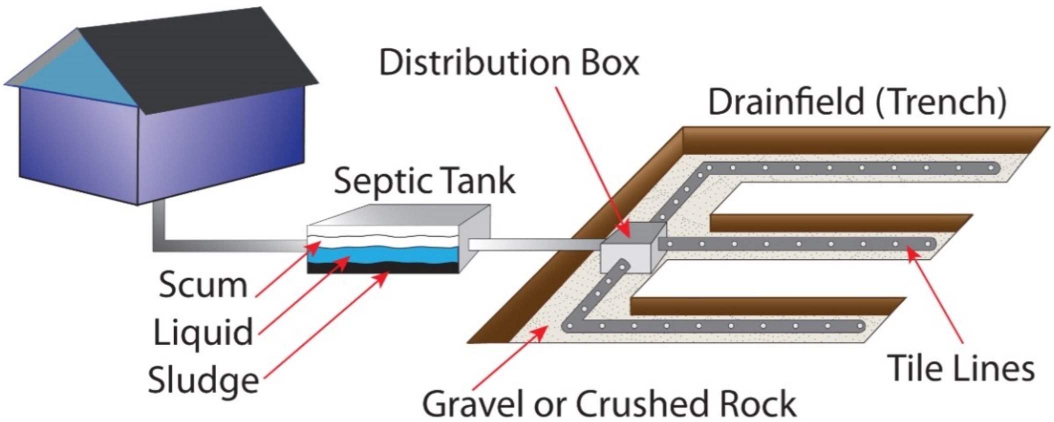


Figure 2.6 Typical OSWTS consisting of a septic tank with a percolation area

A typical layout of a septic tank and percolation area is shown in Figure 2.6 above. The treatment processes that occur in the subsoil include; physical straining, ion exchange, adsorption and attached-growth biological processes (Hazen and Sawyer, 2006) as described in detail in Section 2.3.3. Conventional septic tank treatment systems account for the largest proportion (~87%) of OSWTSs in Ireland (EPA, 2013) and also the recent trend has been a movement towards secondary or alternative treatment systems a study conducted by Gill et al. (2009) showed that when installed correctly in in the appropriate subsoil environment, conventional septic systems can perform as well, if not better than other systems.

2.3.2 Secondary and Tertiary Treatment Systems

Given that the majority of treatment that occurs in a septic treatment system happens in the subsoil there is an increasing tendency to incorporate secondary treatment in order to both reduce organic matter and the level of pathogens. This is usually achieved through the use of a package treatment plant. In addition to reducing organic matter and pathogens, most secondary package treatment plants also produce a partially nitrified effluent. This further level of treatment leads to what is generally considered a 'better quality' effluent for disposal in most cases to the subsoil as before. The main types of secondary treatment systems for small domestic applications are briefly described below.

BAF Systems

Biological Aerated Filter (BAF) treatment systems combine filtration with biological treatment. BAF systems usually consist of a reactor filled with a filter media. This media both supports highly active biomass that is attached to it and filters suspended solids. Typical systems can combine two holding tanks or reactors with one encouraging aerobic conditions promoting ammonia conversion with anoxic conditions in the other which can promote nitrate conversion. These systems tend to have large holding tanks that increase the detention time and allow for more treatment to occur before discharge to a soil treatment area.

RBC Systems

Rotating Biological Contactors (RBCs) are mechanical secondary treatment systems that incorporate rotating disks which support the growth of bacteria and micro-organisms on them. These bacteria and other micro-organisms can then break down and stabilize organic pollutants contained within the wastewater in an aerobic environment which is maintained by a rotating disk which continually brings them into contact with the oxygen in the atmosphere as the disk rotates. Again the treated wastewater is discharged to a soil treatment area once an adequate detention time has been achieved with further treatment occurring in the soil.

Sequencing Batch Reactors

A Sequencing Batch Reactor (SBR) usually combines at least two stages of treatment into one combined treatment cycle. Typically activated effluent is mixed with raw incoming effluent and aerated. The settled sludge is removed to a separate chamber and re-aerated before a proportion is returned to the first stage of treatment. SBR systems tend to require a precise control of timing, mixing and aeration. This required precision means that these systems should have high levels of maintenance and control and this is not always possible or desirable when being utilised in a domestic situation. As with other secondary treatment systems wastewater is discharged to a soil treatment area once an adequate detention time has been achieved.

Activated Sludge Systems

Activated sludge systems usually promote the growth of a biological floc that substantially removes organic material through the addition oxygen. These systems usually consist of two tanks with the first proving aeration to the effluent and then the second allowing the settlement of sludge before discharge to a soil treatment area.

Membrane Bioreactors

Membrane bioreactors (MBR) combine activated sludge treatment with a membrane liquid-solid separation process. The membrane component uses low pressure microfiltration membranes thus eliminates the need for a second sludge settlement stage which can be poor in conventional activated sludge systems.

Fixed-media Filter Systems

A media-filter system consists of a watertight chamber containing a permeable media that supports aerated secondary treatment (Van Geel and Parker, 2003). A pump is usually used to distribute the effluent across the top of the media and the effluent is then collected once it has trickled through the filtration media and can be recirculated if required. The filtered media is then discharged to a soil treatment area. The type of media used in these systems varies but usually consists of sand, peat, foam, or textile with peat being the most popular filter media in Ireland.

Constructed Wetlands

Wetland systems are relatively inexpensive to construct and do not require as costly maintenance as other mechanical secondary treatment systems and have therefore become more popular in Ireland over the past number of years. In Ireland, wetland systems can be used either as a form of secondary treatment (receiving primary effluent) or as a form of tertiary treatment (receiving secondary effluent). The effluent first enters a holding tank and is then allowed into the wetland by gravity or pumping on a timed or dosed regime.

The wetland is constructed using a porous media on which vegetation grows typically reed grasses. There are three main types of constructed wetland systems; free water surface wetlands (FWS), subsurface horizontal flow systems (most common), subsurface vertical flow and also hybrid systems (combinations of some or all of the others). Free water surface wetland systems usually have to be secured with adequate fencing as the effluent is above the porous media and can allow direct contact by humans or animals. These systems have been utilised with limited success in Ireland due to their high dependency on atmospheric conditions (O'Luanaigh et al., 2007).

Tertiary Treatment

Additional treatment to wastewater from secondary treatment systems is defined as tertiary treatment and includes polishing filters, constructed wetlands and packaged tertiary treatment systems. Polishing filters, which typically consist of soil or sand, reduce the number of micro-organisms present in the treated wastewater before discharging to groundwater via the subsoil. Other tertiary treatment systems have the added benefit of further reducing nutrients and micro-organisms and can include constructed wetlands or packaged tertiary treatment systems such as read beds, fine media filters and UV disinfection systems (EPA, 2009).

2.3.3 Treatment in the Subsoil

For nearly all of the treatment systems described above, the method of disposing treated wastewater is to groundwater via the subsoil through soil infiltration systems which are referred to as percolation areas in Ireland, although the preferred term is now Soil Treatment Unit (STU). The unsaturated subsoil that overlies the groundwater table is therefore a critical element in protection against groundwater contamination. Many OSWTS's rely on the ability of the subsoil to further treat the wastewater and in particular to remove the majority of pathogenic bacteria, infectious viruses and protozoa. EPA (2006) noted that the degree of microbial contamination of groundwater in Ireland is very high with up to 30% of groundwater supplies polluted by faecal bacteria. In addition nutrients that can be highly mobile once they reach groundwater have been observed at elevated levels in the south of Ireland (EPA, 2006). Given that the subsoil acts as the final treatment medium for wastewater from OSWTS, it is therefore very important that an understanding of the various treatment and attenuation processes that occur within the subsoil be developed. A recent study by Hynds et al. (2012) investigated private well contaminant from faecal bacteria and the results indicated that contaminant can occur in low vulnerability just as often as in extreme vulnerability areas.

2.3.3.1 Subsoil Suitability

In order for a subsoil to be used as a means for both treatment and disposal of OSWTS wastewater, the hydraulic assimilation capacity or permeability of the subsoil surrounding the base of the percolation trench under saturated conditions must first be accessed. The subsoil must therefore be tested for its permeability or hydraulic conductivity to determine if the Long Term Acceptance Rate (LTAR) is suitable. LTAR (L m^{-2} d^{-1}) is defined as "the amount of pre-treated effluent which the system can infiltrate during its lifetime without water logging or clogging", and is very much based on the measured saturated hydraulic conductivity of the soil (Beal et al., 2006). In Ireland a percolation test is used to ascertain the suitability of a subsoil to receive wastewater effluent (EPA, 2009). This test takes the form of a falling head test which is conducted in-situ and calculates the average time for water to drop 25mm at a depth of 400mm below the invert level of the proposed percolation system pipes. This value is called the "T" value and is measured in units of minutes/25mm leading to the test being referred to as the "T-test". A site is deemed as being acceptable for the installation of a septic tank if the T value is less than 50 or for the installation of a secondary treatment system if the T value is less than 75. If the T value is not within these ranges, a high water table is present at the site or if bedrock outcrops are present an alternative test can be carried out at ground level. This test is essentially the same as the "T-test" in its procedure however it leads to a "P" value for the soil and is therefore referred to as the "P-test". If a soil is deemed suitable based upon the results of a "P-test", it usually leads to the installation of a raised or mounded percolation area.

2.3.3.2 Biomat Development

When wastewater infiltrates into the subsoil via percolation trenches, a clogging layer referred to as biomat develops over time and tends to limit the LTAR due to its low permeability and therefore reduces the overall loading rates that a percolation area can receive (see Figure 2.7) (McKinley and Siegrist, 2010; Mckinley and Siegrist, 2011). Siegrist and Boyle (1987) described the biomat as a "heterogeneous layer comprised of; accumulated suspended solids and organic matter contained in the effluent, a large number of microorganisms and their metabolites and by-products (e.g. extracellular polysaccharides)". Whilst it is recognised that the biomat is a critical part of the soil treatment process that occurs within a percolation area, from a design perspective the poor hydraulic properties of the biomat are of most importance (Kristiansen, 1981). The hydraulic conductivity of the biomat layer has been calculated by Bouma (1975) as approximately 0.6 mm/day for clay soils and 2 mm/day for sandy soils. Beal et al. (2005) have questioned the validity of the commonly used approach which bases the LTAR and percolation trench design on the initial saturated hydraulic conductivity (K_{sx}) of the in-situ soil (the approach used in Ireland), as their research concluded that LTAR was governed by the resistance of the biomat and the sub-biomat soil unsaturated flow regime induced by the biomat and not the existing Ks value of the soil.

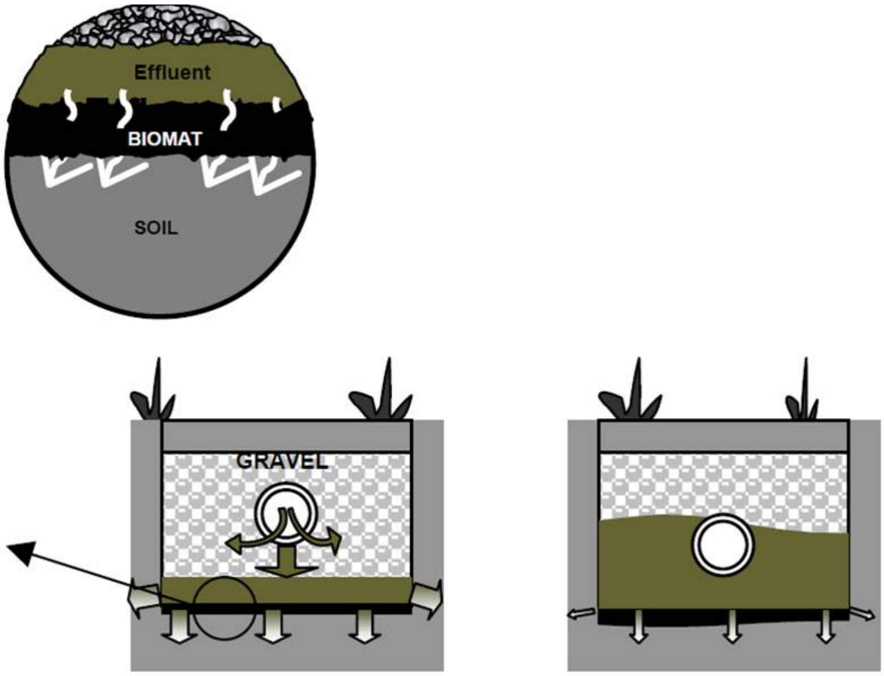


Figure 2.7 Development of the Biomat zone (Beal et al., 2004)

There are a number of factors that affect the development of a biomat layer including; the hydraulic loading rate, the dosing regime, the aeration status of the infiltrative surface and the soil biogeochemical properties (Siegrist and Boyle, 1987; Beal et al., 2006; McKinley and Siegrist, 2011). Beal et al. (2005) has described three phases of biomat development based on previous research carried out by Siegrist and Boyle (1987). Phase one consists of the initial physical clogging of the pores in the infiltrative surface of the in-situ soil and can result in markedly reduced infiltration rates over the first few months after installation. Following this initial phase a period of gradually decreasing infiltration rates takes place with this phase dominated by anaerobic biological activity (Beal et al., 2006). The final phase sees a state of equilibrium being reached, usually with low infiltration (Siegrist and Boyle 1987). The extent and rate of biomat development is also related to the composition of the wastewater (McKinley and Siegrist, 2010). Where subsoil receives a highly treated effluent such as that discharged by a secondary treatment system, the development of a biomat will be significantly retarded due to the low organic content and consequently hydraulic loading rates in percolation areas receiving secondary treated effluent can be higher than those receiving septic tank effluent (Gill et al., 2009).

2.3.3.3 Factors Affecting Subsoil Contaminant Removal Efficiencies

As discussed earlier the permeability of the subsoil is highly important in assessing the LTAR of the soil, however the hydraulic conductivity of the soil will also determine the flow rate of the infiltrating wastewater which in turn determines the contact time between the wastewater and the soil particles and/or biofilms. Maintaining unsaturated conditions in the subsoil between the base of the percolation trenches and the water table is essential as many of the treatment processes that occur in the subsoil are dependent on aerobic conditions and therefore by controlling the hydraulic loading rate these aerobic conditions can be maintained (Beal et al., 2005). The longer the residence time of the wastewater in the unsaturated zone results in more effective the removal rates of pathogens and chemicals.

Apart from the hydraulic conductivity controlling the percolating water's residence time in the subsoil, the main physical treatment process that occurs in the unsaturated zone is filtration. McDowell-Boyer et al. (1986) summarised the three main filtration methods that occur in the soil as surface filtration, straining and physic-chemical filtration. Surface filtration is the main process that results in the formation of the biomat layer as it occurs when particles are too large to penetrate the soil. Straining has been found to be an effective mechanism for filtering wastewater however its effectiveness is related to the grain size of the soil (Siegrist et al., 2000). Canter and Knox (1985) state that filtering begins when larger suspended particles become trapped either at the soil surface or at some depth. As the percolating water moves through the soil individual particles may be blocked in the pore spaces between the soil grains and sometimes several particles may interact to form a bridge in a pore that prevents further movement of these particles in the direction of flow. Canter and Knox (1985) also note that once the movement of larger suspended particles has been blocked, these particles themselves begin to function as a filter and trap successively smaller suspended particles – this process is illustrated in Figure 2.8.

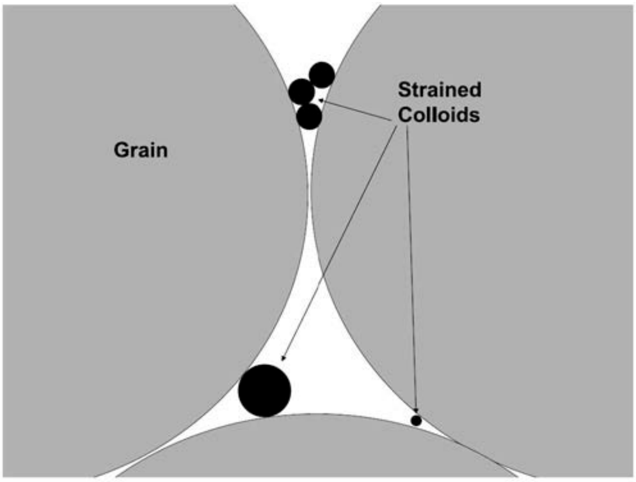


Figure 2.8 Illustration of strained colloids in the smallest regions of the soil pore space formed adjacent to points of grain-grain contact (Bradford et al., 2006)

2.3.4 Fate of Key Contaminants in the Subsoil

Organics and Suspended Solids

The process of organic matter decomposition in the soil has been described in detail by Swift et al. (1979). Micro-organisms present in the subsoil use oxygen as their terminal electron acceptor to convert the organic molecules in the percolating water to carbon dioxide, water and energy – usually heat. This process of microbial decomposition is dependent on both heat and the availability of oxygen (Kätterer et al., 1998) with oxygen levels usually the limiting factor as these micro-organisms thrive in aerobic conditions. Oxygen supply in the vadose zone occurs by diffusion in the soil atmosphere and therefore the oxygen supply can become limited. Under these conditions anaerobic organisms such as methanogenic bacteria become more dominant. Methanogenic bacteria break down insoluble organic compounds to carbon dioxide and methane in the absence of oxygen with only limited bacterial growth. Gray (2004) described the process of organic decomposition as occurring in two stages; the first being non-methanogenic in aerobic conditions and the second being methanogenic under anaerobic conditions. The two main processes that occur in the methanogenic stage are hydrolosis and acidogenesis. The process of biological organic matter decomposition has been summarised in Figure 2.9 below.

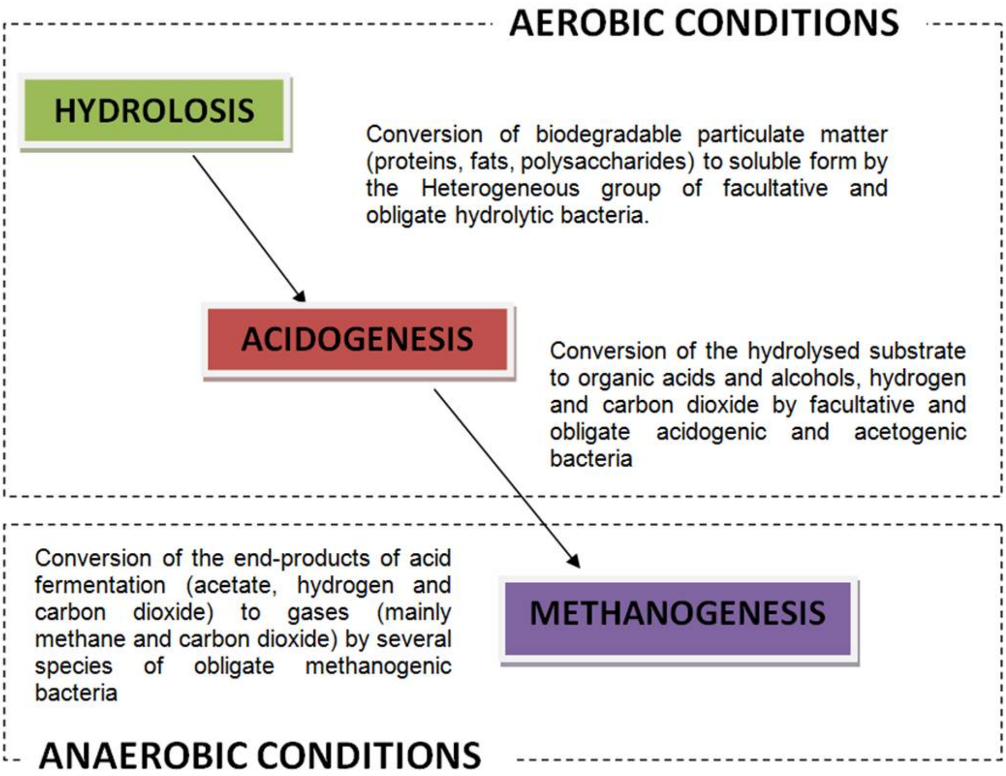


Figure 2.9 Biological organic matter decomposition processes in the subsoil (modified from O'Luanaigh, 2009)

Nitrogen

Wastewater from OSWTS's contains nutrients (nitrogen and phosphorus) with nitrogen being of most concern for groundwater resource protection. Nitrogen content in septic tank effluent that percolates into the subsoil beneath a percolation area typically is in the form of ammonium (NH_4^+) with some organic nitrogen present also. Nitrogen present in the percolating water will typically convert to nitrite (NO_2^-) and nitrate (NO_3^-) soon after entering the subsoil (Beal et al., 2005) by the processes of nitrification. Bouma (1979) described the process of nitrification as an aerobic reaction performed primarily by obligate autotrophic organisms with NO_3^- being the main end product. The process occurs in two steps as shown in [Eq. 2.1].

$$NH_4^+ \longrightarrow NO_2^- \longrightarrow NO_3^-$$

[Eq. 2.1]

Nitrosomonas bacteria, which are a specific group of autotrophic bacteria, first transfer the NH_4^+ into NO_2^-. This reaction is immediately followed by the transformation of NO_2^- to NO_3^-

by autotrophic bacteria called Nitrobacter and Nitrospina – see [Eq. 2.2]. This reaction happens so quickly after the first that it is often omitted when written in the literature.

Canter and Knox (1985) note that denitrification is the only process by which concentrations of NO_3^- in the percolating water can be decreased. The process of denitrification is performed mainly by ubiquitous facultative heterotrophs which convert NO_3^- to either nitrous oxide gas (N_2O), nitric oxide gas (NO) or nitrogen gas (N_2). The process occurs where oxygen, a more energetically favourable electron acceptor, is depleted, and bacteria respire nitrate as a substitute terminal electron acceptor. This process will only occur if favourable conditions for the denitrifying bacteria occur which are typically anaerobic with a supply of readily available carbon in the form of organic substrate. Other conditions such as pH, temperature, degree of oxygen depletion and NO_2^- and NO_3^- concentrations present dictate the proportion of which nitrogen gases are produced.

$$2\ NH_3 + 3\ O_2 \rightarrow 2\ NO_2- + 2\ H_2O + 2\ H^+ \text{ (Nitrosomonas)}$$
$$2\ NO_2^- + 1\ O_2 \rightarrow 2\ NO_3^- \text{ (Nitrobacter, Nitrospina)}$$
$$NO_3^- \rightarrow NO_2^- \rightarrow NO + N_2O \rightarrow N_2 \text{ (g)}$$

[Eq. 2.2]

Phosphorus/Bacteria

Bear and Cheng (2010) have described the processes of chemical treatment in the soil as comprising of adsorption, ion exchange and precipitation and this is the main process through which both phosphorus and bacteria are removed in the subsoil. Adsorption is a process whereby a substance accumulates at a solid-liquid interface. O'Luanaigh (2009) noted that adsorption is the key factor in the removal of phosphates, ammonium, organic compounds, bacteria and viruses from OSWTS effluent in the subsoil Adsorption is most effective in soils containing finer particles, specifically clays, due to the smaller and more angular grain sizes as well as the greater surface area available providing ideal sorption sites. Iron (Fe), aluminium (Al) and hydrous oxides that coat clay minerals as well as the weathered surfaces of ferromagnesium minerals also promote sorption (Miller and Wolf, 1975 citied in O'Luanaigh, 2009). Adsorption occurs due to the differential forces of attraction or repulsion occurring among molecules or ions of different phases at their exposed surface. Adsorption may also occur due to cation exchange taking place in the soil with this process again tending to occur predominantly in clay soils due to the presence of colloids (Fetter, 1993). Clay particles tend to hold electrostatic surface charges which attract ions that are contained within the percolating water and "hold" them due to their opposite charge and this process is called cation exchange (Bouwer, 1984). During this process a cation is released from the clay particle in "exchange" to make room for the more energetically favourable cation in the epercolating water.

The term adsorption mainly applies to organic compounds while ion exchange applies to inorganic compounds however the process by which they occur is essentially the same. Desorption can also occur for a number of reasons including high rainfall events and the changing constituents of the percolating water and therefore this process can be "reversible" and this has been shown in studies involving bacteria and viruses and has been reviewed in detail by Schijven and Hassanizadeh (2000).

Precipitation is the process through which an insoluble solid is formed when two solutions are mixed and can occur in the soil when ions in the percolating water react with compounds either sorped onto the soil particles or dissolved in the percolating water. Typically wastewater from OSWTS's contains high levels of dissolved phosphorus in the form of soluble orthophosphate (PO_4^{3-}). The process through which orthophosphate precipitates out in the unsaturated soil is known as phosphate fixation and is influenced by the quantity and type of cations present and the pH of the soil. Phosphate fixation occurs most efficiently in soils that contain calcium which occur in alkaline areas.

2.3.5 Treatment Systems for Low Permeability Subsoils

Ireland has significant portions of its landscape covered in low permeability (clayey) subsoils which make discharge of effluent from more conventional on-site systems problematic. Hence, in recent years some emerging treatment technologies have been trialled to assess their suitability as follows.

Zero Discharge Willow Constructed Wetland Systems

Research carried out in the past number of years has identified a method for the disposal of wastewater from single households which consist of a novel constructed wetland system that has zero discharge and is very suitable for areas where soil infiltration is not practicable due to the presence of very poor permeability subsoils (Gregersen and Brix, 2001). These systems consist of primary settlement tanks that then disperse the wastewater underground beneath a sealed 'treatment basin' that is planted with willows. Willow 'wastewater cleaning facilities' have zero discharges with the willows evapo-transpiring the water and all nutrients being recycled via the willow biomass. A study is currently being undertaken at Trinity

College Dublin to access their suitability in the Irish climate as much of the previous research carried out using these systems has taken place in Denmark, however initial indications are positive as to their use here as a disposal option. The use of these systems can sometimes be impractical due to the treatment basin area that is required and also given the recent emergence of these systems as a wastewater disposal option there are still some doubts over the long term issue of accumulating salts within the treatment basin (Gregersen and Brix, 2001).

Drip (Trickle) Irrigation

'Drip irrigation' OSWTS's have been used extensively in the USA for the past 30 years however their use in Ireland is not prevalent. A project is underway in Trinity College Dublin with full-scale trials of drip irrigation and LPP (see below) funded by EPA to determine their use in the Irish context. Drip irrigation systems were developed as an alternative to sand evapotranspiration (ET) systems in clay soils with poor percolation characteristics (Church, 1997). Subsurface drip irrigation systems (SDIS) provide uniform dispersal of effluent over the entire soil treatment area and are very successful in enabling wastewater disposal to soils of low permeability. The effectiveness of SDIS has been summarised into four key areas by Hassan et al. (2008):

- Shallow application, enabling effluent to be placed at maximum vertical distance above unsuitable soil horizons or wetness conditions while keeping effluent from being exposed at the ground surface;
- Injection of effluent from emitters at slow rates, which allow for plant water uptake
- Evaporation without the need for temporary storage in a trench or absorption through a trench/soil interface
- The potential to maximise nutrient attenuation by placing the effluent in the most biologically active soil/root zone.

Campos et al. (2000) noted that when receiving secondary treated domestic wastewater the survival of microorganisms was limited and thus the treatment by SDIS was quite effective. Zona et al. (2006) concluded that SDIS offers an alternative when conventional groundwater discharge of effluent is infeasible however a highly treated effluent is required suggesting that secondary and tertiary treatment be included when using this type of disposal system. Cararo et al. (2006) identified that the clogging of the emitters in a SDIS system can be a major issue in the long term and research they carried out concluded that there is no easy way to 'unclog' the system once it has occurred.

Low Pressure Pumped distribution systems

Low pressure pumped distribution systems (LPP) function similarly to a conventional OSWTS drainfield however rather than relying upon gravity, LPP systems are designed to assure that effluent is distributed evenly to all areas of the soil absorption field by pumping. The wastewater is fed from a pump chamber into distribution lines which are installed at a shallow depth in soil and this occurs usually on a timed-dose basis (Stewart and Reneau, 1988). Miles (2007) observed that LPP systems were effective at removing fecal coliform numbers within 60 cm of the infiltrative surface and a study by Hagedom and Reneau (1994) also confirmed satisfactory denitrification rates in LDP systems, however their success was highly dependent on an appropriate loading rate with excessive loading rates (>15 $Lm^{-2}d^{-1}$) leading to almost immediate failure.

2.3.6 Review of Regulatory Guidance for On-site Systems

Prior to 1975 there does not appear to have been any specific regulatory guidance on the suitability of sites or installation standards for the use of OSWTSs in Ireland. Local authorities followed 'best practice'; however it is widely evident that the predominant construction practice was to route grey-water to soakpits with black-water routed to a septic tank followed by a soakpit. The Septic Tank Effluent (STE) was then usually discharged to a soakpit or sometimes to a percolation area that might contain one or more infiltration trenches. In 1975 the National Standards Authority of Ireland (NSAI) produced the S.R.6. (1975) document which was used as the regulatory document for the installation of OSWTS's until 1991 (NSAI, 1975). S.R.6. (1975) "Recommendations for septic tank drainage systems suitable for single houses" contained guidance on site suitability, design considerations and construction and maintenance of septic tank treatment systems. The site suitability test involved a falling head test that established a "T" value for the soil which was the time taken for a 25mm drop of water in the test hole. A "T" value of greater than 60 was deemed to have failed. The percolation area for the septic tank was sized based on the T value for the soil up to a maximum length of distribution piping of 105m for a soil with a T value of 60. Septic tank capacity was sized based on a minimum population equivalent (PE) of 4 with a minimum capacity of 2,720 litres. A minimum distance for the locating of the percolation area from wells and groundwater abstractions was not specified; however a recommendation was made to try to location the percolation area as far away as possible and preferably down gradient.

In 1991 the NSAI revised the S.R.6. document with a number of amendments aimed at improving groundwater quality and reducing the likelihood of systems being installed in unsuitable locations (NSAI, 1991). The main changes that occurred in the 1991 edition of S.R.6. are summarised below:

- More severe site suitability test requirements
- The introduction of a site assessment
- The introduction of minimum distances between percolation areas and groundwater sources
- The inclusion of recommendations for site improvements at sites that have failed the suitability test

The introduction of a site assessment was of major importance as the document now advised that sites may not be suitable for the installation of a OSWTS at all which was not position adopted in the past. The idea of the site assessment was to first identify if a potential development site was suitable for the installation of an OSWTS and only then to proceed further with site suitability tests and a planning application. The suitability assessment consisted of a visual inspection noting the presence of vegetation indicative of wet conditions and also the location of nearby watercourses and water abstractions. An inspection of the trial hole was also required focused on identifying key indicators such as: colour, presence of iron pans, depth to bedrock and soil texture. The site suitability test again consisted of a falling head test leading to a “T” value for the in-situ soil. The percolation area for the septic tank was again sized based on the T value for the soil however T values of less than 5 and greater than 60 were deemed to have failed the test. T values of less than 5 were deemed to indicate that the percolation rate was too fast and could lead to groundwater pollution. A note contained within the document also cautioned against the risk of ponding of effluent at sites that had a T value of between 30 and 60. The required length of piping for the percolation area was somewhat reduced from the previous edition of 1975.

Following the introduction of the EPA in 1993 a new guidance document “Wastewater Treatment Manuals: Treatment Systems for Single Houses” was published in 2000 (EPA, 2000). This document provided detailed guidance on; the assessment of a site with regard to its suitability, the selection of an appropriate treatment option and acceptable treatment system designs and layouts. All proposed OSWTS installations now had to include a Site

Characterisation Form during the planning process which was more in-depth and thorough than the previous site suitability assessment contained within the S.R.6. (1991) document. The EPA (2000) also contained the provision for many new treatment and disposal options including:

- Raised Percolation Areas
- Soil/Sand/Peat and other Intermediate Filter Systems
- Mechanical Secondary Treatment Systems
- Constructed Wetlands

The sizing of the percolation area was now based upon the hydraulic loading rate (i.e. the occupancy of the dwelling) and not the T value of the soil and coupled with this change the required length of trench for infiltration purposes was reduced for the typical occupancy of 4 persons but the maximum length was greatly increased relative to increasing occupancy of the dwelling up to 10 persons.

Following research by Gill et al. (2005) and Gill et al. (2009), a new Code of Practice (CoP) for single house on-site wastewater treatment was introduced in 2009 by the EPA (EPA, 2009). The development of this CoP has been described in detail by Gill (2011) and was based on two large research projects carried out at Trinity College Dublin on behalf of the EPA. The main changes in the new CoP (EPA, 2009) from the previous EPA (2000) document are:

- On-site hydraulic loading rate reduced from 180 L per capita per day down to 150 L per capita per day.
- The range of acceptable subsoils receiving septic tank effluent has narrowed for more highly permeable subsoils
- The range of acceptable subsoils receiving secondary treated effluent has been extended for lower permeability subsoils
- The maximum individual length of percolation trenches receiving secondary effluent has been reduced to 10 m with a trench width of 0.5 m

Most of the guidance documents detailed above contained some provision for maintenance of either the OSWTS or the percolation area or both however there was no national policy for the inspection of OSWTSs in order to ensure that they have not been altered or are not causing pollution to either surface or groundwater. However, following a ruling by the European Court against Ireland, the EPA has introduced a national inspection plan which will begin in late 2013 (EPA, 2013).

2.3.7 Density of On-site Treatment Systems

2.3.7.1 International and Domestic Research

The latest census of Ireland indicates that there are 487,911 decentralised OSWTS's in the country (CSO, 2011). This figure has increased from the previous census of 2006 (447,718) which in turn increased from figures available for 2002 (416,716) (CSO, 2011/2006/2002). In Ireland census data is collected at the level of electoral divisions, whereby the country is broken up into small area populations for the purposes of census data analysis and for voting purposes. There are nearly 3,500 of these electoral divisions (ED) and data is available for the last three censuses on the number of OSWTS at this level. Using ArcGIS; a software application for the management of spatial GIS data (discussed later), it is therefore possible to combining the data available on the number of OSWTS per electorate district with shape-files of the ED boundaries to graphically represent the densities distribution of OSWTS in Ireland and this is shown in Figure 2.10.

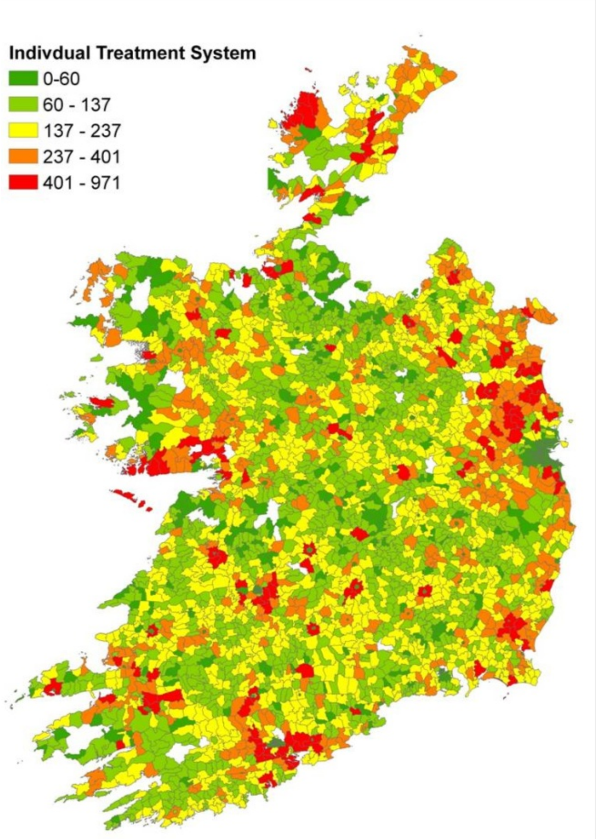


Figure 2.10 Distribution of OSWTS by Electoral Division given by 2011 Census Data

However given that the associated plot size and the local density of these systems vary hugely across the country within electoral divisions, this data is only useful for indicating a national trend.

A limited quantity of research appears to have been carried previously specifically to investigate the impacts of the density of OSWTS on groundwater quality, particularly in the Irish context. The EPA CoP (2009) does not recommend a desirable density for OSWTS, however the groundwater protection scheme (DELG/EPA/GSI, 1999) does have one condition which states "*The authority should be satisfied that, on the evidence of the groundwater quality of the source and the number of existing houses, the accumulation of significant nitrate and/or microbiological contamination is unlikely*". This condition would appear to indicate that a maximum density of OSWTS in an area may exist that if exceeded would lead to groundwater contamination. Gill et al. (2009) estimated empirically that an upper limit of one house every third of a hectare might be appropriate with respect to the density of on-site systems in an area to meet the requirements of the EU Nitrates legislation, however this was not based on research in the area and was only an approximation based upon expected hydraulic loading rates. McCarthy et al. (2010) investigated the impacts of existing OSWTS on surface water quality in a Co. Monaghan catchment. The focus of this study was on shallow groundwater with shallow water-tables present at most of the study sites and relatively fast travel times from the OSWTS outlets to surface water bodies due to the combination of low permeability soils and preferential flow pathways due to the properties of the till subsoil present. This study also noted that some of the OSWTS installed incorporated very poor construction practices with direct discharges of effluent water to drains and surface water bodies. In this regard whilst the study did make reference to OSWTS and their impacts on very shallow groundwater, the receptor of interest was surface water and not productive groundwater aquifers. Bailey et al. (2011) investigated the spatial and temporal effects of intensive dairy agricultural practices on groundwater quality at a large farm in Co. Wexford. Whilst this study was not concerned with OSWTS, the results must be considered as most areas of the country that have high densities of OSWTS are located in areas of intensive agriculture and the study did show nutrients reaching the water table due to the loading at the ground surface.

From an international perspective, Yates (1985) discussed the issue of septic tank density and groundwater contamination in the American context reporting that the USEPA has designated areas with septic tank densities of 1 or more systems per 16 acres (6.48 hectares) as regions of potential groundwater contamination; however the minimum lot size required for the inclusion of a septic tank was c.0.47acres (0.19 hectares). The associated desk study identified incidences of increased nitrate concentrations, with some over the 45mg-N/l standard (the drinking water standard at the time of writing), in Colorado, Delaware, Massachusetts, New Mexico, New York and North Carolina. Yates (1985) concluded that a minimum lot size was required to minimise the potential for groundwater contamination due to septic tanks (OSWTS). Yates (1985) also noted that even though some states did already have minimum lot sizes, they may not be appropriate given the specific hydrogeological conditions and that any lot size guidance should take into account the expected site specific conditions. Pang et al (2006) modelled the impact of clustered septic tank systems on groundwater quality for a study site in Christchurch, New Zealand based on field studies that had previously been carried out in 1977 by Sinton (1982) and in 1986 by Close et al. (1986). Both of these studies had monitored down-gradient groundwater quality and elevated levels of nitrates were found. The Close et al. (1986) study had demonstrated a clear trend of increasing nitrate concentrations down-gradient with increasing up-gradient numbers of septic tank systems. However, neither study found elevated levels of fecal coliforms down-gradient of the study site. Pang et al. (2006) built upon these studies and utilised the associated water quality data to develop a numerical simulation using HYDRUS-2D calibrated against the field data which predicted increases in the down-gradient groundwater concentrations of nitrates (which were below the WHO drinking water guideline of 50mg-N/l). It is important to note however that the septic tank systems that were the subject of these studies discharged wastewater to disposal boulder pits and not percolation fields. In addition these disposal pits were located 4m below the ground surface in an alluvial gravel media which was estimated to have a hydraulic conductivity of 600 m/d.

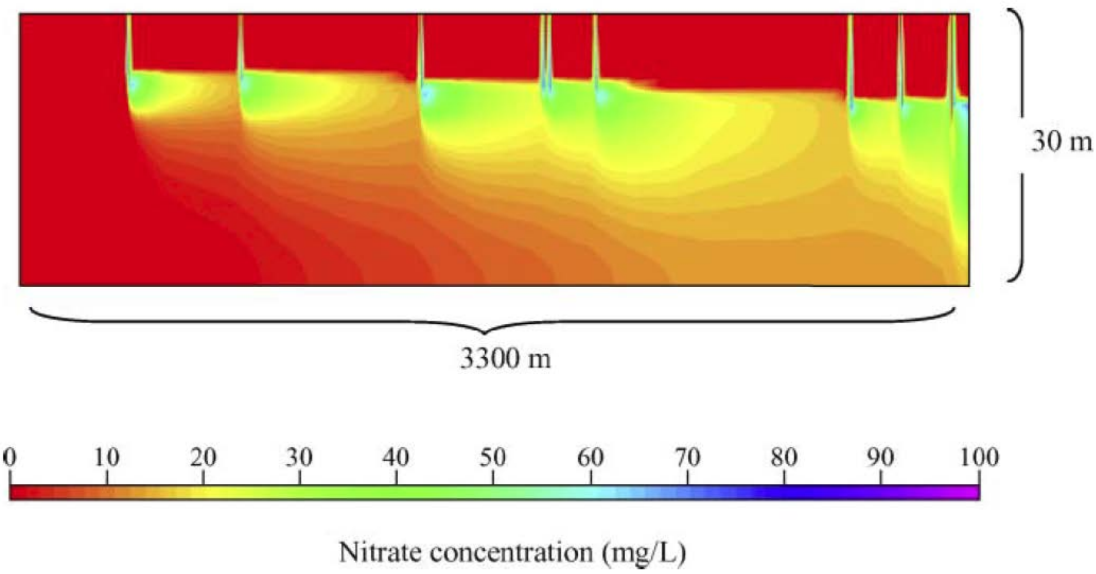


Figure 2.11 Hydrus-2D simulated NO3 plumes developed under the impact of clustered septic tank systems (results at 1000 d) (Pang et al., 2006)

Given that virtually no unsaturated subsoil treatment took place and the corresponding high permeability of the aquifer into which the wastewater was being disposed, it is not surprising that this study area was leading to elevated groundwater nitrate levels as shown in Figure 2.11 above. This situation therefore is quite site specific and not particularly transferable to the Irish context where, even though soak pits have been used in the past for wastewater disposal from OSWTS, a typical OSWTS would incorporate at least some treatment in the subsoil and high permeability aquifers such as the gravel mentioned in this study are very uncommon. Siegrist et al. (2005) investigated the cumulative effects of multiple OSWTSs on water quality within a watershed by incorporating the associated loading into an existing watershed management model. Whilst the study will aid in decision making within similar catchments, one of the major findings was that many gaps in understanding of how OSWTS interact within a watershed or catchment still exist which further research should attempt to fill.

Whilst compiling a number of key parameters for modeling nitrogen transport in the vadose zone, McCray et al. (2005) concluded that whilst dilution may play a part in reducing nitrogen concentrations in groundwater, mixing is not always efficient and plumes of higher concentrations of nitrogen will exist. McCray et al. (2005) based this opinion on studies that had been carried out previously where well-defined nitrate plumes in groundwater were observed originating from wastewater sources, indicating very low transverse and longitudinal dispersivity of nitrogen in groundwater. Gold et al. (1999) concluded that the dilution capacity of groundwater for nitrogen is limited and becomes significantly reduced with increasing density and spatial extent of unsewered development – the view echoed by McCray et al. (2005). Given that dilution is not seen as a long term solution for nitrogen discharges to groundwater, research has been carried out in order to access the existing and potential impacts of OSWTS on groundwater at specific study sites. Chen et al. (2001) examined this issue in the Dillion Lake watershed in Colorado by incorporating discharges from dense OSWTSs into an existing Watershed Analysis Risk Management Framework (WARMF) model in order to take account of the potential for nutrients arising from OSWTS migrating to tributaries of the Lake and causing contaminant. The model had not previously considered loading from OSWTS and it was necessary to account for this associated loading for future planning. The specific focus of this groundwater contaminant research was therefore the surface water receptor with respect to nutrients only in areas where shallow groundwater combined with steep gradients become contaminated and seep out causing surface water contamination (Chen et al., 2001). This study did however simulate significantly higher concentrations of nitrate (up to 19%) in one of the tributaries of Lake Dillion owing to OSWTS densities, with phosphorus loading not being impacted to any noticeable extent due to OSWTSs indicating good retention in the soil.

Andersen et al. (2006) conducted a review of nitrogen loading for OSWTS in the Wekiva area in Apopka, Florida USA. This followed the introduction of a protection scheme for the Wekiva River system which receives much of its baseflow from a series of springs located throughout the Wekiva groundwater basin. These springs had elevated levels of nitrates believed to be caused by the density and poor performance of OSWTS's in the area. This protection scheme required the upgrading of existing wastewater treatment systems that did not meet stringent discharge limits. It was concluded by Andersen et al. (2006) that whilst some proportion of the elevated levels of nitrates could be attributed to OSWTS in the area, other inputs such as fertilizer and atmospheric deposition were in fact contributing up to eight times more nitrogen to the groundwater system than that arising from OSWTS. Meile et al. (2010) examined the natural attenuation of nitrogen loading from dense OSWTSs in Georgia, USA and their potential input to coastal waters with the associated undesirable consequences. The study involved quantifying the numbers and spatial distribution of OSWTS in a coastal area and modeling the impact of increased nitrogen loading at the saltwater-freshwater transition zone. One of the main conclusions demonstrated by Meile et al. (2009) was that sulphide can negatively impact on the availability of O_2 and therefore inhibit denitrification and other anoxic degradation pathways. It is therefore advisable to incorporate adequate setback distances for OSWTSs in coastal areas. A study by Geary (2005) also concluded that plumes of STE can impact negatively on coastal waters and stressed the need for riparian vegetation to limit the movement of plumes of contaminants from OSWTS's. The issue of water table mounding due to cluster and high density wastewater soil adsorption systems has been investigated by Poeter et al. (2005). This study concluded that the relatively large discharges of wastewater that cluster and high density OSWTS's create, coupled with insufficient hydraulic capacity of the subsurface can result in localised mounding of groundwater with undesirable consequences – see Figure 2.12 below for an illustration of this issue.

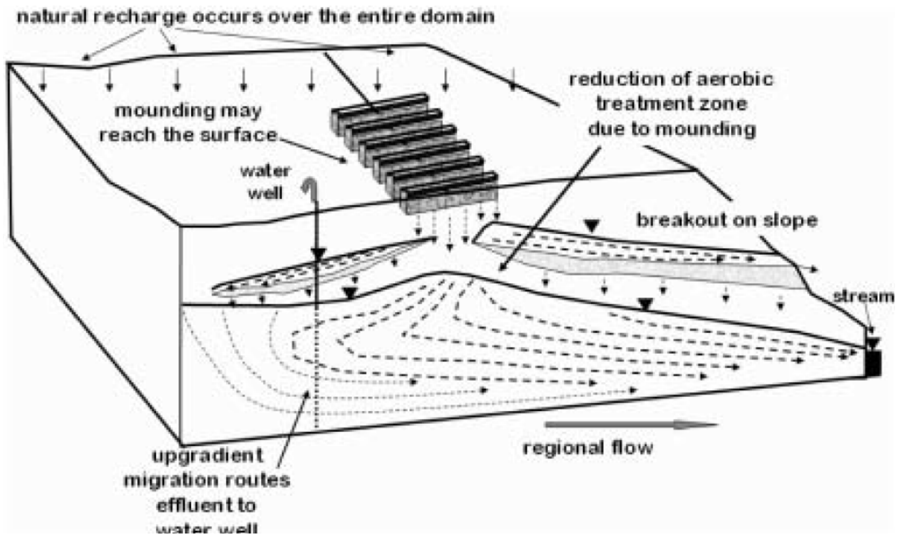


Figure 2.12 Illustration of Groundwater Mounding due to Clusters of OSWTS's (Poeter et al., 2005)

Mounding can be caused by lenses of the low hydraulic capacity subsoil creating localised perched conditions allowing contaminated groundwater move laterally and seep into watercourses or wells. Mounding can also be caused by saturated conditions being created beneath the area receiving the increased wastewater discharges again due to the low permeability of the subsoil. Once these saturated conditions develop, the direction and discharge point of groundwater flow may be altered. Groundwater mounding can have undesirable consequences such as the reduction of aerobic treatment in the vadose zone or seepage of contaminated groundwater at slopes. Poeter and McCray (2008) attempted to model this groundwater mounding numerically with only limited success.

Finally, even if the density of OSWTSs does cause significant elevations in nutrients which can be shown in the Irish context through field studies and/or numerical modeling, Fenton et al. (2011) has shown that a time lag for both vertical and horizontal flushing exists in Irish aquifer with respect to nitrate concentrations. This would mean that even if remedial action were taken immediately, there would be a significant time lag before observed concentrations in the groundwater would reflect any changes in practice with timescales of up to 5 years lag indicated in the Fenton et al. (2011) study.

2.3.7.2 Existing Legislation / Guidance / Local Planning Policy

There have been a number of guidance documents that have issued policy on a national level with respect to the installation and standards of OSWTS and these are usually published by the EPA. The latest of these documents; Code of Practice: Wastewater Treatment and Disposal Systems Serving Single Houses (EPA, 2009), has gone further than any of those preceding it in relation to standards and good practice. However this document provides no guidance on an advisable density of OSWTSs particularly in the popular cluster or ribbon type developments.

The relevant responsible Environmental bodies in Scotland, Ireland and Northern Ireland joined together to form the Scotland and Northern Ireland Forum for Environmental Research (SNIFFER). SNIFFER produced a report in 2009 which reviewed the legislative requirements and responsibilities relating to OSWTSs and their impact on water quality (SNIFFER, 2009). The review found that there is a significant level of microbial contamination in Irish groundwater (~30%) with a similar proportion of groundwater found to have elevated levels of phosphorus. Nitrate levels in Irish groundwater were found to be slightly better however this is only relative to which limit is applied. Again the report also focused on surface water as a target receptor and surface water was found to be at a high risk from OSWTS contamination. Given the number of OSWTSs that are present in Northern Ireland, the SNIFFER report made a number of recommendations in order to mitigate the risk to both groundwater and surface water bodies. These recommendations included; identifying the locations of all treatment systems in order to produce a GIS map, evaluating the condition of the systems through both risk based decision making and through varying degrees of inspection, making it a requirement for regular maintenance of these systems with a view to reducing the overall risk to water bodies and also to regulate new development more stringently with better control and guidance. Whilst this review did mention density of systems as a means of evaluating loading pressure and risk of contamination, density alone was not the focus and no recommendations were made on limiting the density of future development.

The EPA (2013) has introduced a National Inspection Plan which is aimed at addressing the risks as described above and also for Ireland to meets its obligations under European Law. The EPA (2013) report commented that "*a high density of DWWTSs can cause localised plumes with elevated nitrate concentrations in groundwater*" and for this reason areas with a high density of OSWTS have been targeted when they are located in areas that are also deemed to be "high risk". Determining which areas are high risks for the national inspection plan was achieved by combining GIS maps (see Figure 2.13 below) relating areas which are deemed to have poor subsoil permeability with those areas close to sensitive receptors such as Special Areas of Conservations (SAC's). Whilst the EPA (2013) report does identify the density of OSWTS as an increased risk for groundwater contamination, it does not give any guidance on what an appropriate density of these systems should be.

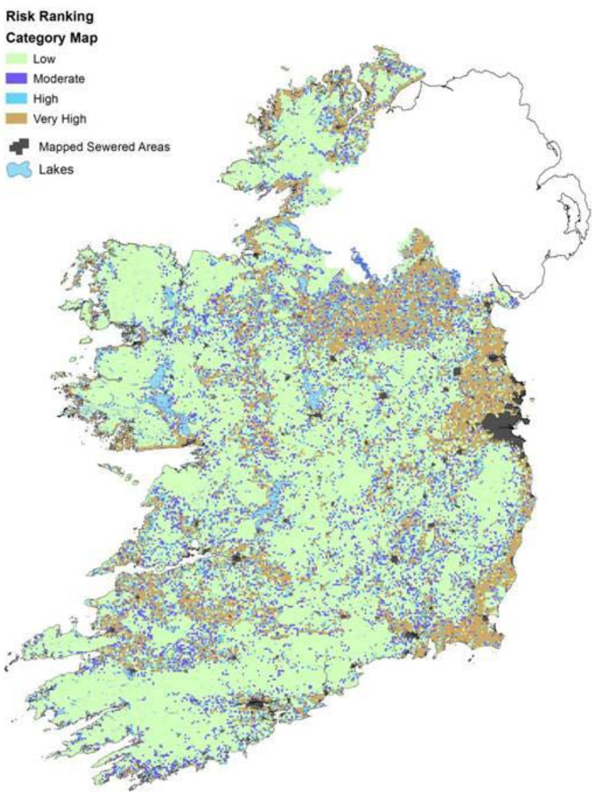


Figure 2.13 Risk Map of Ireland for the New OSWTS Inspection Plan (EPA, 2013)

With respect to the density of OSWTSs, Local Authority Development Plans are usually the only area that gives guidance on what an appropriate density should be. Many of these objectives are not based on any specific research or expected levels of groundwater contamination, but are based more on 'Rules of thumb' that specify an appropriate area required for the proper construction and installation of a treatment system with an appropriately sized percolation area. Figure 2.14 below gives an extract from the Fingal County Council Development Plan which specifies a minimum area of 0.2 Ha as a site size for a development containing an OSWTS. This would yield a density of 5 units per hectare (two units per acre). This is in line with the minimum area specified by the USEPA as reported by Yates (1986).

- Improperly functioning and serviced waste water treatment systems have a potentially serious and negative impact on the quality of the ground water system. The fact that a large proportion of soils in the County are unsuitable for percolation exacerbates the possible polluting impact of waste water systems. The required minimum land-take for a rural house by the Council is 0.2 ha. This minimum area is based on the need to deliver effective operation of the house's waste water treatment system

***Objective* RC02**

Permit only persons with a rural-generated housing need, as defined within this Section of the Development Plan, planning permission for a house within a Rural Cluster where the site size is a minimum of 0.2 hectares for on-site treatment systems, and conforms to the drainage and design standards required by the Council, and 0.125 hectares where connecting to a public sewer.

Figure 2.14 Extract from the Fingal County Council Development Plan specifying the minimum site size for the inclusion of OSWTS (Fingal Co. Co., 2012)

However, this is not a consistent policy and each Local Authority tends to have their own specific method for determining what an appropriate density of OSWTS should be. An example of this inconsistent policy is shown in Figure 2.15 whereby planning permission was refused based on the conclusion that the additional OSWTS associated with the new development would lead to too high a concentration of OSWTS. For this particular example the density of OSWTS in this cluster development was approximately one unit per 0.5 Ha (one per 1.23 acres).

Recommendation: Refuse Permission

3. Taken in conjunction with existing development in the area, the proposed development would result in an excessive concentration of septic tanks and wastewater treatment units in the area and would therefore, be prejudicial to public health.

Figure 2.15 Extract from planning permission decision report recommending refusal by a Local Authority due to the density of OSWTS

Overall, whilst Local Authorities do provide some 'guidance' as to the required density of OSWTS, there is no clear overall national policy on what an appropriate density of OSWTS should be. Any guidance in this area should be based upon field studies and modeling and should take into account the varying groundwater vulnerabilities and subsoil properties across the country.

2.4 Modelling On-Site Wastewater Treatment and Disposal

2.4.1 Introduction

The use of groundwater models has become more prevalent over the last number of decades with the development of sophisticated computer codes and the continuing advances in data processing (Stephens, 1996). Groundwater modelling has become widely used in the fields of Environmental Science, Hydrology, Engineering and Hydrogeology. The development of any mathematical model does however require the input of data for both validation and calibration purposes. The accuracy of any model is therefore governed by the quantity and quality of the input data, which ultimately faces the reality of uncertainty given the highly heterogeneous nature of subsurface geological formations (Bear and Cheng, 2010). Areas that must be considered when developing a conceptual model include; inputs and outputs of water and of relevant contaminants, initial conditions within the domain and boundary conditions which will represent interactions with the surrounding environment (US EPA, 1992).

2.4.2 Water Flow and Contaminant Transport in the Unsaturated Zone

When hoping to understand the movement of groundwater and contaminants with the goal of groundwater management, it is important to first understand the movement of water and/or contaminants in the unsaturated or vadose zone. Infiltrating water (groundwater recharge) may carry with it dissolved contaminants as it moves downward which have arisen from sources at the surface or buried sources in the unsaturated zone such as irrigation systems or drain fields for OSWTS (Bear and Cheng, 2010). Whilst moving downwards with the infiltrating water, contaminants undergo a number of processes including dispersion and adsorption which affect the concentration of these pollutants which will eventually reach the water table (Bear and Cheng, 2010). The main components of the unsaturated or vadose zone are illustrated in Figure 2.16 below.

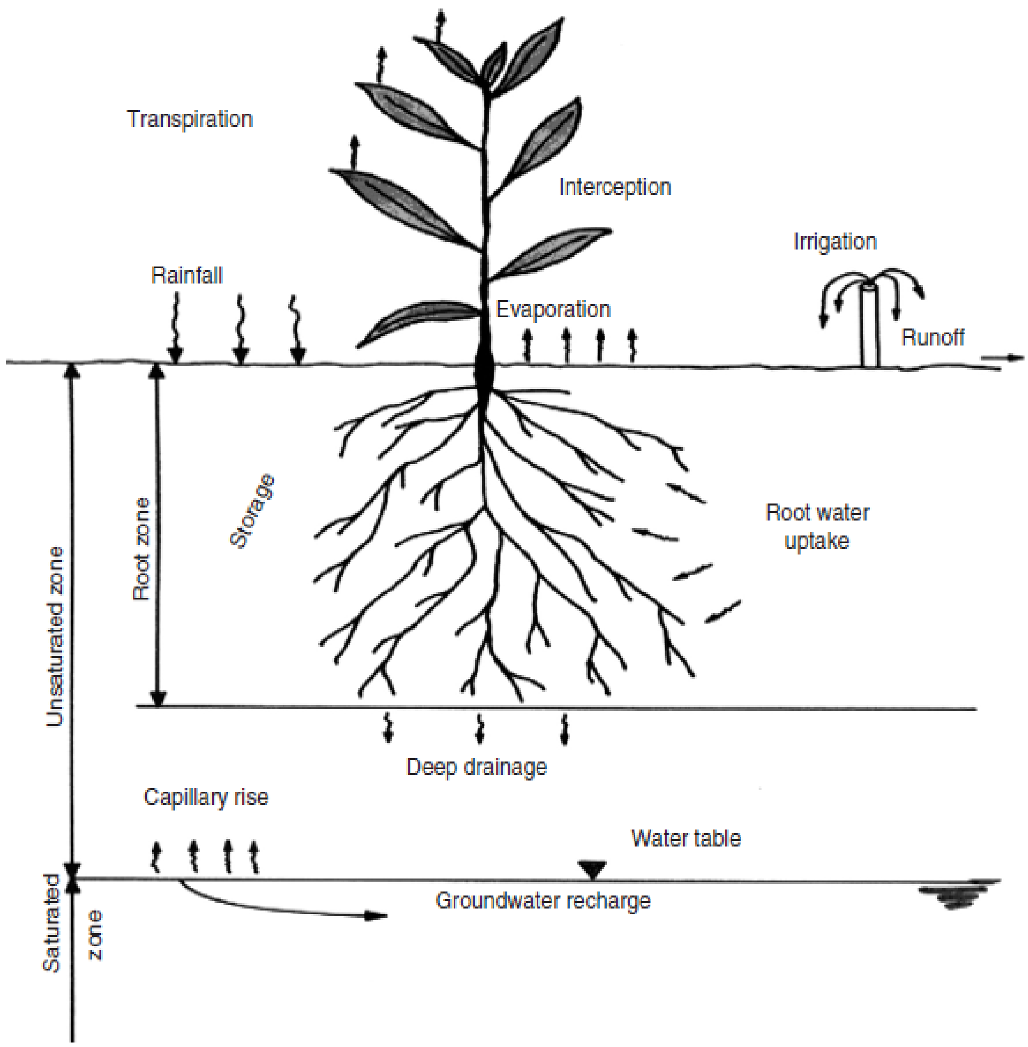


Figure 2.16 Water Fluxes and Components of the Vadose Zone (Šimůnek and van Genuchten, 2006)

Moving upwards from the saturated zone towards the ground surface, soil becomes drier as air replaces water through processes such as internal drainage, and evapotranspiration (Stephens, 1996). In the vadose zone, water is held in the soil pores by capillary forces and adsorption. This leads to a negative (i.e. less than atmospheric) pressure known as suction, tension or matrix head (Mallants et al, 2011). The relationship between water content (θ)

and the height above the water table or the pressure head (ψ) is known as the water retention curve (Marshall and Holmes, 1988). The pore-size distribution of a particular soil textural class will determine how quickly it will lose water; for example the sand in Figure 2.17 loses water much faster than the more fine-textured silt and clay (Marshall and Holmes, 1988). This is due to the larger pore diameters in the more coarse-textured soils and results in sands and gravels draining at relatively small negative pressures. More fine-textured soils such as clays or loams, therefore do not drain until much larger negative pressures are applied (Stephens, 1996).

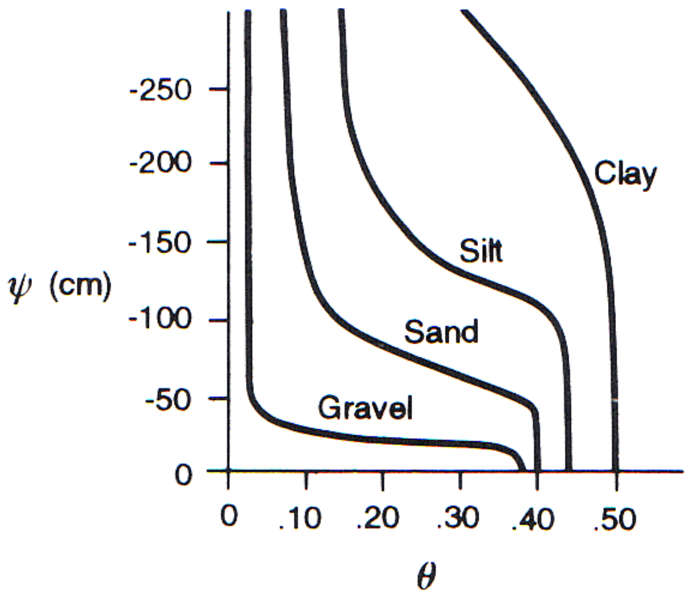


Figure 2.17 Effects of texture on the soil-water retention curve (Stephens, 1996)

One of the most common expressions for the soil-water retention curve, θ(h), is given by Van Genuchten [Eq 2.3], which gives a relatively good description of θ(h) for many soils whilst only requiring a limited number of input parameters (van Genuchten 1980; Mallants et al, 2011). The Van Genuchten soil moisture retention characteristic is defined as:

$$\theta(h) = \theta_r + \frac{\theta_s - \theta_r}{(1 + |ah|^n)^m}$$

[Eq. 2.3]

where:

θr is the residual water content [L^3L^{-3}],

θs is the saturated water content [L^3L^{-3}],

α [L^{-1}]; n [−] and m (= 1−1/n) [–] are shape parameters

However, hysteresis, arising due the water content and pressure head relationship being dependent on the wetting history of the soil, can affect the use that can be made of moisture characteristics (Marshall and Holmes, 1988). The soil-water retention curve given in Figure 2.17 is the relationship when the soil drains from complete saturation. However the soil will behave differently when it partially wets from an initially dry condition or when the soil drains from only partially wetted conditions (Stephens, 1996). This leads to intermediate paths on the soil-water retention curve with scanning curves which are bounded by the main drainage curve and the main wetting curve as illustrated in Figure 2.18 below.

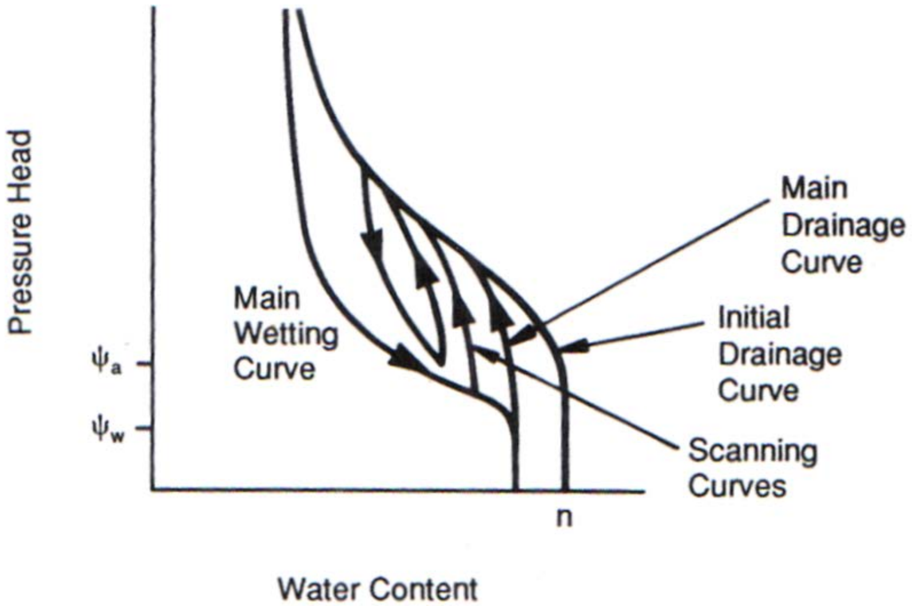


Figure 2.18 Effect of Hysteresis on the soil-water retention curve (Stephens, 1996)

In many cases when considering water flow in the vadose zone, the effects of hysteresis are ignored due to the complicated nature of the relationships involved, however many numerical models include the option to consider its effects such as Hydrus 2D/3D ((Šimůnek et al, 2007).

Another important soil hydraulic property that must be considered when trying to predict water flow in the vadose zone is the unsaturated hydraulic conductivity function which characterizes the ability of a soil to transmit water. The unsaturated hydraulic conductivity of a soil is affected by many soil features including; pore-size distribution shape, roughness, and degree of interconnected of pores. Hydraulic conductivity decreases as the soil becomes less saturated and therefore the unsaturated hydraulic conductivity function yields the dependency of the hydraulic conductivity on the water content, K(θ), or pressure head, K(h) (Mallants et al, 2011). This relationship is given by the van Genuchten-Mualem model [Eq. 2.4] (van Genuchten 1980; Mualem 1976; Wang et al, 2012).

$$K(h) = K_s S_e^l [1 - (1 - S_e^{1/m})^m]^2$$

[Eq. 2.4]

Where:

K_s is the saturated hydraulic conductivity [LT−1]

l is a pore-connectivity parameter

S_e is effective saturation given by : Se = (θ – θr) / (θs - θr)

and m = 1 – 1/n; n > 1

For a soil that is either saturated or unsaturated the flow velocity is given by the Darcy-Buckingham equation [Eq. 2.5]:

$$q = -k(h)\frac{\partial h}{\partial z} + K(h)$$

[Eq 2.5]

Where:

h is water pressure head [L]

z is spatial coordinate [L]

q is hydraulic loading rate or flux [LT^{-1}]

K(h) is the unsaturated hydraulic conductivity function [LT^{-1}]

Water flow in variably saturated rigid porous media can be described in terms of mass balance [Eq. 2.6] (Mullants et al, 2011):

$$\frac{\partial \theta}{\partial t} = -\frac{\partial q}{\partial z} - S$$

[Eq. 2.6]

where:

θ is the volumetric water content [L^3L^{-3}]

t is time [T]

z is the spatial co-ordinate

q is volumetric flux [LT^{-1}]

S is a source/sink term [$L^3L^{-3}T^{-1}$]

Combing [Eq. 2.3] and [Eq. 2.4] results in the Richards equation [Eq. 2.7] which describes water flow in the variably-saturated vadose zone (Richards, 1931; Trimble, 2008; Mullants et al, 2011):

$$\frac{\partial \theta(h)}{\partial t} = \frac{\partial}{\partial z}\left[K(h)\frac{\partial h}{\partial h} + K(h)\right] - S(h)$$

[Eq. 2.7]

This is sometimes referred to as the mixed form of the Richards equation as it contains two dependent variables; the pressure head and the water content (Mallants et al, 2011). This partial differential equation is the equation governing water flow in the unsaturated or vadose zone; however solving [Eq. 2.7] numerically usually involves deriving a simplified analytical solution through methods such as finite differences or finite elements. There are many software packages available that use numerical methods to solve the Richards equation and predict water flow in the vadose zone including; VLEACH (Varadhan, R., and Johnston J. A., 1997), HYDRUS 1D (Šimůnek et al, 1998), HYDRUS 2D/3D (Šimůnek et al, 2007), The UnSat Suite, MODFLOW-SURFACT and VZMOD (Wang et al, 2012) amongst many others.

Contaminant transport in the vadose zone can be described generally by using the mass balance equation [Eq. 2.8] and applying it to the concentration of the contaminant (Šimůnek et al, 1998):

$$\frac{\partial C_T}{\partial t} = -\frac{\partial J_T}{\partial_z} - \phi$$

[Eq. 2.8]

where:

C_T is the total concentration of contaminant in all forms [ML^{-3}],

J_T is the total contaminant mass flux density (mass flux per unit area per unit time)

[$ML^{-2}T^{-1}$],

Φ is the rate of change of mass per unit volume by reactions or other sources (negative) or sinks (positive) such as plant uptake [$ML^{-3}T^{-1}$]

The concentration of the contaminants (C_T) and the source/sink term (Φ) within this definition both incorporate the portions that exist in all three phases (e.g. solid phase, liquid phase and soil gas phase) and equations describing these terms in more detail have been summarised by Batu (2006). Mallants et al (2012) noted that there are three main transport processes that are generally considered to be active in both the liquid and soil gas phases:

- Molecular diffusion
- Hydrodynamic dispersion,
- Advection (convection)

Using Fick's first law for porus media (Batu, 2006) diffusive transport can be given by [Eq. 2.9]:

$$J_d = -\theta D \frac{\partial_c}{\partial_z}$$

[Eq. 2.9]

where

δc/δz is the concentration gradient

D is the effective molecular diffusion coefficient (D_e). [L^2/T]

Due to the tortuous diffusion pathway (i.e. increased path lengths) and given the presence of a solution-solid interface (Batu, 2006), D_e can be approximated to the diffusion coefficient of pure water D_0 by: $D_e = D_0\mathbf{T}$ where $\mathbf{T}$ is a dimensionless tortuosity factor. This tortuosity factor has been found to range from 0.3 to 0.7 for most soils (van Genuchten & Wierenga, 1986).

Similarly hydrodynamic dispersion can be approximated using Fick's first law and therefore [Eq. 2.10] becomes (Mallants et al, 2012):

$$J_h = -\theta D_h \frac{\partial c}{\partial c} = -\theta(D_m + D)\frac{\partial c}{\partial z}$$

[Eq. 2.10]

where: D_h is the hydrodynamic dispersion coefficient [L^2T^{-1}]

D_m is the mechanical dispersion coefficient [L^2T^{-1}]

D is the liquid phase diffusion coefficient [L^2T^{-1}]

Dispersivity is therefore a transport parameter which is usually measured experimentally but can however be estimated based on literature. Equation [Eq. 2.10] only holds for one-dimensional transport. For two and three dimensional transport longitudinal and transverse dispersivities must be incorporated (Bear, 1972) and this has been described in detail by Šimůnek et al (2007). It has been found that dispersivity often changes the larger the path over which contaminants travel (Mallants et al, 2012). Anderson (1984) suggests that a value of one-tenth of the transport distance for the longitudinal dispersivity can be used when no other information is available and similarly a value of one-hundred of the transport distance for the transverse dispersivity can be used again as an initial estimate when values are not known.

Contaminants can also be transported with the moving fluid (i.e. advection) both in the liquid phase (J_{lc}) or the soil gas (J_{gc}) and this is given by [Eq. 2.11]:

$$J_c = qc$$

[Eq. 2.11]

The subscripts l and g are combined to give the more general equation governing contaminant transport due to advection i.e. equation [Eq. 2.11] – since contaminant transport is most dominant in the liquid phase the gaseous phase is often ignored (Radcliffe & Šimůnek, 2010). The total contaminant flux density in both the liquid and gaseous phase incorporating contributions from the various transport processes is given by (Mallants et al, 2012) as [Eq. 2.12]:

$$J = qc - \theta D_h \frac{\partial c}{\partial z}$$

[Eq. 2.12]

where:

D_h is the hydrodynamic dispersion coefficient [L^2T^{-1}] that accounts for both molecular diffusion and mechanical dispersion

Combining all three components of contaminant transport with equation [Eq. 2.8] the mathematical expression for dissolved contaminant transport in the vadose zone can be given by:

$$\frac{\partial(\rho_b s + \theta c + ag)}{\partial t} = \frac{\partial}{\partial z}(\theta D_h \frac{\partial c}{\partial z}) + \frac{\partial}{\partial z}(aD_g^s \frac{\partial g}{\partial z}) - \frac{\partial(qc)}{\partial z} - \phi$$

[Eq. 2.13]

where:

D_h and $D^s{}_g$ are the hydrodynamic dispersion coefficient in the liquid and gaseous phases [L^2T^{-1}]

In the case of one dimensional solute transport in the vadose zone and only considering a non-adsorbing contaminant and steady-state water flow equation [Eq. 2.13] simplifies to:

$$\frac{\partial c}{\partial t} = D_h \frac{\partial^2 c}{\partial z^2} - v \frac{\partial c}{\partial z}$$

[Eq. 2.14]

Given that both equations [Eq. 2.13] and [Eq. 2.14] contain unknowns relating to the solid and liquid phase concentrations, information is needed relating to the equilibrium partitioning between the two phases in order to arrive at a solution (Batu, 2006) and for this reason graphs known as adsorption isotherms are commonly used when solving the advection-dispersion equation. Most isotherm equations that are used in vadose zone advection-dispersion calculations are based on the Freundlich isotherm [Eq. 2.15] (Batu, 2006) which takes the following form:

$$s = K_f c^\beta$$

[Eq. 2.15]

where: K_f and β are constants

The advection-dispersion equations together with the associated equations for initial conditions, boundary conditions and other parameters such as the Freundlich isotherm as described above are a best solved using numerical models (Šimunek and Van Genuchten, 2006). Numerical models generally employ either finite differences or finite elements in order to solve these complicated equations; these methods are discussed in more detail in the following Section 8.1 with respect to groundwater modelling.

Vadose Zone Solute Transport Modelling

The use of any vadose zone model to simulate solute transport and the development and transport of nutrient plumes requires the input of model specific parameters generally relating to nitrogen (N) and phosphorus (P). McCray et al. (2005) have therefore compiled statistically supported reference material to give guidance when choosing model-input parameters. The data is mainly presented in the form of cumulative frequency distributions or data ranges with median values. In addition to nitrogen and phosphorus loading rates, statistical information is also presented for nitrification and denitrification rates as well as linear sorption isotherm constants for phosphorus.

Over the last two decades there have been many advances in the study and understanding of solute transport from on-site effluent percolating through the vadose zone (Šimunek et al., 1998). Hassan et al. (2008) employed the HYDRUS 3D software to compare simulated values of soil water potentials (Ψ_s) and nitrates (NO_3) against observed values at varying depths below the a subsurface drip irrigation (SDIS) system receiving effluent from a sequential batch reactor (SBR). The simulated values were found to compare very favourably with actual values observed at sampling wells, suction lysimeters, and tensiometers installed across the study area. Radcliff and West (2007) used the HYDRUS 2D software to simulate Long Term Acceptance Rates (LTAR) for on-site wastewater treatment systems discharging to a conventional trench percolation field and compared the results to an alternative approach using a simple empirical formula and the results for both methods were found to generally agree. The HYDRUS 2D software was also used by Finch et al. (2008) to investigate the proportion of infiltration of effluent from an on-site wastewater treatment system via the trench sidewall versus the trench bottom. Results from this model indicate that approximately 30% of the effluent water infiltration occurs through the trench sidewall which is contrary to what is commonly assumed. Beal et al. (2008) modelled the effect of biomat development along the sidewalls of a trench in a drainfield for an on-site wastewater treatment and compared the results to a field study of an instrumented drainfield where the water height was manipulated to achieve different hydraulic loading rates. It was found in this study that trench sidewall infiltration is an important factor in providing a buffer zone against ponding or surface surcharging in a drainfield. A good agreement was observed between simulated fluxes through the trench sidewalls and biomat layer with those actually observed in the field study. Maziar & Simunek (2010) investigated the distribution of water around emitters in a subsurface drip irrigation system using the HYDRUS 2D software. The study found a very good agreement between simulated values of a number of parameters that had been refined using the root-mean-square-error method with values observed in the field.

2.4.3 Groundwater Flow Models and Contaminant Transport

In order to model the movement and transport of contaminants in groundwater, it is necessary to first know how the groundwater moves and consequently a groundwater flow model is the first step in developing such a model (Rios et al, 2011). Many of the commonly used groundwater models calculate the three-dimensional movement of groundwater by numerically solving the governing partial-differential equation (PDE) of groundwater flow given below (McDonald and Harbaugh, 1988).

$$\frac{\partial}{\partial x}\left(K_{xx}\frac{\partial h}{\partial x}\right)+\frac{\partial}{\partial y}\left(K_{yy}\frac{\partial h}{\partial y}\right)+\frac{\partial}{\partial z}\left(K_{zz}\frac{\partial h}{\partial z}\right)+W=S_s\frac{\partial h}{\partial t}$$

[Eq. 2.16]

where:

K_{xx},K_{yy} and K_{zz} are values of hydraulic conductivity along the x,y and z coordinate axes which are assumed to be parallel to the major axes of hydraulic conductivity (L/T);
h is the potentiometric head (L);
W is a volumetric flux per unit volume representing sources and/or sinks term (L^{-1});
S_s is the specific storage of the pourus material (L^{-1});
t is time (T)

A common approach taken when modelling groundwater flow is the use of distributed models which break the catchment of interest up in layers or blocks and then use finite elements or finite differences to solve a series of differential equations used to describe the flow in the catchment (such as [Eq. 2.16]). Some of the most common distributed groundwater flow models have been briefly summarised below.

MODFLOW

MODFLOW was developed by the United States Geological Survey (USGS) between 1981 and 1983 using FORTRAN computer language (McDonald and Harbaugh, 1984). MODFLOW implements a modular structure in which similar program functions are grouped together and constructed to be independent of other function groups and and these are referred to as packages. Packages exist that represent flow in bedrock, drains streams and rivers with many others also contained within the overall program structure. Which packages are applied is based upon the specified boundary conditions and initial conditions. MODFLOW has become an international standard groundwater model that has been applied in numerous studies across the world such as the study by Taylor et al. (2012).

MicroFEM

MicroFEM is a finite element groundwater modelling program for multiple aquifer steady state and transient ground water flow modelling. MicroFEM has been used extensively in the modelling of groundwater in various applications, such as the study by Eddebbarh et al. (1996), and is popular due to the ease of input data preparation.

SHETRAN

SHETRAN is a distributed finite difference integrated hydrological, sediment transport and contaminant transport model. The groundwater module (one of five modules contained within the package) is similar to MODFLOW and uses a partial differential-equation to describe the saturation of the cell inside the finite difference array.

FEFLOW

The Finite Element Flow model (FEFLOW) is fully distributed, deterministic hydrogeological model. It has extensive functionality in simulating groundwater flow and multi species reaction transport, with particular proficiency in variable saturated flow (Trefry and Muffels, 2007). Again FEFLOW uses a number of underlying assumptions which are then applied to construct partial differential equations that describe flow in a porous media.

Modelling groundwater in karst

There have been a number of different approaches through which groundwater flow in karst geologic conditions has been simulated such as that taken by Murray and Hudson (2002) whereby a geologic framework model was successfully employed to simulate geologic features in three-dimensions and accurately predict groundwater flow rates and subsurface contaminant transport. Gill et al. (2013) successfully used a pipe network model to represent groundwater flow in a complex karst environment in west Ireland. Traditional partial differential equations that are used to represent groundwater flow in other porous media are not applicable in karst due to the existence of preferential flow paths usually taking the form of caves or conduits.

Contaminant transport in the saturated zone can also be simulated using many of the groundwater modelling packages described briefly above. An example of one contaminant transport package is that used within the MODFLOW software entitled MT3D (Zheng, 1990). MT3D is a modular three-dimensional transport model for simulation of advection, dispersion and chemical reactions of contaminants in groundwater systems, and was revised by Zheng and Wang (1999) to also incorporate the capabilities for simulating advection, dispersion/diffusion, and chemical reactions of contaminants in groundwater flow systems. The partial differential equation describing the fate and transport of contaminants of species k in transient groundwater flow systems is given in [Eq. 2.17] by Zheng and Wang (1999):

$$\frac{\partial\left(\theta C^{k}\right)}{\partial t}=\frac{\partial}{\partial x_{i}}\left(\theta D_{ij}\frac{\partial C^{k}}{\partial x_{j}}\right)-\frac{\partial}{\partial x_{i}}\left(\theta v_{i}C^{k}\right)+q_{s}C_{s}^{k}+\sum R_{n}$$

[Eq. 2.17]

where

θ = porosity of the subsurface medium, dimensionless
C^k = dissolved concentration of species k, ML^{-3}
t = time, T
$x_{i,j}$ = distance along the respective Cartesian coordinate axis, L
D_{ij} = hydrodynamic dispersion coefficient tensor, L2T-1
v_i = seepage or linear pore water velocity, LT-1; it is related to the specific discharge or Darcy flux through the relationship, vi = q_i / θ
q_s = volumetric flow rate per unit volume of aquifer representing fluid sources (positive) and sinks (negative), T^{-1}
$C^k{}_s$ = concentration of the source or sink flux for species k, ML^{-3}
ΣRn = chemical reaction term, $ML^{-3}T^{-1}$

The derivation of the advection-dispersion equation shown in equation [Eq. 2.17] has been outlined in detail by Anderson (1979 and 1984), Anderson and Woessner (1992), Bear (1972 and 1979) and Bear and Cheng (2010) and was summarised for the unsaturated zone in Section 1.4.3 above. The MT3DMS code is capable of handling linear or nonlinear sorption in both equilibrium and nonequilibrium scenarios and is also capable of approximating biodegradation (or radioactive decay) through first-order reaction simulations (Zheng, 1990).

In order to define any groundwater flow and/or contaminant transport model it is necessary to first define initial and boundary conditions across the extent of any model grid. There are three distinct types of boundary conditions that can be applied to any model both in the unsaturated and saturated zone and these are listed below (Bear and Cheng, 2010):

1. A Dirichlet boundary is a specified-head boundary which acts as a source or sink of water entering or leaving the model domain. This would include constant head or general head boundary conditions.

2. A Neumann boundary condition is one where the flux across the boundary is assigned. This is also known as a specified flow boundary. Examples of this type of boundary condition would include the no-flow boundary condition.

3. Cauchy boundary condition. This is a head-dependent flow boundary and therefore defines the flux across a boundary based on head. This type of boundary condition is usually used to represent a river or stream in a flow model.

The assignment of representative boundary conditions will dictate the accuracy and success of any model and is therefore a very important consideration when developing any flow model (Anderson and Woessner, 1992).

Groundwater Solute Transport Modelling

Since the development of the MODFLOW software model in 1983 (McDonald and Harbaugh, 1988), there have been numerous applications developed to both model groundwater flow movement, and more recently, the movement of solutes through an aquifer. Given the high availability of software applications that can model groundwater flow and movement there has been much work published in this area in recent decades. However not all groundwater flow models are based on the MODFLOW code. Rashid et al. (1992) developed an interactive groundwater modelling package that utilises 2-D arrays with nodes that are manipulated by the user. The process is continued until a convergence is observed between the simulated piezometric head values and those observed in the field. This trail-and-error approach can however by very repetitive and time intensive.

In the Irish context, many of groundwater protection zones for public water supply were developed using MODFLOW based software. GSI (2005) describes the use of the MODFLOW software model to develop the groundwater protection zones for the Bog of the Ring public water supply in north County Dublin. A problem often encountered when using a numerical model such as MODFLOW to model groundwater flow at a river basin catchment scale is that the necessary discretization of large catchment areas must be kept relatively coarse in order to minimise computational demands. Hence, Wolf et al. (2008) explored a method that would overcome this issue, in which the catchment specific aquifer geometry properties were modified based on a hydrological catchment drainage analysis. When this new approach was compared with other previously refined models it was shown to provide a good solution to this issue when and is therefore applicable when it is necessary to model groundwater in large catchments.

Zheng (1990) developed a modular three-dimensional transport model which, when used in conjunction with another groundwater flow model such as MODFLOW, can predict the movement and transport of solutes. Since then there have been many developments in this area with a second generation of the original code MT3DMS being developed which incorporates many new features most notably the ability to account for not just solute transport but also incorporating capabilities for simulating advection, dispersion/diffusion, and chemical reactions of contaminants in groundwater flow systems (Zheng & Wang, 1999). Lasserrea et al. (1999) proposed a GIS-linked model for the assessment of nitrate contamination in groundwater with only the effects of advection being considered. A good agreement was found when the GIS model results were compared with those from the MT3D-MODFLOW software for the same study area. Molenat and Gascuel-Odoux (2002) also examined the flow and transport of nitrate using the MT3D-MODFLOW and MODPATH models. The model was developed with a view to achieving better land use planning in the future following elevated levels of nitrates being observed in the groundwater surrounding French Brittany due to intensive agricultural practises. The model developed by Molenat and Gascuel-Odoux (2002) indicated a good agreement with observed field values of piezometric head and nitrate concentrations.

Another aspect of groundwater flow models with solute transport is the ability to predict future concentrations of key contaminants. Given the high levels of nitrates in groundwater that have been detected in northern France, a program called ''Ferti-better' was introduced in 1990 with a view to reducing these levels below European framework limits. Serhal et al. (2009) therefore developed a model based on the MT3D-MODFLOW to simulate the effects of this new program on future nitrate concentrations. The models predicted that by 2015 there would be a reduction in nitrate concentrations in some areas of the study catchment, but that in other areas the program would not be as effective.

A large study, entitled The Pathways Project, has been underway for the last 5 years in Ireland whereby the main hydrological pathways; overland flow, interflow, shallow groundwater and deep groundwater, have been investigated through numerical models and field studies with a view to developing a catchment management tool (CMT) which will model pollutant movement throughout an entire hydrogeological catchment. Work is almost complete on this project and it is hoped that the tool will go live this year (2013).

3 SITE SELECTION AND DESCRIPTION

3.1 Introduction

There were a number of criteria agreed at the outset of the project that needed to be satisfied with respect to site suitability and selection. Existing cluster developments were required and it was desirable that a high proportion of the on-site systems contained within the cluster developments should be well established so that contaminant plumes, if present, would have reached some form of long term quasi-equilibrium. Another key objective was to set up each site for monitoring as early as possible (within the first 6 – 9 months) in order to allow the commencement of sampling. This was necessary due to the constraints on time for the overall project and a minimum required monitoring period of 24 months in order to have representative annual data available which could be used to set up and calibrate numerical models. The key components of the site assessment and selection procedure were a desk study followed by site visits and on-site suitability assessments leading to a decision on which sites to pursue further. The most difficult aspect of the site selection process would inevitably be finding landowners willing to allow drilling of monitoring boreholes and access for sampling given that no incentives were available financial or otherwise.

The site selection process will be discussed in detail in the following sections. A brief description of each of the final study areas will also be given under a number of headings including; land use, topography, surface water hydrology, meteorology, site occupancy, treatment system densities and geology. A more detailed description of the study locations with respect to hydrogeology and conceptual groundwater flow is presented later in Chapter 6.

3.2 Site Selection

3.2.1 Desk Study and Consultation

As outlined in Chapter 1, one of the main aims of this study was to identify four study areas in each of the different groundwater vulnerability zones and to monitor groundwater quality over an extended period of time to identify patterns associated with the density of treatment systems.

The first part of the site selection process involved contacting as many Local Authorities as possible, informing them about the project and asking them if they would be willing to assist in selecting suitable study sites within their administrative areas. In addition to Local Authorities, a number of other professional contacts were also informed about the project and their input was requested. The desk study also involved reviewing Local Authority planning files for recent planning decisions that had been granted for decentralised treatment systems. The Geological Survey of Ireland (GSI) interim groundwater vulnerability and generalised bedrock aquifer maps of Ireland were used as a basis of identifying suitable areas during the desk study. After initial contact with 14 Local Authorities, a number emerged as being interested in the project and were willing to assist further to identify suitable study areas, as follows,

- Offaly County Council
- Laois County Council
- Kilkenny County Council
- Limerick County Council
- Clare County Council
- Wexford County Council
- Fingal County Council

Meetings were arranged with representatives from the appropriate Environment Sections of the above Local Authorities and potential study sites were identified based on local knowledge and the appropriate GSI maps. Once potentially suitable areas had been identified, a more detailed desk study was conducted on the potential study sites. This desk study involved reviewing the potential study sites under a number of suitability criteria as outlined in the flowchart shown in Figure 3.1 below.

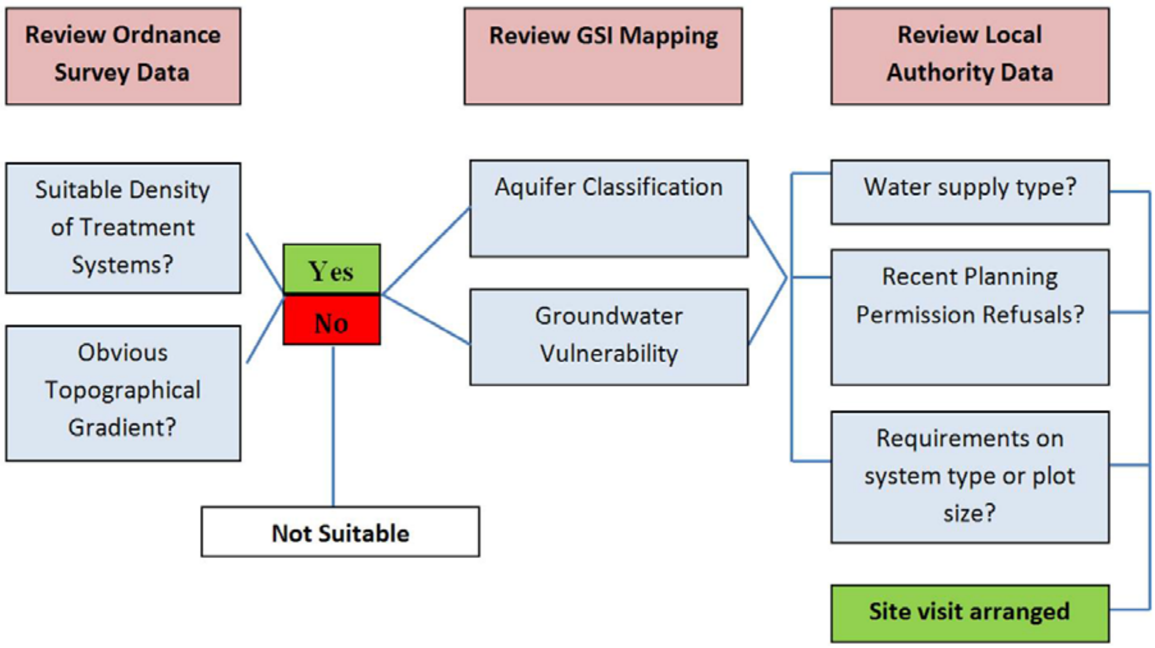


Figure 3.1 Desk Study Review Process for Potential Site Selection

3.2.2 Site Visits and Suitability

Following the desk study and preliminary meetings with Local Authorities and professional contacts a large number of potential study sites were identified across the country. These sites were then visited and re-examined under a further two criteria:

- Site Access for drilling and sampling
- Proximity to local receptors (watercourses/water supply wells)

Table 3.1 summarises the potential study sites that were identified during the site selection process and the main characteristics of the areas.

Table 3.1 Potential Sites Identified for Project

County	Location	Aquifer Class	Groundwater Vulnerability	Approx. No. Units in Cluster
Offaly	Dunkerrin	Locally Imp.	High	14
Offaly	Rhode	Regionally Imp.	Moderate	12
Offaly	Ballynahown	Locally Imp.	High	13
Clare	Toonagh	Regionally Imp - Karstified	Extreme	18
Clare	Killinaboy	Regionally Imp - Karstified	Extreme	8
Clare	Deerpark	Locally Imp.	High	21
Laois	Arless	Regionally Imp. -Karstified	High	26
Laois	Timahoe	Regionally Imp. -Karstified	High	20
Dublin	The Naul	Locally Imp.	Low	17
Dublin	Turvey	Locally Imp.	Low	15
Kilkenny	Danesfort	Locally/Regionally Imp.	Moderate/High	17
Kilkenny	Carrigeen	Regionally Imp. - Fissured	Extreme/High	13
Limerick	Abbeyfeale	Locally Imp.	Extreme	11
Limerick	Faha	Regionally Imp. -Karstified	Extreme/High	14
Limerick	Kilfanane	Locally Imp.	High	17
Wexford	Gusserane	Regionally Imp. - Fissured	High/Moderate	14

3.2.3 Site Selection

Once site visits had been completed for all of the locations listed in Table 3.1, it was decided to prioritise sites that were *Regionally Important* aquifers in order to narrow the selection criteria. This was not possible for the low vulnerability site as both sites identified were within *Locally Important* aquifers. It was then necessary to identify the relevant landowners and seek permission to drill boreholes on their land and also to seek access for the sampling period. Initially permission was agreed with landowners in Rhode, Co. Offaly, Carrigeen Co. Kilkenny, Abbeyfeale, Co. Limerick and The Naul Co. Dublin and preparations were made to begin drilling. Fortunately however, it was subsequently revealed that the cluster site at Abbeyfeale, Co. Limerick had been connected to a local sewer network without the knowledge of the Local Authority some 3 years previously. Agreement then had to be reached with landowners to drill at Faha, Co. Limerick.

Although the initial project objective was to find and monitor 4 study sites across the different groundwater vulnerability classes, it was decided during the project that given the high proportion of cluster developments that exist in the west of Ireland in conduit dominated karst aquifers, that it would be appropriate to identify and monitor a cluster development in this environment. Monitoring boreholes would not be drilled at this location, however a similar study would be undertaken relative to the groundwater flow conditions. This karstic site was selected at Toonagh, Co. Clare. The final locations of the selected study sites are shown in Figure 3.2 below.

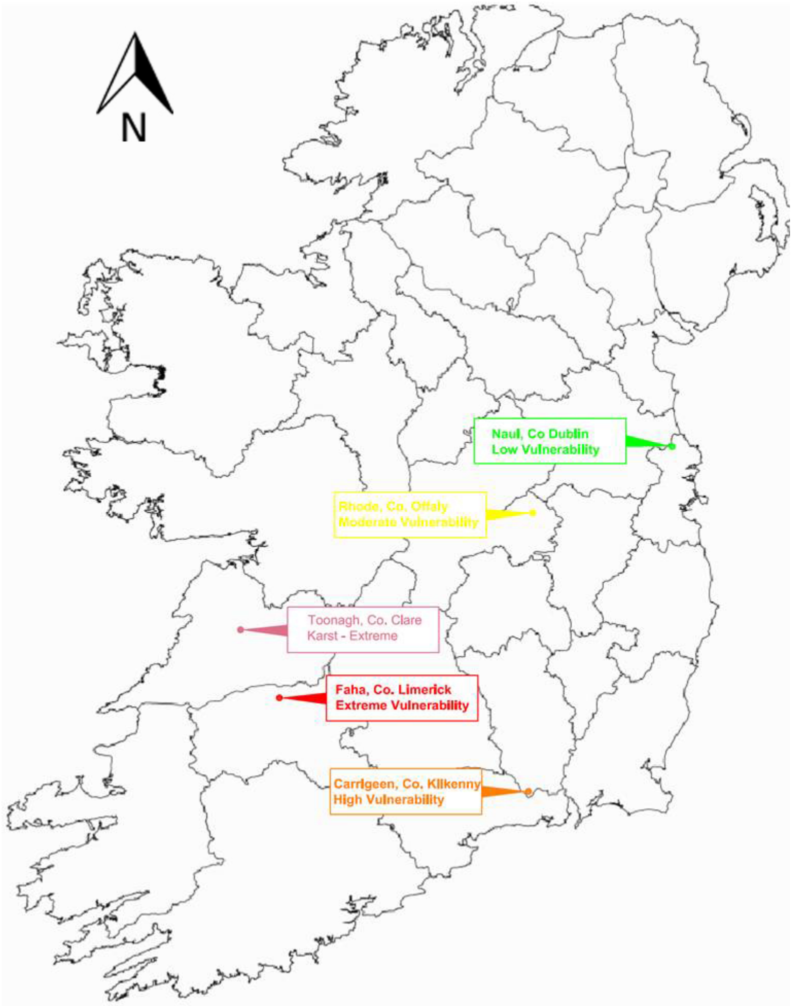


Figure 3.2 Study Site Locations

3.3 Site Description

3.3.1 Land Use, Topography and Surface Water Hydrology

The Naul, Co. Dublin *(Low vulnerability)*

The study site at the Naul, Co. Dublin is located in the townsland of Hazardstown approximately 1.5 km from the village of Naul as shown in Figure 3.3. The cluster development is located along Moonlone lane and is in the administrate area of Fingal County Council. Land use in the area is predominately agriculture and horticulture. To the south-west of Moonlone lane land is pasture and is used for livestock grazing. To the north-west a large area of land is planted with orchards. The study area slopes generally from south to north with a topographical high of 176 mOD at Knockbrack hill to the south falling to approximately 40 mOD to the north of the study area.

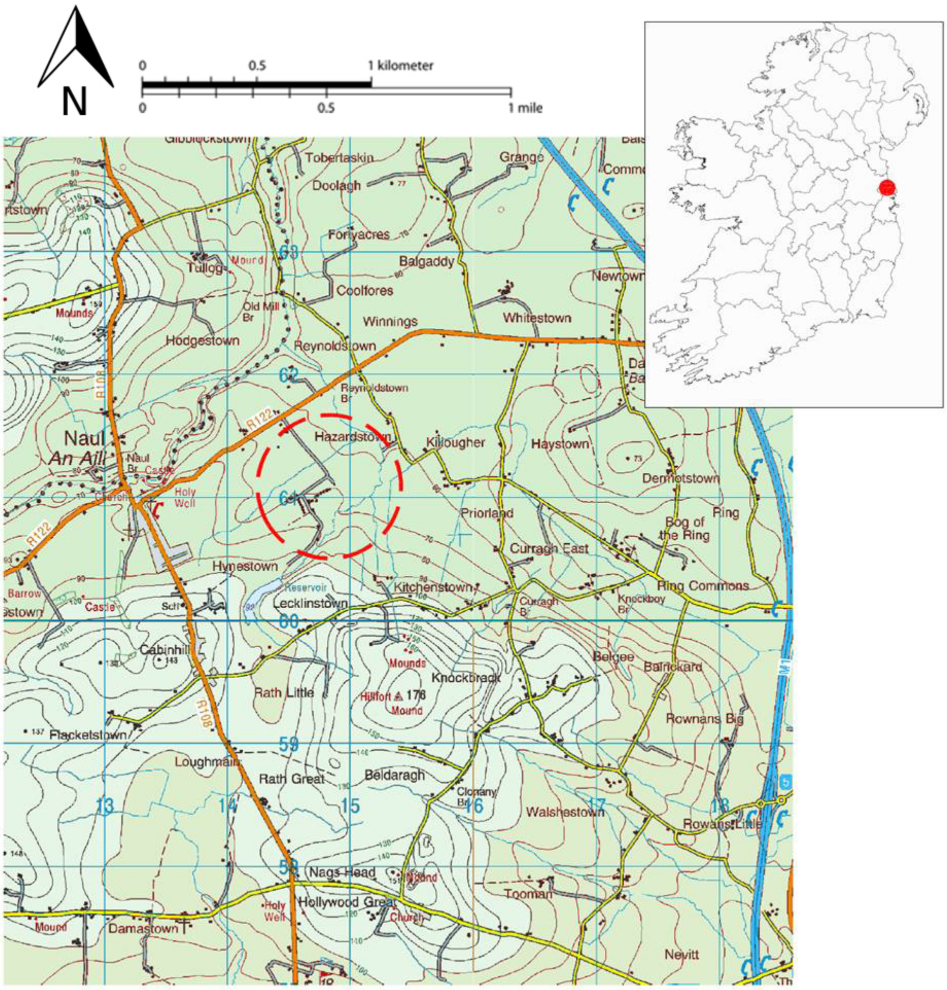


Figure 3.3 Site Location at the Naul, Co. Dublin

Figure 3.4 Aerial view of study area at Naul, Co. Dublin

The area is interwoven with a series of local streams and deep drains which generally flow from south-west to north-east. The study area falls within the surface water catchment of the River Delvin to the north-west to which nearly all of the local watercourses eventually outfall. To the south-west of the study area there is an old water reservoir which is no longer in use. The local Bog of the Ring water supply is located to the east of the study area comprising 4 active boreholes supplying a combined volume of c.3, 500 m^3/d. The dwellings in this cluster development are arranged in a typical 'ribbon' style layout as shown in the aerial view in Figure 3.4 above.

The study area is located in an area described as being Low vulnerability with respect to groundwater. This is due mainly to the extensive low permeability tills or clays that overlay the bedrock aquifers with thickness of up to 40 m present. Bedrock geology at the Naul is quite complex with areas to the south generally underlain by Namurian Mudstones and Sandstones and beneath the study area and to the north underlain by Dinantian limestones with many faulting zones present. Neither of these bedrock formations are extensively faulted and therefore the Namurian bedrock formation to the south is classified as a poorly productive aquifer with the limestone beneath and to the north of the study area classified as locally important aquifers which are moderately productive. Aquifer classifications in the vicinity of the study area at Naul are shown in Figure 3.5 below.

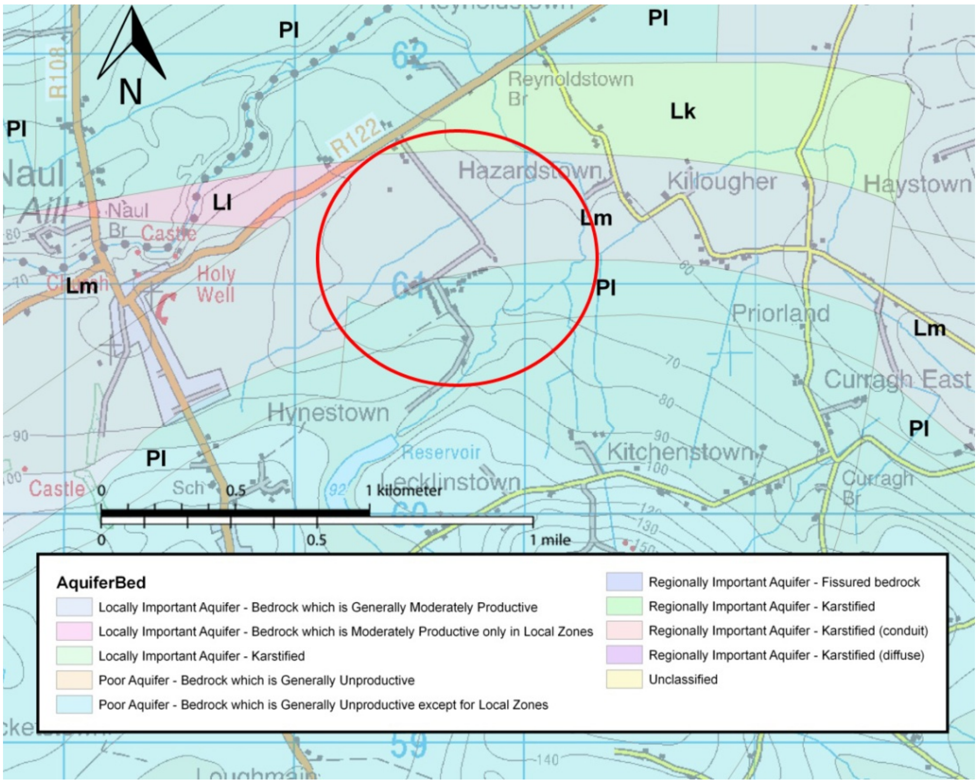


Figure 3.5 Aquifer Classification at Naul, Co. Dublin

As outlined earlier the Bog of the Ring local water supply is located to the east and consequently the study area is located within the outer zone of protection for this supply as shown in Figure 3.6 below. The zone of protection for the Bog of the Ring water supply was determined by a GSI study which involved extensive fieldwork and numerical modelling. Any contamination that occurs within the source protection area may have a direct impact on water quality with the local water supply.

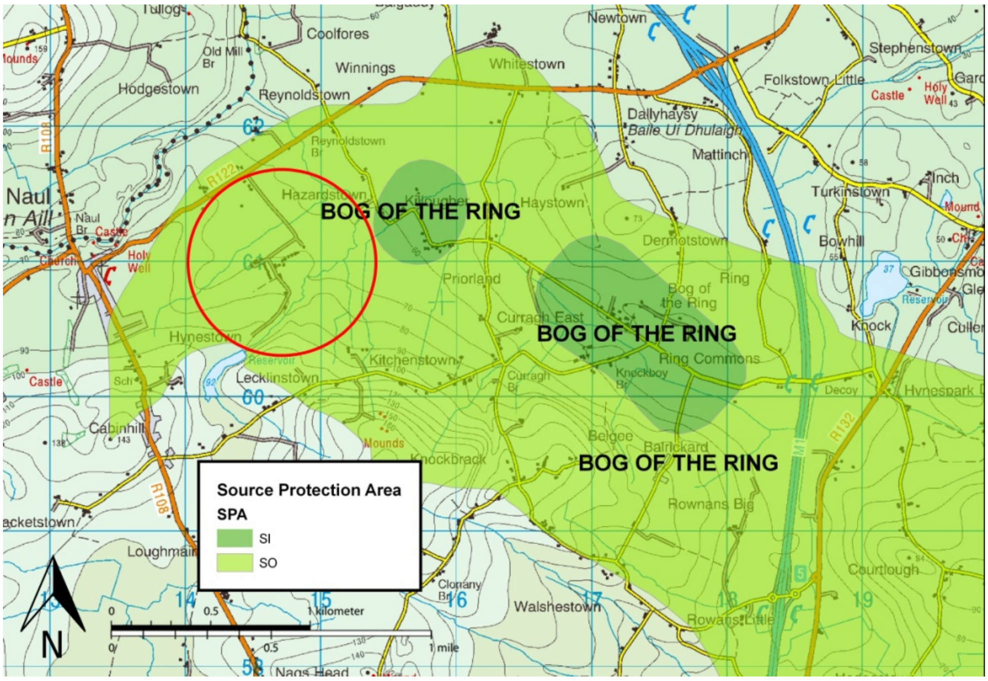


Figure 3.6 Source Protection Area at Naul, Co. Dublin

Rhode, Co. Offaly *(Moderate vulnerability)*

The study site at the Rhode, Co. Offaly is located in the townsland of Ballybrittan approximately 3 km from the village of Rhode. The cluster development is located along a local link road between the R400 and the R441 and is in the administrate area of Offaly County Council as shown in Figure 3.8.

Figure 3.7 View of ‘ribbon’ style development at Rhode, Co. Offaly

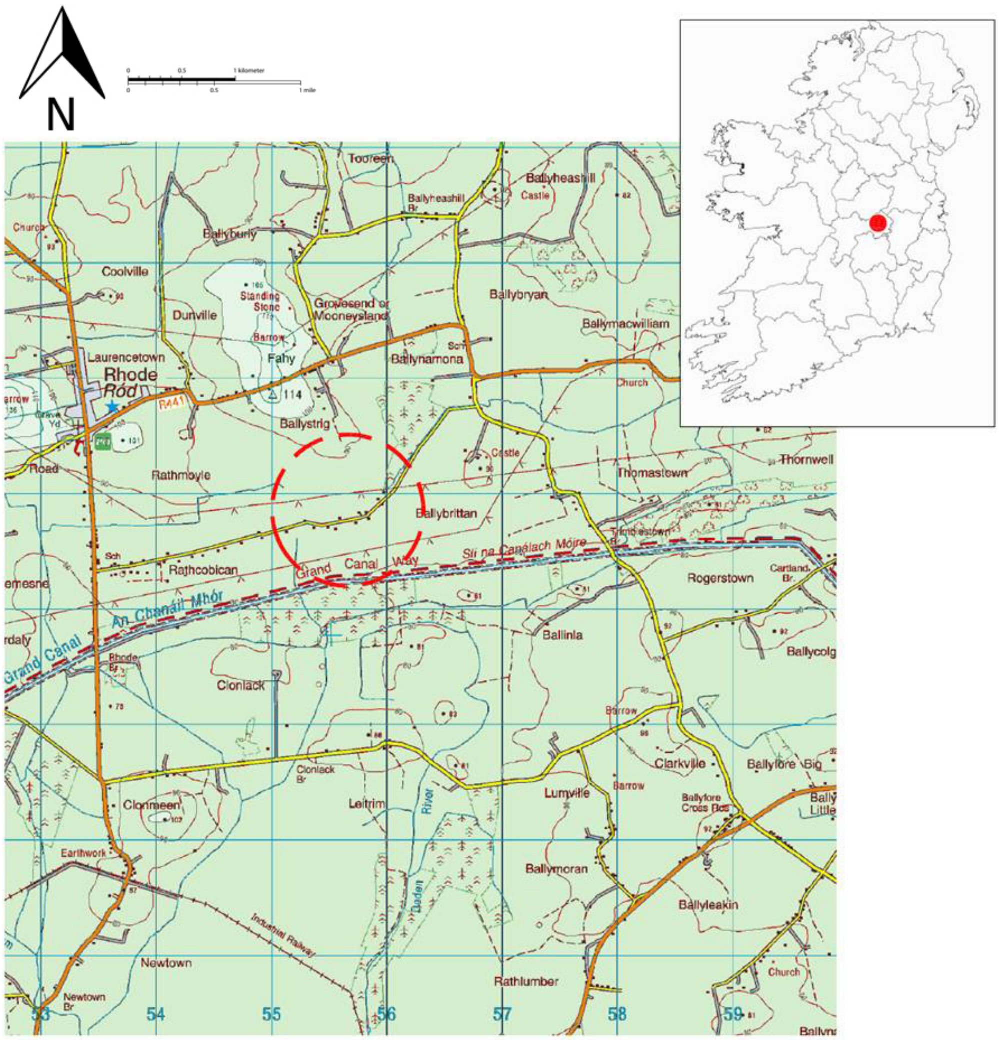


Figure 3.8 Site Location at Rhode, Co. Offaly

The dwellings in this cluster development are arranged in a typical 'ribbon' style layout as shown in the photograph in Figure 3.7 above. Land use in the area is predominately agriculture mainly for tillage and livestock grazing as shown in Figure 3.9 below. There are also extensive areas of forestry to the south and north-east of the study area. The study area slopes gently from north to south with a topographical high of 114 mOD at Ballystrig hill to the north falling to approximately 80 mOD to the south of the study area. Directly in the vicinity of the study area local streams and drains are sparse and any that are present appear to be dry most of the year. The study area is located at the centre of a surface water divide between three catchments; the Yellow River to the north-west, the River Boyne to the east and the Philipstown River to the south. To simplify the surface water catchment can be split into two larger catchments – the Boyne and the Barrow – as the Yellow River eventually joins the Boyne further north and the Philipstown River joins the Barrow further to the south. The Grand Canal flows from west to east directly to the south of the study area.

Figure 3.9 View from downstream borehole at Rhode, Co. Offaly

The study area is located in an area described as being Moderate vulnerability with respect to groundwater. Bedrock aquifers in the area are overlain by low to moderate permeability tills or clays that overlay the bedrock aquifers with subsoil thickness on average of 5 – 15 m present in the area. Bedrock geology in the area surrounding Rhode consists of extensive Dinantian pure bedded limestones which are classified as locally important aquifers which are generally moderately productive – see Figure 3.10. There are no source protection areas in the immediate vicinity of the study area however the Toberdaly local water supply is located to the west.

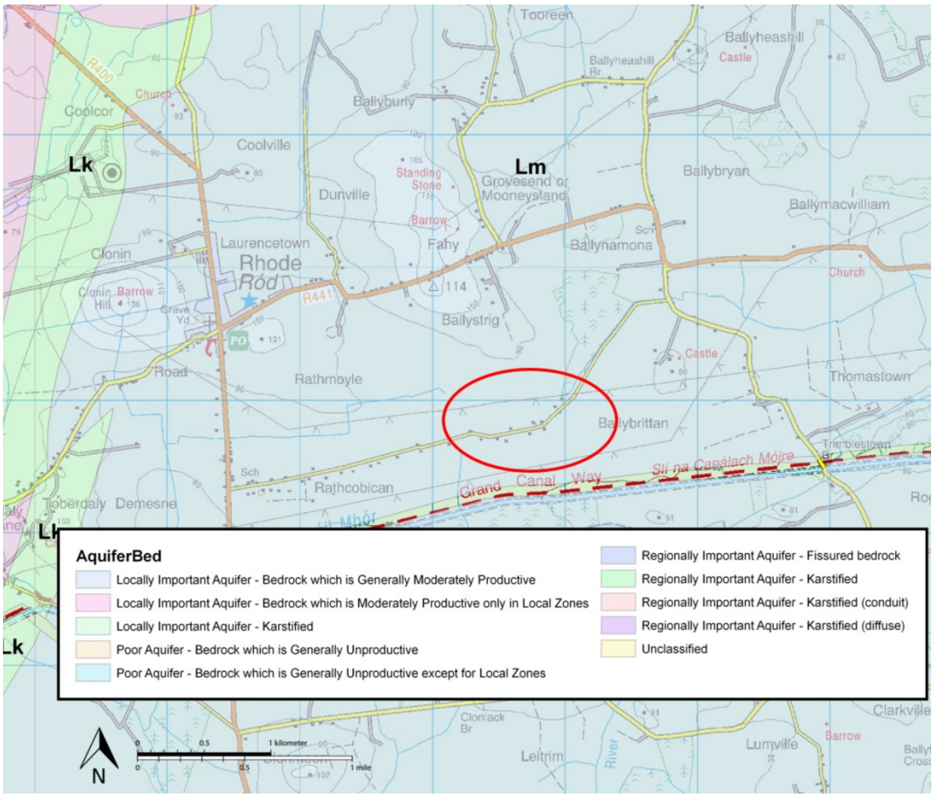


Figure 3.10 Aquifer Classification at Rhode, Co. Offaly

Carrigeen, Co. Kilkenny *(High vulnerability)*

The study area in Kilkenny is located within the village of Carrigeen, Co. Kilkenny near the county border with Co. Waterford as shown in Figure 3.13. The cluster development incorporates a housing development in the village that was built relatively recently with each house having its own treatment system. This study site is unique from the other study areas as it is not the typical 'ribbon' type cluster developments that have grown in huge numbers across rural Ireland in the past number of decades, with 9 dwellings located within a rural cluster housing development similar to small urban housing developments as shown in Figure 3.11 – Figure 3.12 below. This cluster is surrounded by other older dwellings which follow the more typical 'ribbon' style development. This study area provides the 'newest' development of the five study locations, with the majority of the treatment systems being built between 2004 and 2007 with all of these houses containing 'Biocycle' secondary package treatment plants.

Figure 3.11 Study development at Carrigeen, Co. Kilkenny

Land use in the surrounding area is predominately agriculture mainly for livestock grazing. The study area slopes steeply from south to north with a topographical high of 85 mOD at Corluddy Hill to the south falling to approximately 18 mOD to the north of the study area in the direction of Mooncoin. The area is dominated by the River Suir to the south and this forms the boundary between counties Kilkenny and Waterford and the study area lies within the River Suir surface water catchment. Surface water features are sparse in the study area with no visible drains or watercourses. To the east of Corluddy Hill a local stream flows from south to north and joins the Dungooly Stream which then flows from east to west before meeting the River Suir at Ballybrassil.

Figure 3.12 View from downstream borehole at Carrigeen, Co. Kilkenny

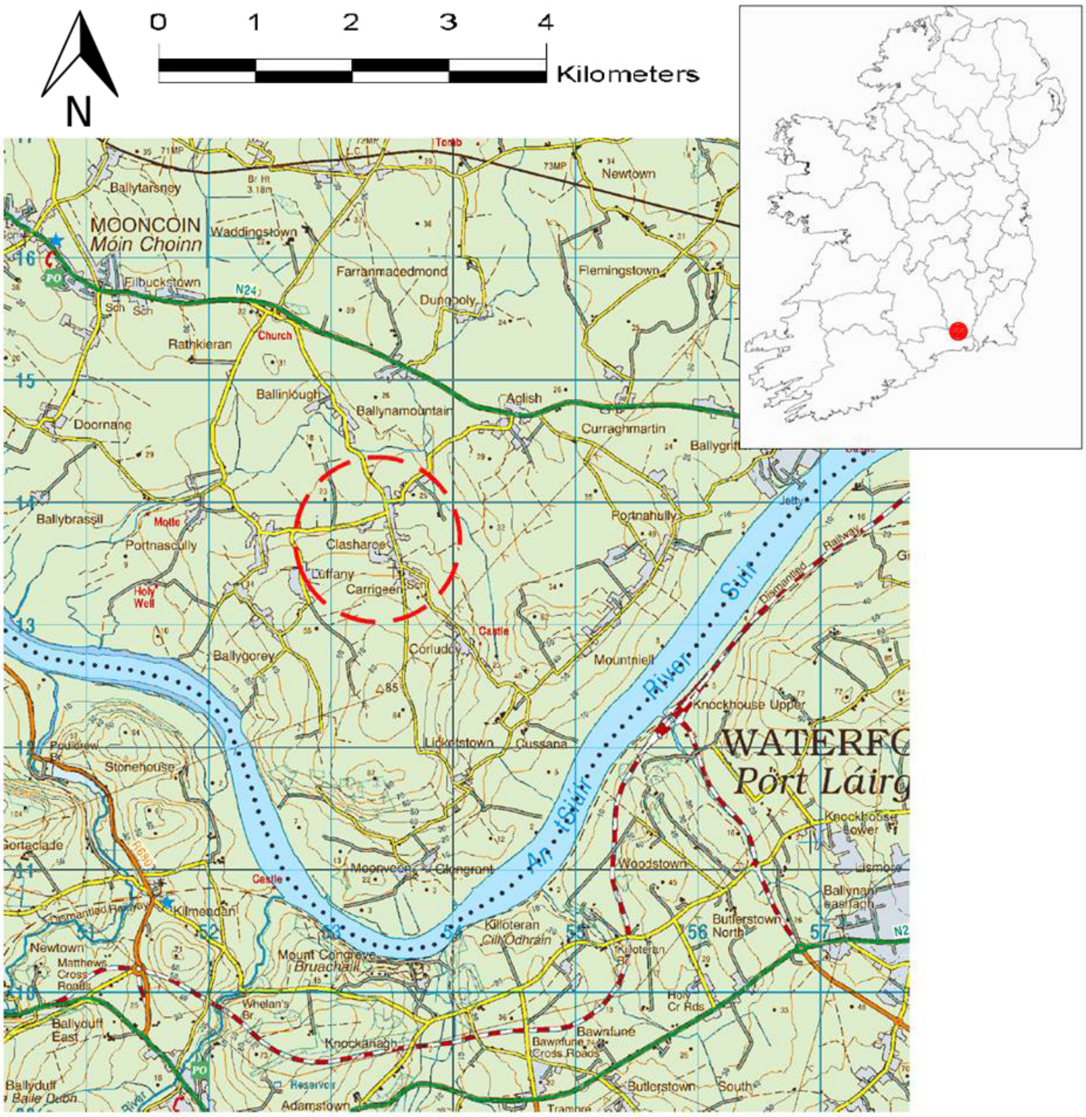


Figure 3.13 Site Location at the Carrigeen, Co. Kilkenny

The study area is located in an area described as being High vulnerability with respect to groundwater. This is due mainly to the presence of shallow higher permeability tills that overlay the bedrock aquifers with thickness of 6 m or less present. As this study area is located at a local topographical high, bedrock geology in the area transitions from Devonian sandstones to the south to Dinantian limestones to the north with a number of bands of intermittent east-west bedrock formations present. Directly beneath the study area is underlain by Kiltorcan-type sandstones. These Devonian sandstones are known to be well faulted and are considered to be one of the best aquifer units present across the country. Consequently the Kiltorcan formation present beneath the study area is classified as being a regionally important aquifer containing fissured bedrock. Further north the limestone formations present are less conductive of water and are classified as locally important which is moderately productive only in local zones. The aquifer classifications in the vicinity of the study area at Carrigeen are shown in Figure 3.14 below. There are no source protection areas in the vicinity of Carrigeen.

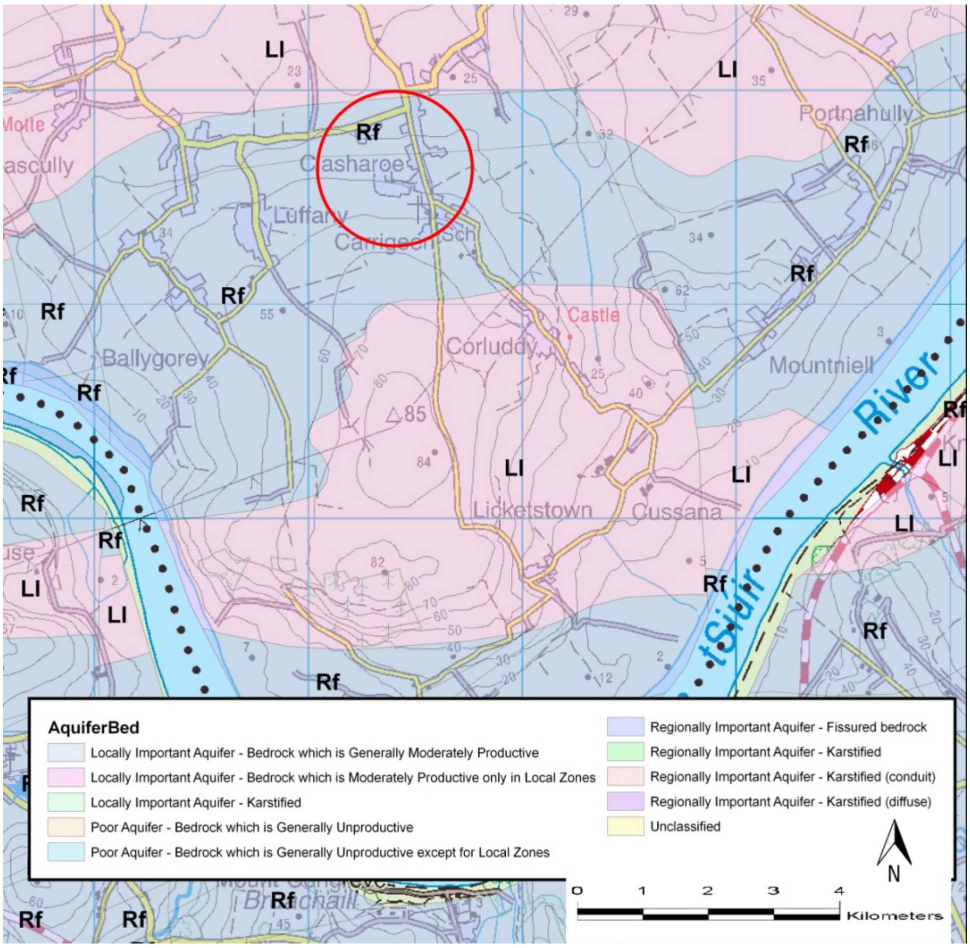


Figure 3.14 Aquifer Classification at Carrigeen, Co. Kilkenny

Faha, Co. Limerick *(Extreme vulnerability)*

The study site at the Faha, Co. Limerick is located approximately 2.5 km from the village of Kildimo in the Shannon estuary region. The cluster development is located along a local road just off the N69 Foynes road and is in the administrate area of Limerick County Council as shown in Figure 3.15 below. The development comprises of dwellings that are constructed in the typical 'ribbon' type layout as shown in Figure 3.16 below.

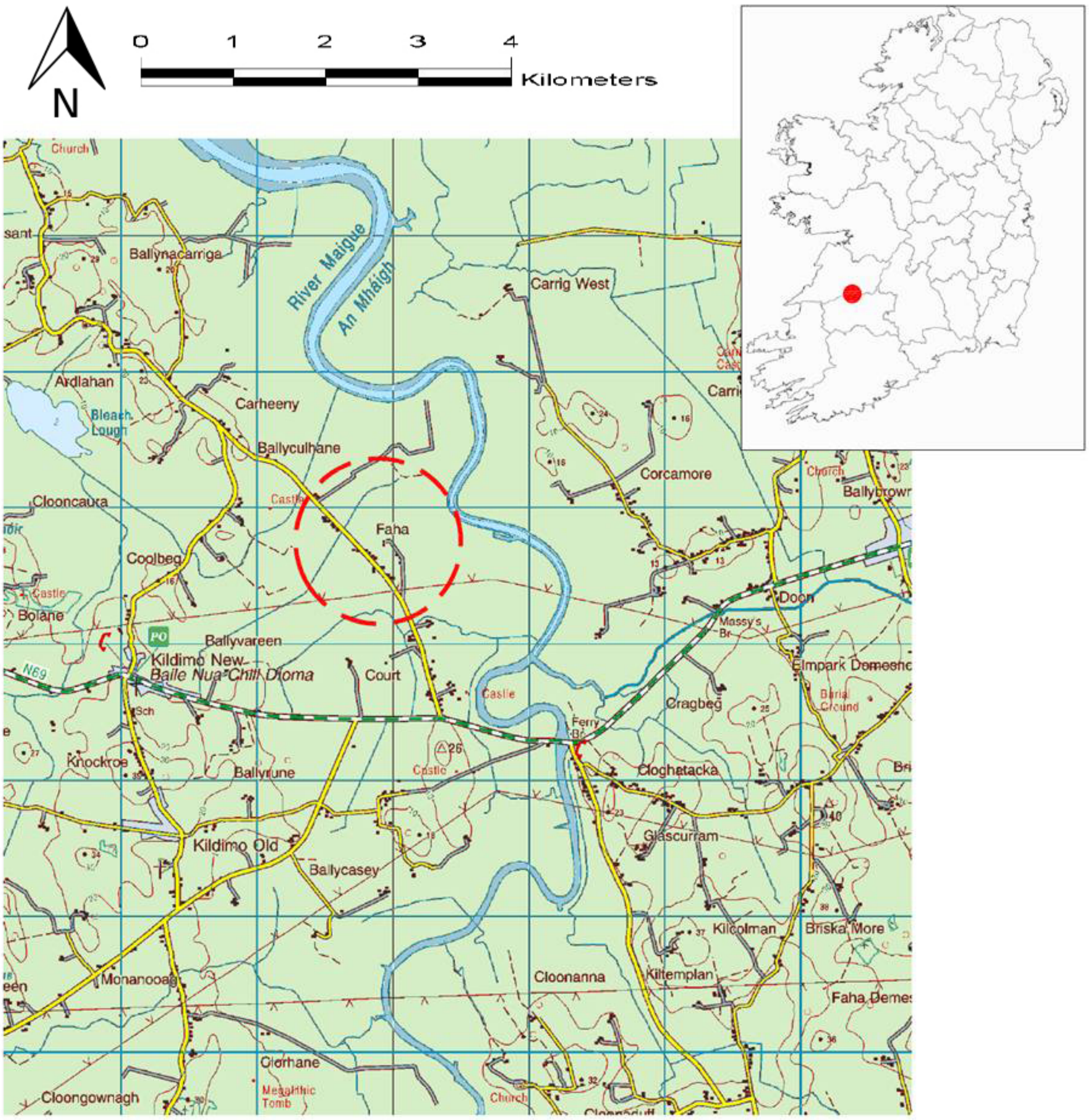


Figure 3.15 Site Location at Faha, Co. Limerick

Figure 3.16 View from downstream borehole at Faha, Co. Limerick

The area is situated in the plains of the River Maigue just 3.5 km upstream from where it joins the Shannon estuary. Land use is predominately agriculture mainly for livestock grazing. The study area is quite flat due to it being situated in a former river flood plain and is generally at an elevation of 3 mOD with a topographical high of 7.4 mOD at a hill located in the centre of the study area.. Flood alleviation work was carried out some 50 years ago and the River Maigue is now heavily embanked along both banks as far upstream as Adare. The entire area is dominated by the River Maigue which meanders gently through the area flowing from south to north. Surface water tends to make it to the Maigue very quickly through a series of shallow drains and streams. Figure 3.17 shows a view along the River Maigue towards Ferrybridge to the west of the study area.

Figure 3.17 View along the River Maigue to the north-east of Faha, Co. Limerick

The study area at Faha is located in an area described as being Extreme vulnerability with respect to groundwater, indicating that bedrock is very close to the ground surface. This is due mainly to the presence of very shallow subsoils that overlay the bedrock aquifers with thickness of 0 – 2 m present. Bedrock is exposed in many locations in the surrounding area and also directly inside the study perimeter. Bedrock in the area consists of Dinantian limestones which are classified as being pure unbedded beneath the study location and pure bedded to the south-east. The GSI lists the aquifer beneath the study area as a regionally important karstified (conduit) aquifer however there are no karst features evident across the local landscape which usually accompanies these conditions. It is assumed that in general the aquifer is not karstified but may be in localised areas where weaker plains of limestone are present. To the south-east the limestone aquifer is classified as being regionally important which is generally moderately productive. The aquifer classifications in the vicinity of the study area at Faha are shown in Figure 3.18 below. There are no source protection areas in the vicinity of the study area at Faha.

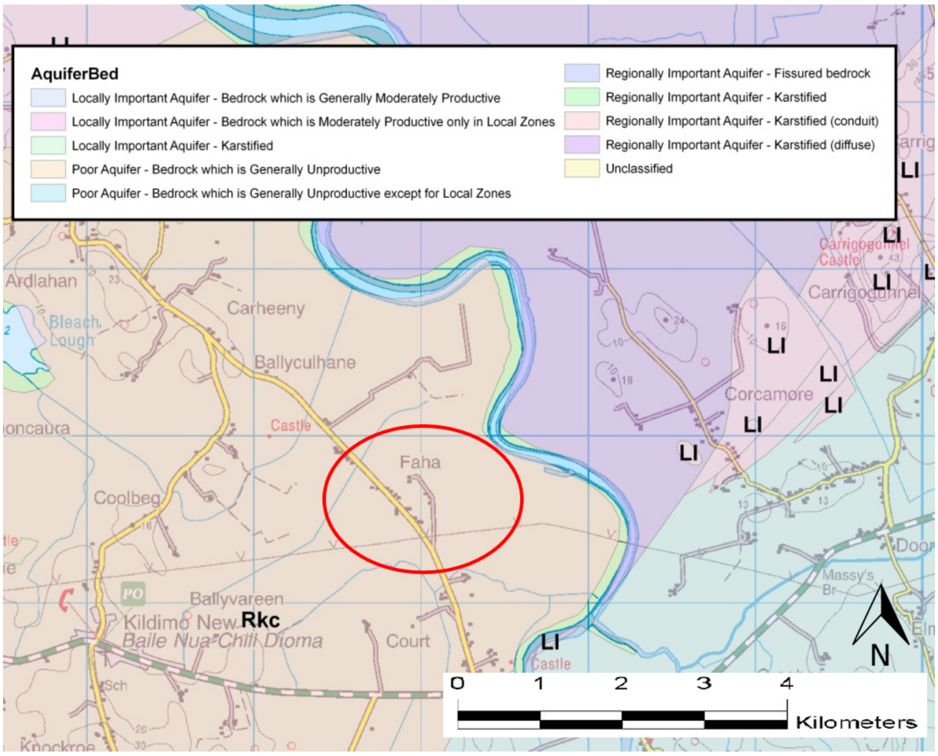


Figure 3.18 Aquifer Classification at Faha, Co. Limerick

Toonagh, Co. Clare *(Extreme vulnerability)*

The study site at the Toonagh, Co. Clare is located approximately 6 km north of Ennis. The cluster development is located on a cul de sac off the R476 Corrofin to Ennis route in the administrate area of Clare County Council as shown in Figure 3.19 below.

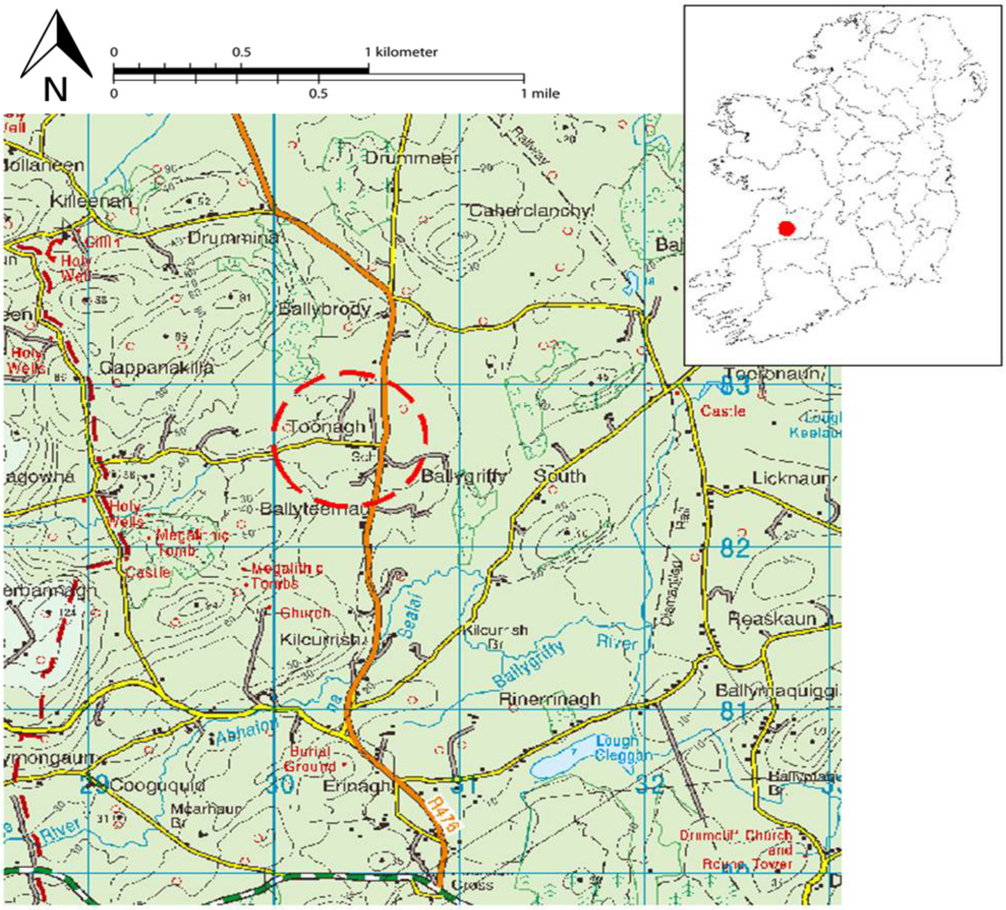


Figure 3.19 Site Location at Toonagh, Co. Clare

This study area is different from the others as the dwellings contained within the rural cluster at Toonagh do not have individual treatment systems that discharge to subsoil – see Figure 3.20. Foul discharge from each of the dwelling is collected into a local sewer which discharges to a small community package treatment plant (PE = 60). The package treatment plant consists of secondary treatment of the foul water followed by discharge of the treated wastewater to swallow hole directly to groundwater. It was obviously thought that this was a considered a safe and convenient method for disposal of the treated wastewater when the plant was constructed in the mid 1970's with the practice occurring quite regularly during this time period in Ireland. Following consultation with GSI records for the area and with members of Clare County Council, it is believed that the swallow hole to which the treatment plant discharges is connected through a series of underground conduit or karst flow systems to a local stream which flows underground to the south of the area reappearing at a local spring. Proof of this connection was one of the elements of this study, as detailed later in Chapter 5.

Figure 3.20 The study development at Toonagh, Co. Clare

The area is within the surface water catchment of the River Fergus which rises to the north of Corrofin and flows through Ennis and into the Shannon estuary to the south. The area is punctuated by many surface water features such as drains, streams and rivers; however all of the surface water features are affected by the dominant karst groundwater system in the area and the surface water hydrology is therefore quite complex. Many of the local streams and rivers disappear into sinkholes and reappear meters or in some cases kilometres away. Depending on rainfall and water levels throughout the area, rivers may also vary from being losing rivers, where river water flows to groundwater through the river bed, to gaining rivers, where the rivers are fed by groundwater (Deakin, 2000). One of these local streams, the Toreen East, disappears into a sinkhole to the west of the study area and reappears to the south-east at the Kilcurrish spring where it then joins with the Shallee River. The Shallee River then joins with the Ballygriffy River and eventually meets the River Fergus at Ballyalia Lough approximately 3.5 km to the south-east. The area is classified as being Extreme vulnerability with respect to groundwater and this is due both to the shallow depths of subsoil that are present in the area (0 – 2 m on average) and the karst nature of the underlying aquifers. Groundwater can move very quickly in karst groundwater systems and thus any pollutants entering the system can migrate a great distance and cause problems downstream in a very short time period. Bedrock in the area consists of Dinantian pure bedded limestones and is karstified with many karst features clearly visible across the local landscape.

The area lies within the inner zone of the protection for the Drumcliff springs public water supply as shown in Figure 3.21. The Drumcliff springs provide public drinking water for the town of Ennis and the surrounding area serving a population of up to 40,000. The supply comprises two springs within 20m of each other located on the northern bank of the River Fergus at Drumcliff just to the north of Ennis town and provides 12,000 m^3/d of drinking water. The springs are regularly bacterially contaminated and this is due to the many point and diffuse sources of contamination within the catchment which can enter the system with very fast travel time to the springs during periods of high rainfall and high water levels (Deakin, 2000).

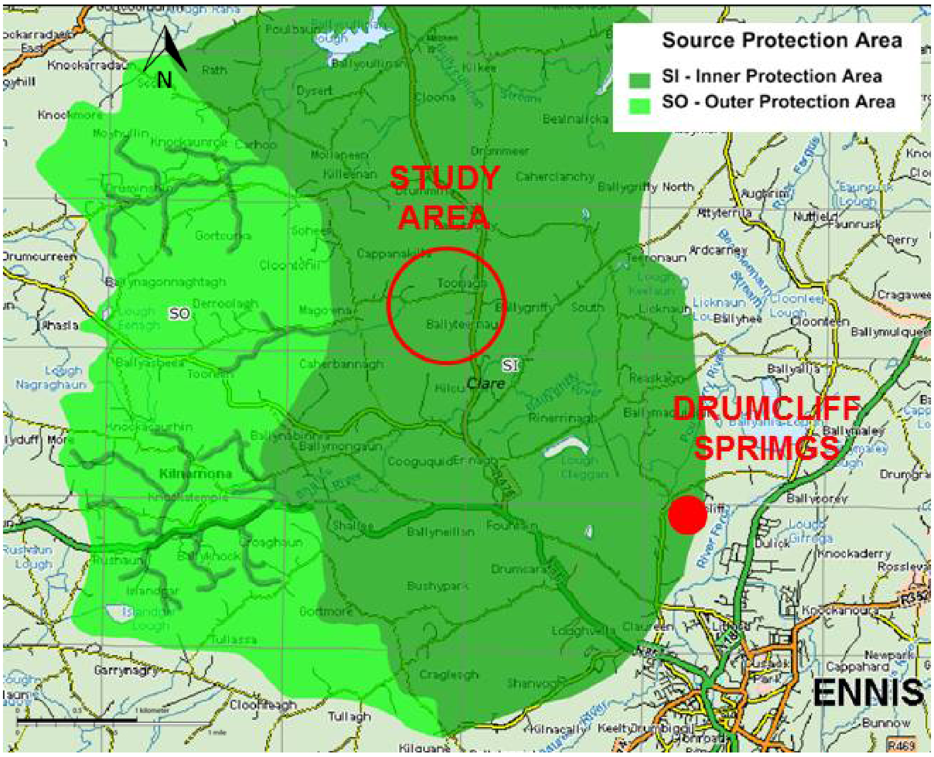


Figure 3.21 Drumcliff Springs Water Supply Zones of Protection

The entire area in the vicinity of the Toonagh is classified as being a regionally important aquifer which is karstified and incorporates conduit flow – see Figure 3.22 below. It can be difficult to extract a water supply, local or otherwise, from a karst aquifer due mainly to the dominance of conduit flow whereby failing to connect with a conduit can lead to very poor yields. Groundwater abstractions in these areas tend to rely on natural springs and this is evidenced by the Drumcliff Springs that were discussed earlier.

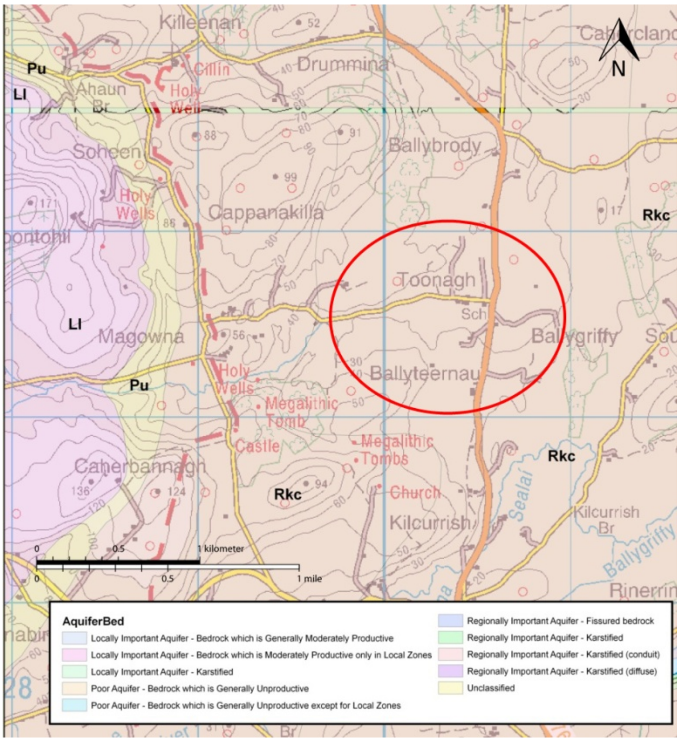


Figure 3.22 Aquifer Classification at Toonagh, Co. Clare

3.3.2 Meteorology

Meteorological data was required for both the analysis of the field results and also later for the numerical modelling. Due to limitations with the project budget, it was not possible to install dedicated weather stations at each of the study locations and therefore it was necessary to rely on Met Eireann data for the closest or most appropriate adjacent observation station. Met Eireann operate three type of weather stations; synoptic weather stations which monitor the full range of meteorological parameters on an hourly basis, climatological stations, which monitor meteorological parameters on an daily basis and rainfall stations which record rainfall on a daily basis. The most common stations across the country are rainfall stations however as these are largely privately operated there can be a significant lag time between collection of the data and the publishing of the data with Met Eireann for public use due to quality control and validation procedures. In this regard it was not possible to acquire all the relevant rainfall data for each of the study sites from the closest rainfall stations. In these instances it was necessary to approximate suitable values based on the closest automatic recording station; this will be discussed in more detail in Chapter 6.

The closest rainfall station to the study area at the Naul was located at Bellewstown, 8.9 km to the north of the study site. 20 km to the south at Dublin Airport there is a fully automatic synoptic recording station. There was also a local private rainfall recording station located in the village of the Naul and records were acquired from this source and compared to available records at Dublin Airport and Bellewstown for the purposes of this study. The location of these rainfall stations is shown in Figure 3.23 below.

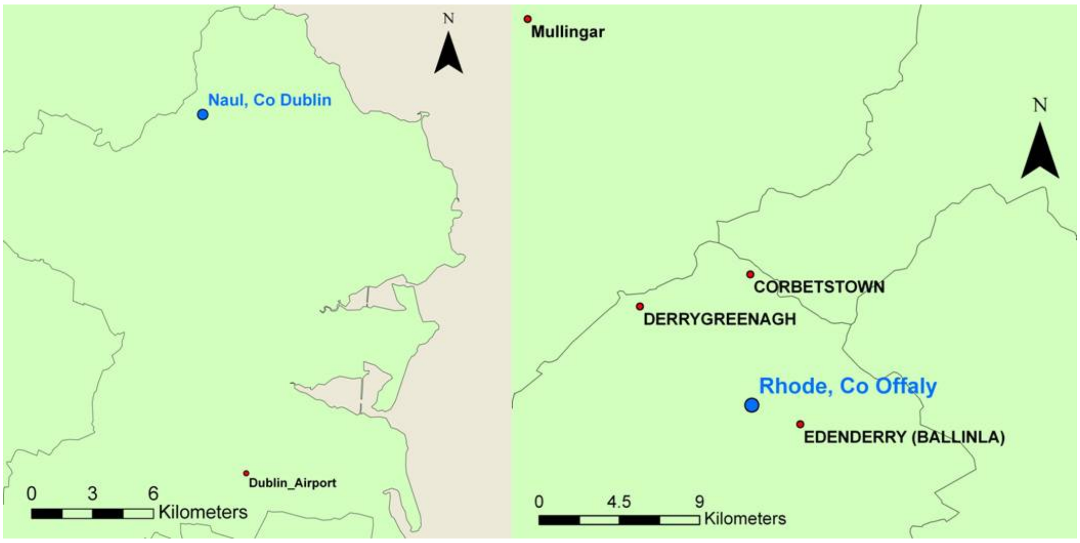


Figure 3.23 Met Eireann Stations in the vicinity of (a) The Naul (b) Rhode

There were two Met Eireann stations within 7 km of the study area at Rhode; a rainfall recording station to the west at Derrygreenagh and a climatological station to the east at Edenderry – see Figure 3.23. Rainfall and other climate data were acquired from the station at Edenderry.

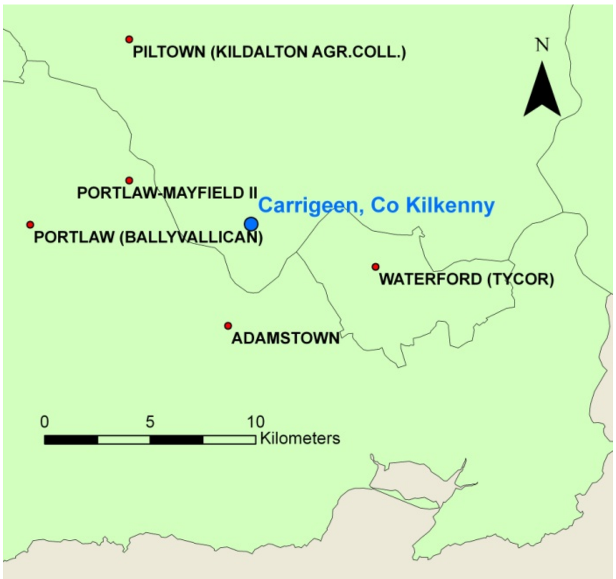


Figure 3.24 Met Eireann Stations in the vicinity of Carrigeen, Co. Kilkenny

There were no obvious Met Eireann recording stations within the locality of the study area at Carrigeen that would provide data that is analogous to the study area. However, to the south and south-west there were two stations; Waterford, a climatological station and Adamstown, a rainfall recording station. However these stations were located to the south of a topographical high at Corluddy Hill and this may reduce the similarity with conditions to the north at Carrigeen. To the north-west there were two rainfall recording stations located at Portlaw (Mayfield and Ballyvallican) and further to the north a climatological station at Piltown. Data was initially acquired from all of these stations and investigated for major differences. Figure 3.24 shows the locations of these Met Eireann stations in relation to the study area.

There were three Met Eireann climatological stations and one rainfall station in the vicinity of the study area at Faha as shown in Figure 3.25. Both the Adare and Shannon Airport stations are located within 7 km of the study site with Castleconnell station located some 13 km away to the north-east. A rainfall recording station was located at Patrickswell which is 5.9 km to the south-west of the study area. Records were acquired Adare, Shannon Airport and Patrickswell and compared to find the most appropriate data for the purposes of this study.

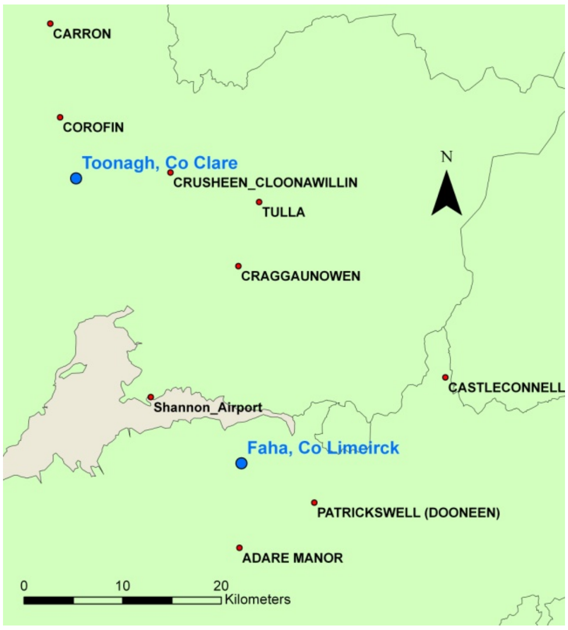


Figure 3.25 Met Eireann Stations in the vicinity of (a) Toonagh and (b) Faha

Finally, rainfall data was available from the Met Eireann rainfall recording station located at Corrofin 6 km to the north of the study area at Toonagh. The closest climatological station was at Shannon Airport some 25 km to the south as shown on Figure 3.25.

3.3.3 Occupancy and Hydraulic Loading Rates

The EPA Code of Practice (CoP) for Wastewater Treatment Systems for Single Houses (EPA, 2009) calculates the typical daily hydraulic loading to an on-site system for single houses as 150 L per capita. This figure was reduced from 180 L per capita from the previous EPA guidance documents based mainly upon research carried out by Gill et al., (2005). This field research carried out on four sites in Ireland, showed domestic wastewater generation measured on all sites to be considerably less than the EPA figure, with observed per capita hydraulic loading rates of between 60 – 100 L/d. However, given that each of the study areas in this research contain treatment systems of differing construction dates and can generally be assumed to be built to very different design standards, the higher value of 150 L per capita per day is most likely a representative hydraulic loading rate to the on-site systems.

All of the study areas contain treatment systems that were built up to 20 – 30 years ago; many are even older. It is therefore likely that the majority of the treatment systems will have been built according to the S.R.6 EPA guidance document (NSAI, 1991) which was first published in 1975 and was then amended in 1991. Systems built before the introduction of S.R.6. (pre 1975) may have included soak pits instead of percolation areas and may have incorporated poor site construction practices such as rainwater being combined with foul water and discharging to the septic tank. It is also likely that many systems built after 1975 may also have been poorly constructed with the most common practice being the inclusion of a single percolation trench instead of the appropriately designed set of multiple adequately sized percolation trenches. The practice of including rainwater in the foul system is also assumed to have continued even with the introduction of the S.R.6. guidelines. Given the above it is likely that different systems will all have varying hydraulic loads associated with them depending on the construction period. An estimation was therefore made for the most appropriate hydraulic loading rates, occupancy rates and the percolation area size for each residential dwelling contained within the cluster developments. Following a review of both of the S.R.6. documents, the EPA Manual for treatment systems for single houses (EPA, 2000) and the EPA Code of Practice (2009) combined with research by Gill et al. (2005); an assumption has been made regarding appropriate hydraulic loading rates as set out in Table 3.2 below. Occupancy will be assumed as 3 persons per dwelling (CSO, 2012) for all cluster developments which is consistent with all of the available guidance documents. The numbers of each type of system outlined above and given in Table 3.3 and are summarised by each study area. This will be considered in more detail in Chapter 7 during the vadose zone modelling where an occupancy figure of 4 persons per dwelling was decided upon in order to assume a "worst-case" scenario.

Table 3.2 On-site Treatment System Design Standards and Loading Rates for a typical 4 PE house

Treatment System	Construction Standard	Percolation Area* (m^2)	Wastewater (L/capita/d)	Hydraulic Loading Rate ($L/m^2/d$)
Septic Tank	Pre 1975	20**	220	44
	S.R.6. (1975)	58	180	12.4
	S.R.6. (1991)	44	180	16.4
Septic Tank	EPA (2000)	36	180	20
Secondary	EPA (2000)	32	180	22.5
Septic Tank	EPA CoP (2009)	32	150	18.75
Secondary	EPA CoP (2009)	30	150	25

*Based on trench width and length of trench required
**Assumed as conservative estimate based on possibility of soak pit being installed

Table 3.3 Treatment System Breakdown for each study area

	Number of Systems Present					
	Septic Tank			Secondary Treatment		
Location	Pre 1991	EPA 2000	EPA 2009	Pre 1991	EPA 2000	EPA 2009
Naul, Co. Dublin	17	1	0	0	3	0
Rhode, Co. Offaly	10	0	0	0	1	0
Carrigeen, Co. Kilkenny	1	2	0	0	14	0
Faha, Co. Limerick	12	1	0	0	6	1
Toonagh, Co. Clare	2	0	0	*60 PE Secondary treatment plant*		

The expected contaminant concentrations for both conventional septic systems and secondary treatments systems have been summarised by the EPA (2013) and are given in Table 3.4 below. This will be discussed further when developing the vadose zone model in Chapter 7.

Table 3.4 Typical Pollutant Concentrations from On-site Systems (EPA, 2013)

Pollutant	Conventional Septic Tank	Secondary Treatment System
Faecal Coliforms	> 1 million/100ml	> 5 – 10,000/100ml
Nitrogen (mg/l N)	30 – 80	20 – 35
Phosphorus (mg/l P)	5 – 20	1 – 5
BOD (mg/l)	150 – 500	20 – 50

3.3.4 Treatment System Density

The ultimate aim of this research is to establish if the density of on-site treatment systems has an impact on groundwater quality. Local Authorities decide whether a system should be allowed during the planning application process and their planning policy will dictate the density of these systems. The Department of the Environment provide general guidelines on planning policy; however each relevant Local Authority usually addresses the issue of cluster developments in their own differing manner and in many cases with very different approaches. During the site selection process for this study it was found that whilst most Local Authorities have general policies on rural clusters contained within their county development plan, many had their own very 'individual' methods for determining whether density of on-site treatment systems will cause adverse effects for groundwater, as discussed in Chapter 2. The combined result of these circumstances is that there is no consistency at country level regarding plot size for one-off developments or for density of on-site systems within cluster developments.

For each of the study locations, an estimate has been made as to the 'extent' of the area contained within the cluster development and this will be discussed in more detail in Chapter 4. An average plot size has also been estimated for each location. Based on the number of systems within the cluster and the overall area the following general estimates for treatment system density has been calculated as outlined in Table 3.5.

Table 3.5 Density of On-site Systems at Study Areas

Site Location	Average Plot Size (hectare)	No. Units in Cluster Development	Approx. Area of Cluster (hectares)	Treatment System Density (Units/hectare)
The Naul, Co. Dublin	0.35	21	11.6	1.82
Rhode, Co. Offaly	0.30	11	10.6	1.04
Carrigeen, Co.	0.18	17	6.9	2.44
Faha, Co. Limerick	0.20	20	9.9	2.04

Figure 3.26 Treatment Plant with view of rotating media disc (insert) at Toonagh, Co. Clare

As discussed earlier, the study area at Toonagh, Co. Clare differs from the others as each of the dwellings contained within the cluster do not have their own private treatment system and all of the houses discharge their wastewater to a larger decentralised wastewater treatment system which has a Population Equivalent (P.E.) of approximately 60. This small community size treatment system comprises a primary settlement tank with secondary treatment by way of a Rotating Biological Contractor (RBC) as shown in Figure 3.26. Loading rates for the treatment plant were determined through consultation with Clare County Council records and typical outflow effluent pollutant concentrations determined through a number of samples that will be analysed during the sampling period of the project.

3.4 Geology

3.4.1 Geology – Site at the Naul, Co. Dublin

Bedrock Geology

Bedrock in the vicinity of the study area is quite complex as given in Figure 3.27. To the south-west there are a number of complex faults and folds and this trend also exists both to the north and north-east of the study area. Bedrock geology in the immediate vicinity of the study area is slightly more straightforward. To the south and in the vicinity of Knockbrack Hill, the area is underlain by the Walshestown formation (WL) comprising Shales, thin sandstones/siltstones with occasional thin limestones. Moving northwards the Balrickard formation (BC) forms a thin band of coarse micaceous sandstone with shale interbeds and this is then bounded by a band of shaly limestone, with bands of brown limestone. This shaly or 'Calp' limestone forms the Loughshinny formation and it is from this formation that the local Bog of the Ring groundwater supply is extracted via 4 production wells. Bedrock in the area has been described extensively by the GSI in the Bog of the Ring Groundwater Protection Zone Report produced in 2005 (GSI, 2005).

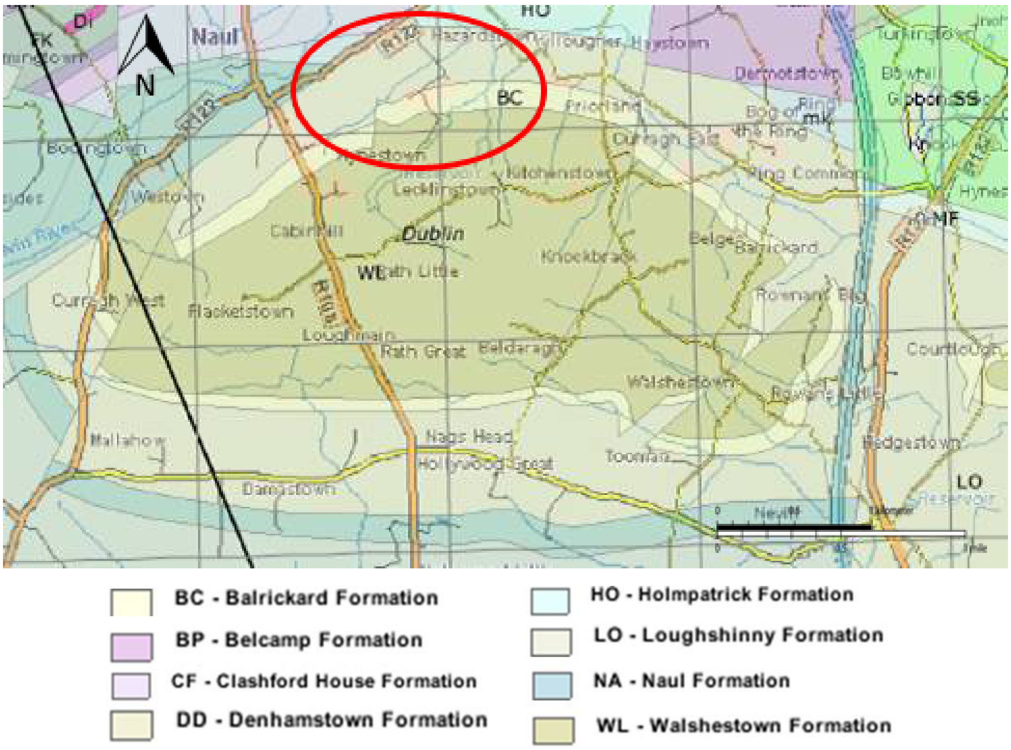


Figure 3.27 Bedrock 100k Solid Geology of the Area at the Naul, Co. Dublin

Subsoils (Quaternary) Geology

Subsoils in the area surrounding the study site at the Naul, Co. Dublin are dominated by sandstone and shale tills with matrix of Irish Sea Basin origin (IrSTLPSsS) as shown in Figure 3.28 below. To the south tills are also present however these tills are derived from shales and sandstones (TNSSs). There are also rock outcrops present to the south of the study area. Local Authority planning application records were consulted for this area in order to identify the BS 5930 classifications that were observed. Soils in the area have generally been classified as CLAY.

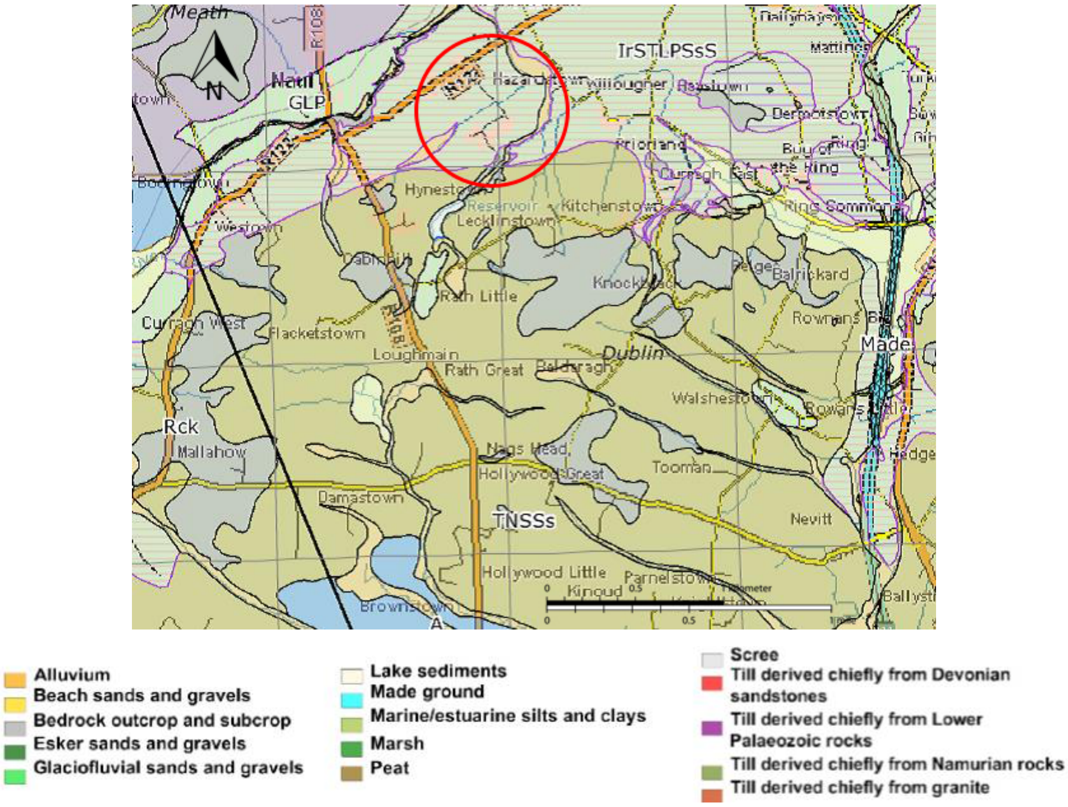


Figure 3.28 Subsoils map of the study area at The Naul, Co. Dublin

Both the of the till subsoil units that dominate the area surrounding the study site are 'clayey' in texture and are therefore are expected to display low permeability properties.

Soils

Soils in the study area are described as Grey Brown Podzolics. An individual county map has not been completed for Dublin, however soils and subsoils were described by Meehan (2004). The parent materials of soils in this area are limestone glacial till to the south and Irish Sea origin with Limestone and shale to the north. These soils are not generally permeable and lead to high runoff rates which are evidenced by the presence of a large network of drains and streams in the area..

Depth-to-bedrock

Depth-to-bedrock is greater than 10 m beneath and to the north of the study site. To the south and in the vicinity of Knockbrack hill, bedrock is much shallower and is exposed in areas. 3 No. boreholes were drilled in the area during the course of this study and bedrock was encountered at depths of 15 m, 35 m and 40 m BGL with details of the associated borehole logs given in Section 4.2.1. A large amount of bedrock data was available from the 2005 GSI study with data available for a number of production and monitoring boreholes. This information together with the draft depth-to-bedrock GSI map (see Figure 3.29) were combined to produce a contoured map of generalised depths to bedrock for the surrounding area.

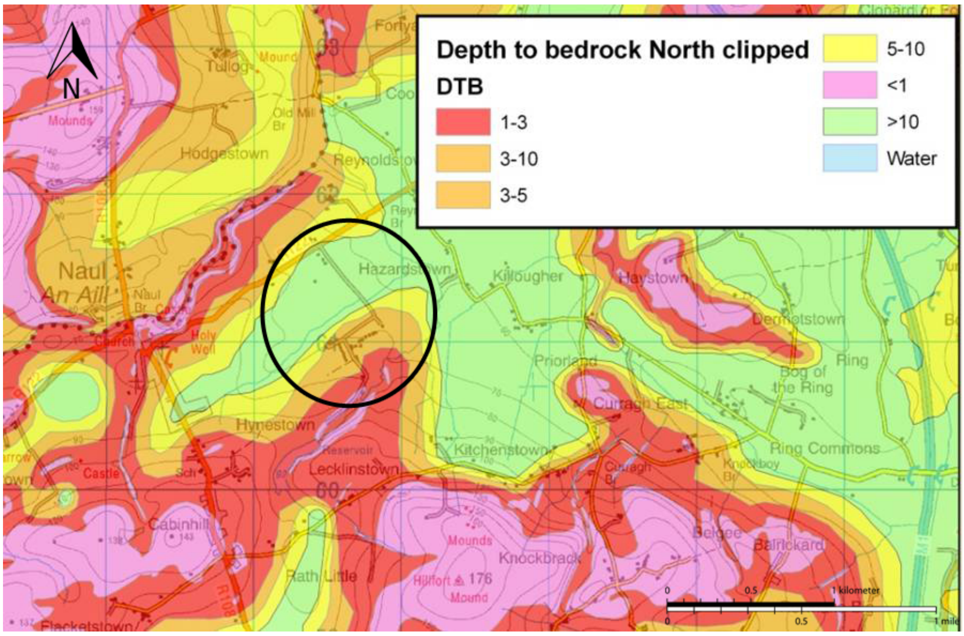


Figure 3.29 Depth-to-bedrock map of the study area at the Naul, Co. Dublin

3.4.2 Geology – Site at Rhode, Co. Offaly

Bedrock Geology

The area surrounding the study area at Rhode, Co. Offaly is underlain by the Edenderry Oolite formation (AWed). This limestone has been described as medium-dark grey, coarse grained oolitic limestone (Oolitic limestones are composed of small spherical grains) which is generally well bedded (Hudson, 1996) – see Figure 3.30. To the west there is a north-south trending outcrop of the Allenwood limestone formation (AW) approximately 500 m wide. The Allenwood limestone is dark grey, coarse grained and crystalline with a well-developed bedding and jointing network.

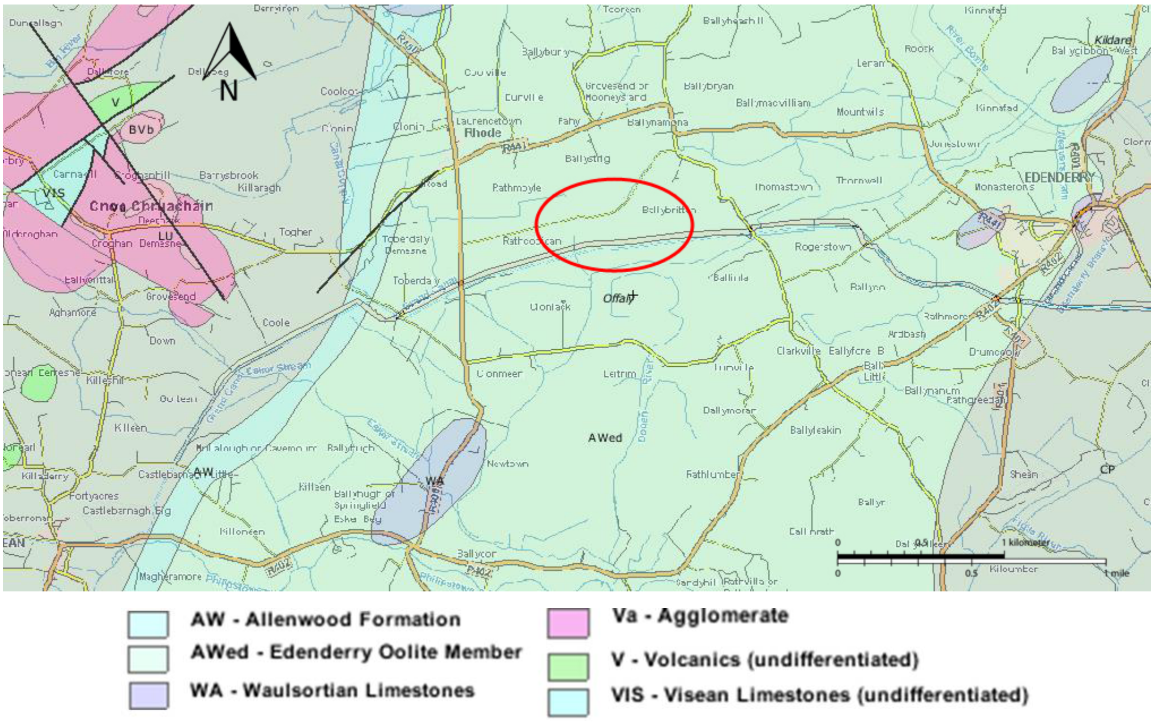


Figure 3.30 Bedrock 100k Solid Geology of the Area at Rhode, Co. Offaly

Further west there is an extensive area of Calp limestone of the Lucan formation (LU) with outcrops of volcanic rocks surrounding Croghan Hill. To the south-west there is an outcrop of Waulsotian limestone (WA) which is generally clean, pale grey and massive.

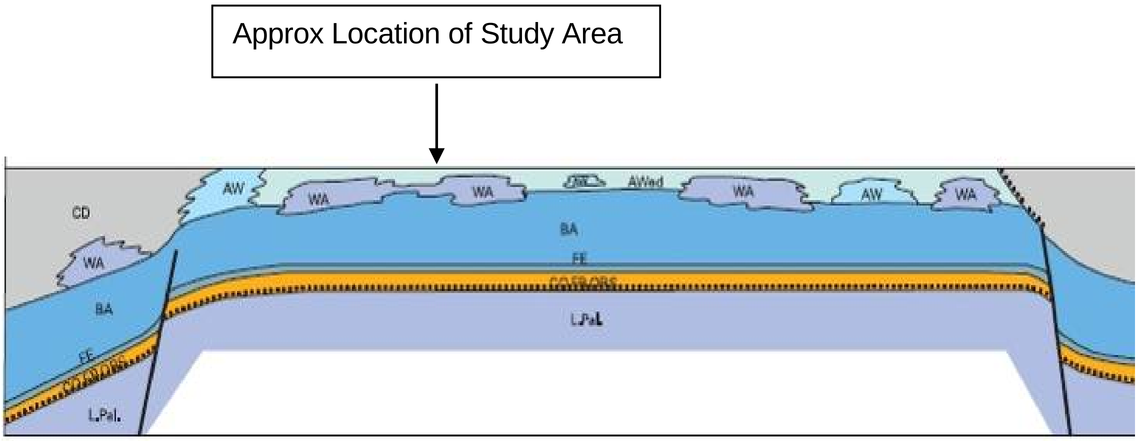


Figure 3.31 Geological cross-section adapted from Sheet 16 (GSI, 1994a)

This Waulsotian limestone (WA) is thought to extend in a number of areas at varying thicknesses beneath the Edenderry Oolitic formation as shown in Figure 3.31. To the east another formation of Calp limestone outcrops quite extensively.

Subsoils (Quaternary) Geology

Subsoils in the area surrounding the study location are classified as tills derived chiefly from Carboniferous limestone. These tills are moderately permeable due to the high content of sands, gravels and larger cobbles and boulders. In contrast to this, areas surrounding the study location dominated by peat or cutaway peat subsoils which are low permeability and tend to be marshy and poorly drained. To the north and east there are alluvium deposits along the course of the Yellow and Boyne rivers – see Figure 3.32. Local Authority planning application records were consulted for this area in order to identify the BS 5930 classifications that were observed and a large database of records were available for this area. Soils in the area appear to vary spatially over short distances, however most records available contained CLAY in some combination with either silts, sands or gravels.

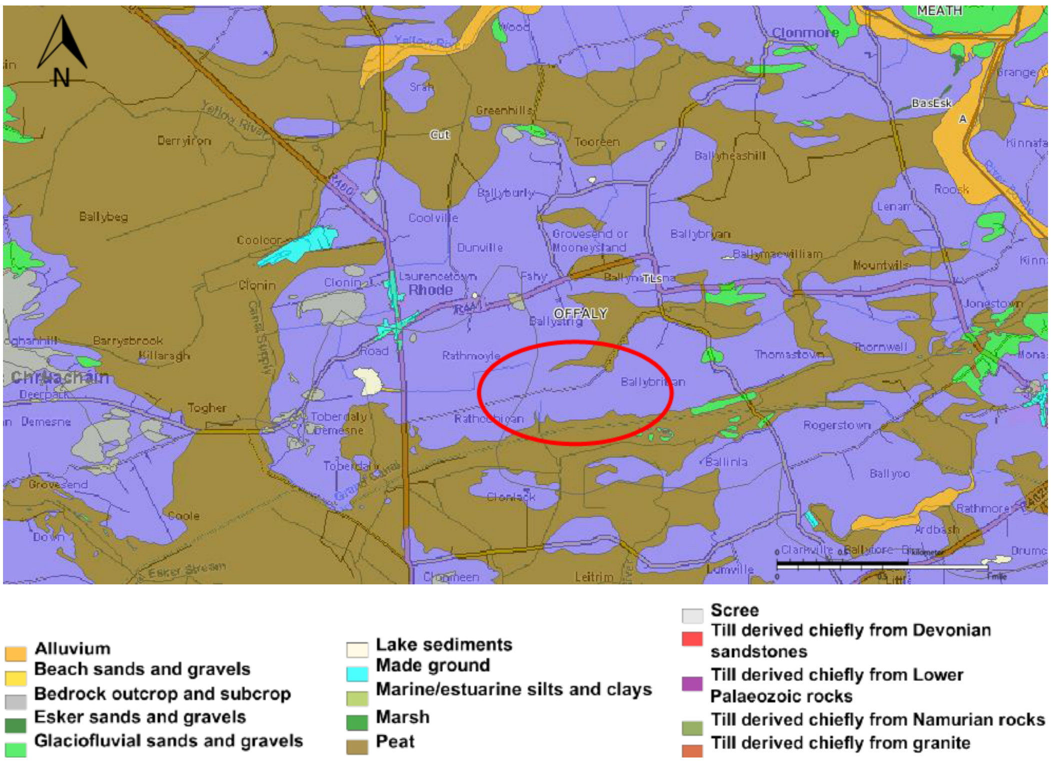


Figure 3.32 Subsoils map of the study area at Rhode, Co. Offaly

Soils

Soils in the study area are dominated by the Elton series which is a Grey-brown Podzolic whose parent material is calcareous drift composed mainly of limestone with some sandstone and shale. The Elton series in this area also contains pockets of Bouldery Phase and Ballintemple series soils both of which are very similar to the Elton series soils. To the north and south the Elton series is bounded by the Banagher Deep Phase peat whose parent material is humified fen peats composed of variable amounts of sedges, mosses, reed and wood remains. To the east and west the Elton series is bounded by raised bog which has been both milled and machined extensively.

Depth-to-bedrock

Bedrock is overlain by thick deposits of tills in the area surrounding the study site. Depth-to-bedrock in the area is on average greater than 10 m, with bedrock typically greater than 20 m below ground level (BGL). 3 boreholes were drilled in the area during the course of this study and rock was encountered at depths of 30 m, 11 m and 17 m BGL, and details of the associated borehole logs are given in Section 4.2.1. Geological Survey of Ireland (GSI) records for the area were also consulted and a number of depth-to-bedrock records were available in the vicinity of the study area with at least 6 of these records in close proximity to the northern boundary of the study area. This information was combined with the draft depth-to-bedrock GSI map (see Figure 3.33) and refined locally with the additional information available from the 3 TCD boreholes in order to produce a contoured map of generalised depths to bedrock for the surrounding area. Depth-to-bedrock decreases to the north of the study area in the vicinity of the Grand Canal and this is most likely due to the presence of a transition zone between the till and peat subsoil boundaries.

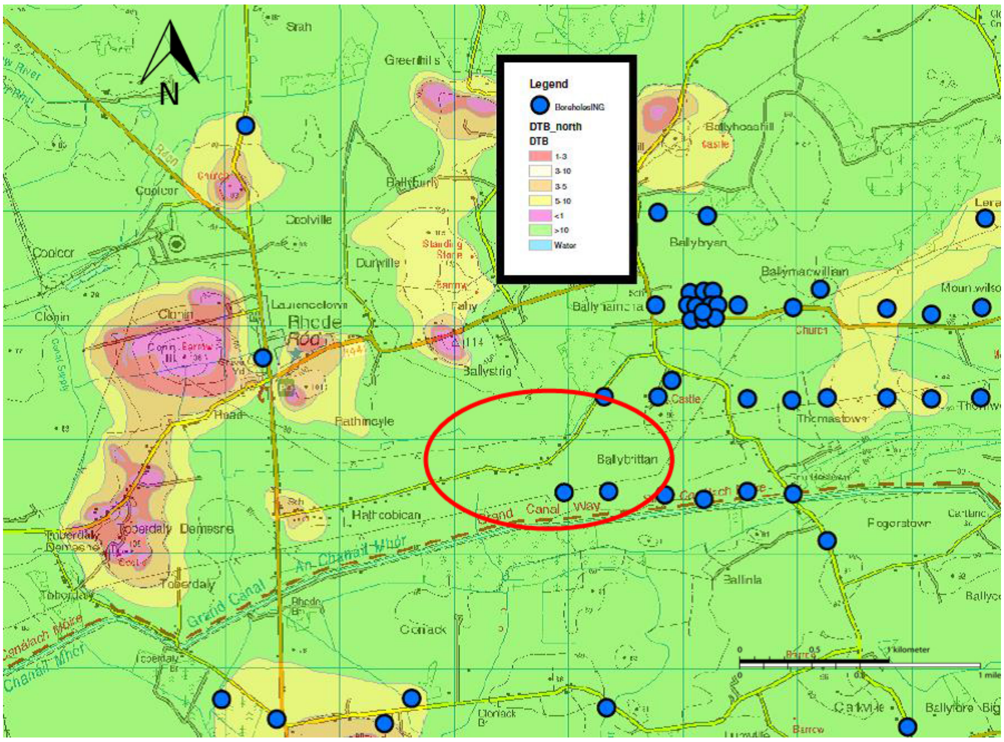


Figure 3.33 Depth-to-bedrock map of the study area at Rhode, Co. Offaly

3.4.3 Geology – Site at Carrigeen, Co. Kilkenny

Bedrock Geology

Bedrock geology in the vicinity of the study area at Carrigeen, Co. Kilkenny is complex with many rock formations cropping out, faults present and bedrock geology changing a number of times within 500 – 1500 m (see Figure 3.34). In order to fully understand the geology of the area it is possible to simplify the geology of the area in terms of rock age.

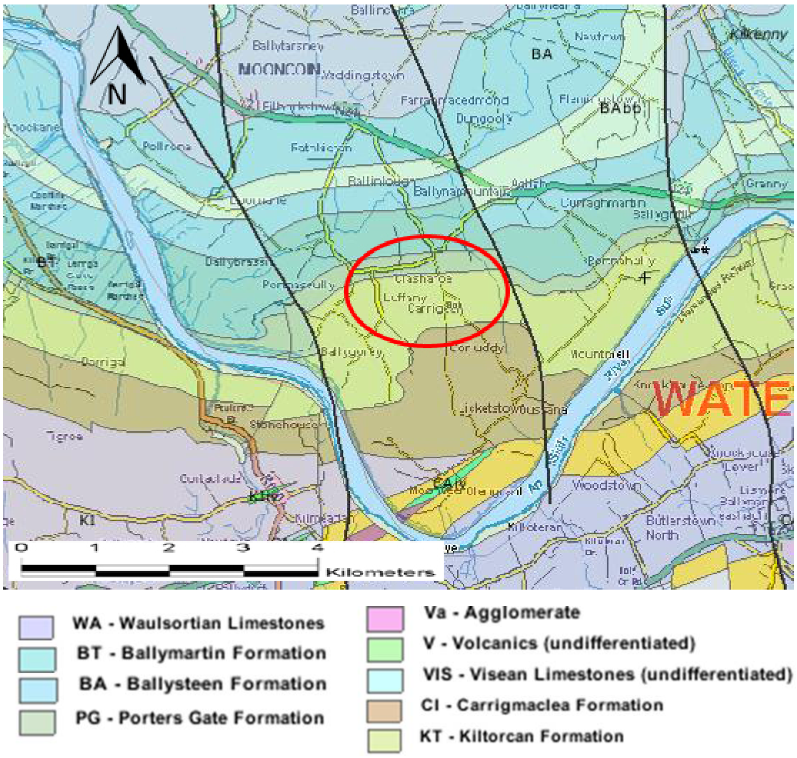


Figure 3.34 Bedrock 100k Solid Geology of the Area at Carrigeen, Co. Kilkenny

To the south of the River Suir are older Lower Palaeozoic rock outcrops of the Ordovician period including Slates, Siltstones and Volcanics. Further north and beneath the study area there is an outcrop of Upper Palaeozoic Devonian rocks chiefly Old Red Sandstone (ORS). This outcrop, mainly comprising of the Carrigmaclea and Kiltorcan formations, effectively forms a belt around a marine shelf or ramp in a depression which enabled the formation of Carboniferous rocks including limestones, mudstones and siltstones. Bedrock in this basin consists mainly of limstone with Cherty, Calcareous shale and Waulsotian limestone formations all present. The cross-section shown in Figure 3.35 illustrates the surrounding geology more clearly.

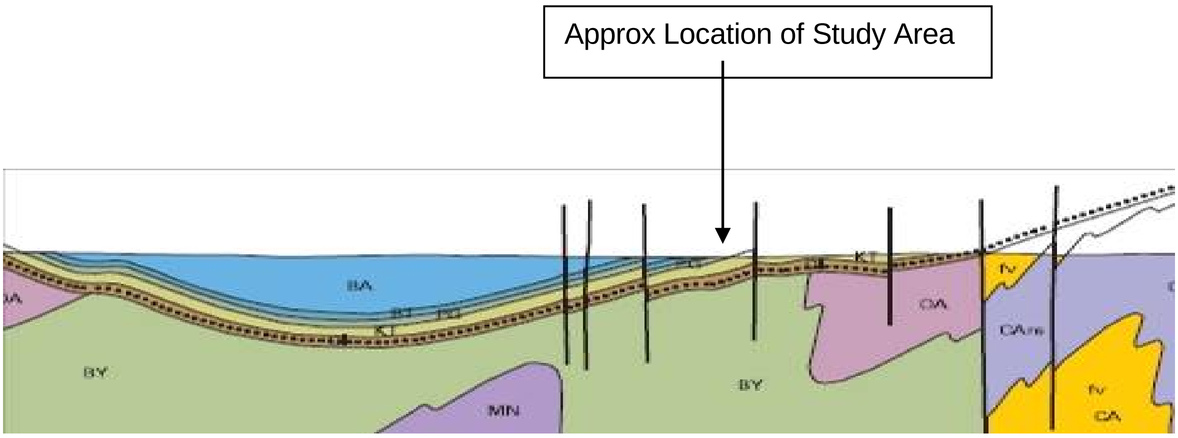


Figure 3.35 Geological cross-section adapted from Sheet 23 (GSI, 1994b)

Subsoils (Quaternary) Geology

Subsoils in the area surrounding the study site at Carrigeen in Co. Kilkenny are generally tills derived from Devonian sandstones as shown in Figure 3.36. In this area these tills are moderately permeable with the presence of cobbles and sand and gravel lenses in places. To the east of the study area there are Alluvium deposits along the course of the Dungooly stream. Bedrock outcrops to the south and the south-east of the study area near the summit of Corluddy Hill.

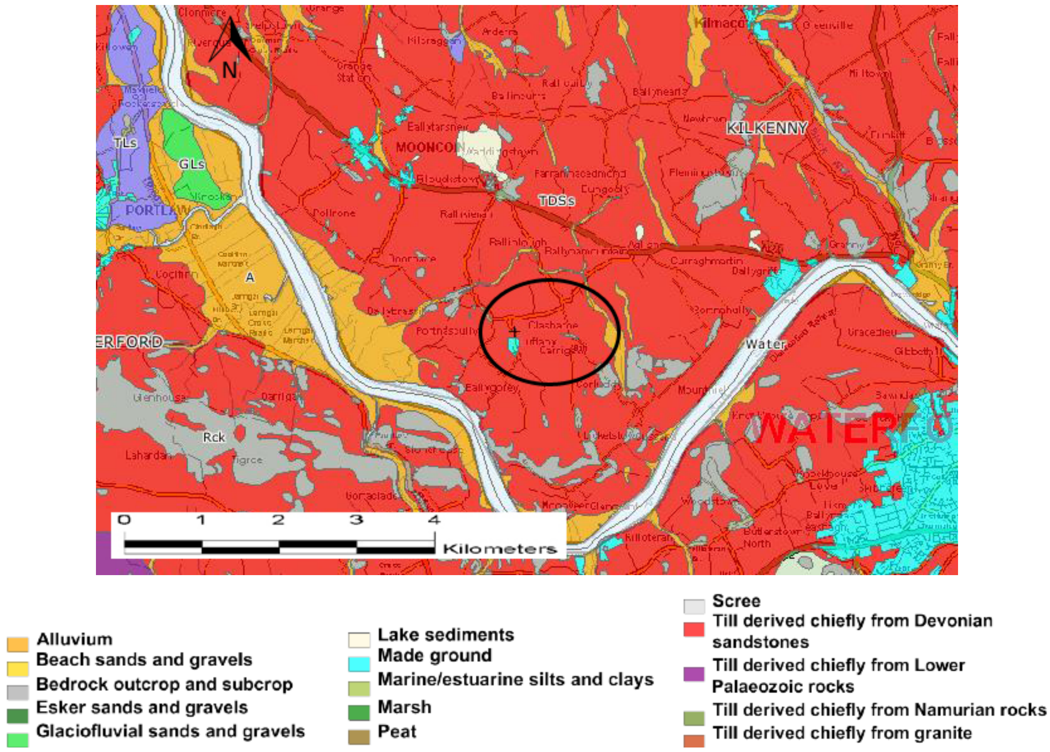


Figure 3.36 Subsoils map of the study area at Carrigeen, Co. Kilkenny

Local Authority planning application records were consulted for this area in order to identify the BS 5930 classifications that were observed. Soils in the area have generally been classified as silty CLAYS or silty/sandy clays.

Soils

Soils in the study area are described as Acid Brown Earths. An individual county map has not been completed for Kilkenny and this categorisation is based on the Teagasc General Soil Map of Ireland. The parent material of the acid Brown Earths is classified as being a mixed sandstone/limestone glacial till. These soils are generally quite permeable due to the presence of coarser parent material throughout the soil matrix.

Depth-to-bedrock

Depth to bedrock varies greatly in the surrounding areas of the study site. To the south of the study area bedrock is very shallow and is exposed in areas near the summit of Corluddy Hill. To the north of the study area bedrock is overlain by thicker deposits of tills and alluviums. There was no GSI depth-to-bedrock boreholes in the vicinity of the study are. 2 boreholes were drilled in the area during the course of this study. Bedrock was encountered at a depth of 6 m BGL in one of these boreholes however the second borehole was abandoned at 20 m BGL with competent bedrock not having being encountered. Information was available from local residents with depth-to-bedrock for their residential water supply wells provided at two locations. This information together with the draft depth-to-bedrock GSI map (see Figure 3.37) were combined to produce a contoured map of generalised depths to bedrock for the surrounding area.

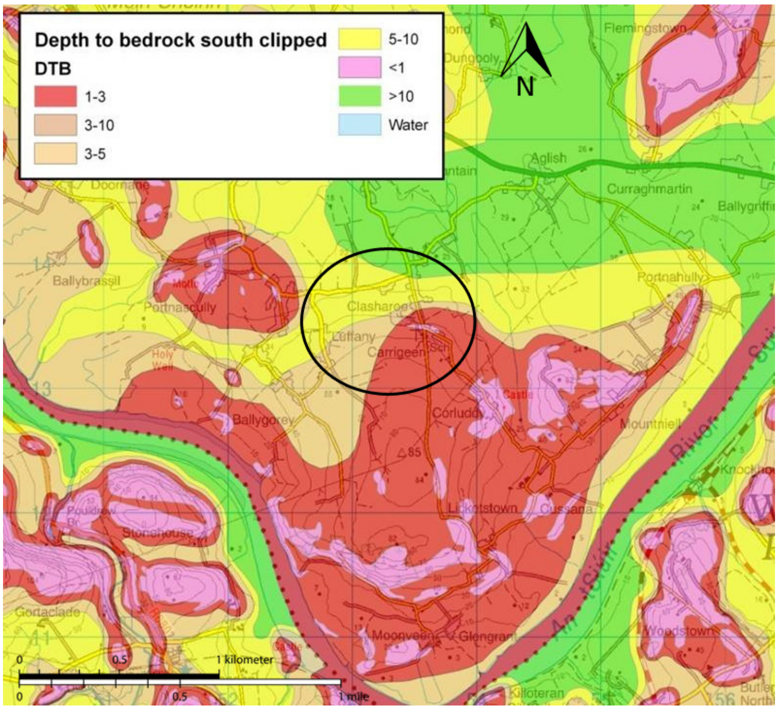


Figure 3.37 Depth-to-bedrock map of the study area at Carrigeen, Co. Kilkenny

3.4.4 Geology – Site at Faha, Co. Limerick

Bedrock Geology

Bedrock geology in the area surrounding the study site at Faha, Co. Limerick is dominated by Dinantian Carboniferous rocks (GSI, 1999). The study site is underlain by Waulsortian Mudbank, pale-grey massive limestones (WA) – see Figure 3.38.

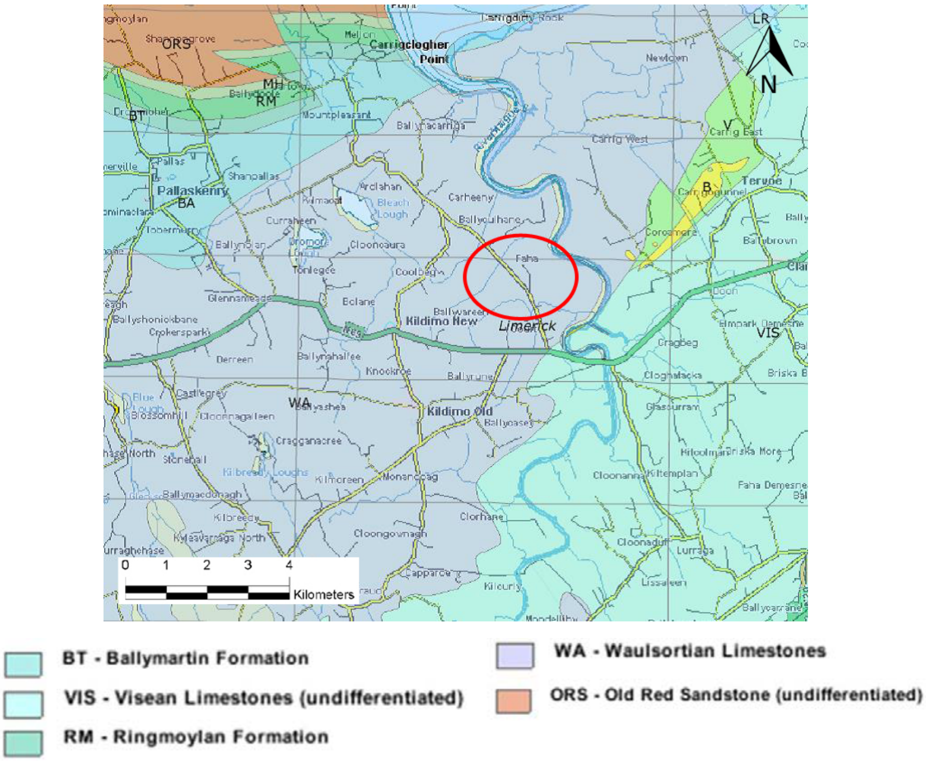


Figure 3.38 Bedrock 100k Solid Geology of the Area at Faha, Co. Limerick

These Waulsortian limestones, which are sometimes referred to as “reef” limestones, dip generally to the northeast (Deakin, 1995). To the north-west the shaly limestones of the Ballysteen Formation (BA) crop out, however this outcrop is not extensive. Deakin (1995) reported that the Ballysteen formation is magnesian in places, which suggests that dolomitisation has occurred. Further to the north-west there are a number of smaller outcrops comprising of several different formations including; the Ballymartin Formation which comprises limestone and dark-grey calcareous shale (BT), the Ringmoylan Formation comprising calcareous shale and crinoidal limestone (RM) and the Mellon House Formation comprising of siltstone, sandstone and calcareous shale.

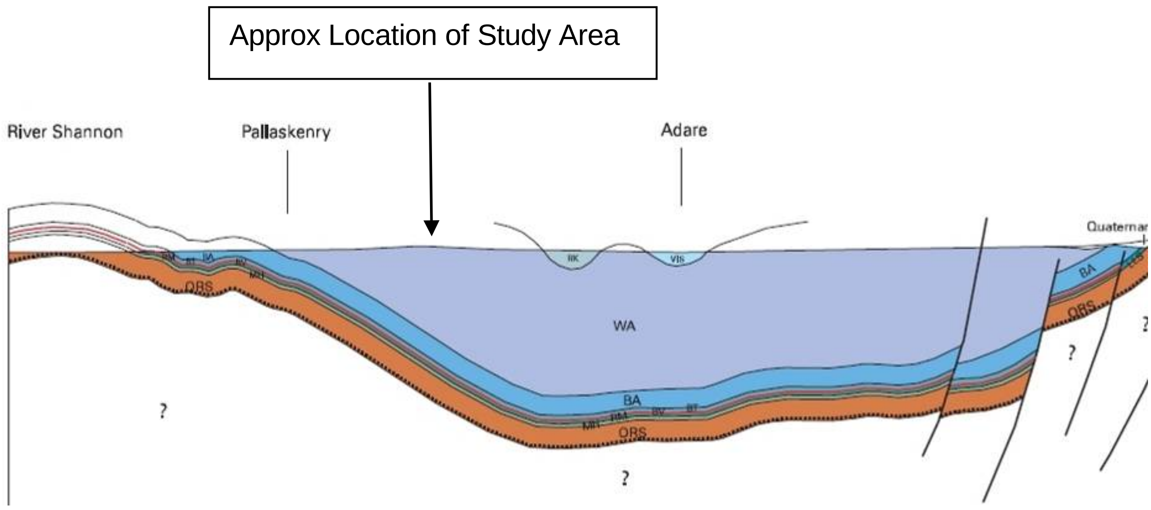


Figure 3.39 Geological cross-section for Shannon area - Adapted from Sheet 17 (GSI, 1999)

Directly adjacent to the river Shannon there is an outcrop of Old Red Sandstone (ORS). This Devonian Old Red Sandstone was laid down before the younger Carboniferous rocks and extends below these formations as illustrated in the generalised cross-section given in Figure 3.39. To the south-east the Waulsortian limestone is overlain by Visean limestones (VIS) which are generally described as pale grey, clean, medium to coarse-grained, bedded limestones.

Subsoils (Quaternary) Geology

Subsoils in the area surrounding the study location are classified as till derived chiefly from (Carboniferous) limestone (see Figure 3.40); these tills tend to have a sandy and/or silty matrix. In the areas adjacent to the course of the river Maigue, subsoils are classified as undifferentiated alluvium. These alluvium deposits extend into the study area and further south-west towards Kildimo. To the west, and north-west bedrock is close to the surface with thin till deposits where there is no outcrop and in many areas there are exposed bedrock outcrops present. To the north-west surrounding Bleach and Dromore Lough there are lake sediment deposits and to the west there are small areas of fen-peat deposits. As discussed in Chapter 2, when applying to build a new dwelling together with an on-site wastewater treatment system, applicants are required to excavate a trial hole and classify the soil to BS 5930. Local Authority planning application records were consulted for this area and soils had been classified as stony sandy CLAY in nearly all available records.

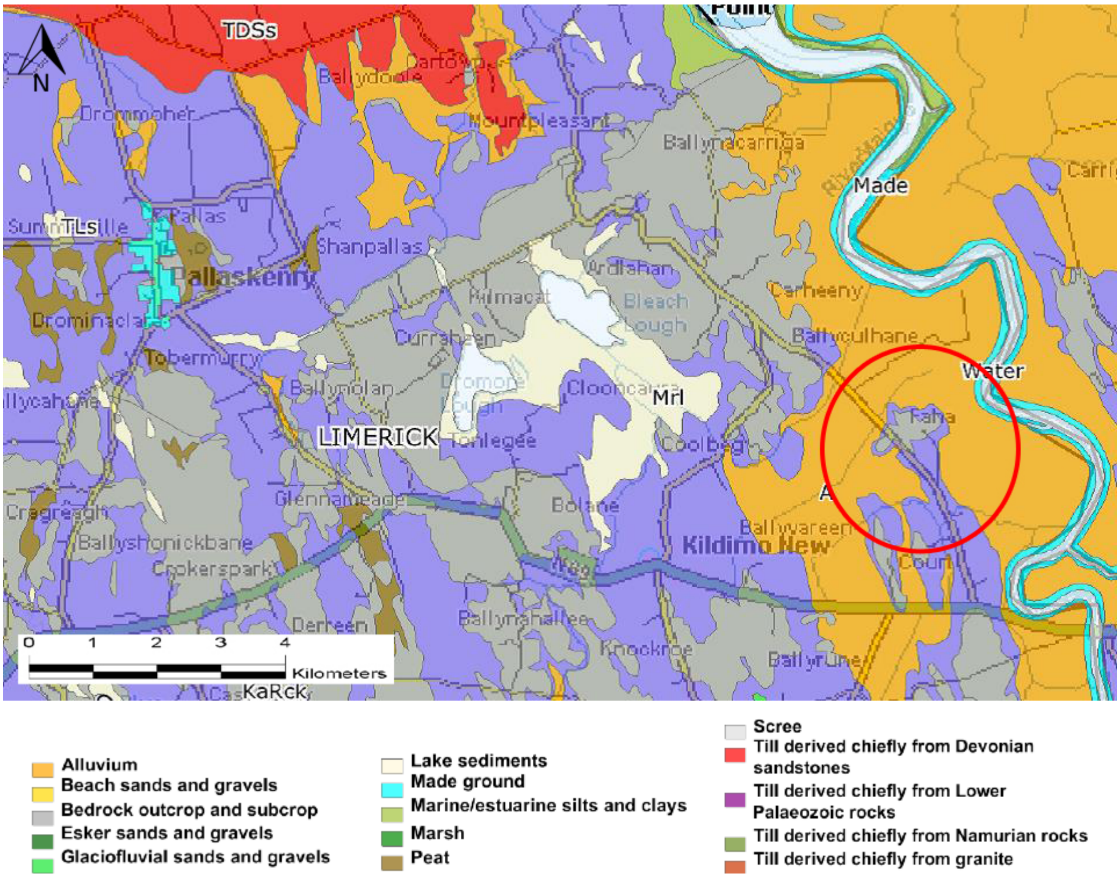


Figure 3.40 Subsoils map of the study area at Faha, Co. Limerick

Soils

Soils of the area have been summarised by Finch and Ryan (1966) and are shown on the accompanying published soils map of Co. Limerick. In the study area, soils are generally of the Shannon series and are classified as gleys comprising estuarine alluvium and are described as being fine-textured and base-rich (Finch and Ryan, 1966). These gleys are heavy textured and are very poorly drained due to the high silt content and also due to the low-lying nature of the area which is situated in the flats of the river Maigue. There are also areas of the Howardstown gley series which is of glacial drift origin containing mostly limestone with some shale, sandstone and volcanic. The study area also contains a small extent dominated by the Rineanna complex which contains soils of the Ballincurra, Elton and Rineanna Series and are classified as Brown Earths, Grey-Brown Podzolics and Lithosols respectively. To the west of the study area the Rineanna complex is very extensive with smaller extents of the Elton and Howardstown series.

Depth-to-bedrock

The area surrounding the study site contains many areas where rock is at the surface and is cropping out. Depth-to-bedrock in the area is on average less than 5 m, with bedrock typically less than 2.5 m below ground level (BGL). As part of this study 3 boreholes were drilled in the area and rock was encountered at depths of 5.5 m, 5 m and 2.5 m BGL, and details of the associated borehole logs are given in Section 4.2.1. Geological Survey of Ireland (GSI) records for the area were also consulted and a number of depth-to-bedrock records were available both to the north and west of the study area. This information was combined with the draft depth-to-bedrock GSI map (see Figure 3.41) and with the information gained from drilling at the study location to produce a contoured map of generalised depths to bedrock for the surrounding area.

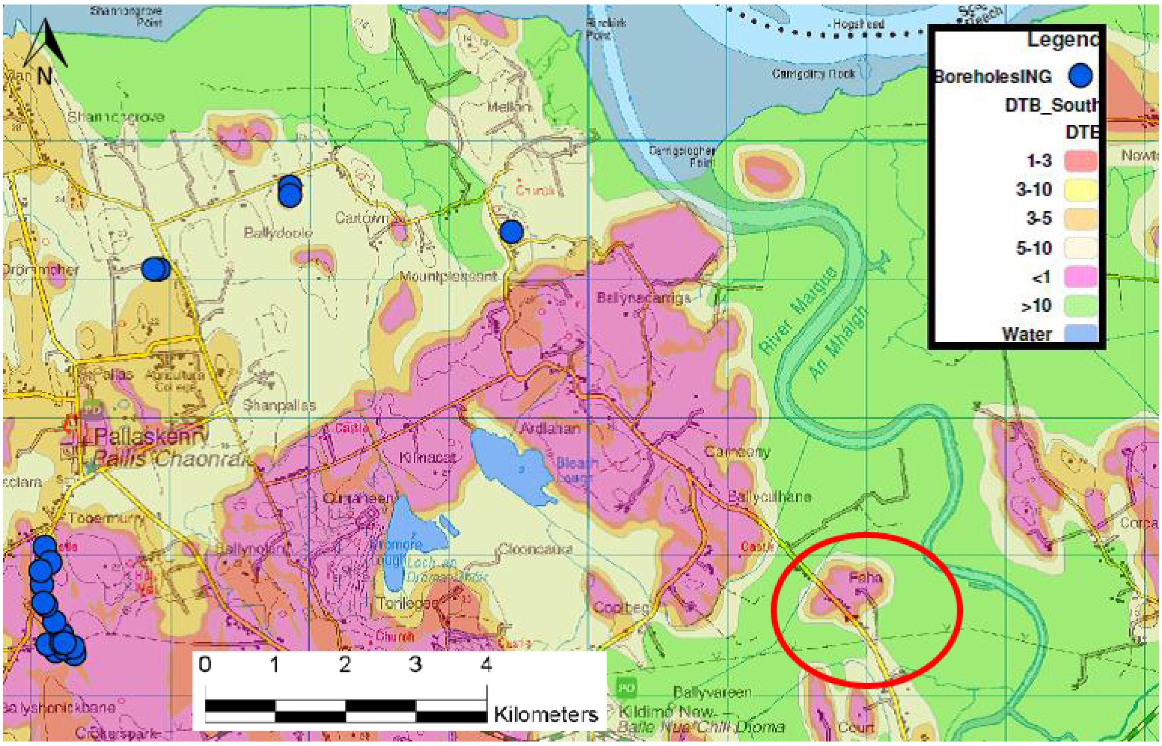


Figure 3.41 Depth-to-bedrock map of the study area at Faha, Co. Limerick

3.4.5 Geology – Site at Toonagh, Co. Clare

Bedrock Geology

The area surrounding Toonagh, Co. Clare is dominated by karst features (see Figure 3.42). This is due to the bedrock geology consisting of permeable Dinantian pure bedded limestones. Beneath the study site the Burren formation is segregated into the Ailwee Member (BUaw) which forms a band of highly fossiliferous limestone with clay bands, whilst to the east of the study site the Burren formation (BU) comprises pale grey clean skeletal limestone as shown in Figure 3.43. To the west a number of formations including the Gull

Island formation (GI) comprise Namurian Sandstones and therefore there are no karst features to the west of the study site.

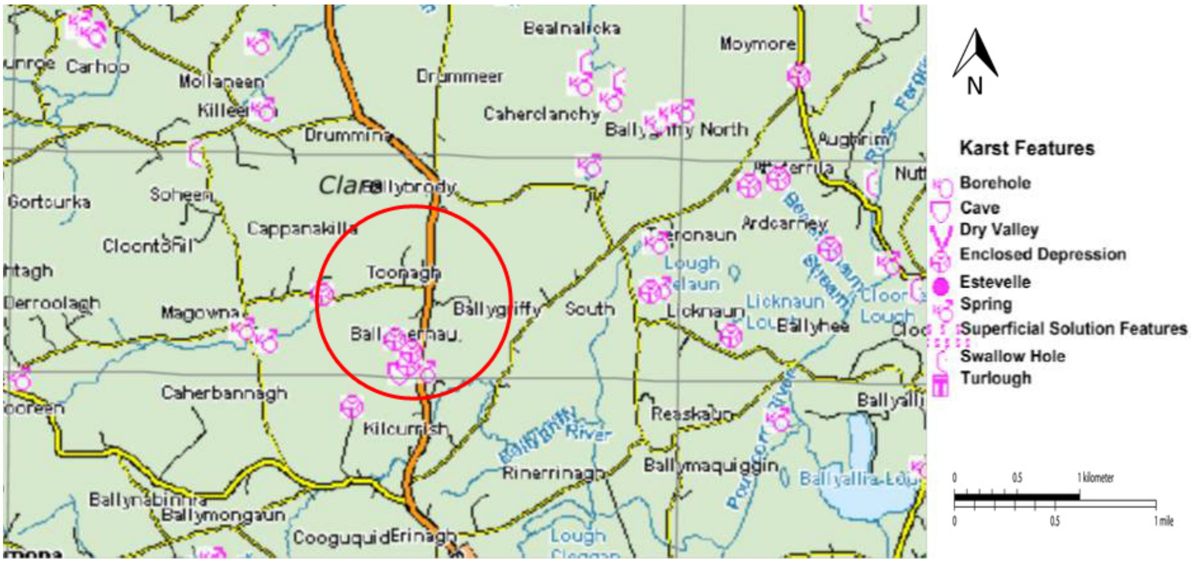


Figure 3.42 Karst Features near Toonagh, Co. Clare

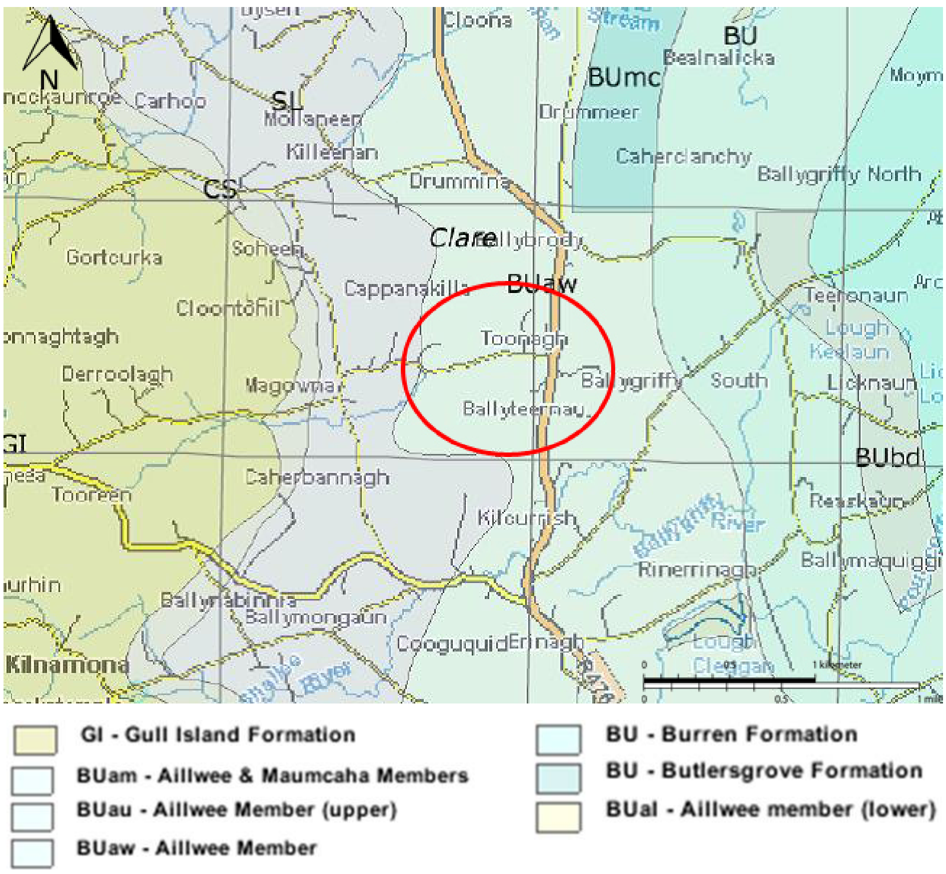


Figure 3.43 Bedrock 100k Solid Geology of the Area at the Toonagh, Co. Clare

Subsoils (Quaternary) Geology

Subsoils in the area surrounding the study site at Toonagh, Co. Clare are dominated by limestone tills as shown in Figure 3.44 below. This changes to sandstone and shale tills to the west given the changing bedrock units. However, the area as a whole is dominated by rock being exposed at the surface and this is common across the region with the area only 10km from the Burren National Park.

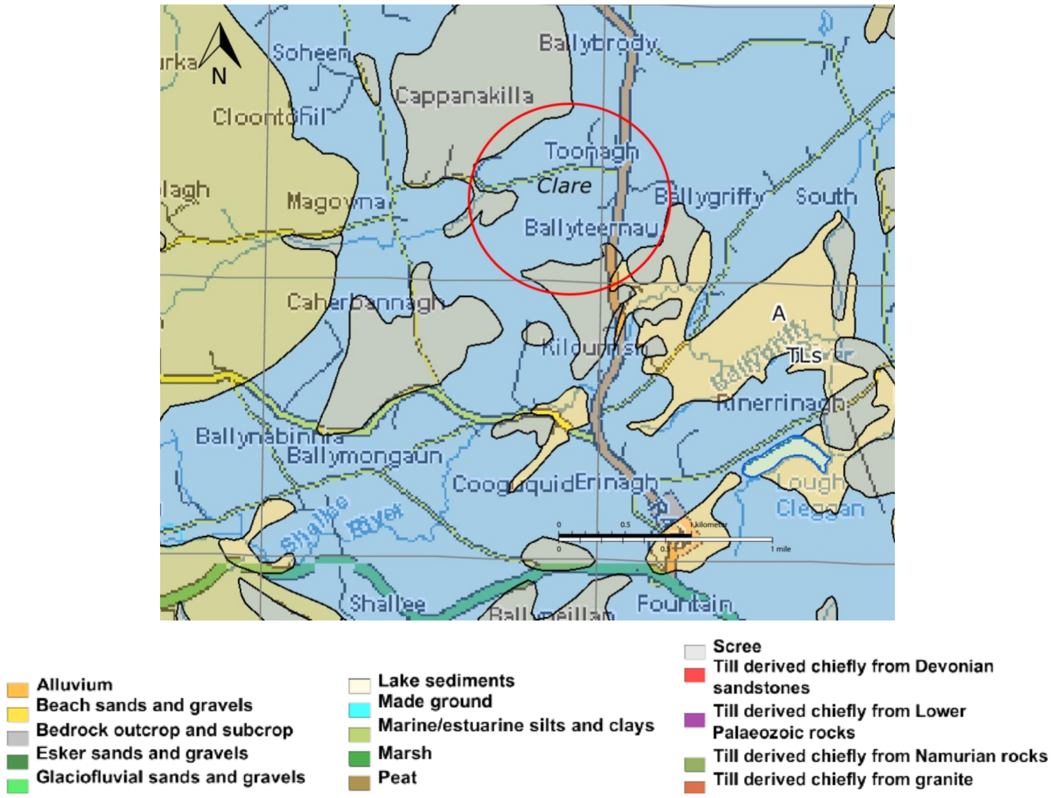


Figure 3.44 Subsoils map of the study area at Toonagh, Co. Clare

Soils

Soils in the area are generally Grey Brown Podzolics of the Patrickswell series and to the west soils are described as *gleys arising from the Kilrush series*. Both parent groups are tills and vary based on the underlying bedrock. Soils in the immediate area of the study site are highly permeable due to their shallow thickness and the high proportion of coarser grained particles.

Depth-to-bedrock

Bedrock in the area is very shallow with many outcrops visible and large areas of exposed rock predominantly in limestone areas. GSI depth-to-bedrock mapping is not available for the area however bedrock is typically less than 3 m below ground level.

4 FIELDWORK METHODOLOGY

4.1 Introduction

This study involved a significant amount of fieldwork given that there were five study sites with elements of on-site and laboratory analysis required at each location. This chapter outlines the site set up and layout for each of the study locations. The methods used for sample collection, laboratory and field analysis are also described.

4.2 Site Layout and Setup

4.2.1 Site Layout

As described in Chapter 3, each of the study locations comprised of a cluster of residential dwellings of varying age and size located in different groundwater vulnerability and aquifer classifications. Each of these dwellings had an on-site treatment system and the type of treatment system generally varied with the age of the dwelling, for example older dwellings tended to have a traditional septic tank with percolation area, whilst newer properties tended to have secondary treatment systems installed; where possible Local Authority records have been consulted to determine same. In many instances site access was not possible and in these cases an estimate has been made as to the exact location of the systems. Where possible the type of treatment system was also identified.

Boreholes were drilled upstream and downstream of each cluster system in order to provide sampling locations for groundwater monitoring. A nested array of piezometers was then installed in each of the boreholes typically containing three horizons; one in the saturated zone above the bedrock (if present), one in the bedrock subsoil interface (transition zone) and one approximately 5 m into the bedrock. This setup would enable monitoring of contaminants as they moved downwards into the aquifer as well as spatially as they travelled with the groundwater local gradient. Determining what orientation the groundwater table followed at each of the study sites and therefore the locating of the boreholes was based upon estimation from ground surface topography and surface water features. In general the groundwater table profile is a subdued replica of the surface topography and it was on this basis that the locations of monitoring boreholes was decided with the exception of the study site at the Naul where a previous study carried out by the GSI (Hunter Williams et al., 2005) provided guidance. Agreement from landowners to drill had already been agreed as detailed in Chapter 3, and this also influenced the locating process. Once the desired locations for drilling of the monitoring boreholes were established a full health and safety appraisal was carried out in conjunction with the drilling contractor. This included the identification of hazards such as overhead power cables, site access and other potential hazards at the drilling locations. Based on the outcome of this health and safety review the final locations for drilling were agreed upon with the contractor and drilling could then commence. Figure 4.1 – Figure 4.4 give summaries of the geological conditions encountered at each of the boreholes that were drilled as part of this project. The final drilling locations for the groundwater monitoring boreholes are shown on the site layout plans given in Figure 4.5 – Figure 4.9 below. Full drilling logs from the contractor are also contained within Appendix A.

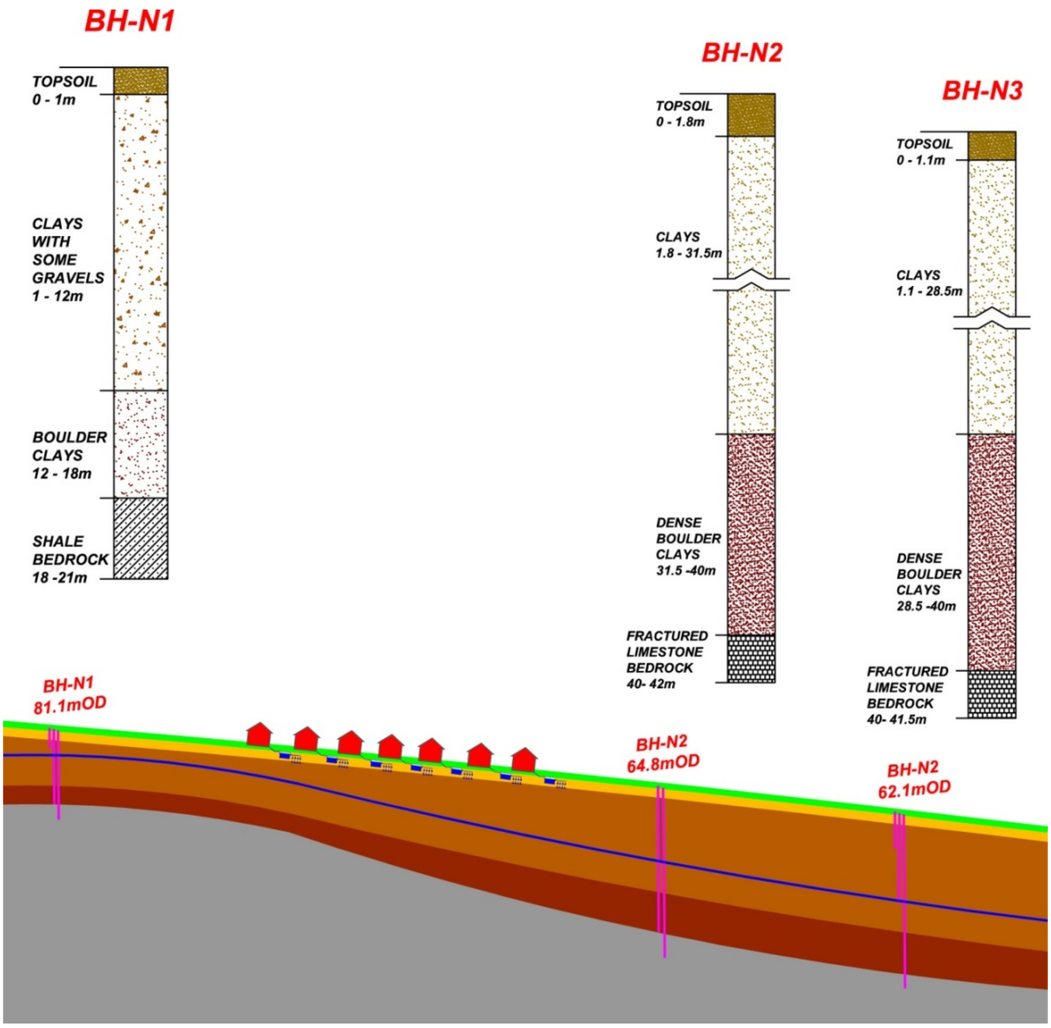


Figure 4.1 Summary borehole logs with schematic cross section at Naul, Co. Dublin

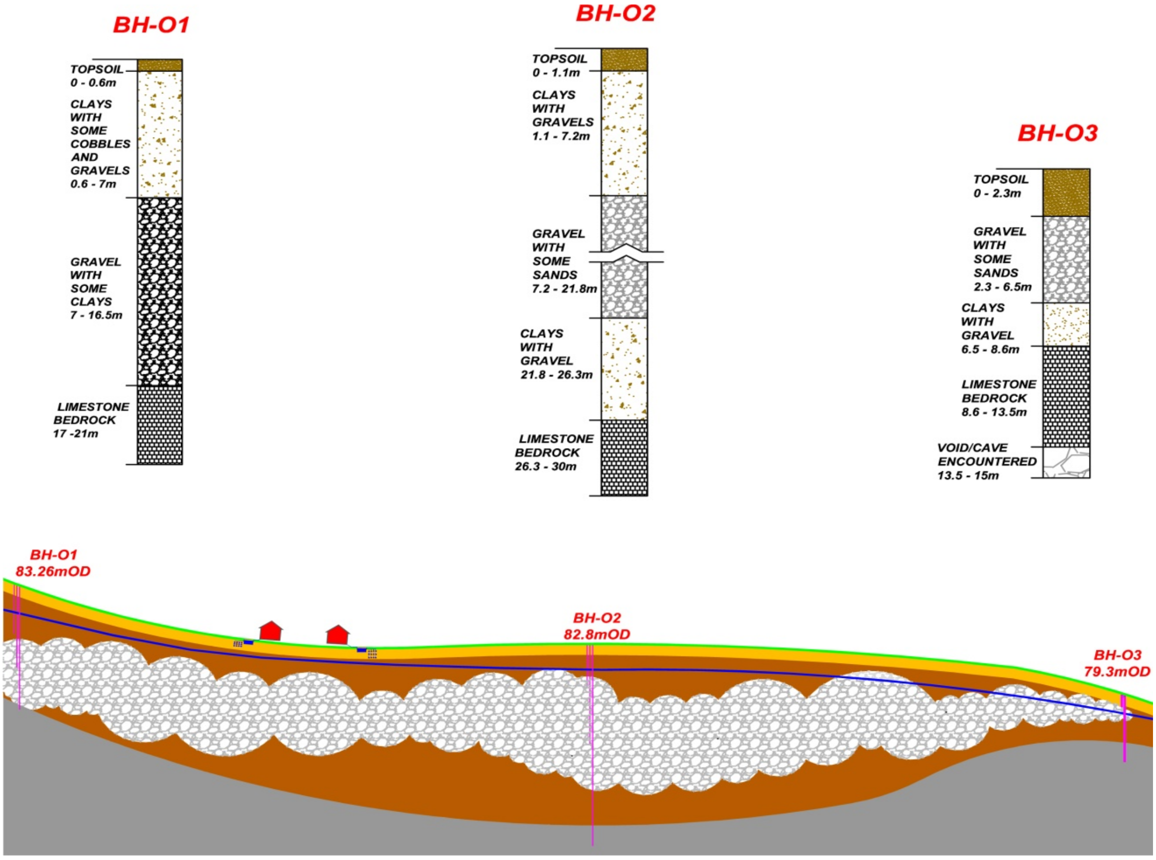


Figure 4.2 Summary borehole logs with schematic cross section at Rhode, Co. Offaly

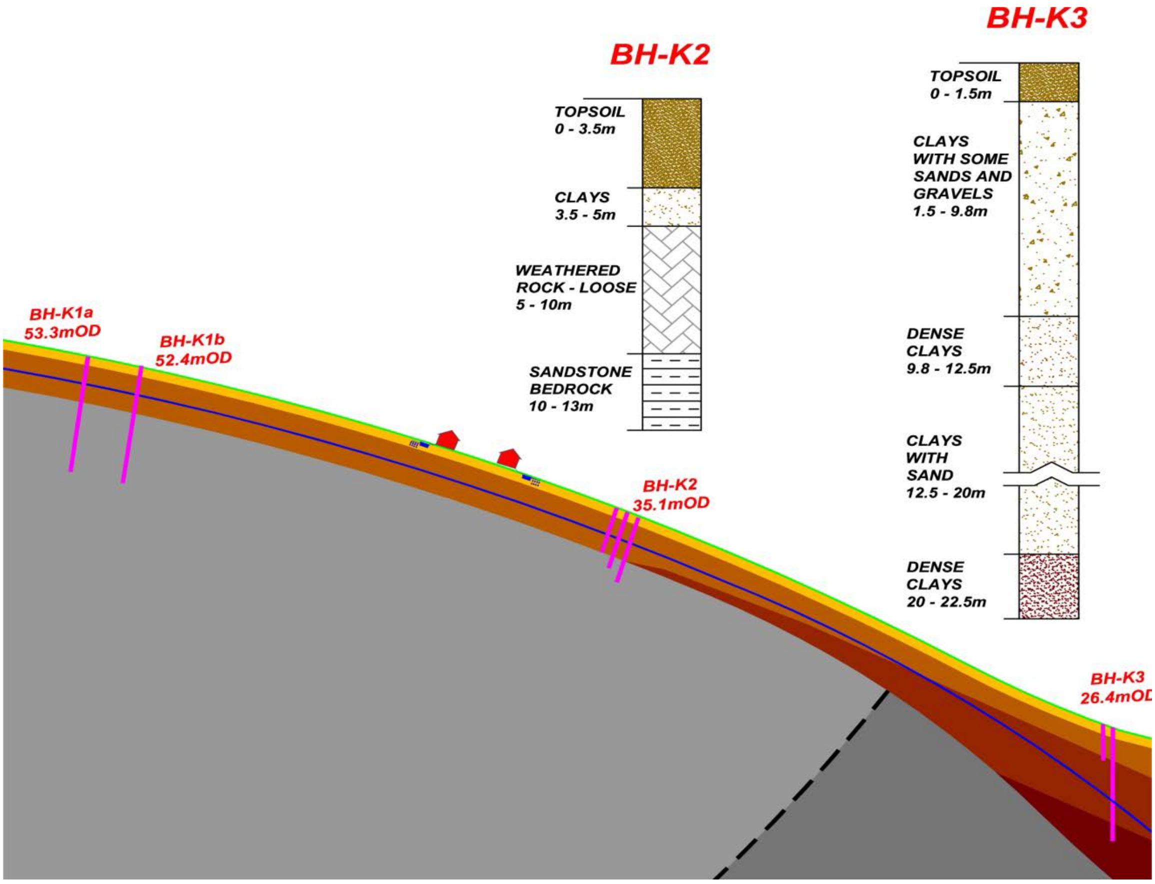


Figure 4.3 Summary borehole logs with schematic cross section at Carrigeen, Co.Kilkenny

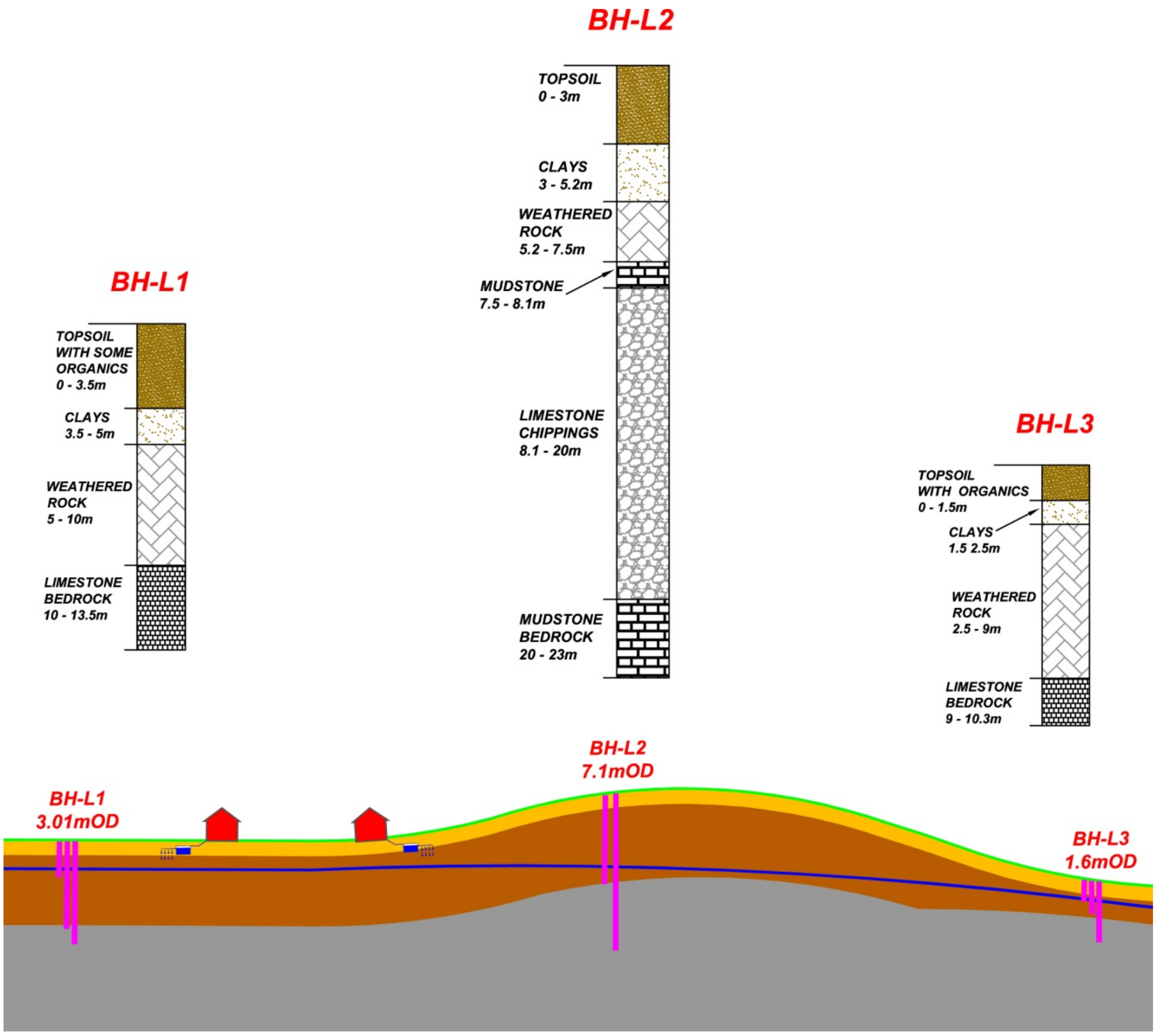


Figure 4.4 Summary borehole logs with schematic cross section at Faha, Co. Limerick

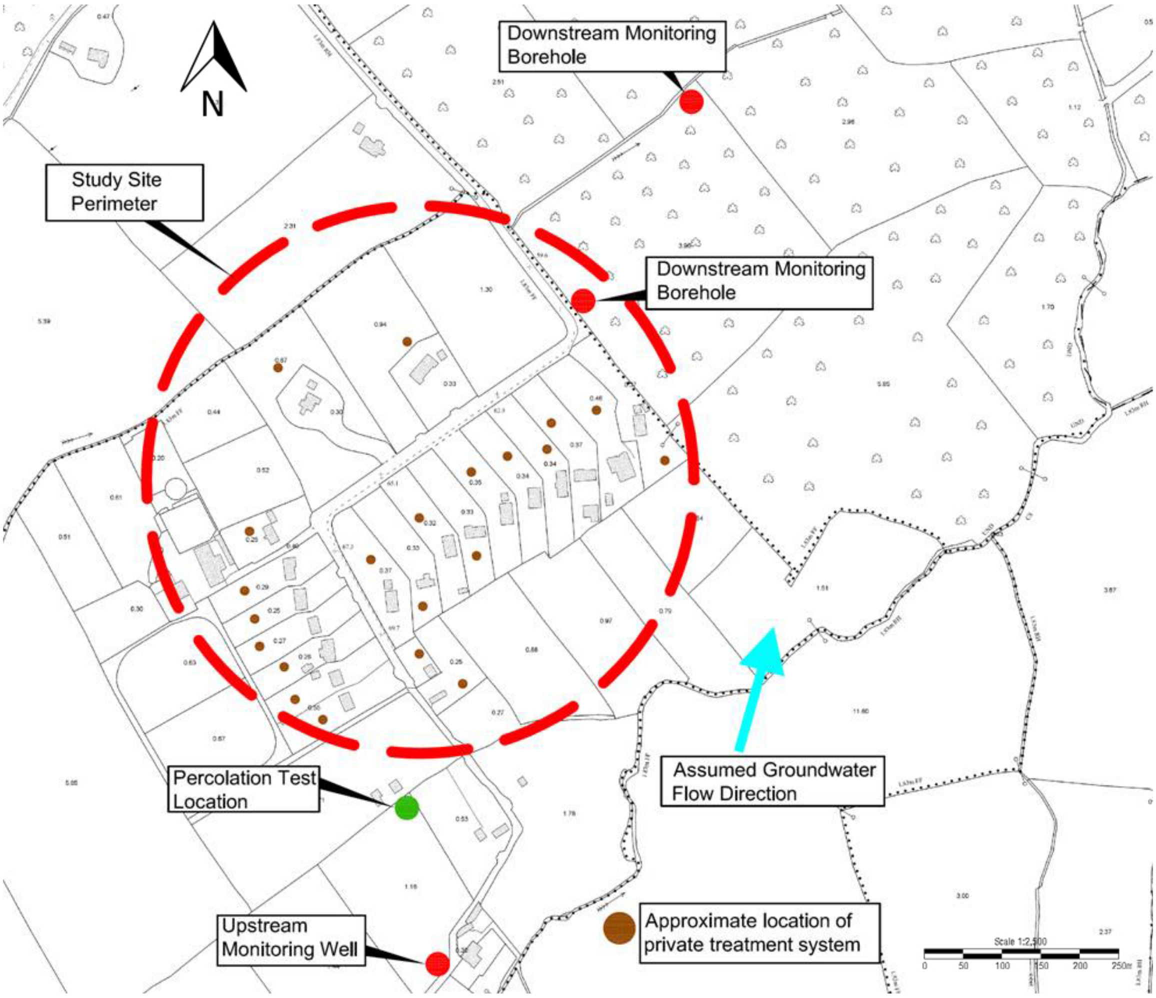


Figure 4.5 Site Layout – Naul, Co. Dublin

The study site at the Naul shown above in Figure 4.5 included 3 No. TCD drilled boreholes. BH-N1 was located upstream at the entrance to a pasture field but was in an area of fenced off wasteland. Three piezometers were installed with two in the bedrock and one in the subsoil in case a high water table was present during the wetter months of the year. BH-N2 and BH-N3 were both located in an apple orchard north of the study area. Both boreholes contained two piezometers; one in the bedrock interface and one in the subsoil in order to access the presence of higher water tables. A percolation test was carried out at the site and was located as close as possible to the study area that agreement could be achieved.

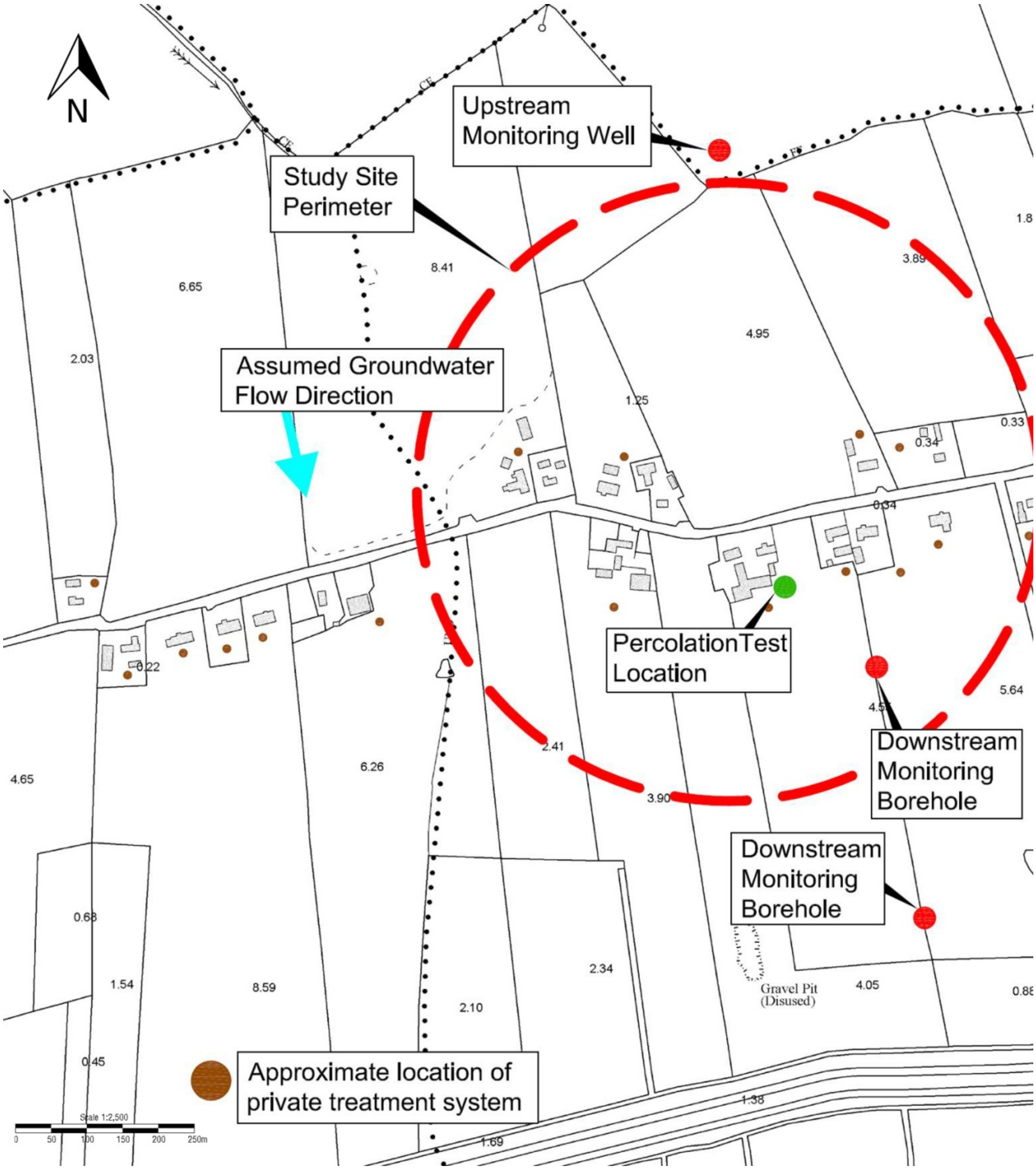


Figure 4.6 Site Layout – Rhode, Co. Offaly

The study site at Rhode shown in Figure 1.6, included 3 No. TCD drilled boreholes. BH-O1 was located upstream at the bottom of a field used for agriculture. Three piezometers were installed with two in the bedrock and one in the subsoil. BH-O2 and BH-O3 were both located in a field used for agriculture south of the study area. BH-O2 contained three piezometers; one in the bedrock interface one in the bedrock and one in the subsoil. BH-O3 contained two piezometers both in the bedrock.

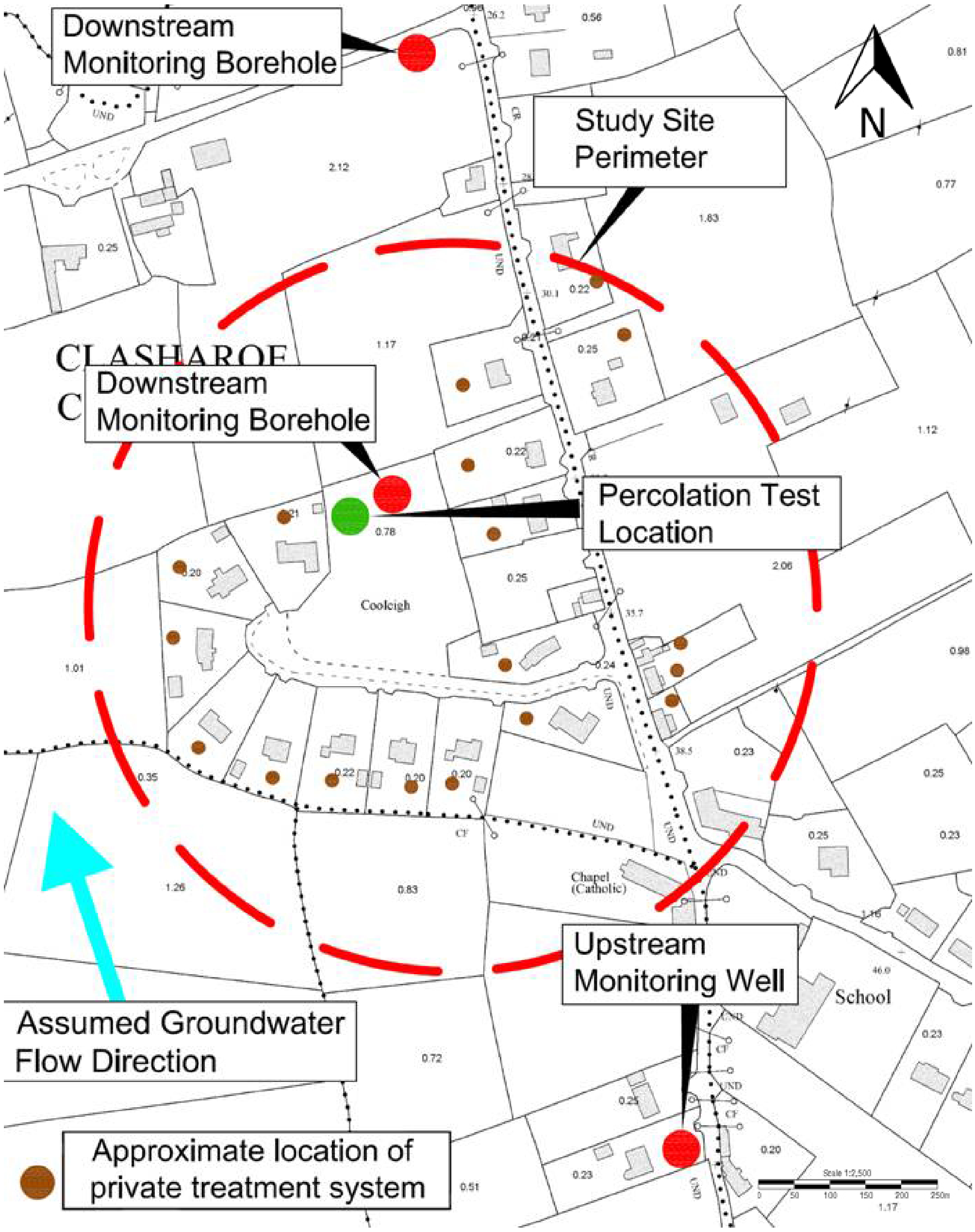


Figure 4.7 Site Layout – Carrigeen, Co. Kilkenny

The study site at Carrigeen shown in Figure 1.7 included 2 No. TCD drilled boreholes. Two additional private wells were monitored upstream instead of a borehole being drilled. This was due partly to budget constraints and also to agreement not being reached with landowners. BH-K2 was located downstream at the bottom of a vacant development site inside the study areat. Three piezometers were installed with two in the bedrock and one in the subsoil. BH-K3 located adjacent to a local road and contained two piezometers both in the subsoil as bedrock was not encountered before drilling was abandoned. A percolation test was carried out at the site also within the vacant development site inside the study area.

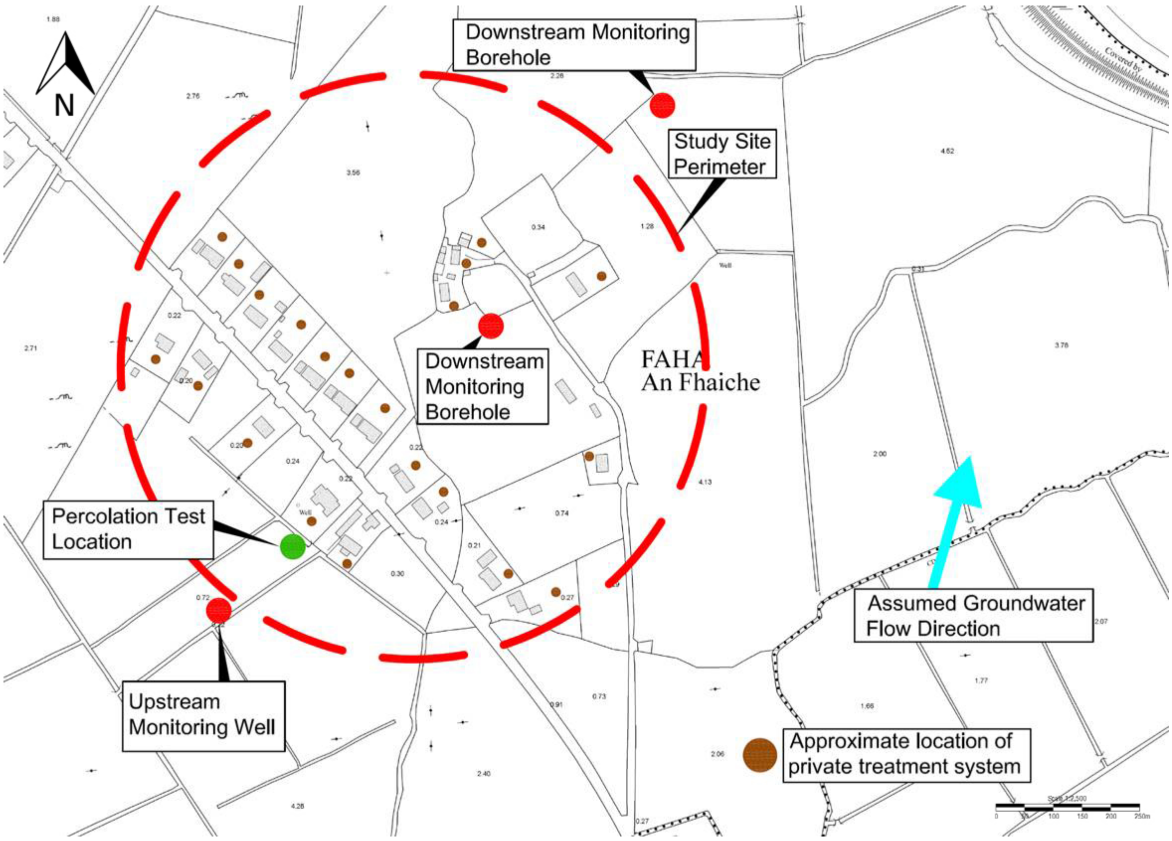


Figure 4.8 Site Layout – Faha, Co. Limerick

The study site at Faha shown in Figure 1.8 included 3 No. TCD drilled boreholes. BH-L1 was located upstream at adjacent to an access road to agriculture land. Three piezometers were installed with two in the bedrock and one in the subsoil. BH-L2 and BH-L3 were located in agricultural fields downstream of the study development. BH-L2 contained two piezometers both in the bedrock. Three piezometers were installed in BH-L3 with two in the bedrock and one in the subsoil. A percolation test was carried out at the site in an agricultural field adjacent to the location of BH-L1.

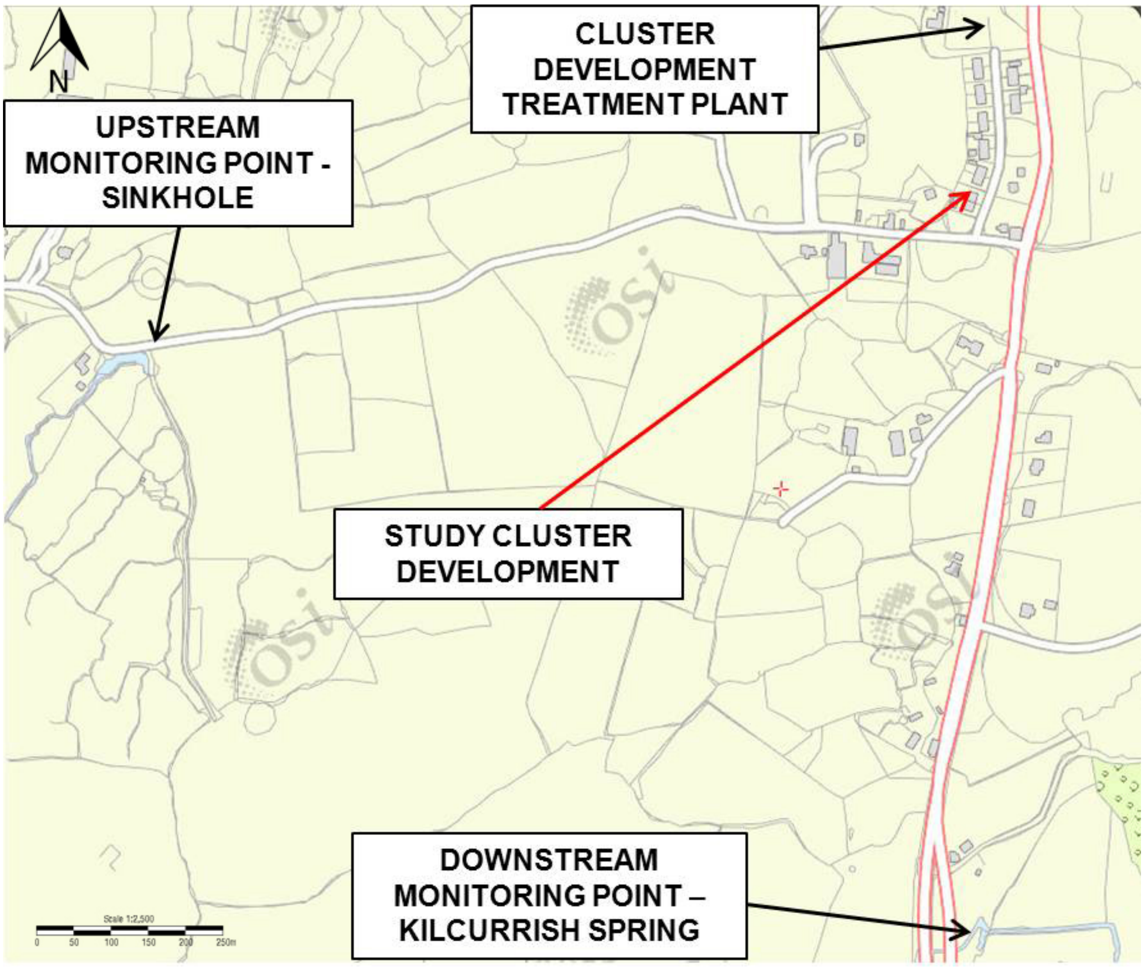


Figure 4.9 Site Layout – Toonagh, Co. Clare

The study site at Toonagh did not include any monitoring boreholes due to the karstified nature of the bedrock and the low likelihood of encountering a conduit or preferential groundwater flow path during drilling. Two monitoring points were decided upon for water quality, tracer studies and discharge. As outlined earlier it was reported locally that a connection existed between the treatment plant at Toonagh, which is discharging to a sinkhole, and the Kilcurrish spring. Samples were also taken from the treatment plant outflow in order to quantify the typical discharge water quality determinants.

4.2.2 Borehole Construction

As detailed in Chapter 1, the main project objective was to monitor the quality of groundwater in a downward profile at each of the study sites to determine the level of both chemical and microbiological contamination from the on-site effluent from the clusters and the degree of any natural attenuation within the subsoil and/or bedrock. In order for this to be achieved a borehole construction which includes nested piezometers was utilised. This allows one larger diameter borehole to be drilled which can then have a number of smaller diameter piezometers installed at different depths that are sealed and independent of each other. A piezometer consists of a plastic pipe with a slotted section in the area that water is to be sampled from with the rest of the pipe having solid walls. Figure 4.10 shows the piezometers prior to and after installation and Figure 4.11 shows the typical piezometer installation setup. Details of the piezometer installation depths at each of the study locations are given in Table 4.1 below.

Figure 4.10 (a) Piezometers prior to installation with geo-membrane covering slotted section (b) Completed Borehole with raised cover and concrete plinth (c) Installed piezometers during borehole construction

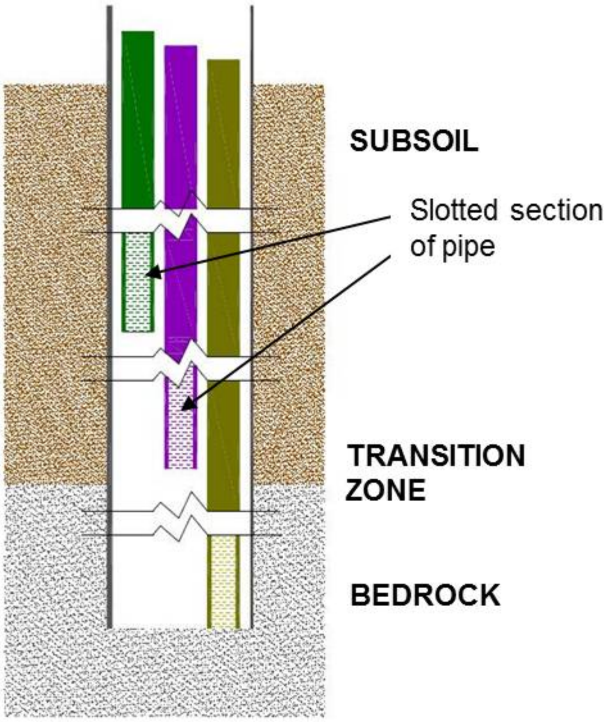


Figure 4.11 (a) Typical Nested Piezometer Setup. (b) A Borehole being drilled with Beretta Rig

The boreholes were drilled using a 203 mm diameter Air Rotary Casing Hammer drilling rig – shown in Figure 4.11 above. 52 mm diameter piezometers were then installed at different depths in the aquifer the deepest being 5 m into the bedrock, another horizon typically being in the transition zone and, if present, one in the subsoil water table. For ease of identifying each of the boreholes at the study locations they will be named in the following manner:

BH-O3a

Identifies that this is a borehole

Identifies the location i.e. O for Offaly, L for Limerick, D for Dublin and K for Kilkenny

Identifies specific borehole and piezometer; 1 is upstream, 2 is first downstream and 3 is second downstream. a, b and c identify specific piezometer in the nested setup

Table 4.1 Details of piezometers installed at study locations

Location	Piezometer Elevation* (mOD)	Length of piezometer (m)	Slotted Section (m)	Elevation at centre of slotted section (mOD)
Naul				
BH-N1a	81.224	2	1	73.724
BH-N1b	81.332	13.5	3	63.332
BH-N1c	81.425	18	3	58.925
BH-N2a	64.801	31	3	20.801
BH-N2b	64.850	40	3	11.75
BH-N3a	62.101	10	3	39.601
BH-N3b	62.160	31	3	18.66
BH-N3c	62.126	35	3	14.626
Offaly				
BH-O1a	83.264	18	3	66.764
BH-O1b	83.287	10	3	74.787
BH-O1c	83.246	29.5	3	55.246
BH-O2a	82.558	6.5	3	77.558
BH-O2b	82.536	17	3	67.036
BH-O2c	82.802	21	3	63.302
BH-O3a	79.308	2.5	1	77.308
BH-O3b	79.301	10	3	70.801
Kilkenny				
BH-K1a**	54.308	n/a	n/a	n/a
BH-K2a	34.608	12.5	3	23.608
BH-K2b	34.538	16.5	3	19.538
BH-K2c	34.584	20	3	16.084
BH-K3a	26.106	4	3	23.606
BH-K3b	26.111	14.6	3	13.011
Limerick				
BH-L1a	2.983	5	1.5	-1.267
BH-L1b	3.091	9	2	-4.909
BH-L1c	3.326	13	1	-9.174
BH-L2a	7.148	12	3	-3.352
BH-L2b	7.151	23	6	-12.849
BH-L3a	1.615	2.6	1	-0.485
BH-L3b	1.593	5	1.2	-2.807
BH-L3c	1.625	10.3	1	-8.175

*Level shown here is top of piezometer – subtract length to find elevation of base

**Data not available as this is a private well that was used as upstream monitoring location

4.3 Field Monitoring and Sample Collection

4.3.1 Sample Collection

Samples were collected from each of the monitoring boreholes on a monthly basis from November 2010 to November 2012. However, sampling did not begin until May 2011 at the Naul due to issues with pump equipment, as discussed later. At each site visit, each piezometer was purged of three times the borehole volume to remove stagnant water and draw in fresh groundwater from the surrounding aquifer. For a number of the piezometers at the different study sites it was not possible to remove three borehole volumes of water due to their very slow recovery rate and in these instances only one volume was purged. These included:

- BH-O3a and BH-O3b
- BH-L2a and BH-L2b
- BH-K3a and BH-K3b

For boreholes with a depth not greater than 15 m a Waterra WaSP-P3 pump with LDPE tubing was used to recover water samples. This pump was made from plastic and provided good chemical resistance as well as being very convenient for field monitoring as it powered from a 12V battery. All field measurement equipment was thoroughly rinsed with distilled water between sampling events. The pump is shown in Figure 4.12 and can also be seen in use at one of the study sites.

Figure 4.12 (a) Waterra WaSP-P3 Pump (b) Sample being taken using the Waterra pump

For boreholes that were greater than 15 m in depth a Grundfos MP1 pump was used to recover water samples. This was a much more cumbersome setup as a power inverter, pump controller and 6.5 KVA generator were all required in order to take samples using this pump. Due to the stainless steel construction of the MP1 pump it provides very good chemical resistance and is very durable. Again all field measurement equipment was thoroughly rinsed with distilled water between uses. The entire pump setup involved for the MP1 pump is shown in Figure 4.13 below.

Figure 4.13 MP1 Pump and equipment

The water samples were transferred into 500 ml sterile autoclaved plastic sample bottles, labelled and transferred to a cooler box for transportation back to the Trinity College Laboratory. Sample bottles were filled to the brim so to allow as little air to be enclosed as possible that could react with the water during transportation. Additional samples were taken for the determination of Total phosphorus and these samples required 25 ml of water to be transferred to sterile 50 ml borosilicate glass bottles using a sterilised 25 ml graduated cylinder. Both the glass and plastic sample bottles were cleaned thoroughly between uses with dilute phosphate free detergent and then sterilised at 121°C for 20 minutes using a Hirayama HV-25 Autoclave.

4.3.2 Field Measurements

A number of water sample parameters were measured on-site so as to ensure accurate measurement. These parameters included:

- Temperature (°C)
- Electrical Conductivity (EC, μS cm^{-1} at 25 °C)
- pH (potential of hydrogen)
- Water level in Piezometers – measured using a dipmeter

On-site analysis was undertaken, in accordance with USGS National Field Manual for the Collection of Water Quality Data (2005). A Hanna Instruments HI-98129 Waterproof pH/Conductivity/TDS Tester was used to record all parameters except water level. Again, between different sampling sources the electrodes were thoroughly rinsed with distilled water.

Temperature

Temperature was used as a useful parameter to help to distinguish "true" groundwater and from surface water. All sample temperatures were taken using a Hanna HI-98129 Combo tester and were recorded in °C.

Electrical Conductivity

Electrical Conductivity (EC), like temperature, is a useful parameter due to ease of measurement and its application in determining the origin of water. All groundwater samples had EC measured in the field, immediately after sample collection. The Hanna Hi-98129 Combo tester was calibrated the day prior to sample collection on site using a known solution of EC. All values of Electrical Conductivity were recorded in μS cm^{-1} at 25°C.

pH

The pH of water determines the solubility (amount that can be dissolved) and biological availability (amount that can be utilized by aquatic life) of chemical constituents such as nutrients (phosphorus, nitrogen, and carbon) and heavy metals (lead, copper, cadmium, etc.) (Hynds et al. 2012). The Hanna Hi-98129 Combo tester was calibrated the day prior to sample collection on site using two known solutions of pH (4.1, 7.01) and pH measurements taken immediately after sample collection.

4.4 Laboratory Methods and Analysis

4.4.1 Bacteriological Analysis

At the outset of the project a number of key chemical and bacterial indicator pollutants were identified to focus upon during this study. Contamination of groundwater by pathogenic microorganisms (bacteria and viruses) can cause human diseases such as diarrhoea, cramps, nausea, headaches and other symptoms (Ozler and Aydm, 2006). The microbiological quality of water is assessed by monitoring for non-pathogenic bacteria of faecal origin referred to as Faecal Indicator Bacteria (FIB). The presence of these bacteria indicate that faecal pollution of the water is likely and thus pathogenic bacteria may also be present. The coliform group of bacteria are one the most commonly occurring and can be found in the aquatic environment, in soil and in vegetation. Relatively simple laboratory tests exist to enumerate total coliforms in water samples, however it has been concluded that total coliforms do not always originate from faecal sources and thus their occurrence does not conclude a health risk (Youn-joo and Breindenbach, 2005). Within the coliform group of bacteria, the presence of *Escherichia coli* (*E. coli*) has been found to indicate contamination by faecal material from either humans or other warm blooded animals. Members of the genus *Enterococcus* bacteria are also commonly used as an indicator of faecal contamination of water specifically *Enterococcus faecalis*. The European Union drinking water directive (1998) lists *E-coli* and *Enterococci* as the only two groups defined as obligatory microbial parameters. In addition it has been suggested that the use of dual indicator bacteria groups provides a more robust outcome and that when using a dual indicator monitoring programme, *E-coli* and *Enterococci* are complimentary (Anderson et al., 2005). It was therefore decided that both *E-coli* and *Enterococci* would be monitored during the course of this study. It must be noted that Group D *Streptococcci* has been reclassified in the genus *Enterococcus* (including *Enterococcus faecalis*, *Enterococcus faecium*, *Enterococcus durans*, and *Enterococcus avium*) and therefore *Streptococcus faecalis* is now referred to as *Enterococcus faecalis*.

Determination of Enterococci faecalis

Samples were analysed for the presence *Enterococcus faecalis* (formerly referred to as Group D S*treptococci*) using the membrane filtration method and following the procedure as set out by the American Public Health Association (1992) in the Standard Methods for Examination of Water and Wastewater. The medium chosen for this method was Slanetz & Bartley culture media using the direct plating method. Prior to sampling, and usually the previous day, petri dishes were made up containing the Slanetz and Bartley medium. The required volume of the Slanetz & Bartley medium was made up according to the manufacturer's requirements using sterilised glassware and was poured into sterile 85 mm diameter petri dishes and allowed to cool. The cooled petri dishes were then inverted and stored in a laboratory fridge until required for use. Prior to use the petri dishes were removed from the fridge and allowed to warm to room temperature.

The membrane filtration method required 100 ml of the water sample to be filtered through a sterile 45 µm 47 mm diameter filter using a previously sterilised bottle top filtration unit and a vacuum pump. Sterilised tweezers were used at all times to both place and remove filters. Once the sample volume had passed through the filter by means of the vacuum a small volume of distilled water was used to wash the filtration unit, in case any bacteria still remained on the sides of the unit. The filter was then removed from the filtration unit and placed onto the surface of the Slantez and Bartley medium inside a pre-prepared petri dish and the lid replaced. The petri dish was then inverted, labelled and incubated at 44°C for a period of 44 hours. After incubation the filter was examined, with a hand lens in a good light, and all red or maroon colonies were counted as presumptive *Enterococci*. A petri dish containing Slantez and Bartley medium and an incubated filter with *Enterococci* colony growth is shown in Figure 4.14.

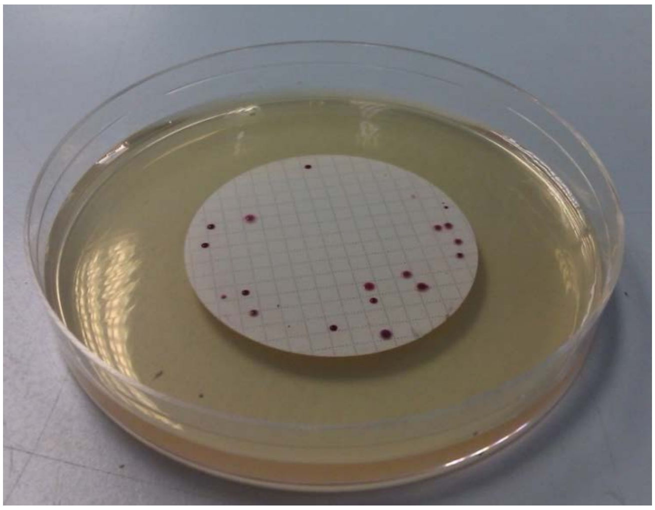


Figure 4.14 Petri dish containing Slantez and Bartley medium and an incubated filter with Enterococci colony growth

Determination of Escherichia coli (E-coli)

Two methods were used for the determination of *E. coli*. Initially a membrane Lauryl Sulphate Broth was used again with the membrane filtration method with direct plating. Later in the project, and also as a means of quality control, *E. coli* were determined using the Colilert-18/Quanti-Tray method.

Membrane Lauryl Sulphate Broth was made up according to the manufacturer's requirements using sterilised glassware and was then stored in a glass bottle and refrigerated until required. Adsorbent pads were dispensed into sterile 55 mm diameter petri dishes and the pads were then soaked with approximately 2.5 – 3 ml of the Lauryl Sulphate Broth using a pipette and disposable pipette tips – the adsorbent pads were kept sterile through the use of a pad dispenser. Samples were then filtered using 0.45 µm 47 mm diameter filters by the same method as described for *Enterococci* above. The filter was then transferred to a pre-prepared petri dish using sterile tweezers and the dish was then closed, labelled and transferred to an incubator and kept at 44°C for 18 hours. After incubation the filter was examined, with a hand lens in a good light, and all yellow colonies were counted as presumptive *E. coli*. A petri dish containing an incubated filter with *E. coli* colony growth is shown in Figure 4.15. Colilert-18 Quanti-Tray analysis is approved by the US-EPA and the tests were carried in accordance with the manufactures recommended procedure. Both of these test methods followed the procedures as set out by the American Public Health Association (1992) in the Standard Methods for Examination of Water and Wastewater

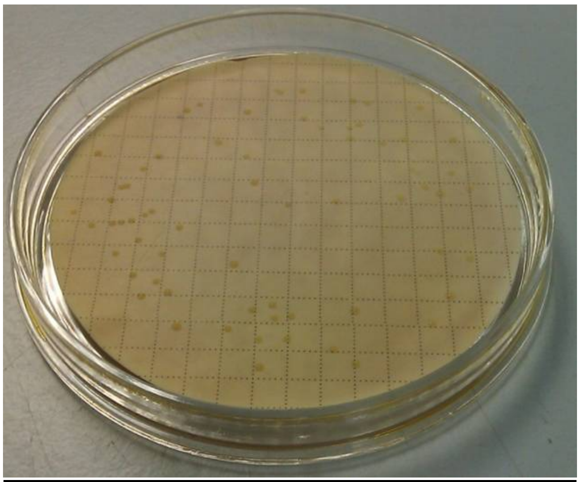


Figure 4.15 Petri dish containing Membrane Laurel Sulphate Broth medium and an incubated filter with *E. coli* colony growth

4.4.2 Chemical Analysis

Throughout the duration of the site monitoring period water samples were analysed for ammonium (NH_4-N), nitrate (NO_3-N), nitrite (NO_2-N), total dissolved nitrogen (N) and chloride (Cl) using a Merck Spectroquant Nova 60 spectrophotometer and the associated

US-EPA approved reagent test kits. Total dissolved phosphorus was also sampled for the duration of the project however due to issues with implementing the laboratory method; sampling of phosphorus only began a number of months into the site monitoring period. The Merck Spectoquant test kits were used only once they had first been deemed suitable for the expected range of values to be found in the water samples and with the water samples themselves being within the recommended pH range for their use. Other parameters were analysed in a less frequent or 'once-off' basis including bromide (Br) and calcium (Ca). All of the chemical analysis was carried out in the Environmental Engineering Laboratory at Trinity College Dublin.

Bromide (Br)

Katz et al. (2011) investigated the use of Cl/Br ratios as an indicator to assess potential impacts on groundwater quality from septic systems. Bromide was therefore analysed on samples for two monthly sampling cycles in order to investigate the ratios present. A Bromide Ion-Selective Electrode (ELIT 8271) was used to access bromide content along with a reference Electrode (ELIT 003n), temperature probe and an ELIT ISE/pH 4 channel Ion Analyser. Standard bromide solutions with concentrations of 0.1, 0.25, 0.5 and 1 mg/l Br were made up using a stock solution of concentration 1000 mg/l Br. The stock solution was made by dissolving 1.489 g of anhydrous potassium bromide (KBr) in distilled water and making up to 1 Litre with distilled water in a 1000 ml volumetric flask. The instrument was calibrated before use using the stock solutions and each of the samples were then analysed using set conditions such as sample volume, depth of probe into sample and sample treatment.

Total dissolved phosphorus

Due to the low concentrations of phosphorus expected in groundwater samples collected during this study, a different analytical procedure was used than the test kits described above. The ascorbic acid test method that was selected for the determination of total dissolved phosphorus and the test procedure was carried out following the standard method outlined by Murphy and Riley (1962). This test method utilises a digestion whereby organically bound phosphorus is completely oxidised by acid persulphate at 120°C. Phosphate ions (PO_4) react with sodium molybdate and antimony potassium tartrate; the resulting compound is reduced by ascorbic acid to form 'molybdenum blue'. The sample is then coloured blue in direct proportion to the amount of phosphate in the sample – the final concentration is then determined spectrophotometrically.

A calibration plot was then prepared using the known PO_4-P solution concentrations and their measured absorbance – see Figure 4.16 below. This plot was then used to calculate the total dissolved phosphorus concentration contained within each of the samples.

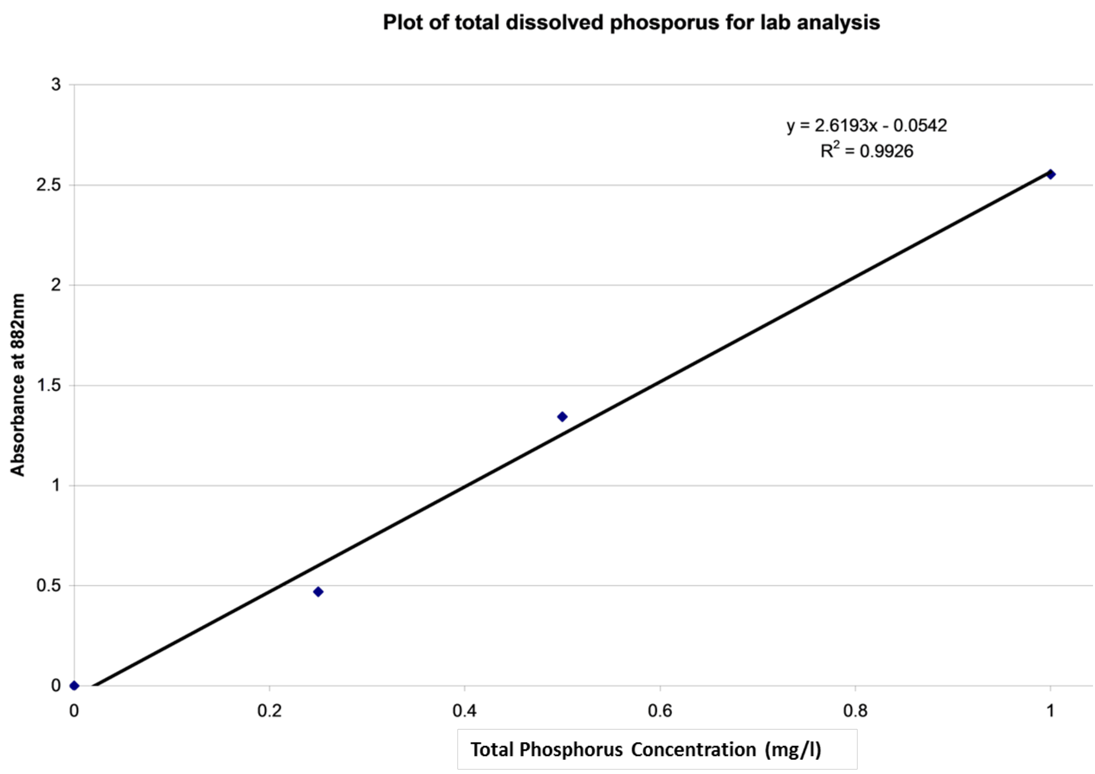


Figure 4.16 Calibration Plot for determining total dissolved phosphorus

4.4.3 Soil Particle Size and XRD Analysis

Soil samples were taken from each of the trial holes that were excavated in order to complete the falling head percolation tests at each of the study areas and a particle size distribution analysis was undertaken in order to determine the contents of clay, sand, gravel and silt. The proportions of each particle size were determined by wet sieve analysis to B.S. 1377 1990 Clause 9.2. This analysis was completed in the Geotechnical Engineering Laboratory at Trinity College Dublin. The sieve set-up and a sieved sample are shown in Figure 4.17 below. X-ray Diffraction Analysis (XRD) was also carried out on the soil samples by the Geology Department in Trinity College Dublin. Approximately 1 g of each soil sample was ground to a fine powder and analysed for mineralogy. Soil mineralogy is important due to its strong influence on soil behaviour, its use in soil classification, and its relevance to soil genetic processes. Of particular interest are the attenuation properties of certain crystalline particles and their role in the removal of certain dissolved chemicals in the percolation water of single wastewater treatment systems. X-ray diffraction is a useful technique that yields detailed information about the atomic structure of crystalline substances and also aids in the identification of minerals in rocks and soils. The XRD analysis was carried out as per the procedure set out by Harris ad White (2007) by the Senior Experimental Officer in the TCD Geology Department.

Figure 4.17 (a) A fully sieved sample from Carrigeen, Co. Kilkenny showing quantities passing the different sieves (b) A full stack of sieves from 2.0 mm down to 0.063 mm

4.4.4 Tracer Probe Calibration and Setup

As outlined previously, it was necessary to determine whether the treatment plant at Toonagh, Co. Clare was connected through groundwater flow paths in the underlying karst bedrock system to the Kilcurrish spring downstream. Previous studies in karst areas such as those carried out by Morales et al. (2007) and Smart et al. (1986) have used tracers to both determine links between karst features and estimate underground flow rates. It was hoped to use fluorescent tracers as part of this study to initially show that a groundwater link exists and subsequently estimate the quantity of groundwater and flow rates between the two karst features. Both Rhodamine WT and Fluorescein fluorescent dyes were used as tracers during this study. Fluorescein was detected downstream using an AquaFluor

Handheld Fluorometer which was calibrated using a single standard which had been made up accurately in the laboratory previously. Due to limitations in the use of this meter and the nature of the tracer studies undertaken, it was decided early on to concentrate on using Rhodamine WT for the majority of tests during the project. It should be noted that the Rhodamine WT dye was used only once agreement had been reached with Clare County Council and local landowners. A Turner Cyclops-7 probe was used both in the field and in the laboratory for the determination of the Rhodamine concentration present in a sample. In order to use the probe it had to first be calibrated against a set of known standards. This was carried out in the TCD Environmental Laboratory and a satisfactory calibration was achieved at all gain settings. The details of this calibration are given in Table 4.2 and Figure 4.18 – Figure 4.20 below.

Table 4.2 Calibration of Cyclops-7 Rhodamine Probe

		Gain Setting		
Raw RWT Concentration (ppb)	**Active RWT Concentration (ppb)**	**x1**	**x10**	**x100**
0	0	0.026	0.025	n/a
10	2	0.028	0.047	0.239
20	4	0.032	0.092	0.696
50	10	0.04	0.165	1.43
100	20	0.055	0.313	2.88
250	50	0.0987	0.747	n/a
500	100	0.17	1.469	n/a
1000	200	0.306	2.895	n/a

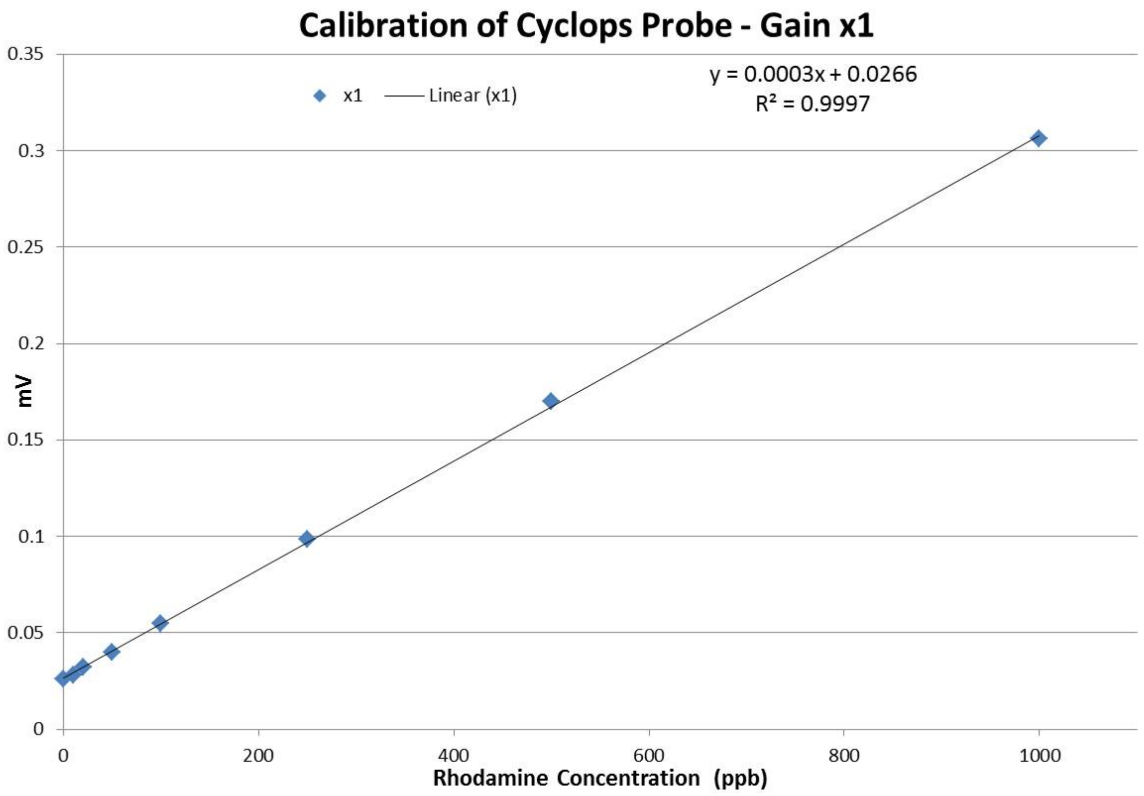


Figure 4.18 Calibration Regression Plot of Cyclops Rhodamine Probe at x1 Gain Setting

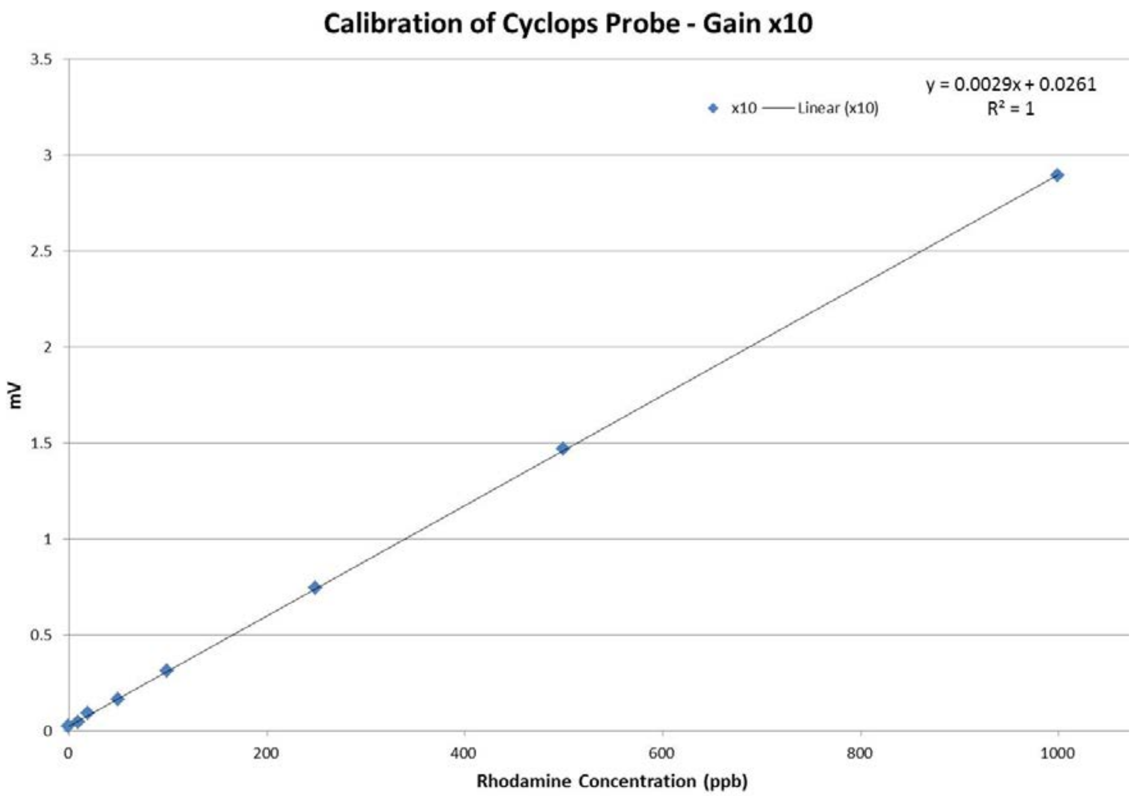


Figure 4.19 Calibration Regression Plot of Cyclops Rhodamine Probe at x10 Gain Setting

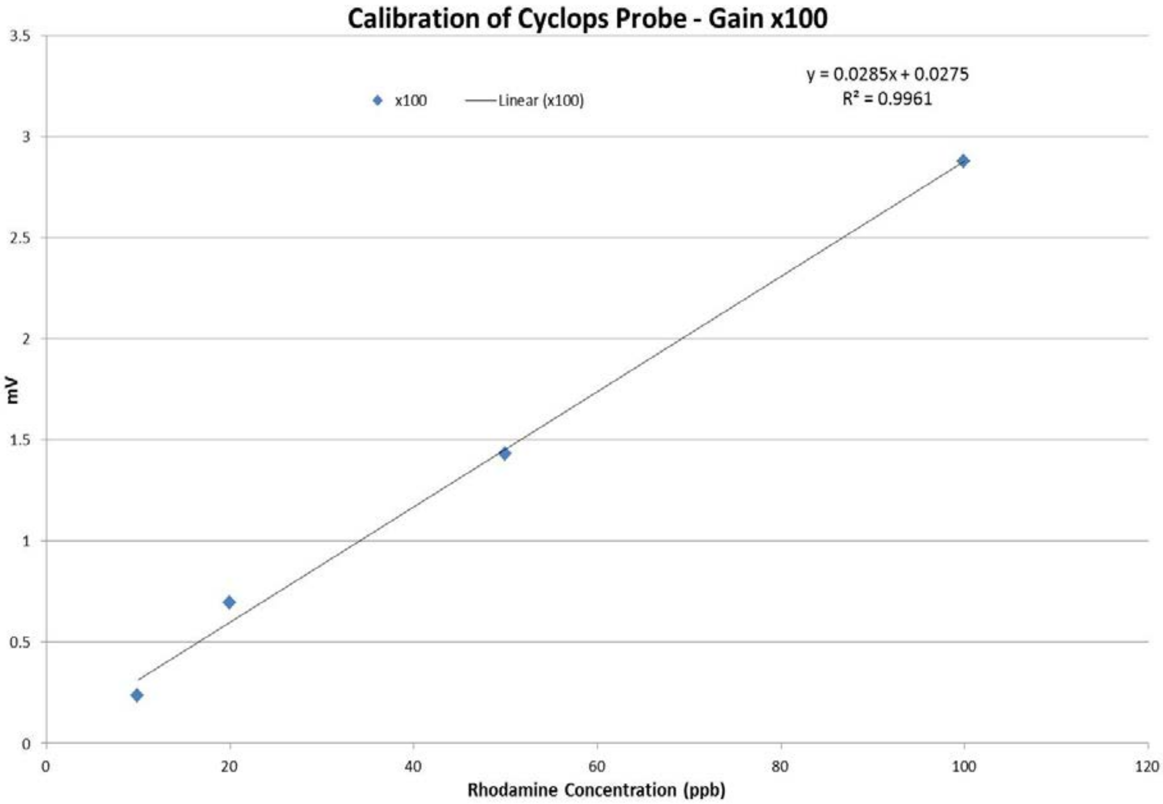


Figure 4.20 Calibration Regression Plot of Cyclops Rhodamine Probe at x100 Gain Setting

It was anticipated that the Rhodamine tracer would take at least 2 – 3 days to reach the Kilcurrish spring which served as the downstream sampling point due to anticipated flow rates. In order to record the progression of the tracer through the groundwater system the Cyclops-7 rhodamine probe was installed at the downstream sampling point together with a TinyTag data logger and left for a period of 5 days to accurately define the tracer breakthrough curve. The instrument was set-up using the x10 gain setting and the datalogger was set to record voltage at 1 minute intervals. The instrument set-up is shown in Figure 4.21 below.

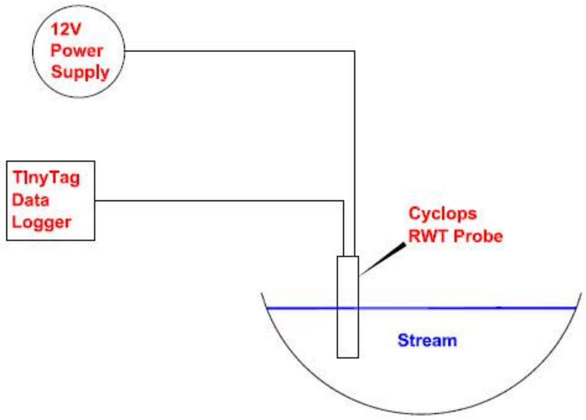


Figure 4.21 Setup of Cyclops Rhodamine Probe in the Field

In addition to the use of a Rhodamine WT probe in the field, charcoal bags were also used in order to confirm the presence of the dye at the downstream monitoring point. Given the highly porous nature of the activated charcoal used, the dye is attracted and absorbed as it passes the monitoring location. The dye is then removed from the charcoal using an eluting solution in the laboratory. The elution solution was made up as described by Aley (2008) and was a mixture of 5% aqua ammonia and a 95% isopropyl alcohol solution with sufficient potassium hydroxide flakes to saturate the solution. The isopropyl alcohol solution was 70% alcohol and 30% water and the aqua ammonia solution was 29% ammonia. The potassium hydroxide flakes were added until a super-saturated layer was visible at the bottom of the container. After recovery from the field charcoal bags were soaked in the eluting solution for a 24 hour period in the laboratory and the solution then be analysed using the rhodamine probe.

4.4.5 Quality Control Procedures

Due to budget limitations on the project it was not possible to have samples reviewed by an external accredited water laboratory. Samples were therefore duplicated on a random basis and the test kits were regularly checked against known standards. In addition *E. coli* was checked for a number of months using two methods as outlined in Section 4.4.2 above with a very good agreement found in all cases. The method for determining total dissolved phosphorus was checked by the Environmental Sciences Laboratory in Trinity College. Duplicate samples were analysed in the Environmental Engineering and Environmental Science Laboratories for a random selection of samples and again a very good agreement was found between both. In addition the quality of results was ensured by following the highest quality laboratory procedures possible which included:

- The use of dedicated glassware for the various analysis procedures
- Constant sterilisation of sample bottles and laboratory equipment between and during sample runs
- Bacterial contamination was restricted as all sample equipment and sample bottles were autoclaved at 121°C before use
- No equipment was shared in the laboratory with other operators in order to preserve the quality control procedures
- Test kits were regularly checked using the recommended Merck combi-check kits
- All laboratory equipment was regularly calibrated and checked
- All equipment was calibrated either the day of or the day previous to a site visit

4.5 Field Methods

4.5.1 Pumping Tests – Piezometer Quality Control

Pumping tests were undertaken at each of the nested piezometers in order to ensure the independence of each sampling horizon. By design each piezometer must be kept independent from each other through the use of bentonite seals and grouting, however in some cases this seal may not be effective due to poor construction practices. In order to assess the independence of the piezometers a pumping test was carried out which involved pumping one of the piezometers for an extended period of time whilst monitoring the water level in the others to check for any fluctuations in their water level. If the water level changes significantly in adjacent piezometers while pumping another then it indicates that an effective seal was not achieved and the piezometers are not independent of each other.

Pump Test Results

Each of the piezometers was pumped separately for an extended duration until steady state drawdown was achieved; this duration varied for each borehole from 30 minutes up to many hours. Results of the pump tests for the *MODERATE*, *HIGH* and *EXTREME* vulnerability sites are given in Table 4.3 – Table 4.5 below. The results for the LOW vulnerability site are not shown here as there was no movement in any of the adjacent piezometers following pumping of each of the piezometers individually. It can be seen that when each of the piezometers were pumped only very slight drops in adjacent piezometers were recorded typically less than 5 mm. Given that the localised water table was being drawn down due to pumping in a single piezometer it was reasonable to expect some small change, however as the magnitude of these changes was small it was concluded that each of the seals between the piezometers had been achieved satisfactorily. The independence of the individual piezometers can be verified further given the different water chemistry results observed – see Chapter 5 for details.

Table 4.3 Pumping test results at Rhode, Co. Offaly

Borehole Purged	Drawdown (m)	Change in rest water level (m)		
		BHO1a	**BHO1b**	**BHO1c**
BHO1a	0	-	0	0
BHO1b	0	0.006	-	0
BHO1c	0	0.005	0	-
		BHO2a	**BHO2b**	**BHO2c**
BHO2a	0.016	-	0.006	0
BHO2b	0.015	0	-	0
BHO2c	9.93	0	0	-
		BHO3a	**BHO3b**	
BHO3a	5.39	-	0	
BHO3b	0.1	n/a	-	

Table 4.4 Pumping test results at Carrigeen, Co. Kilkenny

Borehole Purged	Drawdown (m)	Change in rest water level (m)		
		BH-K2a	**BH-K2b**	**BH-K2c**
BH-K2a	0.05	-	0	0
BH-K2b	0.45	0	-	0
BH-K2c	0.38	0	0	-
		BH-K3a	**BH-K3c**	
BH-K3a	dry	-	n/a	
BH-K3c	1.23	n/a	-	

Table 4.5 Pumping test results at Faha, Co. Limerick

Borehole Purged	Drawdown (m)	Change in rest water level (m)		
		BHL1a	BHL1b	BHL1c
BHL1a	0.024	-	0	0
BHL1b	0.56	0	-	0
BHL1c	6.39	0.06	0.08	-
		BHL2a	BHL2b	
BHL2a	4.22	-	0	
BHL2b	1.98	0.04	-	
		BHL3a	BHL3b	BHL3c
BHL3a	0	-	0	0
BHL3b	0	0	-	0
BHO3c	0.03	0	0	-

4.5.2 Slug Tests

Slug tests were carried out on all piezometers located in the bedrock horizon with a view to estimating a value for transmissivity and ultimately calculating an estimate for the hydraulic conductivity (k) of the aquifer needed later in order to develop the numerical models. Slug tests were carried out in accordance with Section 7.2 of BS ISO 14686:2003 (BSI, 2003). A slug was manufactured of stainless steel in the TCD materials laboratory. For all tests it was assumed that the water level during the test would remain above the screened section of the piezometer.

The procedure used in conducting the slug tests was as follows:

- Rest depth to water level was recorded
- The slug was secured to the borehole up-stand using high tension cable
- The depth that the slug would be allowed into the water was fixed at 1 m by measuring and marking the suspension cable (see diagram in Figure 4.22)
- A Groundwater Datalogger or Diver was installed into the piezometer at a position well below the depth of the test (~5m) and secured at the surface to the borehole up-stand

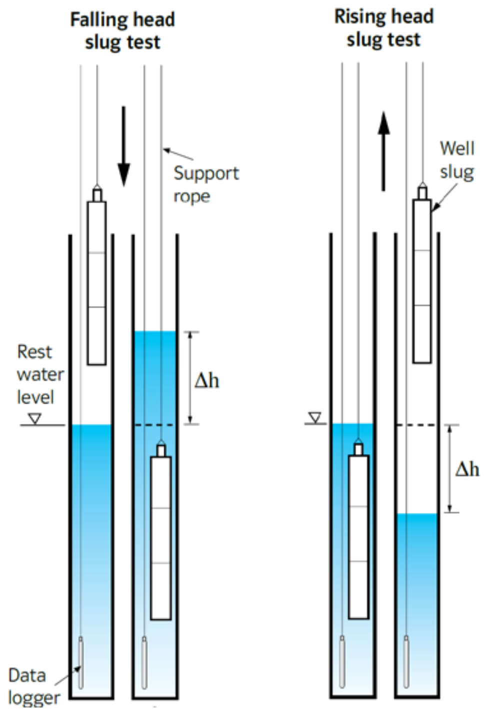


Figure 4.22 Setup of Slug Tests (Waterra, 2010)

Slug Test Part (a): Falling Head Test

- The slug was then dropped to the position 1 m below the rest water level as quickly as possible

- The displaced water level was checked at regular intervals at the surface using a dip-meter until either the water level returned to its rest position or if the rate of recovery was too long at least 70% recovery had occurred.

The diver was set to record data at 1 second intervals from 30 minutes prior to the start of the test. It is recommended that a data-logger be used to record the test as in higher permeability formations full recovery can be within seconds. In lower permeability formations the test can take hours or even days and in these instances a dip-meter would be sufficient.

Slug Test Part (a): Rising Head Test

- Once Recovery had been achieved the slug was pulled out of the as quickly as possible

- Again the displaced water level was checked at regular intervals until recovery had occurred

4.5.3 Falling Head Soil Percolation Tests

Falling Head Soil Percolation Tests (T-tests) were carried out at all of the study sites in order to gain an estimate of the field saturated hydraulic conductivity of the subsoil. These tests were carried out in accordance with the EPA Code of Practice: Wastewater Treatment Systems for Single Houses (EPA, 2009) – see Figure 4.23. The test procedure involves the excavation of a hole adjacent to the likely location of the percolation areas and at a depth representative of where the infiltrating effluent would enter the subsoil. This hole was then filled with clean water 24 hours prior to undertaking the test to provide pre-soaking. The hole was then refilled with clear water up to the 400 mm the time noted. The water was then allowed to drop to 300 mm and the subsequent time required for the water to drop to 200 mm was recorded. The hole was then refilled up to 300 mm and the time required for it to drop to 200 mm recorded again. This procedure was then repeated. The average time required for the water to drop from 300 mm to 200 mm was divided by four to give the time required for a fall of 25 mm or the t-value. A second hole was also excavated at some of the study sites and the modified T-test method was employed as outlined in the EPA Code of Practice. This method is similar to the standard method however the time of drop of the water is recorded only once in 50 mm intervals from 400 mm to 100 mm and the t-value is calculated for each 50 mm drop using various conversion factors and averaged to form an estimate for the T-value.

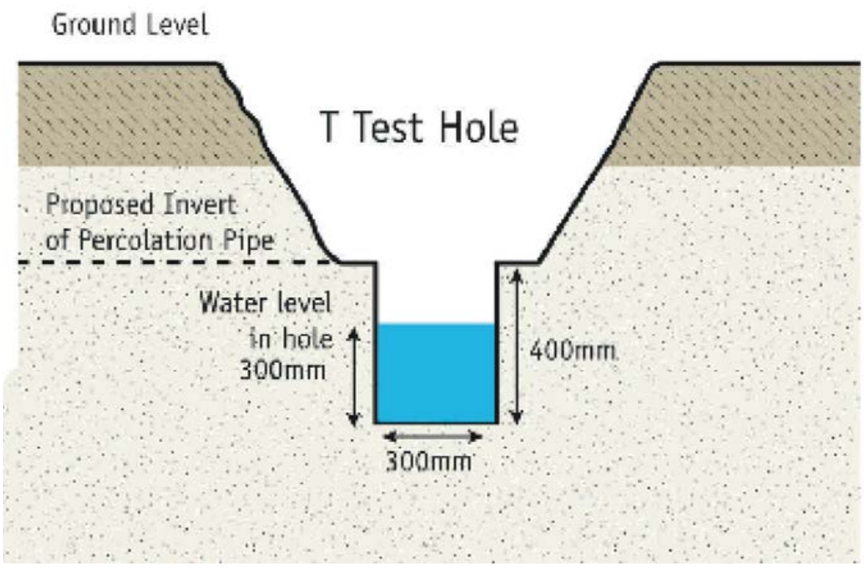


Figure 4.23 Setup of T-test (EPA, 2010)

4.5.4 Topographical Surveying

A topographical survey was carried out at all of the study locations in order to determine accurate co-ordinates and elevations of:

- Tops of monitoring boreholes (and therefore accurate depths of piezometers)
- Riverbeds/Streambeds (where applicable)
- Locations of Treatment systems/percolation area (where possible)

In addition to surveying the above, topographic points were also recorded at regular intervals inside the general area of the study sites in order to develop accurate contours of elevation and to supplement data requirements for groundwater modelling (see Chapter 8).

A Trimble 4700 GPS System was used to carry out the topographic surveys. The GPS system consisted of a Trimble R6 Rover and Base Station with a Trimble TSC2 Survey controller. At each of the surveyed areas the base station was levelled and its exact topographic location was obtained via a mobile phone connection to the Ordnance Survey Ireland's headquarters in Dublin. Once the base station's Eastings and Northing co-ordinates relative to the Irish National Grid and elevation relative to Malin Ordnance Datum (i.e. mOD) was established, the survey controller then calculated all of the local topographic data based on the known co-ordinates of the base station. All of the topographic points were recorded using the portable rover which was connected to the base station by means of radio communication. The data was stored on the survey controller and was then downloaded onto a computer for processing once the survey was completed.

4.5.5 Stream Gauging

Salt Dilution Gauging

It was necessary to determine stream discharge for the site at Toonagh, Co. Clare. Given that low flow in the stream was typically at a depth of 0.1 m or less it was not always possible to use a flow-meter. It was decided to use salt dilution gauging to determine discharge in the stream in these instances. This method involves a known weight of salt being injected into the stream at an upstream point and then some distance downstream the Electrical Conductivity (EC) is monitored over time in order to produce a breakthrough curve. This method is based upon the fact that electrical conductivity is a measure of the ease with which an electrical current can travel through water and therefore at low salt concentrations electrical conductivity in stream-water tends to vary linearly with the salt concentration as given by:

$$C = k(EC - ECbg)$$

where:

C is the concentration of the salt in stream water
EC is the electrical conductivity of the stream-water-injection solution mixture
ECbg is the background or electrical conductivity of the stream-water
k is a proportionality constant, to be determined by calibration.

A plot of electrical conductivity and concentration against time produce what is referred to as a break through curve. A conversion factor (0.0047 for NaCl) is used to change from measured electrical conductivity values to concentrations in g/l of salt. The flow in the stream is then calculated by integrating the area beneath the breakthrough curve using Simpson's rule or otherwise in a spread sheet. A typical breakthrough curve is shown in Figure 4.24 below.

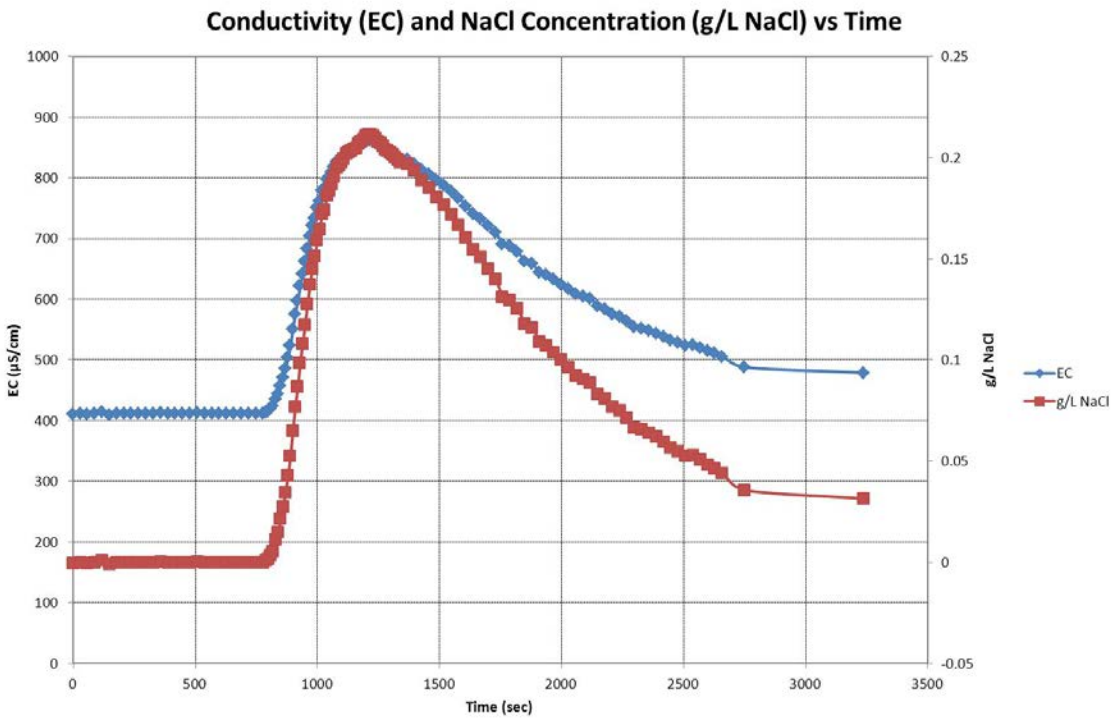


Figure 4.24 A breakthrough curve for salt dilution gauging

Figure 4.25 A flow-meter being used at Toonagh, Co. Clare

Digital Flow-meter

A digital flow meter was used when the river gauge was at a suitable level for its use. Flow was measured at two thirds of total depth and was measured at a number of points across the width of the stream. The discharge in the stream was then averaged from all these measurements. Figure 4.25 shows a flow-meter being used at Toonagh during a period of high flow in the Toreen east stream.

5 ANALYSIS OF FIELD RESULTS

5.1 Introduction

This chapter presents an analysis and interpretation of the field results obtained during the course of this study. Summary values either in tabular or time series plot format are presented in each of the appropriate sections. Complete breakdowns of all results collected during this study are given in Appendix C.

Analysis of all field data will be presented first including field tests undertaken and laboratory analysis resulting from these field tests such as soil analysis. The chemical and bacterial results will then be presented and discussed and lastly the results of the tracer study at Toonagh Co. Clare will be presented. Chemical Oxygen Demand (COD) test results are not included in this Chapter due to the reasons outlined in Chapter 4, but are listed in Appendix C.

5.2 On-site Assessment

5.2.1 Low Vulnerability Site – Naul, Co. Dublin

Falling Head Percolation Test

Falling head percolation tests were carried out at each of the study sites as described in Chapter 4. No obvious evidence of mottling or segregation of strata was observed at Naul, Co. Dublin during excavation of the trial hole which is shown in Figure 5.1 below and was excavated to a depth of 1.2m. The T-test was carried out using both the standard and modified method and the results are shown in Table 5.1 and Table 5.2 below. The standard T-test indicated a T value of 23.3 mins/25mm for the subsoil with the modified T-test indicating a slightly lower value of 21.2 mins/25mm (see Appendix B for details of all site characterisation forms). The results of the T-tests indicate that the subsoil provides very favourable conditions for the installation of a percolation system for the disposal of wastewater in this area.

Figure 5.1 (a) Percolation Test and (b) Soil Profile at the Naul, Co. Dublin

Table 5.1 Percolation Standard T-test Results for Naul, Co. Dublin

Standard T-test			
Fill No.	Start time (300mm)	Finish time (200mm)	Δt (hours)
1	11:31:00	12:46:00	01:15:00
2	12:51:00	14:20:00	01:29:00
3	14:25:00	16:22:00	01:57:00
	Average Δt (hours)		01:33:40
	Average Δt (mins)		93
T-value = 23.25 (minutes/25mm)			

Table 5.2 Percolation Modified T-test Results for Naul, Co. Dublin

Modified T test							
Depth (mm)	Time	Fall of water (mm)	Time of fall (hours)	Time of fall (mins)	Time factor (T_f)	K_{fs}	T Value
300	11:36:00	300 - 250	00:31:00	31	8.10	0.26129	17.03086
250	12:07:00	250 - 200	00:47:00	47	9.70	0.206383	21.56186
200	12:54:00	200 - 150	00:53:00	53	11.90	0.224528	19.81933
150	13:47:00	150 - 100	01:23:00	83	14.10	0.16988	26.19504
100	15:10:00						
Results of T-test = 21.2 (minutes/25mm)							

<u>Soil Textural Classification</u>

As per BS 5930:1999 and EPA (2009) there exists a flow chart for describing subsoils based on the texture of the subsoil. This classification is based upon a thread test and a ribbon test. A dilatency test is then used to differentiate between silts and clays. At Naul, Co. Dublin, 4 threads were formed with ribbons that were 110mm in length. The dilatancy tests were uncertain leading to the classification of the subsoil as a SILT/CLAY. Given the gritty feel of the samples in the field this classification was further refined to a sandy SILT/CLAY.

Particle Size Analysis

Particle Size Distribution analysis (PSD) can be used both to classify the soil samples and also to give an indication of the estimated saturated hydraulic conductivity (K_s). Figure 5.2 taken from CEN/TR 12566-2:2005 Appendix E provides a method of estimating the soil classification based only upon the results of the PSD. PSD results for Naul are given in Table 5.3 below. Using the soil content proportions given in Table 5.3 the soil can be classified using Figure 5.2 as clay LOAM (the location for the Naul is shown with a blue dot).

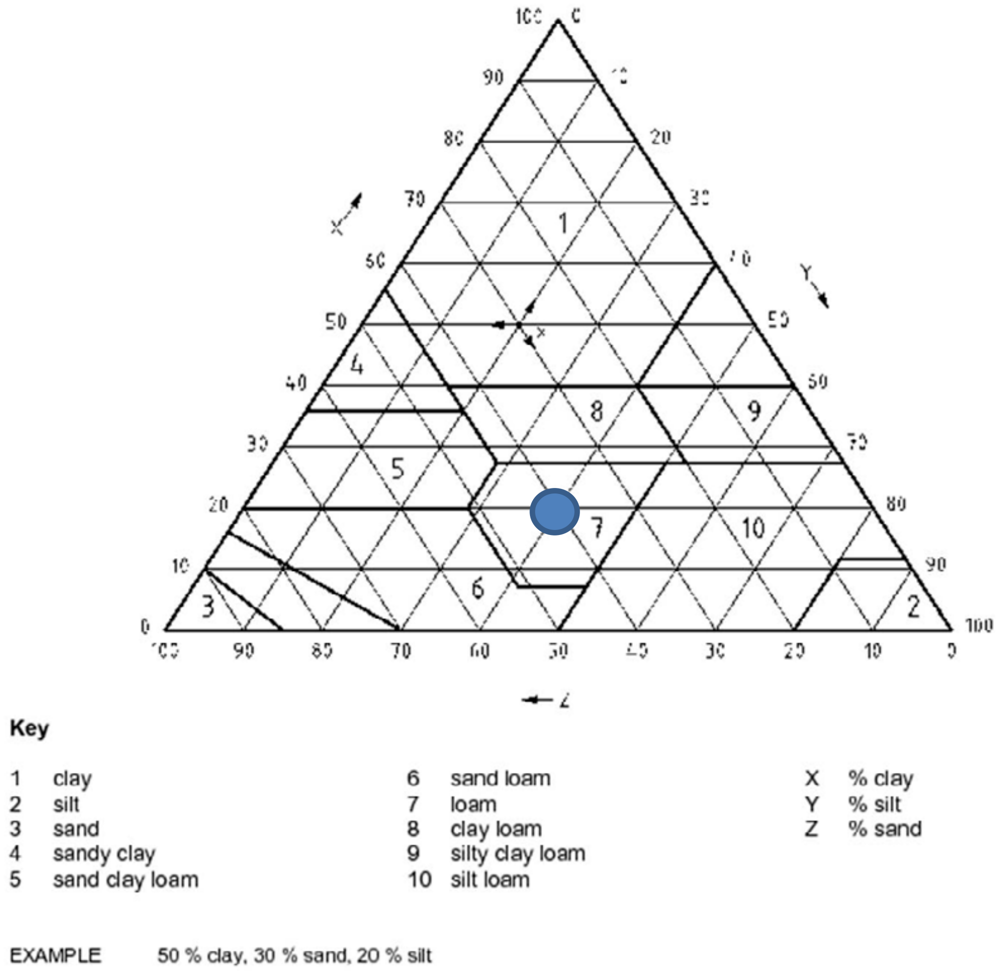


Figure 5.2 Soil Classification chart with Naul site shown (CEN/TR 12566-2:2005 (E))

Given that the soil maps for the area (discussed in more detail in Chapter 3) classify the area as tills or boulder clays, it is surprising that there is such a high proportion of sands and silts present. It is possible that due to the intensive agriculture in the area that the topmost strata of subsoil consist of made ground to improve the quality of horticulture output. Another possible explanation for the high sand/silt content is the highly heterogeneous nature of till deposition with sorting of fines common in many areas. During drilling at this location heavy impermeable black boulder clays were encountered almost constantly from 1 – 2 m down to depths of 35 – 40 m and it is therefore likely that the clay LOAMs that were encountered during excavating of the trial holes are not extensive, and that the predominant subsoil deeper down is in fact a CLAY.

Table 5.3 Particle Size Analysis of Soil Samples from Naul, Co. Dublin

Depth of sample below ground level	% Gravel	% Sand	% Silt	% Clay
1.1m	12.2	42.3	13.2	32.3

Empirical equations have been developed that relate PSD results to soil saturated permeability and the most widely used of such equations is the modified Hazen equation (Marshall and Holmes, 1988). The HYDRUS unsaturated zone modelling software uses a model called Rosetta to estimate K_s with two basic types of Pedotransfer Functions (PTF) (Class PTF and Continuous PTF), allowing the estimation of van Genuchten water retention parameters using limited (textural classes only) or more extensive (texture, bulk density and one or two water retention measurements) input data. Using the Hazen equation the K_s for soils at Naul, Co. Dublin has been estimated at 0.0026 m/day whereas the Rosetta model fitted a value of 0.121 m/day. Given that the T-value for the soil indicates a K_s of 0.2 m/day (Mulqueen and Rodgers, 2001), it can be seen that the three estimations are slightly at odds. A possible explanation for this could be the presence of localised preferential flow paths in the soil in the vicinity of the area where the T-test was carried out.

X-ray Diffraction Analysis (XRD)

X-ray diffraction (XRD) analysis was carried out on the soil samples described above. Interpretive plots were produced are contained in Appendix D. The soil was found to contain lithian muscovite with some quartz and ordered albite. Both the muscovite and albite minerals contain aluminium oxide hydroxide and this has been found to adsorb phosphorus due to the presence of the hydroxyl groups (Tanada et al., 2003). It is likely therefore that this soil will provide good attenuation of phosphorus, as has generally been found in previous field trials in Ireland on on-site effluent.

5.2.2 Moderate Vulnerability Site – Rhode, Co. Offaly

Falling Head Percolation Test

During excavation of the trial hole at Rhode, Co. Offaly a reddish-brown strata of soil that resembled a "Ironpan" was encountered at approximately 0.6m below ground level as shown in Figure 5.3 below. Given the colour and texture of this layer it is likely that this dense matrix was formed by iron oxides and perhaps calcium carbonate. Ironpans (or hardpans) usually form just below the topmost layer of soil in areas that have a specific fine grained soil structure (usually clays) and are acidic in nature. Hardpans can form naturally or can also form due to agricultural practises whereby repeated ploughing and trafficking of the topsoil can compact the soil causing the formation of the hardpan layer. Hardpan layers are usually very impervious to water and can reduce infiltration of rainwater recharge and cause the topsoil to become poorly drained. However, given the lands at Rhode, Co. Offaly are visibly well drained, it is possible that the hardpan encountered may be quite localised. No evidence of mottling was encountered in the trial hole and neither was the water table. Due to the presence of the hardpan soil samples for laboratory analysis were taken both above and below 0.6 m in order to identify any changes in subsoil properties in the vicinity of the soil infiltration systems. Both standard and modified T-tests were carried out and the tests resulted in T values of 34.5 mins/25mm and 25.6 mins/25mm for the soil in this area. These T values indicate very favourable conditions for the installation of soil infiltration wastewater disposal systems at a depth of 1.1m in the subsoil.

Soil Textural Classification

Soil classification was carried out on both samples above and below 0.6 m depth. Above 0.6 m 1 - 3 threads were formed with ribbons that were 80 mm in length. And the soil was dilant classifying the soil as SILT. Below 0.6 m 2 – 4 threads formed with threads of up to 110 mm in length and there was difficulty with dilatancy leading to the classification of a SILT/CLAY. Again given the gritty nature of the samples in the field this can be classified further as a sandy SILT/CLAY with the sample above 0.6m being further classified as a sandy SILT.

Figure 5.3 Soil Profile at Rhode, Co. Offaly

Particle Size Analysis

As outlined earlier samples were taken at two depths below ground level at Rhode Co. Offaly due to the presence of a hardpan at approximately 0.6 m below ground level. Results of the two particle size analysis are shown in Table 5.4 below. The soil above the hardpan had a higher percentage of sand and gravel with the soil below the hardpan having a higher proportion of clay present.

Table 5.4 Particle Size Analysis of Soil Samples from Rhode, Co. Offaly

Depth of sample below ground level	% Gravel	% Sand	% Silt	% Clay
0.5m	37.5	37.4	14.8	10.3
1.2m	29.3	33.5	14.4	22.8

Using the soil content proportions given in Table 5.4, the soil can be classified using Figure 5.2 as a silt LOAM above 0.6m and a CLAY below 0.6m. This classification compares well with the classification based on the soil textural analysis.

The Rosetta software package in HYDRUS calculated K_s values for soils at Rhode, Co. Offaly 0.09 m/day above 0.6 m and 0.13 m/day below 0.6 m depth. Given that the T-value for the soil was 34.5 mins/25mm, implying a K_s of 0.12, the values are in a similar range to those estimated by the percolation tests.

X-ray Diffraction Analysis (XRD)

X-ray diffraction (XRD) analysis was carried out on the soil samples described above. Interpretive plots were produced are contained in Appendix D. The soil above 0.6 m was found to contain calcite, lithian muscovite and quartz with some ordered albite. Below 0.6 m the soil contained less calcite, quartz and muscovite. Due to the presence of the aluminium oxide hydroxide it is likely that this this soil will provide good attenuation of phosphorus. The calcite content is to be expected due to the till be derived chiefly from limestone rock.

5.2.3 High Vulnerability Site – Carrigeen, Co. Kilkenny

Falling Head Percolation Test

Both a standard and a modified T-test were carried out at Carrigeen, Co. Kilkenny which is shown in Figure 5.4 below. No evidence of mottling or a high water table were encountered when excavating the trial hole. The standard T-test indicated a T value of 20.25 mins/25mm with the modified T-test indicating a T value of 18.3 mins/25mm. The calculated T values at Carrigeen indicate that very favourable conditions exist in the area for the installation of soil infiltration wastewater disposal systems.

Figure 5.4 Percolation Test at Carrigeen, Co. Kilkenny

Soil Textural Classification

Soil textural classification was carried out the subsoil sample with 1 - 2 threads being formed with ribbons that were 60 - 80 mm in length. Given that the soil was only dilatant with difficulty it can be classified as a SILT/CLAY. The soil was highly gritty leading to the further classification as a sandy SILT/CLAY.

Particle Size Analysis

Results of the soil particle analysis for samples recovered at Carrigeen Co. Kilkenny are shown in Table 5.5 below. Using the soil content proportions given in Table 5.5, the soil can be classified using Figure 5.2 as a sandy LOAM. This classification compares well with the classification based on the soil textural analysis given that a loam is defined as a mixture of silt clay and sand and the predominant fraction is sand.

Using the Rosetta software in HYDRUS the K_s for soils at Carrigeen, Co. Kilkenny has been estimated at 0.144 m/day. Given that the T-value for the soil was 20.25 mins/25mm implying a K_s value of 0.207 m/day, it can be seen that the Rosetta package (which is based upon soil textural information only) predicts a value in a similar range to that estimated by the percolation tests.

Table 5.5 Particle Size Analysis of Soil Samples from Carrigeen, Co. Kilkenny

Depth of sample below ground level	% Gravel	% Sand	% Silt	% Clay
0.9m	15.5	38.3	19.6	26.6

X-ray Diffraction Analysis (XRD)

X-ray diffraction (XRD) analysis was carried out on the soil sample described above. Interpretive plots were produced are contained in Appendix D. The soil was found to contain calcite, lithian muscovite and quartz with some ordered albite. As was found with the previous soil samples it is likely that this this soil will provide good attenuation of phosphorus mainly due to the albite and muscovite minerals contained within the soil matrix.

5.2.4 Extreme Vulnerability Site – Faha, Co. Limerick

Falling Head Percolation Test

During excavation of a trial hole at Faha, Co. Limerick two distinct soil strata were encountered. From ground level to approximately 0.55 m the soil was very fine textured with a high proportion of organics (~15%). Below 0.55 m the soil became very wet and soft with a high proportion of fines in particular silts again with a large amount of soils resembling peats at places. Samples were taken from both strata for laboratory analysis in order to identify any difference in soil properties. Below 0.68 m water began to ingress into the trial hole and filled to quickly reach a level of 0.65 m below ground level as shown in Figure 5.5 below. This is consistent with the elevation of the water table that was encountered in the area at the three monitoring boreholes. In this context the area is not suitable for a conventional soil infiltration wastewater disposal system. Any treatment system installed in the area would need to include a raised or mounded percolation area. This is consistent with what was observed in the area with nearly of the dwellings built on areas that appear to have been in-filled to raise the entire sites by at least one meter. Due to the high water table a T-test could not be carried out and a standard P-test was instead carried out. The P-test indicated a P value for soils in this area of 38.75 mins/25mm. This P value would indicate that the soil is suitable for the installation of a soil infiltration wastewater disposal system subject to the inclusion of a raised or mounded disposal system in order to achieve the appropriate thickness of unsaturated subsoil above the water table (i.e. 0.9 m for secondary treated effluent or 1.2 m for septic tank effluent).

Figure 5.5 Soil Profile at Faha. Co. Limerick Showing Ingress of Water

Soil Textural Classification

Soil textural classification was carried out on both samples above and below 0.55 m depth. Above 0.55 m 1 - 3 threads were formed with ribbons that were 80 mm in length. And the soil was dilatant classifying the soil as SILT. Below 0.55 m 5 threads formed with threads of up to 130 mm in length and there was difficulty with dilatancy leading to the classification of a CLAY. Below 0.55 m the soil was gritty leading the further classification of a sandy CLAY.

Particle Size Analysis

Samples were taken at two depths below ground level at Faha Co. Limerick as it appeared clear that the upper layer of soil in the trial hole had very different properties to soil at a greater depth. Results of the two particle size analysis are shown in Table 5.6 below which revealed that the two soil samples were in fact more similar than had been expected. The upper soil had a higher percentage of sand and gravel with the soil at greater depth having a higher proportion of silt and clay present. Given that the area was historically a floodplain for the nearby River Maigue the high proportion of silt in both soil horizons is expected. Using the soil content proportions given in Table 5.6 the soil at both depths can be classified using Figure 5.2 as a silt LOAM. This classification is at variance with the soil textural classification of a sandy SILT/CLAY. However, both classifications would indicate a low soil permeability.

The Rosetta software estimated the K_s for soils at Faha, Co. Limerick at 0.06 m/day for below 0.55 m and 0.11 m/day above 0.55 m. Given that a T-test was not carried out it is difficult to make a comparison with the results of the P-test which was carried out only 400 mm below the soil surface. This value does give an indication that subsoil permeability in the area is low due to the presence of a very fine silt and clay soil matrix.

Table 5.6 Particle Size Analysis of Soil Samples from Faha, Co. Limerick

Depth of sample below ground level	% Gravel	% Sand	% Silt	% Clay
0.4m	10.7	30.1	47.9	11.4
0.75m	2.9	21.9	58.4	16.8

X-ray Diffraction Analysis (XRD)

X-ray diffraction (XRD) analysis was carried out on the soil sample described above. Interpretive plots were produced with the plot for the sample above 0.55 m shown in Figure 5.6 below. The second plot for the sample below 0.55 m is contained in Appendix D. Both soil samples were found to contain calcite, lithian muscovite and quartz with some ordered albite – again likely indicating good phosphorus attenuation. However, the sample above 0.55 m was found to contain dolomite. The bedrock permeability of the area will be discussed in more detail in Chapter 6, however the presence of this dolomite in the sample indicates that estimation that the limestone bedrock in the area has become partially dolomitic in nature is supported given the identification of this mineral in the subsoil profile.

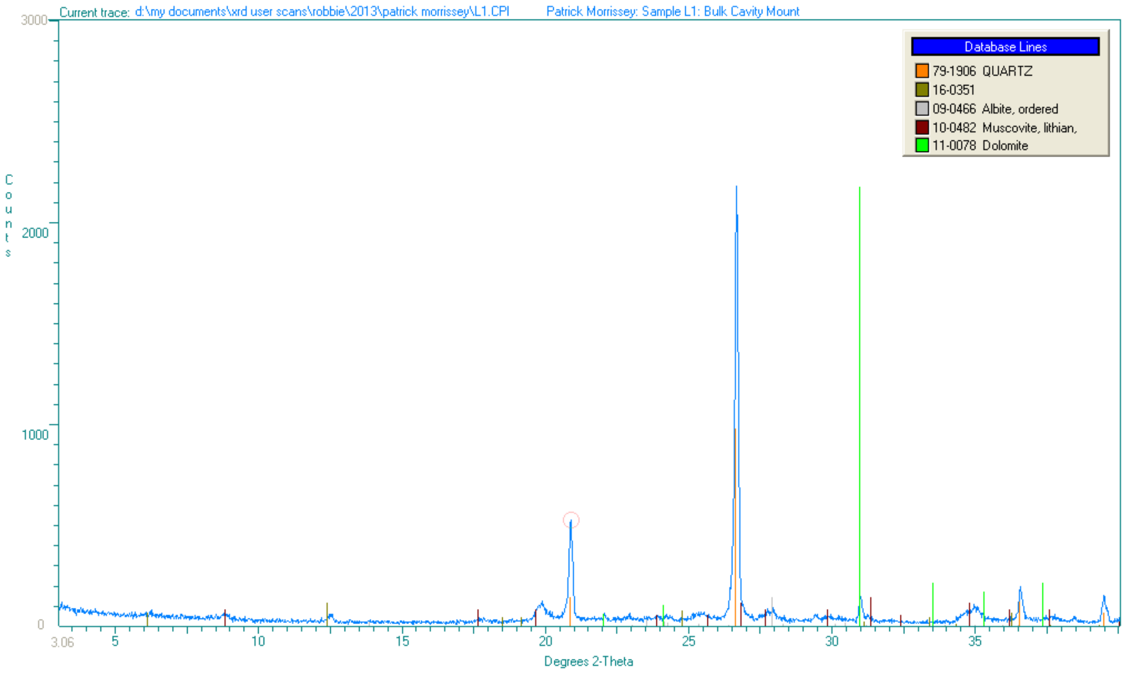


Figure 5.6 XRD Trace Results for soil sample above 0.55m at Faha, Co. Limerick

5.3 Field Tests and Data Analysis

5.3.1 Slug Tests

Slug tests were carried out at each of the sites where boreholes were drilled with the exception of the low vulnerability site at Naul, Co. Dublin, in order to determine an estimate for the hydraulic conductivity of the underlying aquifer. These tests were not carried out at the Naul due to the high quantity of available data for the area that had been carried out previously. Given the issues that can occur with slug tests (described below) it was considered that the data already available would be a more accurate and reliable reflection of bedrock permeability in the area. Slug tests were carried out on the deepest piezometer at each of the boreholes at the other three sites. Whilst some of the slug tests gave satisfactory result, the majority did not yield any meaningful or useful results and were on a whole not successful. Research by Black (2010) confirms the difficulties in conducting such tests particularly when such small diameter piezometers are involved. In most cases no discernible changes in water levels could be identified on the water level logger and therefore results could not be calculated. Overall three of the seven tests carried out yielded some form of results however given the issues that were encountered in carrying out the tests as well (as the arguments set out by Black, (2010)) the results shown here must be treated with a high degree of caution and used in context with other available information for the area.

Results for the slug test were calculated using the Hvorslev method as described by Campbell et al. (1990). The method involves determining the ratio H/Ho, where Ho is the distance the water rises/declines after the addition/removal of the slug. Using a semi-log plot of H/H_o against elapsed time the time taken for the water level to recover to 37% of its initial level can be determined (i.e. $H/H_o = 0.37$). Hydraulic conductivity is then determined using:

$$K = \frac{r^2 ln\left(\frac{L}{R}\right)}{2LT_0}$$

where:

r is the piezometer diameter

L is the screen length

R is the borehole diameter

T_o is the time required for the water level to recover to 37% of initial

The Hvorslev method is generally only suitable for wells that have a length that is eight times larger than the radius of the well screen. This condition was met for all of the boreholes drilled as part of this study and therefore the method was applicable in principle. Results of the slug tests carried out at Rhode, Co. Offaly are shown in Table 5.7 below. The estimated hydraulic conductivity for the bedrock aquifer in which the piezometer is located is c.11.5 m/day and this is reflected in the extremely fast response time of the water level during the test indicating a highly transmissive aquifer. This value compares well with listed values in the draft aquifer properties database of Ireland currently being prepared for the GSI and EPA by Tobins Consulting Engineers. It also compares well with values assumed by the GSI when preparing the Toberdaly Group Water Scheme (Hudson, 1996) who assumed a K value of 5 – 10 m/d for the 'calp' limestone (see Chapter 3 for further details).

Table 5.7 Slug test results for BH-O2c at Rhode, Co. Offaly

Slug in		Slug out	
Piezometer diameter – (r)	0.052 m	Piezometer diameter – (r)	0.052 m
Borehole diameter – (R)	0.416 m	Borehole diameter – (R)	0.416 m
Screened length – (L_e)	3 m	Screened length – (L_e)	3 m
Time taken for water to fall 37% of initial change – (T_o)	10	Time taken for water to rise 37% of initial change – (T_o)	5
Hydraulic conductivity – (K)	8.9E-05 m/s	Hydraulic conductivity – (K)	0.000178 m/s
	7.692833 m/day		15.38567 m/day
Average hydraulic conductivity	**11.53925 m/day**		

Results of the slug tests carried out at Carrigeen, Co. Kilkenny are shown in Table 5.8 below. The estimated hydraulic conductivity for the bedrock aquifer in which the piezometer is located is 1.68 m/day. This value is on the lower end when compared to those listed in the draft aquifer properties database of Ireland which gives values of 1 – 80 m/day for the sandstones in the area.

Table 5.8 Slug test results for BH-K2c at Carrigeen, Co. Kilkenny

Slug in		Slug out	
Piezometer diameter – (r)	0.052 m	Piezometer diameter – (r)	0.052 m
Borehole diameter – (R)	0.416 m	Borehole diameter – (R)	0.416 m
Screened length – (L_e)	3 m	Screened length – (L_e)	3 m
Time taken for water to fall 37% of initial change – (T_o)	52	Time taken for water to rise 37% of initial change – (T_o)	41
Hydraulic conductivity – (K)	1.71E-05 m/s	Hydraulic conductivity – (K)	2.17E-05 m/s
	1.479391 m/day		1.876301 m/day
Average hydraulic conductivity		**1.677846 m/day**	

Results of the slug tests carried out at Faha, Co. Limerick are shown in Table 5.9 below. The estimated hydraulic conductivity for the bedrock aquifer in which the piezometer is located is 11.25 m/day and again an extremely fast response time of the water level was observed during this test. Given that the bedrock is similar to the bedrock at Rhode, Co. Offaly it is notable that the two results are similar. Deakin, 1995 carried out a groundwater protection plan for the local Croom water supply and quoted K values of 3 – 6 m/day for the area which is similar to the value obtained here.

Table 5.9 Slug test results for BH-L1c at Faha, Co. Limerick

Slug in		**Slug out**	
Piezometer diameter – (r)	0.052 m	Piezometer diameter – (r)	0.052 m
Borehole diameter – (R)	0.416 m	Borehole diameter – (R)	0.416 m
Screened length – (L_e)	1	Screened length – (Le)	1
Time taken for water to fall 37% of initial change – (T_o)	13	Time taken for water to rise 37% of initial change – (T_o)	7
Hydraulic conductivity – (K)	9.12E-05 m/s	Hydraulic conductivity – (K)	0.000169 m/s
	7.881 m/day		14.63614 m/day
Average hydraulic conductivity	**11.25857**	**m/day**	

5.3.2 Stream Gauging

Discharge in the Toreen East stream in the karst catchment in Co. Clare was calculated upstream and downstream of the Toonagh study cluster on four separate occasions using both salt dilution gauging and a digital flowmeter. Calculated values for discharge in the stream are summarised in Table 5.10 below. Figure 5.7 – Figure 5.8 illustrate the breakthrough curves obtained during the salt dilution gauging. Full details of the salt dilution gauging are given in Appendix E.

It is likely that the flow at the spring is delayed from that observed at the sink due to the travel time and distance involved assuming a plug flow type scenario from high intensity short duration storm events and this is illustrated graphically in Figure 5.9 below. This may limit the ability of making direct comparisons between the two flow rates. For this reason an examination of rainfall quantities in the preceding days was carried out for each of the dates that flow monitoring was carried out – see Figure 5.10 and Figure 5.11 for details. It can be seen that whilst the previous 24 hours on the 18th of January had seen extremely high levels of rainfall, both other days on which flow monitoring was carried out (i.e. 28th May 2012 and 21st January 2013) there had not been significant rainfall in the three previous days and therefore it can be assumed that no lagged flow effect could be interfering with comparisons between the upstream and downstream flow rates. On average the downstream discharge in the Toreen East stream was found to be approximately double that in the upstream measuring point prior to entering the underground karst limestone system as shown in Table 5.10 below.

Table 5.10 Recorded Discharges in the Toreen East Stream

Date	Location	Discharge (L/s) Salt Dilution	Discharge (L/s) Flowmeter
28-05-12	Upstream	9.2	-
28-05-12	Downstream	14.6	-
18-01-13 (AM)	Upstream	438	-
18-01-13 (AM)	Downstream	958	-
18-01-13 (PM)	Upstream	349	384
18-01-13 (PM)	Downstream	873	833
21-01-13	Upstream	-	74
21-01-13	Downstream	-	181

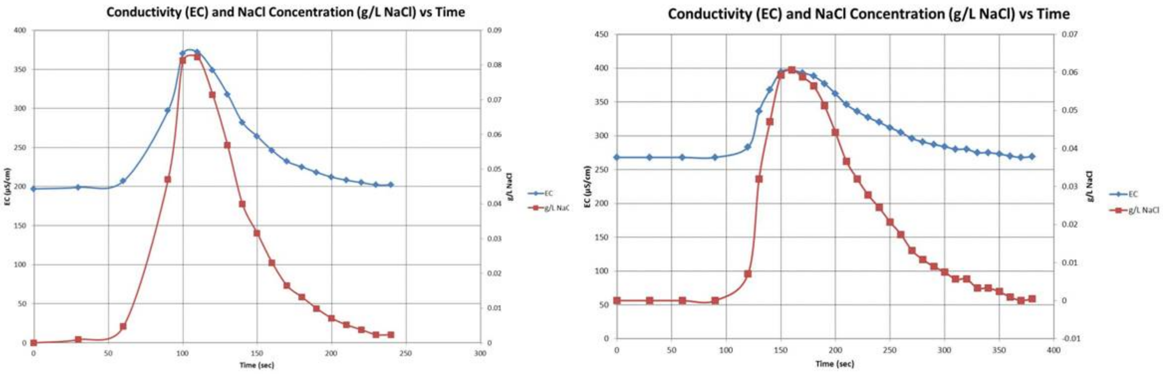


Figure 5.7 Breakthrough curves obtained from upstream dilution gauging on 18-01-13 at (a) morning and (b) evening

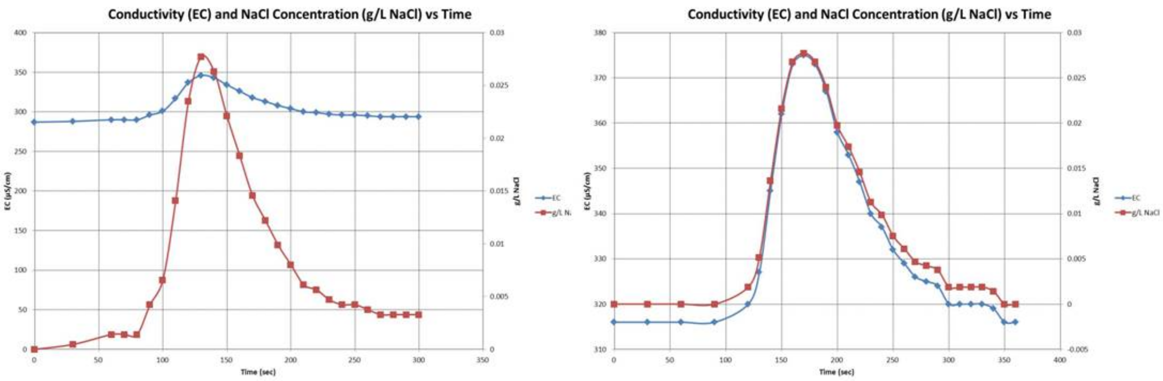


Figure 5.8 Breakthrough curves obtained from downstream dilution gauging on the 18-01-13 at (a) morning and (b) evening

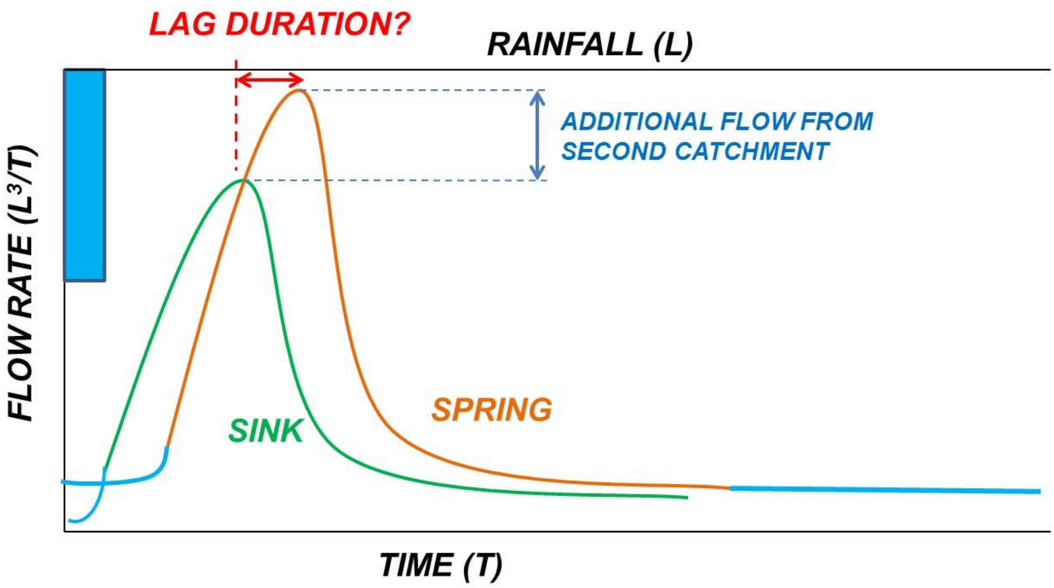


Figure 5.9 Illustration of possible lag between sink and spring for flow comparisons

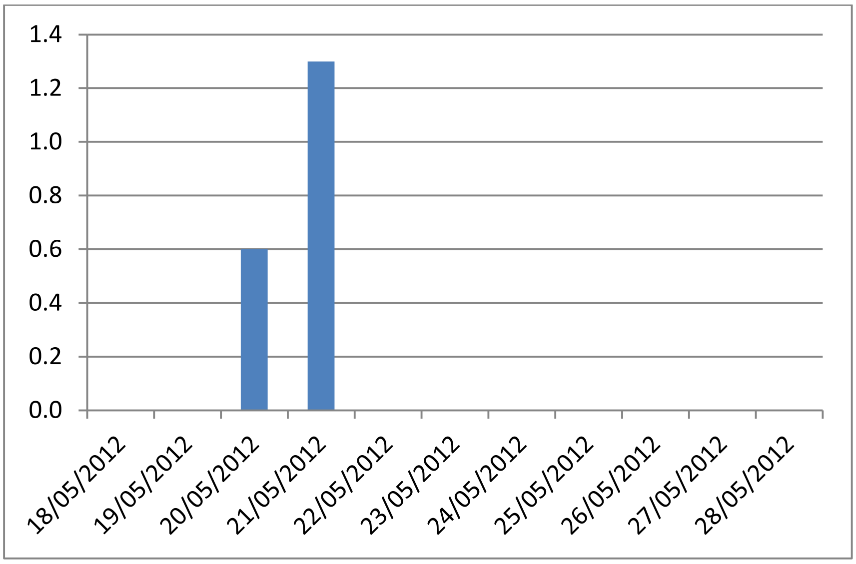


Figure 5.10 Rainfall (mm) at Toonagh 10 days prior to flow monitoring on the 28th of May 2012

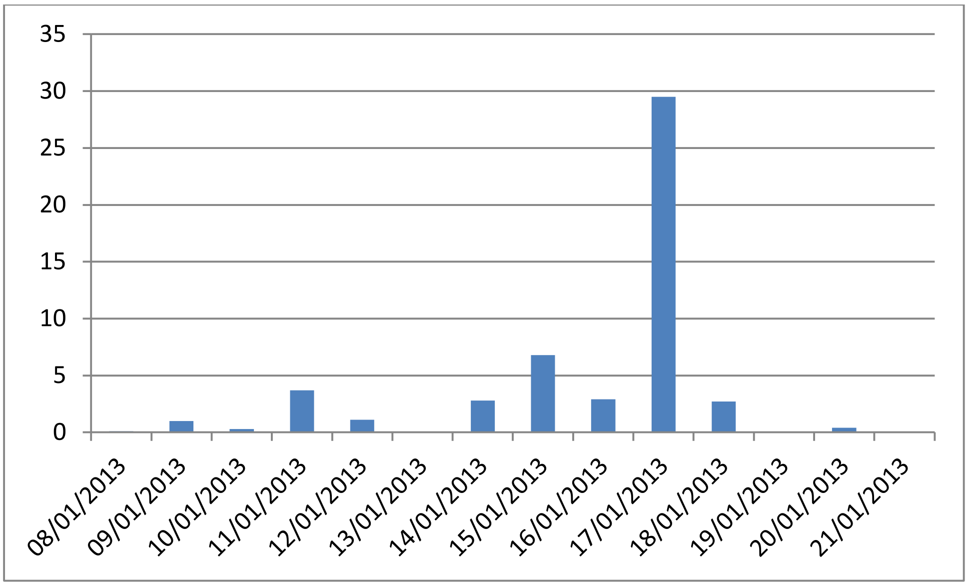


Figure 5.11 Rainfall (mm) at Toonagh 10 days prior to flow monitoring on the 18th and 21st of January 2013

5.3.3 Water Level Data Analysis

Water levels were recorded in each of the piezometers on a monthly basis from November 2010 until October 2012. Levels were recorded in meters below ground levels on site (mBGL) and the corresponding water levels in meters to Ordinance Datum were then calculated using the values surveyed during the topographical survey using the GPS surveying equipment. Daily rainfall data was obtained from Met Eireann for the closest rainfall recording station (see Chapter 4 for details) and summed to give monthly totals. Water level data were then plotted against total monthly rainfall for all study locations with the exception of Toonagh, Co. Clare. The resulting plots are shown in Figure 5.13 – Figure 5.19 below. Schematic cross-sections of each of the sites are also given.

Water levels at Naul, Co. Dublin are not directly affected by rainfall quantities (as might be expected for such a low permeability area) with levels quite static during the course of the study period. It must be noted that BH-N1a was installed at a very shallow depth (2 mBGL) to monitor for a high seasonal water table but this piezometer remained dry for the entire duration of the monitoring period. Another of the piezometers, BH-N3b, located in the second downstream borehole was installed in the subsoil just above the bedrock interface. Water levels were intermittent at this horizon with water levels recorded on some of the visits and no water present on other occasions. This indicates that during high rainfall events a perched water table occurs which recedes slowly after the rainfall events have passed. This is common in low permeability clay soils.

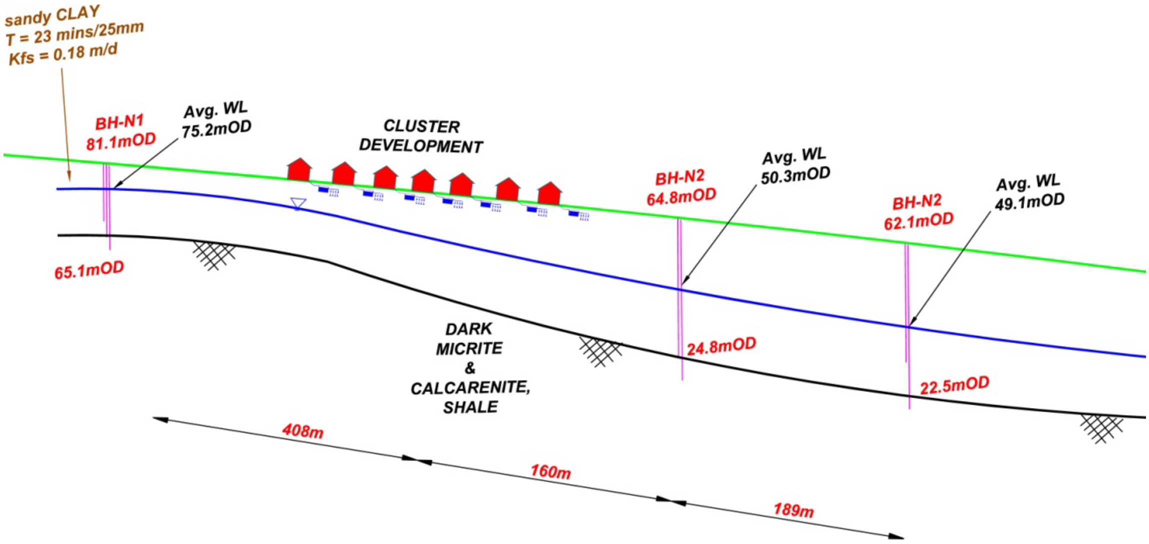


Figure 5.12 Schematic cross-section of study area at Naul, Co. Dublin

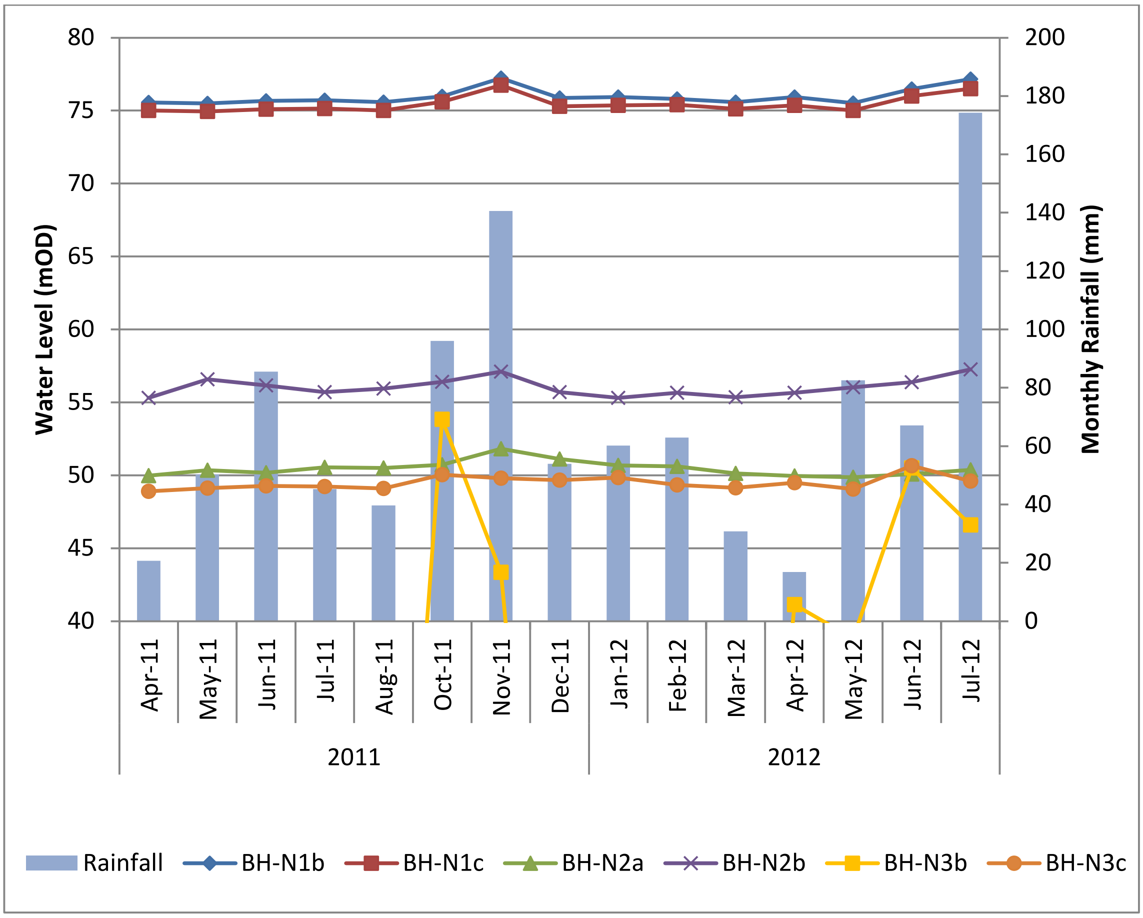


Figure 5.13 Water level response to rainfall at Naul, Co. Dublin

Water levels at Rhode, Co. Offaly do appear to be related to rainfall with perhaps a lag involved. No significant seasonal trend can be seen in water level variations but muted responses in water elevations during or following months with high rainfall can be observed when the data is shown in a time series. Given that there is between 5 – 15 m of clays present across the area, this is consistent with the time required for rainfall to infiltrate based on an assumed K_s of 0.2 – 0.5 m/d which is consistent with values for this type of subsoil (Misstear and Brown, 2008) and with the T-value obtained during this study. It is possible that the areas that are at a higher elevation to the north where subsoil cover is shallower allow recharge to the aquifer more quickly than the flatter plains to the south where the subsoil is more extensive. This will be discussed further in Chapter 6.

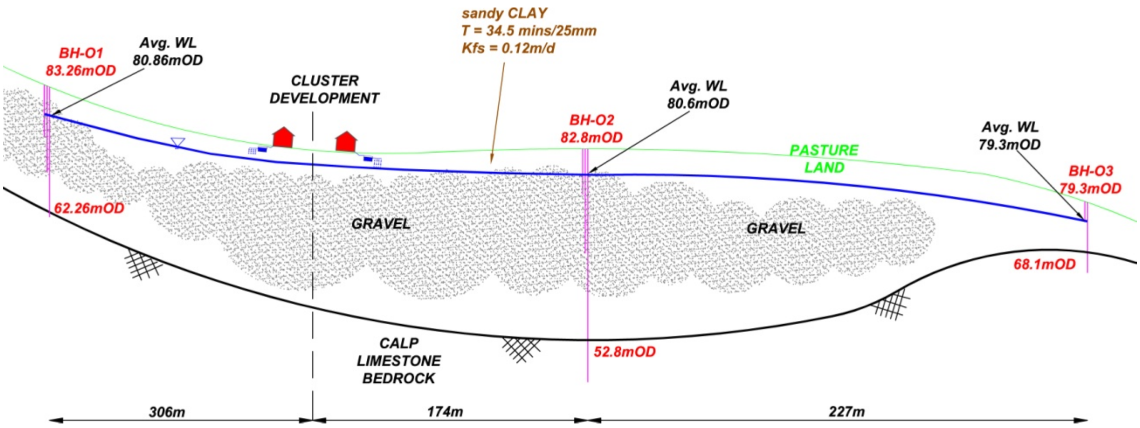


Figure 5.14 Schematic cross-section of study area at Rhode, Co. Offaly

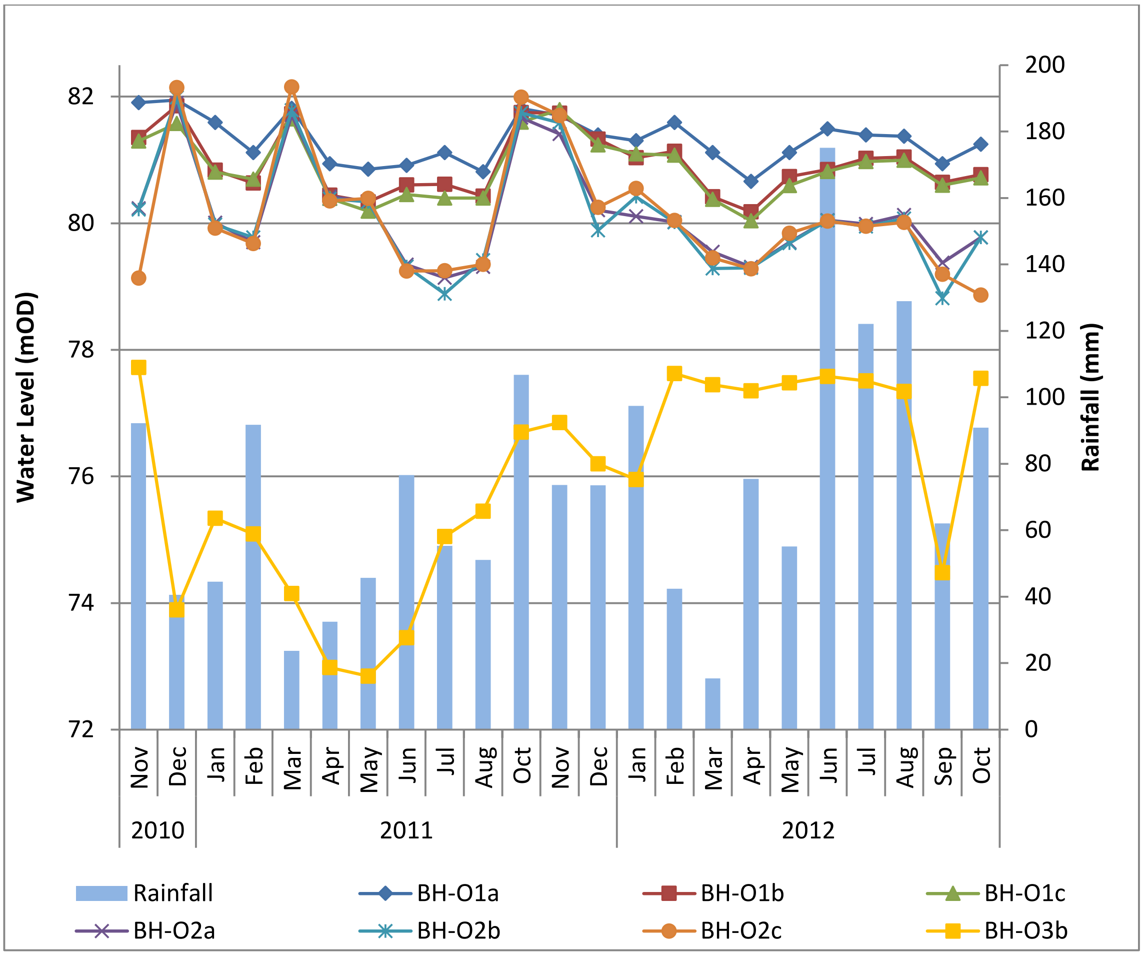


Figure 5.15 Water level response to rainfall at Rhode, Co. Offaly

Water levels at Carrigeen Co. Kilkenny appear to be highly responsive to rainfall. Figure 5.17 shows the variance of water elevations as recorded at the monitoring boreholes with total monthly rainfall. This indicates that recharge to the underlying aquifer occurs relatively quickly and this is due to the shallow free draining sandy subsoil in the area and the fractured nature of the aquifer which allows the bedrock to accept the infiltrating water the nature of subsoils and bedrock geology in the area will be discussed in more detail in Chapter 6. Direct access was not available to the upstream monitoring wells as they were in use as potable water supplies and it was therefore not possible to monitor water elevations at these locations.

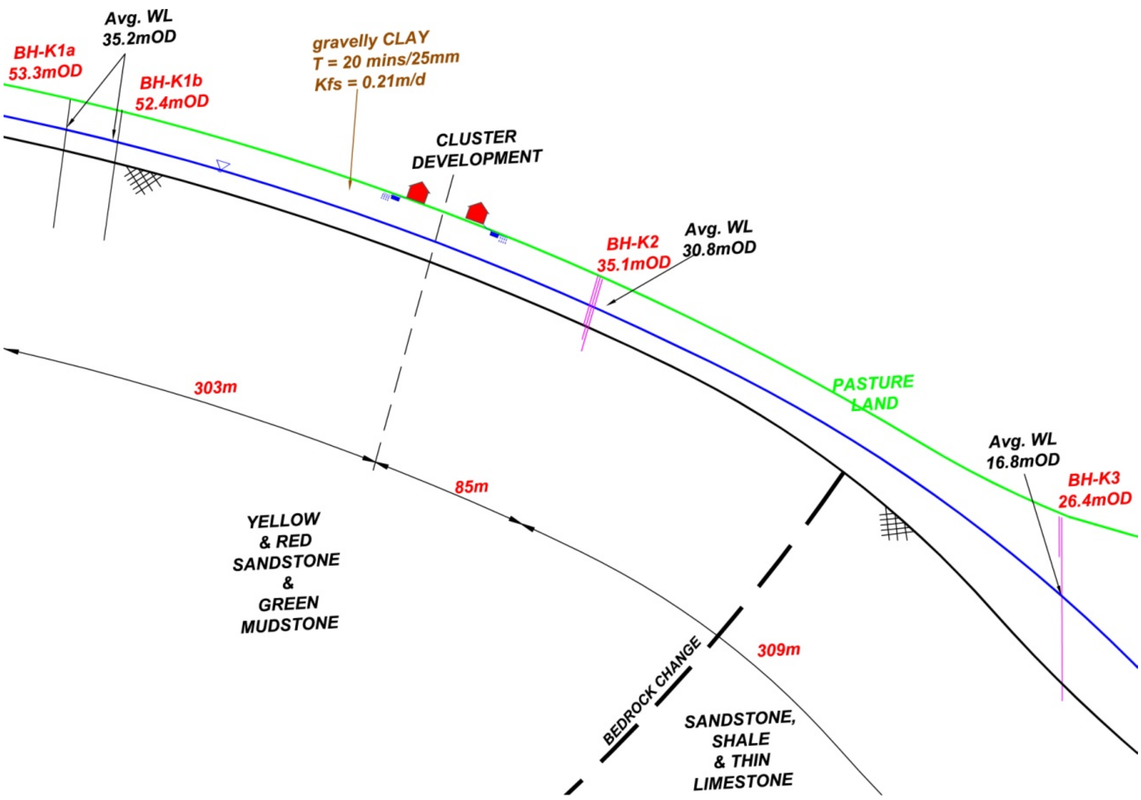


Figure 5.16 Schematic cross-section of study area at Carrigeen, Co. Kilkenny

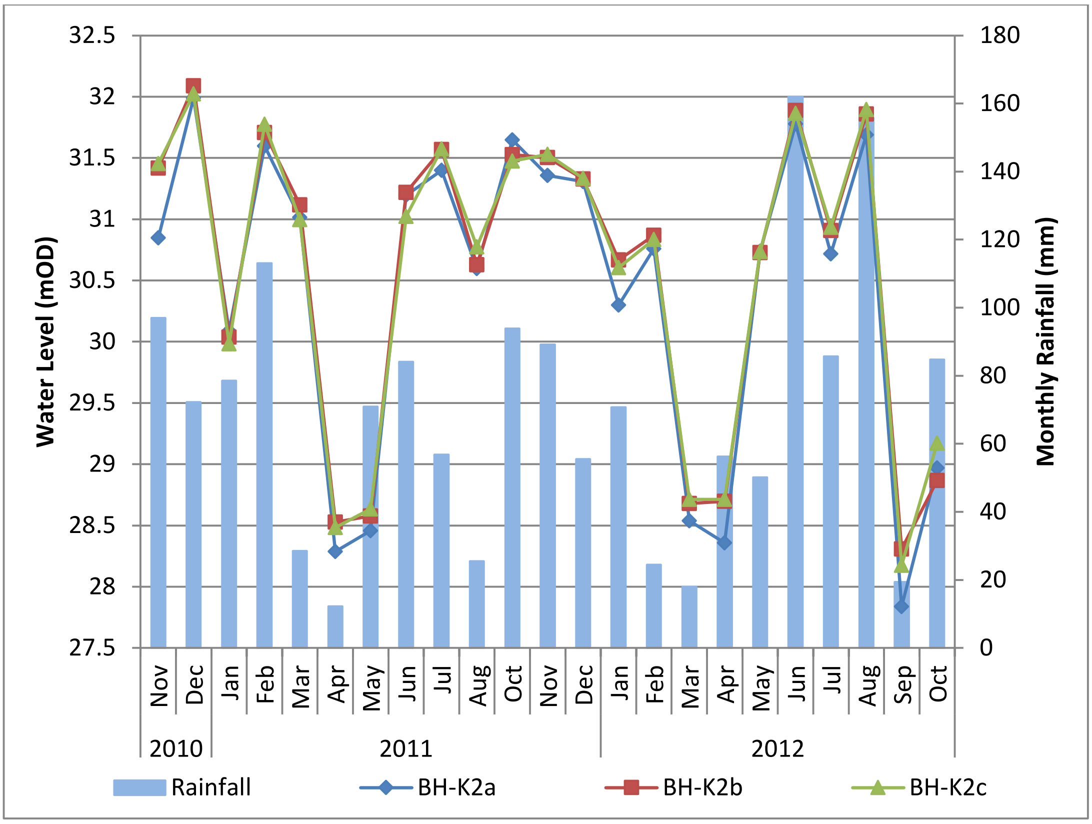


Figure 5.17 Water level response to rainfall at Carrigeen, Co. Kilkenny

Water levels at Faha, Co. Limerick appear to show two distinct opposing trends with time and rainfall quantities. The time series of water elevations with total monthly rainfall indicates that on some occasions water levels at some or all boreholes increase during periods of low rainfall and decrease during periods of high rainfall. Conversely water elevations follow an opposite trend at other times, which vary in a more conventional manner with levels increasing during periods of high rainfall and decreasing during periods of lower rainfall. Both show a close response to rainfall quantities however the two different trends indicate that groundwater movement in the area may not be straightforward.

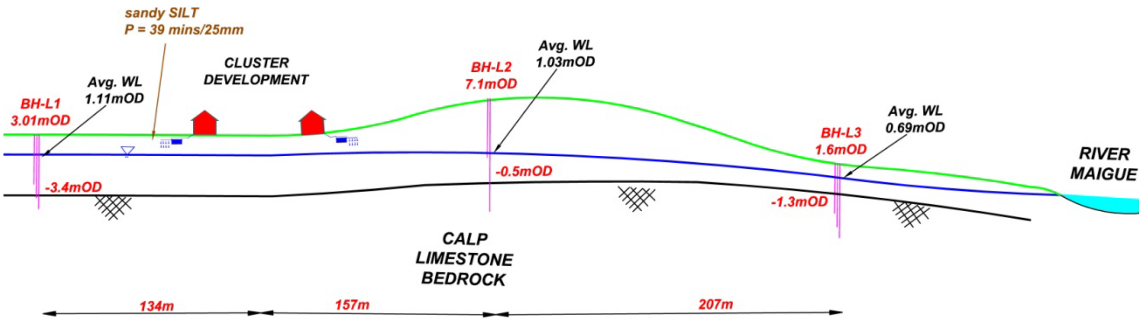


Figure 5.18 Schematic cross-section of study area at Faha, Co. Limerick

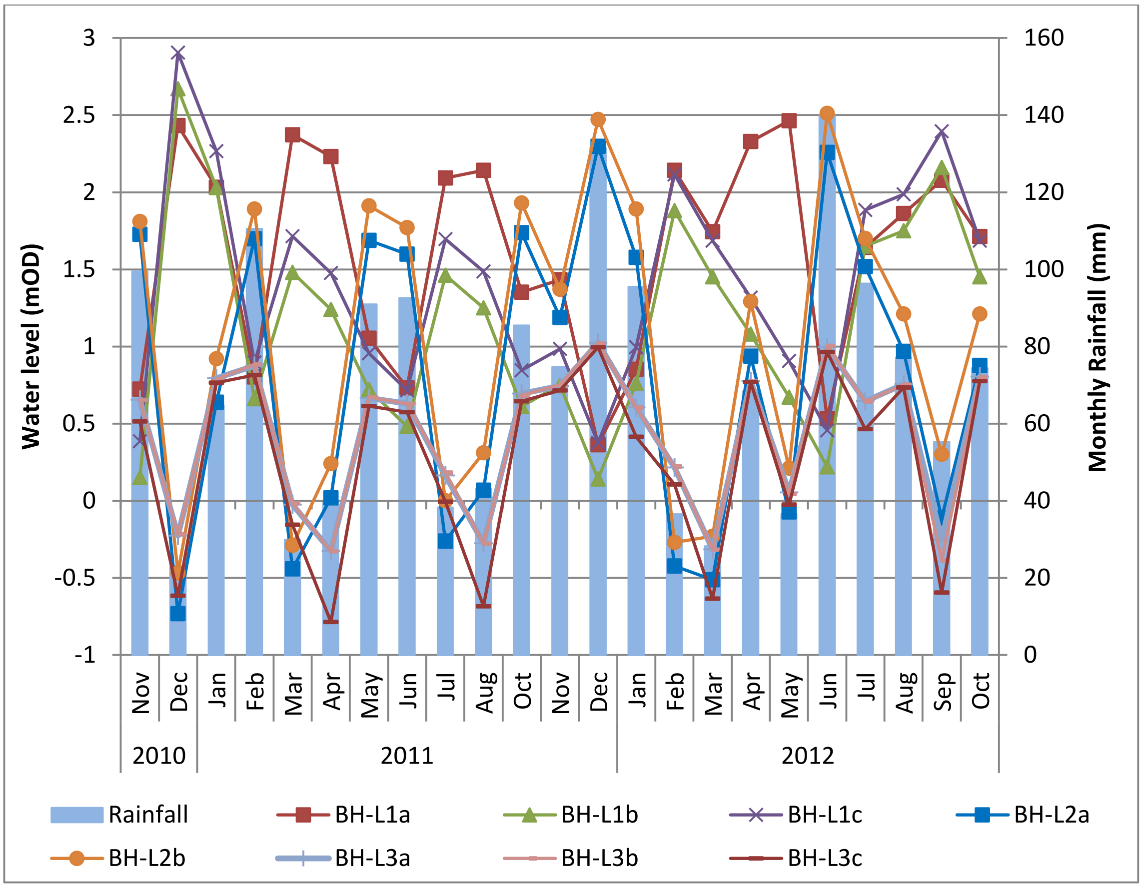


Figure 5.19 Water level response to rainfall at Faha, Co. Limerick

Given the close proximity of the River Maigue (which is tidal in this area) to the study area, it is likely that the tidal fluctuations of the river have an impact on groundwater levels in the area. Data were available for water levels in the River Maigue at Ferrybridge from the Office of Public Works (OPW). Data were also acquired from the Irish Marine Institute of modelled tidal levels in the Shannon estuary downstream of the mouth of the River Maigue. Pearson correlation analysis was carried out on levels in the boreholes and the Maigue and tidal levels for the times at which the borehole water levels were recorded each month. The correlation coefficients (r^2) showed very little evidence of any relationship between the levels observed in the boreholes and with those in the River Maigue. A very weak correlation was observed with the observed groundwater levels and the tidal data (r^2 = 0.1 – 0.2). It must be noted that the tidal data obtained from the Marine Institute only takes into account tidal levels in the Shannon estuary and does not take account of levels in the River Shannon or inflowing rivers such as the Maigue. It is possible that this is why a weak correlation was observed between the tidal data and none at all with the Maigue data. Hence, the time lag between the tidal levels and the Maigue levels was first investigated against which any expected lag behind in the levels observed at the boreholes could then be compared. A series of time shifted (15 minute intervals) data sets were produced for the OPW River Maigue data and again Pearson's correlation analysis was carried out between these shifted values and the Tidal data. The results of this analysis are shown in Table 5.11 below. It can be seen that without any time lag a moderate correlation between the data exists of 0.55. However, as the data is 'shifted' this increases to a very strong correlation of 0.96 for a time lag of 120 minutes.

Table 5.11 Correlations (R) between OPW River Maigue data and Marine Institute modelled tidal data for the Shannon estuary

	Maigue OPW Data
Marine Institute Tidal Data	1
Maigue OPW Data	0.554464065
Maigue OPW Data shifted 15min	0.649628315
Maigue OPW Data shifted 30min	0.734460332
Maigue OPW Data shifted 45min	0.807586293
Maigue OPW Data shifted 60min	0.867821519
Maigue OPW Data shifted 75min	0.914190555
Maigue OPW Data shifted 90min	0.945944286
Maigue OPW Data shifted 105min	0.962575359
Maigue OPW Data shifted 120min	0.963822092
Maigue OPW Data shifted 135min	0.94967449
Maigue OPW Data shifted 150min	0.920367203
Maigue OPW Data shifted 165min	0.876381722
Maigue OPW Data shifted 180min	0.818430153

Further correlation analysis was carried out between the 120 minute shifted Maigue OPW data with the levels observed at each of the boreholes based on the times that each was dipped. This was to take account for the fact that any change in levels in the river Maigue would again be ahead of the associated change in borehole levels due to the time taken for it to propagate inland. The analysis matched the time that each of the boreholes was dipped against the corresponding shifted Maigue OPW data and a correlation between the corresponding elevations (mOD) was obtained with associated optimal time shift required to yield the highest correlation factor. Summaries of the results are shown in Table 5.12 below.

Table 5.12 Optimal correlations between 120 min shifted Mague data and borehole water levels. All elevations were matched for the corresponding times that a borehole water elevation was recorded

	Time shift	*Correlation (R)*
BH-L1a	150 mins	0.673564064
BH-L1b	150 mins	0.589550126
BH-L1c	150 mins	0.589550126
BH-L2a	100 mins	-0.837486506
BH-L2b	100 mins	-0.858798712
BH-L3a	70 mins	-0.891317448
BH-L3b	70 mins	-0.868610547
BH-L3c	70 mins	-0.846121019

It can be seen that the 120 minute shifted River Maigue data gives very strong negative correlation values for BH-L2 and BH-L3 and strong/moderate correlation values for BH-L1. The weaker correlation at BH-L1 is to be expected as it is further inland and the effects of the river and the tidal influence dissipate the further inland. The opposing magnitude of the correlations is explained by the time series plot of water levels in the River Maigue (see Figure 5.20) over a 24 hour period for one of the sampling visits with the times that each of the boreholes was dipped shown. It can be seen from Figure 5.20 that when BH-L1 was dipped, the tidal influence was positive (downward crest of tidal level) but for the other two boreholes it was negative corresponding to a trough in tidal levels. These tidal influences explain the unusual trends in the time series plot of rainfall and water levels shown Figure 5.19 and discussed earlier.

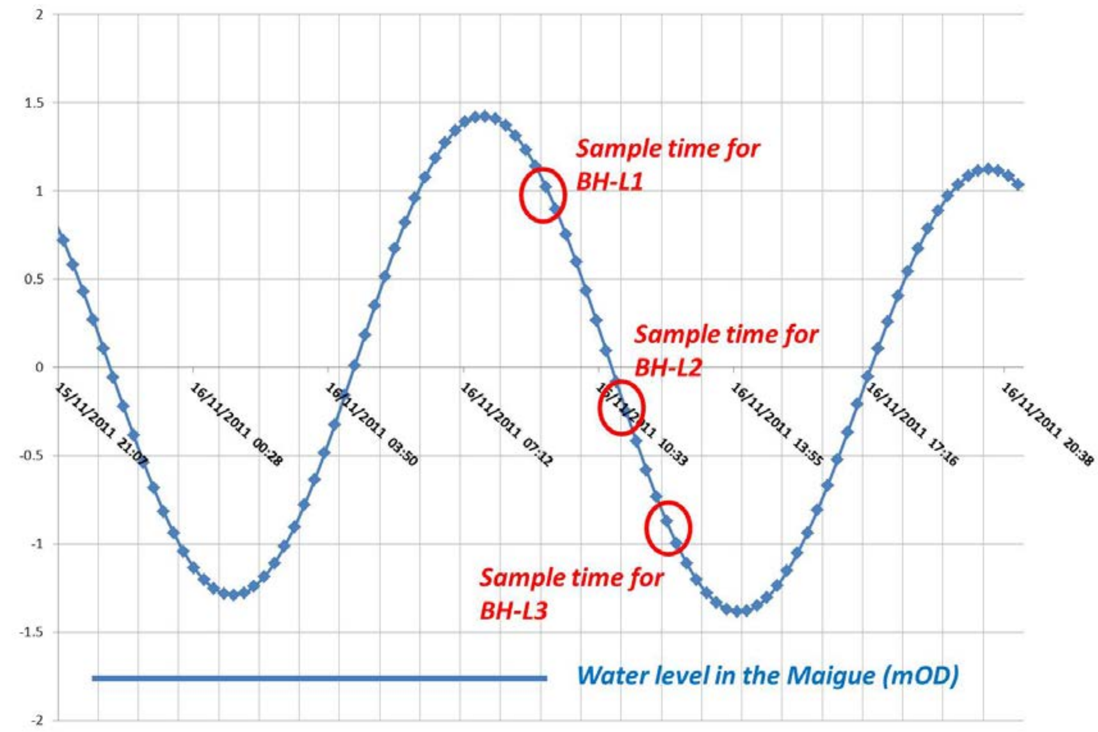


Figure 5.20 Plot of water levels in the River Maigue over a 24 hour period on the date of a sampling visit with the times each of the borehles were dipped shown

The area is underlain by limestone bedrock and the highly heterogeneous nature of limestone as a bedrock aquifer may explain the opposing correlations. Given that the aquifer classification for the area is Rkd – Regionally Important Aquifer – Karstified (Diffuse), the karstified nature of the bedrock may be allowing water flow very quickly in some areas through large conduits in the epikarst. Given that groundwater flow in the area is not straightforward (particularly given the tidal influences shown above); it may be difficult to draw conclusions from the chemical and bacterial analysis.

5.3.4 Temperature, Electrical Conductivity and pH

Samples were analysed for temperature, electrical conductivity (EC) and pH in the field using portable probes throughout the duration of the monitoring period. The recorded values are shown in Appendix C. The EPA has recommended guideline or 'trigger' values for both pH and electrical conductivity (EPA, 2003) outside of which should trigger action to mitigate potential contamination of the groundwater body. For pH the acceptable range is 6.5 – 9.5 and for electrical conductivity the trigger is values observed above 1000 $\mu S cm^{-1}$.

At the Naul, Co. Dublin samples had an average temperature of 10.7°C, average EC values ranging from 360 – 500 µScm^{-1}and pH ranging from 7.02 – 7.17. Individual monthly values varied from these averages but no values that were indicative of groundwater pollution were observed during the course of the study. A plot of EC values with rainfall is shown in Figure 5.21 below. No clear pattern with rainfall trends can be seen with observed EC values. There is a weak indication that EC values are lower during the winter months but monitoring over a more extended period would be required to validate this.

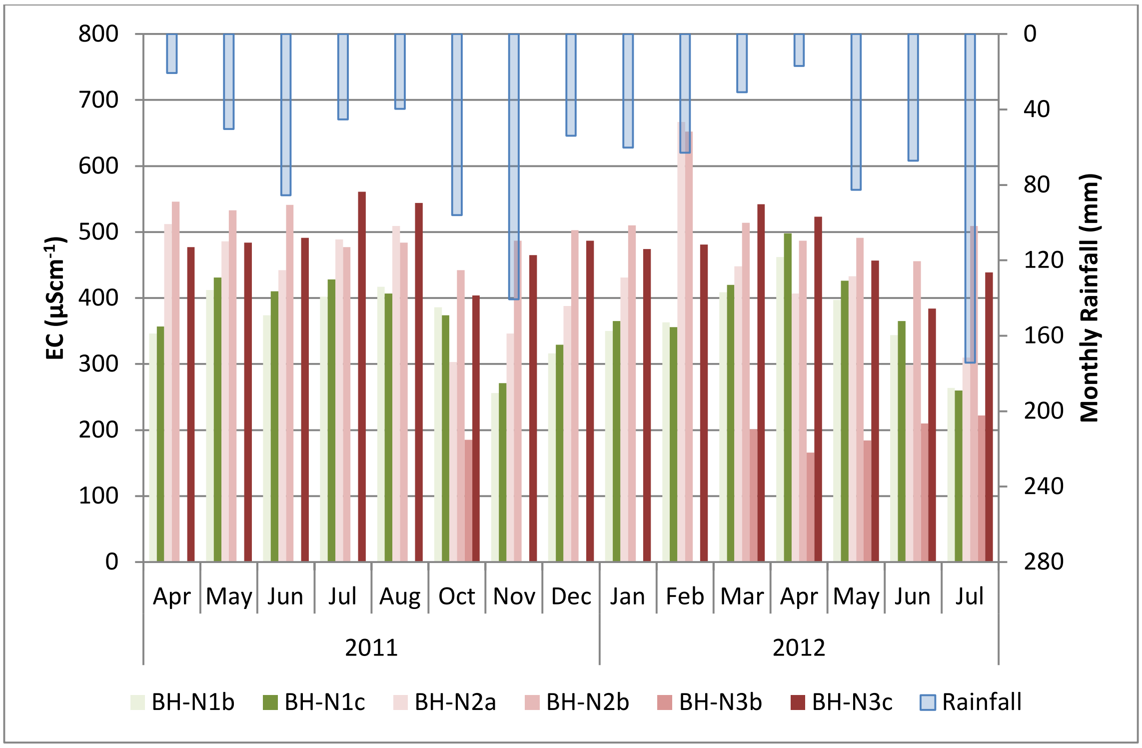


Figure 5.21 Electrical conductivity values with rainfall at Naul, Co. Dublin

Average groundwater temperature at Rhode Co. Offaly was 10.4°C with average electrical conductivity values ranging from 600 – 800 µScm^{-1} and the average pH was 7.15. Whilst the pH and temperature values are in the range that is typical for groundwater in Ireland, the average electrical conductivity range is slightly higher than would be expected. Examining the individual monthly values highlights individual monthly spikes that are far higher than background values with individual piezometers recording values of up to 1551 µScm^{-1} (BH-O2a – Dec 2010). The majority of the very high values (>1000 µScm^{-1}) were observed in the shallower piezometers and these values are highlighted in Figure 5.22 below; however values of 800 – 1000 µScm^{-1} were recorded in the deeper piezometers on some months. None of the individual monthly pH values for each of the piezometers were outside the range of 6 – 8 which is considered normal for groundwater. Given the fact that the elevated values of electrical conductivity in the shallower piezometers were not constant throughout the year it is more likely that there were caused by agricultural practices than as a direct results of OSWTS's which would produce a more constant loading throughout the year. No clear trend linking rainfall and EC can be observed; however there does appear to be a weak seasonal trend in EC values. From observations on-site and discussions with farmers this seasonal trend does generally match cattle grazing seasons

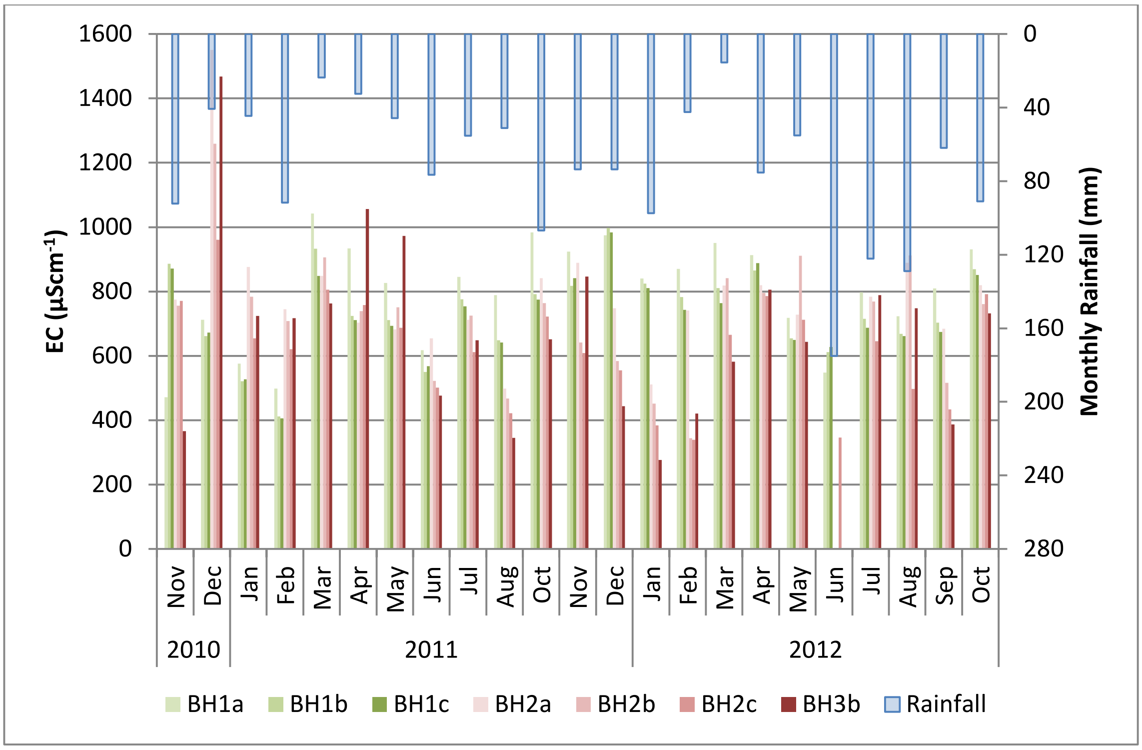


Figure 5.22 Electrical conductivity values with rainfall at Rhode, Co. Offaly

Average groundwater temperature at Carrigeen Co. Kilkenny was 11.5°C with average electrical conductivity in the range of 300 – 400 µScm^{-1} and an average pH of 7.05. These average values are typical for groundwater and do not indicate anthropogenic pollution in the area. A plot of EC values with rainfall is given in Figure 5.23 below. No clear seasonal trends can be seen.

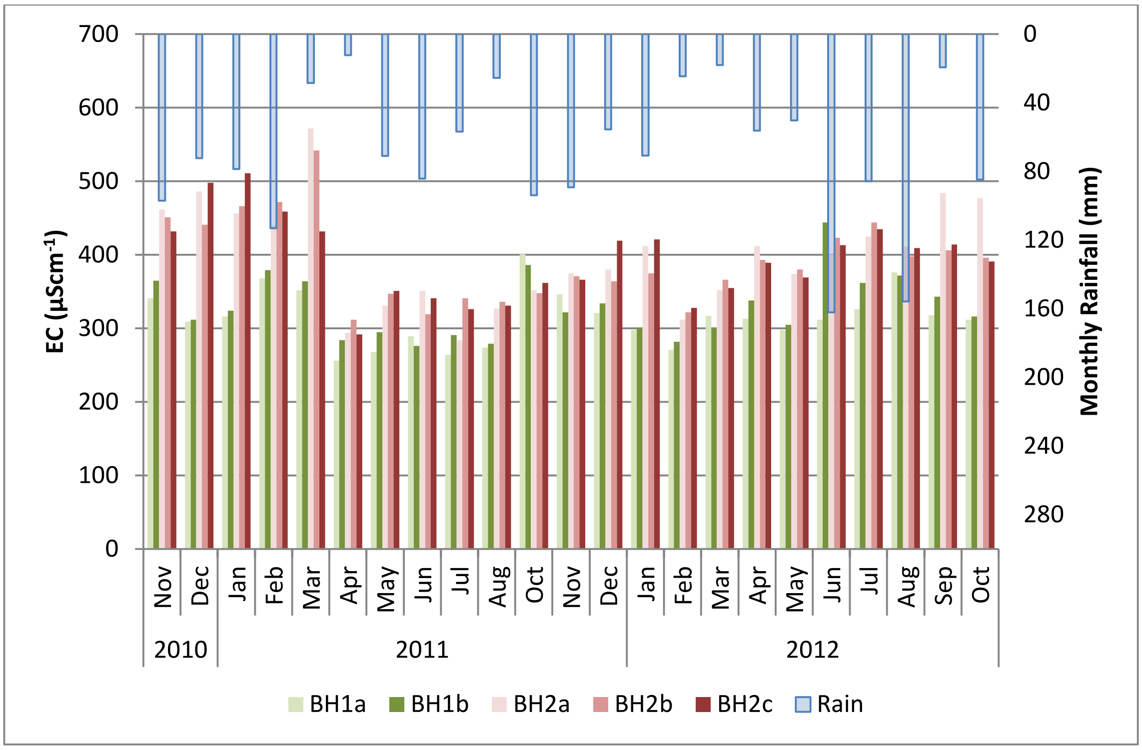


Figure 5.23 Electrical conductivity values with rainfall at Carrigeen, Co. Kilkenny

Average groundwater temperature at Faha, Co. Limerick ranged from 10 – 11°C with average electrical conductivity values ranging from 700 – 900 µScm^{-1} and pH values ranged from 7 – 7.5. Individual monthly values of electrical conductivity exceeded 1000 µScm^{-1} on a number of occasions which would indicate anthropogenic pollution of the aquifer. Examination of the monthly values for electrical conductivity shown in Figure 5.24 appears to indicate that the higher values (>900 µScm^{-1}) occur from March to July corresponding to the grazing period for cattle in the area. It is noted however, that values are high throughout the year indicating that background values in the entire area may be elevated suggesting that there may also be a more constant contaminant loading source in the area.

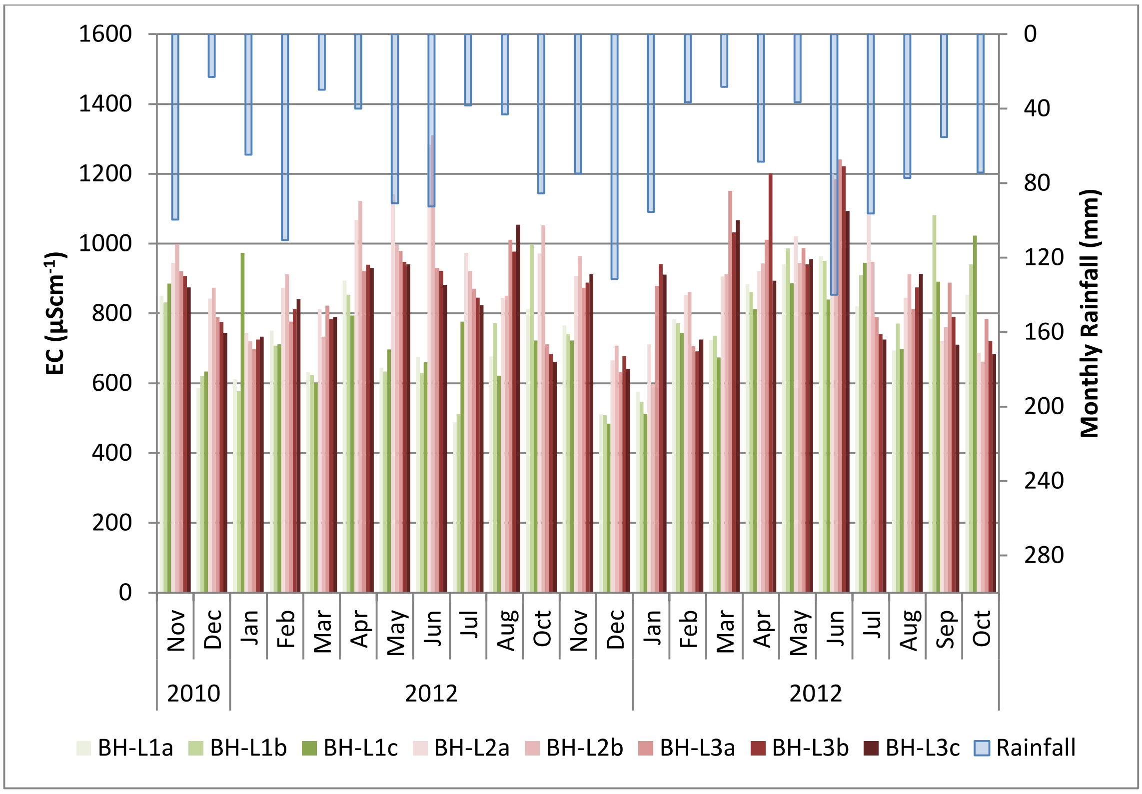


Figure 5.24 Electrical conductivity values with rainfall at Faha, Co. Limerick

Average water temperature in the Toreen east stream at Toonagh Co. Clare reflected ambient outdoor temperatures and ranged from 7 – 15°C. Downstream water temperature was more typical of groundwater with much less fluctuation during the monitoring period. Higher and lower upstream water temperature changes accompanied much more muted changes in downstream water temperatures which could have been underground for hours or days depending on its origin. Similarly electrical conductivity values tended to be higher upstream than at the Kilcurrish spring with upstream values in the range of 240 – 810 µScm^{-1} and downstream values ranging from 220 to 730 µScm^{-1}. Measured pH values were slightly basic ranging between 7.3 and 8.7. Bedrock in the area is limestone which is calcareous and water in similar limestone areas tends to therefore have higher pH values – however there does seem to be some indication of raised pH levels in the area. A summary of the recorded values at Toonagh, Co. Clare is given in Table 5.13 below

Table 5.13 Temperature, Electrical Conductivity and pH values observed at Toonagh, Co. Clare during the monitoring period

	Upstream	Downstream

Year	Month	EC (μScm^{-1})	pH	T (°C)	EC (μScm^{-1})	pH	T (°C)
2012	Jun	248	7.66	15.2	222	7.48	13.6
	Jul	361	7.82	14.9	509	7.26	13.2
	Aug	624	8.08	13.4	587	7.97	11.4
	Sep	442	8.34	11.6	484	8.13	10.6
	Oct	335	7.96	12.2	478	7.82	10.9
	Nov	808	8.69	7.5	729	8.19	10.1
	Dec	713	8.51	9.3	581	8.21	10
2013	Jan	239	8.49	7.8	219	8.19	9.4
	Feb	745	8.22	8.9	586	7.94	10.2

5.4 Chemical and Bacterial Results and Analysis

A number of different statistical tests were used in order to access the data collected as part of this study. Determining whether sample means are in the long run the same or statistically different usually involves first establishing a null and alternative hypothesis and then calculating a test statistic which is compared to a critical value. If the test statistic exceeds the critical value the null hypothesis is rejected and the means are said to be statistically significantly different from each other i.e. in the long run the means are not the same and come from sets of data that are statistically significantly different.

Where two sample means and the associated variance were being compared, the two-sample t-test was used. The two sample t-test was accessed assuming unequal sample variances and was two-sided with this most likely reflecting the situation whereby two boreholes or piezometers may have sample means that are not statistically different but still may have different variances. For these statistical tests the null hypothesis (H_o) stated that the difference between sample means was zero in the long run. The alternative hypothesis (H_1) stated that the difference between the means is not zero in the long run.

$$H_o: \mu_1 - \mu_2 = 0$$

$$H_1: \mu_1 - \mu_2 \neq 0$$

In order to access whether a numbers of samples means (>2) are statistically different from each other, a different statistical test is required which was Analysis of Variance 2(ANOVA), whereby the data are assessed using sums of squares and the associated F-test. For these statistical tests the null hypothesis (H_o) stated that the sample means were all the same. The alternative hypothesis (H_1) stated that the means are not all the same in the long run.

$$H_o: \mu_1 = \mu_2 = \mu_3 .... = \mu_N$$
$$H_1: \mu_1 \neq \mu_2 \neq \mu_3 ....\neq \mu_N$$

ANOVA was used in two differing methods and applications during this analysis. Single factor ANOVA was used to access statistical significance between more than two sample means for a single parameter of interest i.e. nitrate between piezometers in a single borehole or between piezometers at the same horizon both upstream and downstream of a study site. In order to access the statistical significance between both sample means and two parameters and more importantly their interaction with each other either upstream or downstream two factor ANOVA with replication was used. Both Microsoft Excel and the Minitab statistical software packages were used to carry out the statistical analysis of the data. For all tests the significance level (α) was chosen to be 0.05 giving 95% confidence in the outcome of the test. For the ease of illustrating the results of tests that will be contained below t-values and p-values that are deemed to be statistically significant will be highlighted as shown below. The usual underlying assumptions of independence, data normality and homoscedasticity were assumed to hold for all datasets.

Statistically insignificant outcome	(blue)
Statistically significant outcome	(orange)
Not statistically significant at 0.05 significance level; but likely to be so at a 0.1 significance level	(green)

Correlation coefficients will also be referred to in this chapter. These coefficients are interpreted using the standard Pearson 'rule of thumb' which groups ranges of correlation coefficients (r) into varying strengths of relationship as shown below:

Pearson's rule of thumb

r =	+0.70 or higher Very strong positive relationship	(orange)
	+0.40 to + 0.69 Strong positive relationship	(yellow)
	+0.30 to + 0.39 Moderate positive relationship	(green)

+0.20 to + 0.29 weak positive relationship	
+0.01 to + 0.19 No or negligible relationship	
-0.01 to - 0.19 No or negligible relationship	
-0.20 to - 0.29 weak negative relationship	
-0.30 to - 0.39 Moderate negative relationship	
-0.40 to - 0.69 Strong negative relationship	
-0.70 or higher Very strong negative relationship	

5.4.1 Low Vulnerability Site – Naul, Co. Dublin

<u>Nitrogen</u>

Nitrate is often a good indicator of pollution from on-site effluent or agriculture as organic nitrogen (org-N) and ammonia (NH_3) in effluent as nitrification will normally occur in the unsaturated subsoil as the pollutants pass through (as detailed Chapter 2). Nitrate tends to be much more mobile in groundwater (EPA, 2000) leading to its use as an indicator of anthropogenic activities. Correspondingly, significant ammonia concentrations in groundwater are evidence of very close pollution sources.

Nitrogen in all forms was either not present or only present in very low concentrations often below the limit of detection at Naul, Co. Dublin. Nitrite and ammonium were below the limits of detection of the laboratory methods (<0.02 mg/l NO_2; <0.05 mg/l NH_4) for the initial six months due to time and budget constraints it was decided not to sample for these parameters beyond this period. Nitrates were detected above the lower detection limit in a sporadic manner during the sampling period as shown in Figure 5.25 below. Averaged nitrate concentrations over the study period are given in Table 5.14. The highest recorded value of nitrate at the Naul site was 2.3 mg-N/l which is considerably below the EPA threshold limit of 10 mg-N/l. No significant difference (statistically) was found between sample means at different sampling horizons at any of the three boreholes (see Table 5.15 below) indicating no difference in long run mean nitrate concentrations with depth into the aquifer. More interestingly from the perspective of this study, single factor ANOVA tests between the two sampling horizons both upstream and downstream also did not show a significant difference between mean nitrate concentrations with p-values of 0.566 for horizon 2 and 0.878 for horizon 3 as shown in Table 5.16 and Table 5.17 below (using a p-value of 0.05 or below to indicate a statistically significant difference between sample means as per convention).

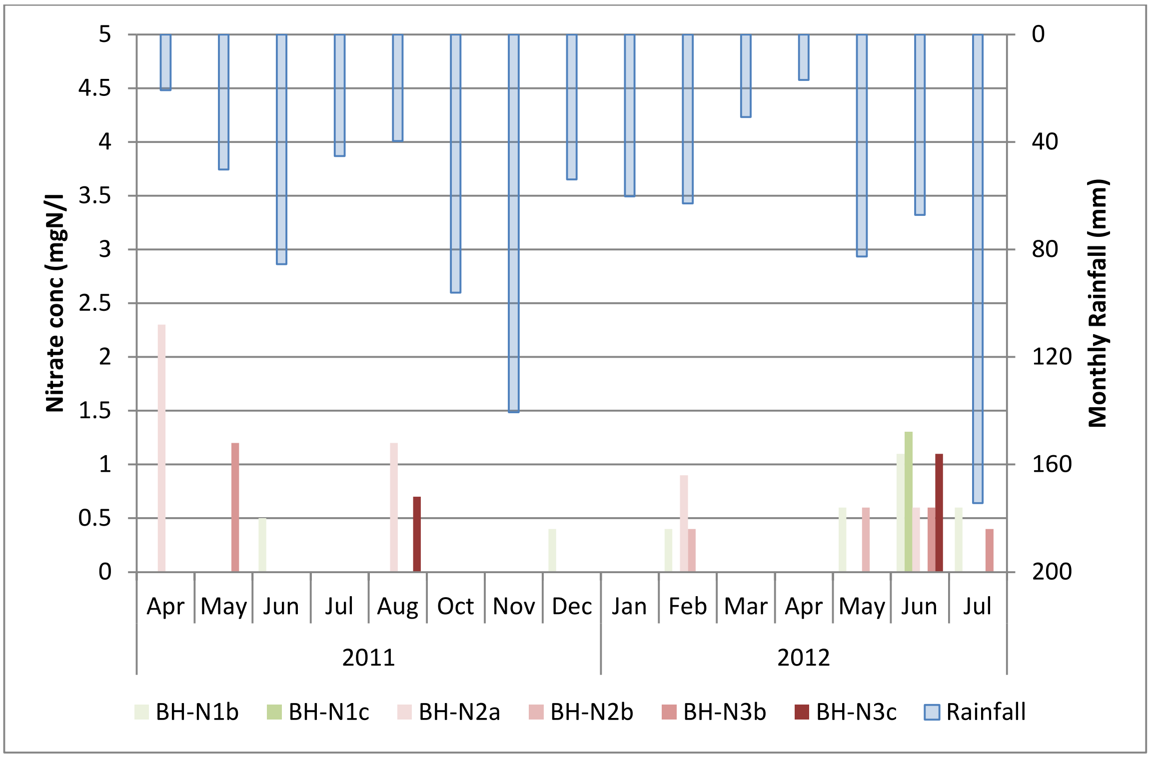


Figure 5.25 Nitrate concentrations with rainfall at Naul, Co. Dublin

Table 5.14 Average Nitrate concentrations at Naul, Co. Dublin (all values in mg/l)

Horizon	BH-N1	BH-N2	BH-N3
Horizon b	024	0.33	0.147
Horizon c	0.087	0.067	0.12

Table 5.15 T-test output for BH-N1 for the parameter Nitrate (t-Test: Two-Sample Assuming Unequal Variances)

Nitrate	*BH-N1b*	*BH-N1c*
Mean	0.24	0.0867
Variance	0.116857	0.1127

Observations	15	15
Hypothesized Mean Difference	0	
df	28	
t Stat	1.239562	
P(T<=t) one-tail	0.112713	
t Critical one-tail	1.701131	
P(T<=t) two-tail	0.225427	
t Critical two-tail	2.048407	

Table 5.16 ANOVA output for Horizon 2 and Nitrate at Naul, Co. Dublin

Anova: Single Factor Nitrate Low Vulnerability - Horizon 2

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-N1b	15	3.6	0.24	0.116857
BH-N2a	15	5	0.33333333	0.445238
BH-N3b	15	2.2	0.14666667	0.116952

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.26133	2	0.13066667	0.577279	0.565816	3.2199423
Within Groups	9.50667	42	0.22634921			
Total	9.768	44				

Table 5.17 ANOVA output for Horizon 3 and Nitrate at Naul, Co. Dublin

Anova: Single Factor Nitrate Low Vulnerability - Horizon 3

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-N1c	15	1.3	0.08666667	0.112667
BH-N2b	15	1	0.06666667	0.032381
BH-N3c	15	1.8	0.12	0.106

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.02178	2	0.01088889	0.130121	0.878341	3.2199423
Within Groups	3.51467	42	0.08368254			
Total	3.53644	44				

Phosphorus

As outlined in Chapter 4, the samples were analysed for total dissolved phosphorus. In terms of water quality, the parameter of main interest is reactive phosphorus (or orthophosphate) as this will be the most available as a nutrient for uptake in surface water bodies. Any particles that were not dissolved in the original water sample but that were fine enough to pass through the filtration process (<0.45 µm) would therefore be incorporated in the results for total dissolved phosphorus as they would be fully dissolved during the chemical/thermal digestion. However, such fine particulate phosphorus may not be mobile through the aquifer and even if it was mobile and could enter surface water bodies it may not be available for uptake as a nutrient and would ultimately settle out. In this context the results present here are indicative of orthophosphate however the results may "overestimate" the actual reactive form of phosphorus available. Phosphorus concentrations observed at Naul, Co. Dublin are shown in Figure 5.26 below together with rainfall for the sampling period. Average phosphorus concentrations over the study period are given in Table 5.18. Mean phosphorus concentrations

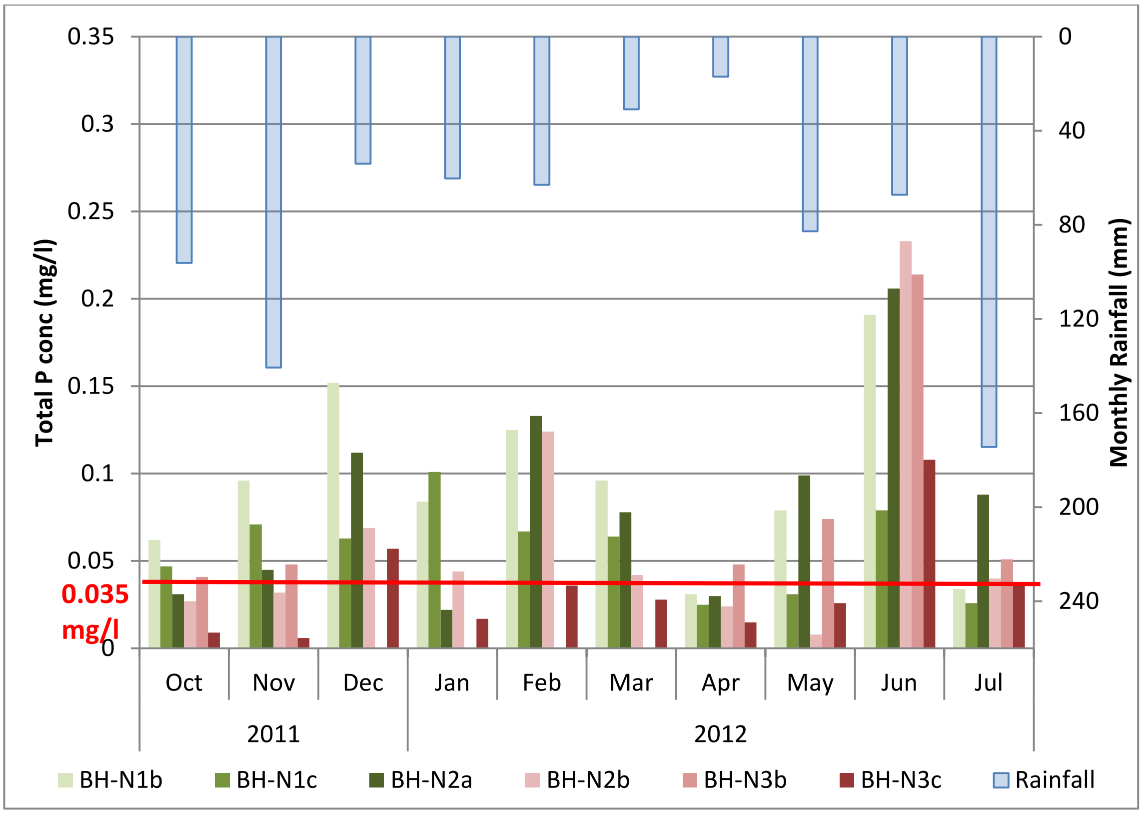


Figure 5.26 Total Phosphorus concentrations with rainfall at Naul, Co. Dublin

were above the EPA limit of 0.035 mg-P/l (shown as solid red line) at all piezometers. Two sample t-tests were carried out to compare phosphorus concentrations between sampling horizons at each of the borehole locations. No significant difference was found between mean phosphorus concentrations during this analysis indicating that depth does not affect long run mean phosphorus concentrations. Equally, single factor ANOVA analysis between borehole locations at the two sample horizons did not show a significant difference between upstream and downstream phosphorus with p-values of 0.174 and 0.303 respectively (See Table 5.19 and Table 5.20 below). The highest recorded value was 0.233 mg-P/l recorded in June 2013 at BH-N3b following a month of very high rainfall. Given that this piezometer appeared to be located in a zone that acts as a perched water table it is more likely that these values were due to particles washed into the groundwater and not orthophosphate travelling from an upstream source. Overall the levels of phosphorus observed at the Naul study location would indicate there may be a case of elevated phosphates; however, given the low values of nitrogen recorded during the sampling period it is likely that this is a characteristic of the aquifer and not a result of any specific anthropogenic activities.

Table 5.18 Average Phosphorus concentrations at Naul, Co. Dublin (all values in mg/l)

Horizon	BH-N1	BH-N2	BH-N3
Horizon b	0.095	0.084	0.047
Horizon c	0.002	0.064	0.034

Table 5.19 ANOVA output for Horizon 2 and Phosphorus at Naul, Co. Dublin

Anova: Single Factor — Total P - Horizon 2 - Low Vulnerability

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-N1b	10	0.95	0.095	0.0025167
BH-N2a	10	0.844	0.0844	0.0032705
BH-N3b	10	0.476	0.0476	0.0041672

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.01	2	0.006189	1.8652019	0.1742767	3.3541308
Within Groups	0.09	27	0.003318			
Total	0.1	29				

Table 5.20 ANOVA output for Horizon 3 and Phosphorus at Naul, Co. Dublin

Anova: Single Factor — Total P - Horizon 3 - Low Vulnerability

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-N1c	10	0.574	0.0574	0.0006178
BH-N2b	10	0.643	0.0643	0.0045305
BH-N3c	10	0.34	0.034	0.0009093

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.01	2	0.002522	1.2490567	0.3028228	3.3541308
Within Groups	0.05	27	0.002019			
Total	0.06	29				

Chloride (and Bromide)

Chloride values observed at Naul, Co. Dublin during this study ranged from 3.8 to 45.7 mg Cl/l and are shown graphically in Figure 5.27 below with average concentrations over the study period given in Table 5.21. Two sample t-tests were carried out which revealed no significant difference in mean chloride concentrations with depth at each of the three boreholes.

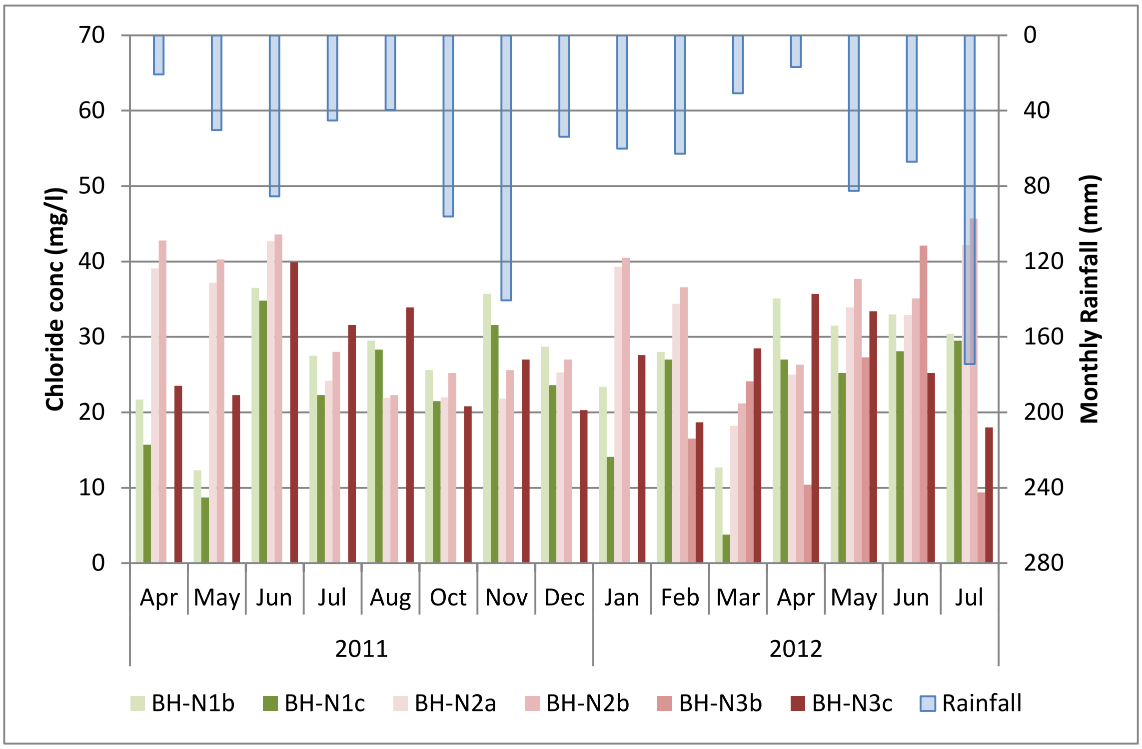


Figure 5.27 Chloride concentrations with rainfall at Naul, Co. Dublin

Table 5.21 Average Chloride concentrations at Naul, Co. Dublin (all values in mg/l)

Horizon	BH-N1	BH-N2	BH-N3
Horizon b	27	30	27
Horizon c	22	33	30

ANOVA analysis was used to access the statistical difference in mean chloride concentrations upstream and downstream of the study development. A p-value of 0.114 resulted from the analysis for horizon 2 which is not significant at the 0.05 significance level but does indicate some significance at a higher (less sensitive) significance level. A p-value of 0.0034 resulted from the analysis at horizon 3 indicating a highly significant result, showing therefore that the long run means at each of the borehole locations were not the same. The output of these ANOVA analyses is shown in Table 5.22 and Table 5.23 below. The output of the ANOVA at horizon 3 identifies that BH-N2 (the first downstream borehole) has higher mean chloride concentrations than both BH-N1 and BH-N3.

Table 5.22 ANOVA output for Horizon 2 and Chloride at Naul, Co. Dublin

Anova: Single Factor — Chlorides - Low Vulnerability - Horizon 2

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-N1b	15	411.6	27.44	55.03114
BH-N2a	15	460.1	30.67333	70.30495
BH-N3b	6	129.8	21.63333	151.8547

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	353.9377	2	176.9689	2.323	0.113794	3.284918
Within Groups	2513.979	33	76.18117			
Total	2867.916	35				

Table 5.23 ANOVA output for Horizon 2 and Chloride at Naul, Co. Dublin

Anova: Single Factor — Chlorides - Low Vulnerability - Horizon 3

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-N1c	15	341.2	22.74667	75.29695
BH-N2b	15	497.9	33.19333	70.95352
BH-N3c	15	406.4	27.09333	44.65067

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	826.1818	2	413.0909	6.491699	0.003494	3.219942
Within Groups	2672.616	42	63.63371			
Total	3498.798	44				

Chloride and bromide ions have been used to differentiate between various sources of anthropogenic and naturally occurring contaminants in groundwater due to their conservative nature. Previous studies have suggested that the ratio of chloride to bromide (Cl/Br) ions by mass can point to the source of the ions in a water sample. Katz et al. (2010) suggested that ratios between 400 to 1100 can be attributed to chemical constituents associated with human activities - in this case pollution owing to OSWTS. As part of this study bromide concentrations were investigated in addition to chloride concentrations between August and October 2012 and November and December 2012 for the site at Toonagh, Co. Clare in order to quantify the associated Cl/Br ratios. Resulting bromide concentrations and the associated Cl/Br ratios for Naul, Co. Dublin are shown in Table 5.24 below. None of the sampling horizons had Cl/Br ratios in the range that indicates anthropogenic pollutants.

Table 5.24 Bromide ratios for Naul, Co. Dublin

Date	Sample	Bromide (mg/l)	Cl/Br Mass Ratio
Aug-12	BH-N1a	0.117	234
	BH-N1b	0.126	221
	BH-N1c	0.121	202
	BH-N2a	0.067	339
	BH-N2b	0.074	262
	BH-N3c	0.086	307

E-coli

Results of the enumeration of *E. coli* at the Naul, Co. Dublin during the sampling period are shown in Figure 5.28 below with average concentrations for the study period given in Table 5.25. *E-coli* were generally absent from samples taken during the study and no clear trend of occurrences with either time or rainfall emerges. No significant difference was found using two sample t-tests between sample means at any of the three borehole locations indicating that mean *E-coli* numbers do not vary with depth. Equally an analysis using ANOVA comparing upstream and downstream *E-coli* numbers did not yield any significant differences in the long run means at each borehole location with p-values of 0.853 and 0.376 for horizons 2 and 3.

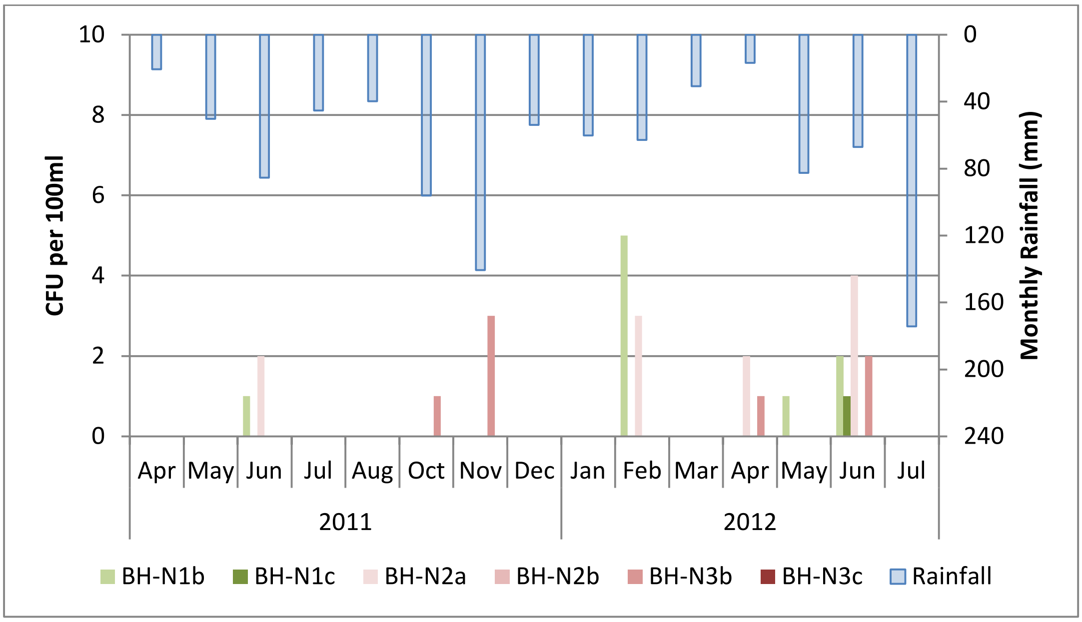


Figure 5.28 E-coli numbers with rainfall at Naul, Co. Dublin

Table 5.25 Average E-coli concentrations at Naul, Co. Dublin (all values in cfu/100ml)

Horizon	BH-N1	BH-N2	BH-N3
Horizon b	0.6	0.73	0.46
Horizon c	0.06	0	0

<u>Enterococci</u>

Figure 5.29 below illustrates graphically the numbers of *Enterococci* that were found during the study at Naul, Co. Dublin. As with the *E-coli* results seen previously, *Enterococci* bacteria were generally absent from samples taken during the study period. There was no significant difference between sample means at either BH-N2 or BH-N3 with depth. A significant difference was found however between horizons BH-N1b and BH-N1c with a t-value of 2.43 as shown in Table 5.27 below. BH-N1b had mean *Enterococci* numbers of 0.467 CFU/100ml with a standard deviation of 0.743 whereas BH-N1c had both a mean and standard deviation of zero during the study period.

Table 5.26 Average Enterococci concentrations at Naul, Co. Dublin (all values in cfu/100ml)

Horizon	BH-N1	BH-N2	BH-N3
Horizon b	0.46	0.6	0.4
Horizon c	0	0.13	0

As described in Chapter 4 previously, the land use in the area surrounding BH-N1 (upstream at the Naul) is agricultural whereas the land use at both of the downstream boreholes is horticulture (orchards). It is likely therefore that the higher mean *Enterococci* numbers found at the shallower horizon BH-N1b when compared statistically to the deeper horizon BH-N1c is due to cattle grazing in the area particularly given the steep groundwater gradient in the area and the lack of any OSWTS's upstream of BH-N1.

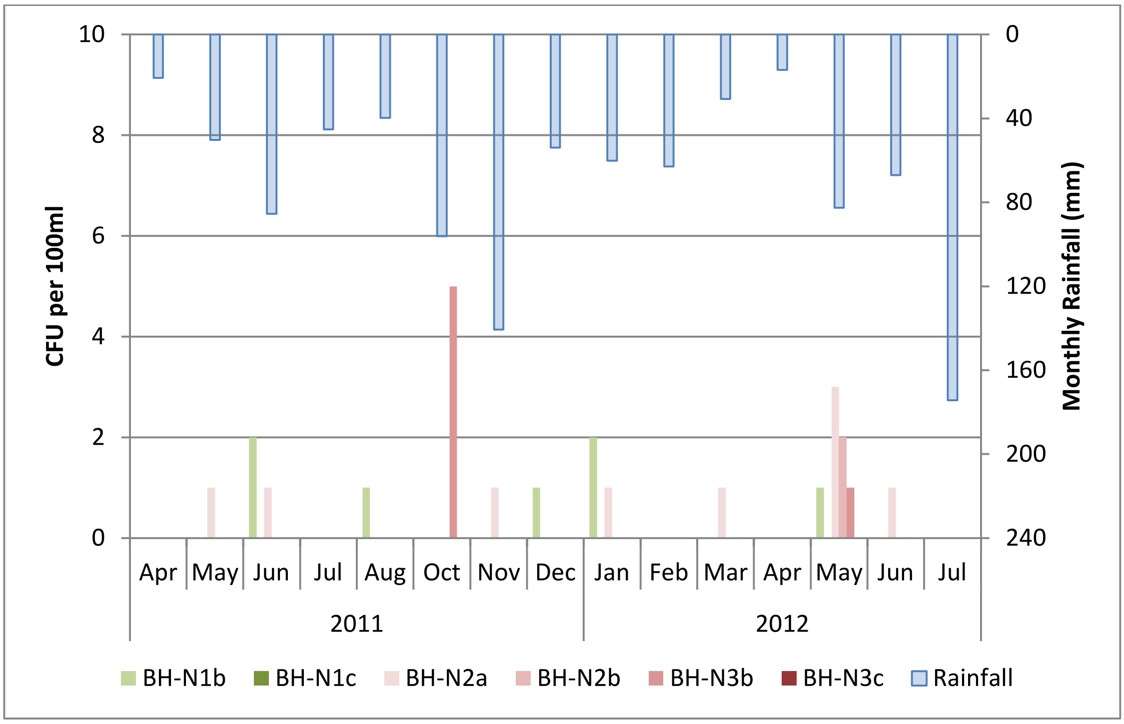


Figure 5.29 *Enterococci* numbers with rainfall at Naul, Co. Dublin

Table 5.27 T-test output for BH-N1 for the parameter *Enterococci* (t-Test: Two-Sample Assuming Unequal Variances)

Enterococci	BH-N1b	BH-N1c
Mean	0.466666667	0
Variance	0.552380952	0
Observations	15	15

Hypothesized Mean Difference	0
df	14
t Stat	2.431829168
P(T<=t) one-tail	0.01452037
t Critical one-tail	1.761310136
P(T<=t) two-tail	0.02904074
t Critical two-tail	2.144786688

Single factor ANOVA analysis comparing upstream and downstream *Enterococci* numbers did not yield any significant differences in the long run means between each borehole location with p-values of 0.836 and 0.376 for horizons 2 and 3 respectively.

<u>Low flow stream sampling at Naul, Co Dublin</u>

Due to the high overall groundwater quality observed at the Naul low vulnerability study site, it was decided to sample the surrounding surface water drainage system as it may be the more 'at risk' receptor in this area due to the low overall permeability of the subsoil. It is possible that wastewater disposed of in a conventional percolation area could travel laterally to the nearest surface water feature, given the likely decrease in permeability with depth in the till subsoil, which is usually a local drainage ditch or stream. In order to allow for normal seasonal variation in surface water quality it was decided that the most appropriate time to sample would be during low flow conditions as this would allow for the least amount of dilution of any nutrients entering the surface water system and that the main nutrients would be the focus of the study. A summary of the results recorded during this sampling event along with the locations where samples were taken are given in Figure 5.30 below. It should be noted that sampling locations 7 and 8 were outflows from land drain pipes that terminate in the local stream which are serving to drain the local apple orchard.

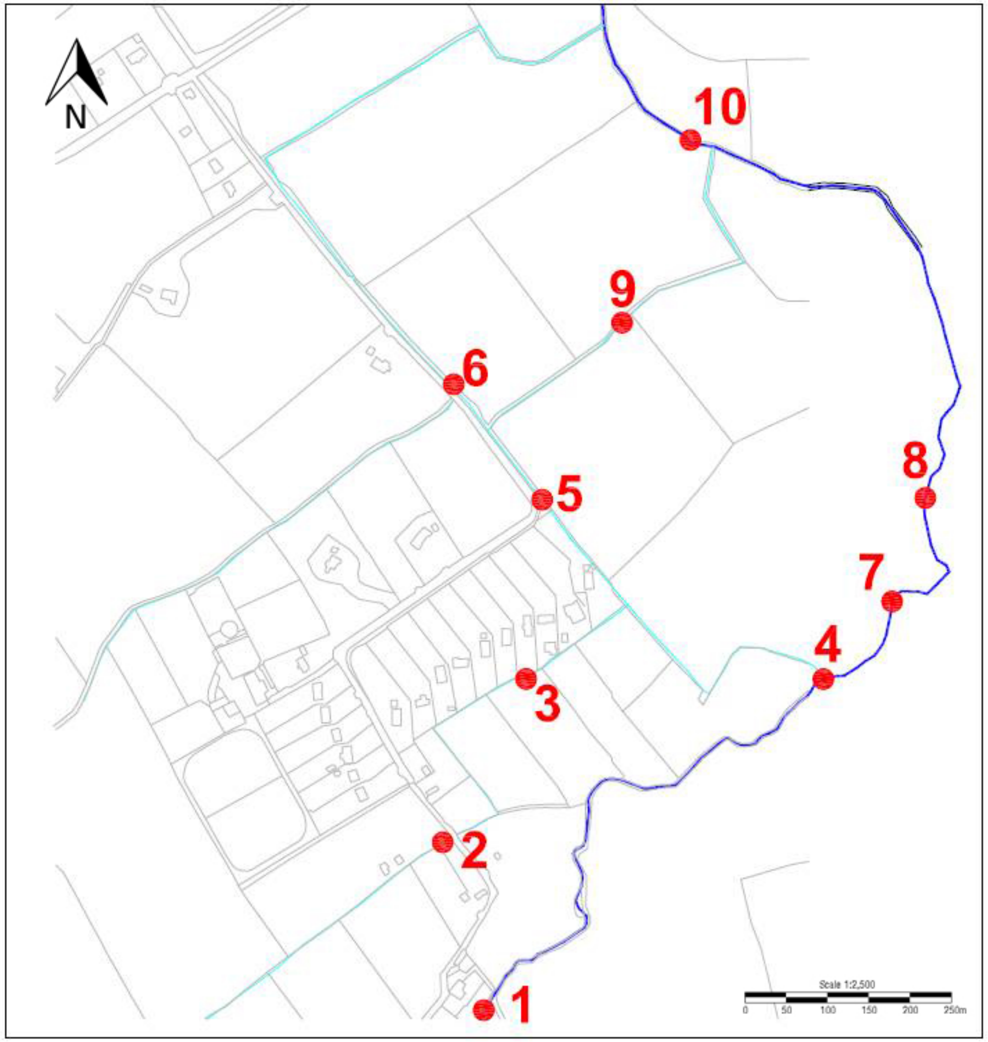
Mean nitrate and phosphorus concentrations were higher downstream (1.26 mg N/l; 0.133 mg P/l) than upstream (0.767 mg N/l; 0.125 mg P/l) of the study development, but neither were significantly different from each other as shown in Table 5.28 and Table 5.29 below. The land drains appear to be contributing a significant proportion of these increases, which could correspond to fertiliser application in the orchard. It is not known what, if any, fertiliser application occurs at the orchards and in this regard further investigation would be required to clarify. It can be concluded that surface water in the area does not appear to be impacted negatively by the cluster development.

Table 5.28 T-test output for low flow sampling of nitrate (t-Test: Two-Sample Assuming Unequal Variances)

Nitrate	*Upstream*	*Downstream*
Mean	0.766666667	1.26
Variance	0.023333333	0.928
Observations	3	5
Hypothesized Mean		
Difference	0	
df	4	
t Stat	-1.12185616	
P(T<=t) one-tail	0.162357332	
t Critical one-tail	2.131846786	
P(T<=t) two-tail	0.324714664	
t Critical two-tail	2.776445105	

Table 5.29 T-test output for low flow sampling of phosphorus (t-Test: Two-Sample Assuming Unequal Variances)

Total P	*Upstream*	*Downstream*
Mean	0.125197	0.133482421
Variance	0.002221	0.001331161
Observations	3	5
Hypothesized Mean		
Difference	0	
df	3	
t Stat	-0.26113	
P(T<=t) one-tail	0.405445	
t Critical one-tail	2.353363	
P(T<=t) two-tail	0.81089	
t Critical two-tail	3.182446	



SAMPLING POINT NUMBER	NITRATE CONCENTRATION (MG/L)	TOTAL PHOSPHURUS CONCENTRATION (MG/L)
1	0.8	0.153
2	0.5	0.101
3	0.6	0.140
4	0.6	0.102
5	0.9	0.179
6	0.8	0.102
7	2.1	0.096
8	1.9	0.099
9	1.6	0.093
10	2.8	0.179

DITCH

STREAM

Figure 5.30 Low flow sampling locations and results at Naul, Co. Dublin (March 2012)

5.4.2 Moderate Vulnerability Site – Rhode, Co. Offaly

Nitrogen

As for the low vulnerability site at Naul, nitrite and ammonium values at Rhode Co. Offaly (moderate vulnerability) were predominately below the laboratory limits of detection with only a small proportion of samples registering values above the detection limits (<0.02 mg-N/l NO_2; <0.05 mg-N/l NH_4). Nitrate values ranged from 0.5 to 13.2 mg-N/l during the sampling period and average concentrations over the study period are given in Table 5.30. Figure 5.31 below shows average nitrate concentrations grouped by quarter for each of the sampling locations with upstream piezometers shown in green shading and downstream piezometers shown in purple shadings. The highest nitrate concentrations occurred at BH-O1a with lowest concentrations tending to occur at BH-O2c and BH-O3b. A clear trend can be seen with higher nitrate concentrations occurring during the winter months with lowest values recorded in the autumn months. This is consistent with previous studies in Ireland including a recent large EPA study on nitrate leaching to groundwater (EPA, 2006).

Table 5.30 Average Nitrate concentrations at Rhode, Co. Offaly (all values in mg/l)

Horizon	BH-O1	BH-O2	BH-O3
Horizon a	6.82	3.43	-
Horizon b	4.08	3.24	-
Horizon c	3.79	2.89	3.34

A highly significant difference between sample means at different sampling horizons was found at the upstream monitoring borehole BH-O1 as shown in the ANOVA output in Table 5.31 below. It can be seen from Table 5.31 that mean nitrate concentrations decrease with depth into the aquifer at BH-O1. A similar test at BH-O2 did not show a significant difference between mean nitrate concentrations with depth into the aquifer with a p-value of 0.433. Single factor ANOVA tests between the two sampling horizons both upstream and downstream showed a significant difference between mean nitrate concentrations at the shallower horizon (horizon 1) with a t-value of 4.826 as shown in Table 5.32.

Table 5.31 ANOVA output for BH-O1 and Nitrate at Rhode, Co. Offaly

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-O1a	23	157	6.8261	9.269289
BH-O1b	23	93.9	4.0826	2.471502
BH-O1c	23	87.3	3.7957	2.97498

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	128.7	2	64.371	13.12295	1.6E-05	3.13592
Within Groups	323.7	66	4.9053			
Total	452.5	68				

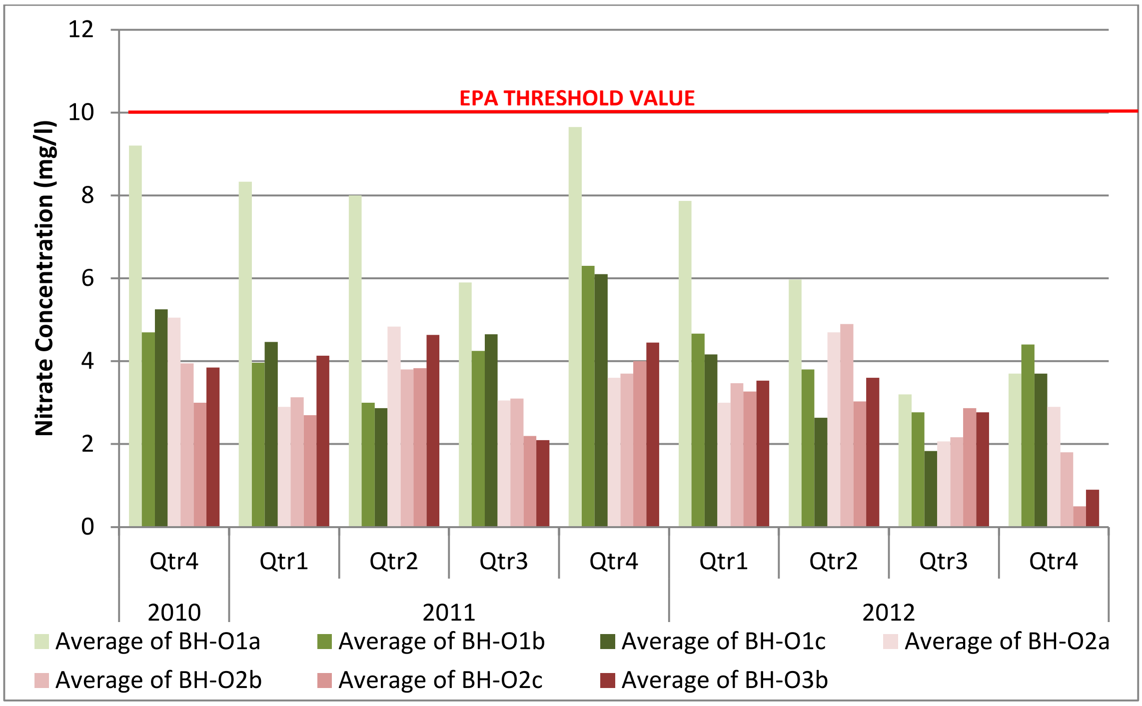


Figure 5.31 Seasonal Nitrate levels at Rhode, Co. Offaly

However, this is due to BH-O1a having higher mean nitrate concentrations (6.826 mg N/l) than BH-2a (3.438 mg N/l) and so it can be seen that mean nitrate concentrations therefore reduce downstream of the study development. The higher upstream concentrations are likely due to the intensive grazing of cattle in the area and the spreading of slurry. Similar statistical tests comparing upstream and downstream mean nitrate concentrations for horizons 2 and 3 did not result in significant differences indicating that in the long run the samples have similar mean nitrate concentrations upstream and downstream of the cluster development.

Table 5.32 T-test output for BH-O1 and Nitrate at Rhode, Co. Offaly

Nitrates	*BH-O1a*	*BH-O2a*
Mean	6.826	3.4348
Variance	9.269	2.0869
Observations	23	23
Hypothesized Mean Difference	0	
df	31	
t Stat	4.826	
P(T<=t) one-tail	2E-05	
t Critical one-tail	1.696	
P(T<=t) two-tail	4E-05	
t Critical two-tail	2.04	

Phosphorus

Phosphorus concentrations observed at Rhode, Co. Offaly together with rainfall are shown in Figure 5.32 below with average concentrations over the study period given in . Mean phosphorus concentrations exceeded the surface water limit of 0.035 mg P/l for all piezometers.

Table 5.33 Average Phosphorus concentrations at Rhode, Co. Offaly (all values in mg/l)

Horizon	BH-O1	BH-O2	BH-O3
Horizon a	0.053	0.048	-
Horizon b	0.049	0.049	-
Horizon c	0.052	0.053	0.054

Phosphorus concentrations appear to follow a trend with higher values recorded during the period March – June and lower values during the rest of the year. There is a weak indication that total dissolved phosphorus concentrations tended to fall during or following periods of high rainfall. Statistical tests were carried out to compare mean concentrations with depth at each of the borehole. ANOVA results showed p-values of 0.874 and 0.181 for BH-N1 and BH-N2 respectively indicating that there is no significant difference between the mean phosphorus concentrations with depth into the aquifer. Comparing mean phosphorus concentrations upstream and downstream of the cluster development yielded similar statistically insignificant results indicating that there is no mean difference between upstream and downstream phosphorus concentrations over time. The cluster development does not appear therefore to be negatively affecting phosphorus concentrations in the surrounding aquifer.

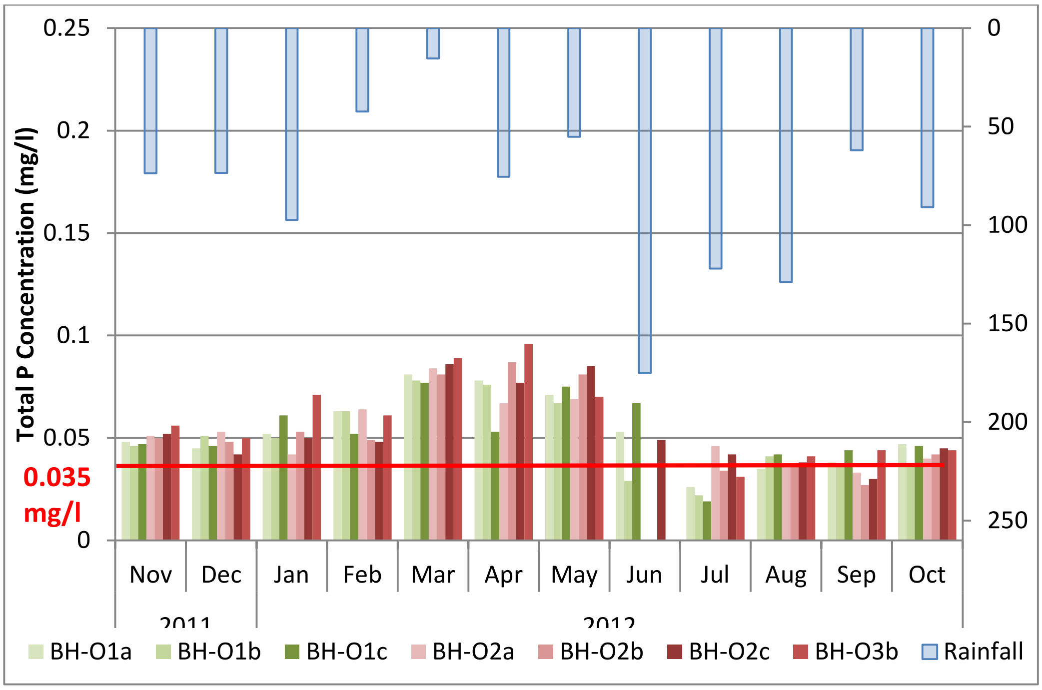


Figure 5.32 Total Phosphorus concentrations with rainfall at Rhode, Co. Offaly

Chloride (and Bromide)

Chloride values observed at Rhode, Co. Offaly during this study are shown graphically in Figure 5.33 below, with averaged values given in Table 5.34. Values observed during this study ranged from 9.4 to 136 mg/l. Values of chlorides appear to show a trend of higher values in December and February/March and this seems to coincide with lower rainfall quantities, although soil moisture would be higher during the winter time. All of the sampling horizons yielded Pearson correlation coefficients in the range -0.5 – 0.6, indicating a moderate to strong correlation between rainfall and chloride concentrations.

Table 5.34 Average Chloride concentrations at Rhode, Co. Offaly (all values in mg/l)

Horizon	BH-O1	BH-O2	BH-O3
Horizon a	44	39	-
Horizon b	40	35	-
Horizon c	30	29	47

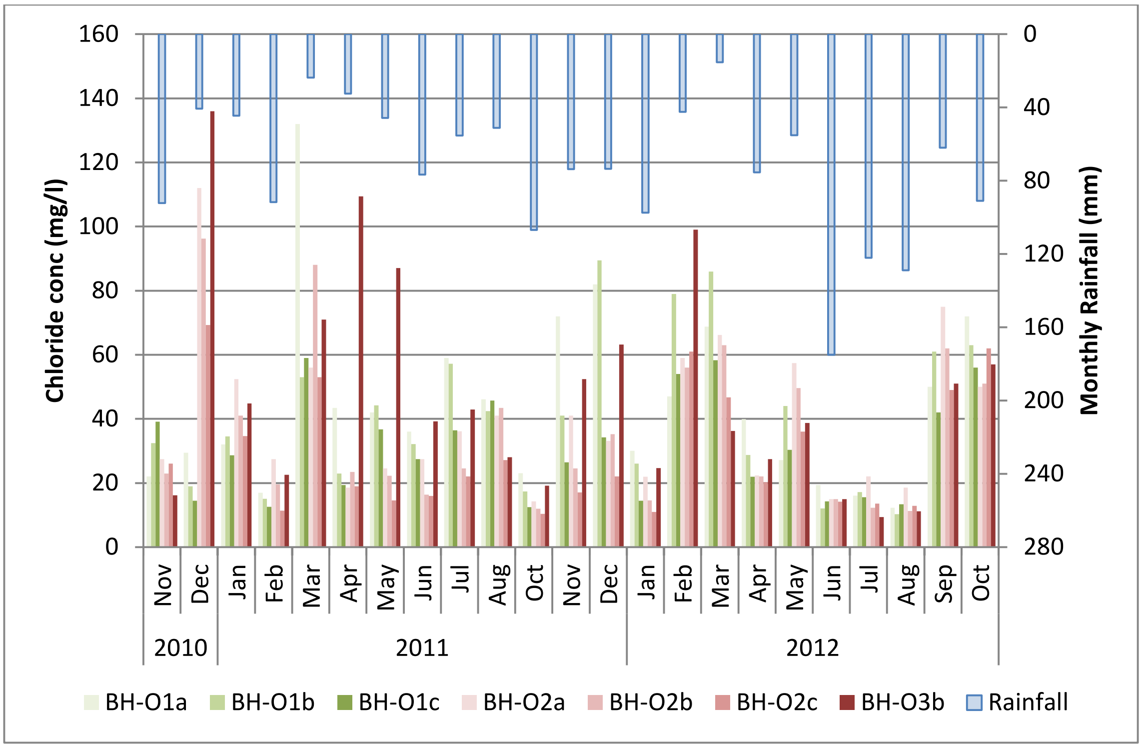


Figure 5.33 Chloride concentrations with rainfall at Rhode, Co. Offaly

Two sample t-tests were carried out on the data which revealed no significant difference in mean chloride concentrations with depth at each of the three boreholes. However, at BH-O1 the p-value, when comparing means with depth for chloride, was 0.133 as shown in Table 5.35 below which, whilst this value is not significant, it would be at a less conservative significance level. As seen earlier mean nitrate concentrations decreased with depth at BH-O1 and this trend is repeated here for mean chloride concentrations.

Table 5.35 T-test output for chlorides at BH-O1 (t-Test: Two-Sample Assuming Unequal Variances)

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-O1a	23	1018.7	44.291	754.6054
BH-O1b	23	928	40.348	548.4108
BH-O1c	23	712.9	30.996	248.4059

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	2145	2	1072.5	2.073948	0.1338	3.1359
Within Groups	34131	66	517.14			
Total	36276	68				

In order to investigate if chloride concentrations were higher downstream of the study development statistical tests between mean concentrations measured at similar horizons at each borehole location were carried out. The tests at horizon 1 and 2 did not yield significant results however the ANOVA analysis for horizon 3 resulted in a p-value of 0.017 – a highly significant result – as shown in Table 5.36 below. It can be seen that mean values at BH-O1c and BH-O2c are similar (~30 mg/l) and the mean concentration at BH-O3b (47.8 mg/l) is therefore resulting in the outcome of the test. As outlined in Chapter 4, BH-O3b is located within a depression with a depth-to-bedrock of less than 10 m. This shallower cover of subsoil provides less protection to contaminants entering the aquifer in this area and may explain the higher mean concentration observed.

Table 5.36 ANOVA output for chloride in horizon 3 at Rhode, Co. Offaly

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-O1c	23	712.9	30.996	248.4059
BH-O2c	23	669.4	29.104	345.5959
BH-O3b	23	1101.4	47.887	1119.793

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	4919.5	2	2459.8	4.305834	0.0175	3.1359
Within Groups	37703	66	571.26			
Total	42623	68				

Bromide concentrations and the associated Cl/Br ratios for Rhode, Co. Offaly are shown in Table 5.37 below. None of the sampling horizons had Cl/Br ratios in the range that indicates nearby anthropogenic pollutants. There does seem to be a trend of increasing ratios from August to October however a more prolonged study would be required to establish any definite seasonal trends.

Table 5.37 Bromide ratios for Rhode, Co. Offaly

Date	Sample	Bromide (mg/l)	Cl/Br Mass Ratio
Aug-12	BH-O1a	0.224	55
	BH-O1b	0.168	61
	BH-O1c	0.151	89
	BH-O2a	0.155	120
	BH-O2b	0.148	76
	BH-O2c	0.187	69
Sep-12	BH-O1a	0.324	154
	BH-O1b	0.292	209
	BH-O1c	0.279	150
	BH-O2a	0.345	217
	BH-O2b	0.324	191
	BH-O3b	0.226	226
Oct-12	BH-O1a	0.271	266
	BH-O1b	0.192	328
	BH-O1c	0.134	417
	BH-O2a	0.164	305
	BH-O2b	0.181	281
	BH-O2c	0.190	327

E-coli

The occurrences of *E.-coli* at Rhode, Co. Offaly over the monitoring period are shown in Figure 5.34 below, with average values for the study period given in Table 5.38. *E-coli* occurrences were either very low or absent altogether during the course of the study with the exception of a number of peaks at BH-O3b. Given that BH-O3b is located in an area of shallower subsoil cover and the isolated nature of these peaks it is most likely to be a result of agricultural practices given that OSWTS provide constant loading of contaminants.

Table 5.38 Average E-coli concentrations at Rhode, Co. Offaly (all values in cfu/100ml)

Horizon	BH-O1	BH-O2	BH-O3
Horizon a	0.78	0.56	-
Horizon b	0.17	0.39	-
Horizon c	0	0.30	2.21

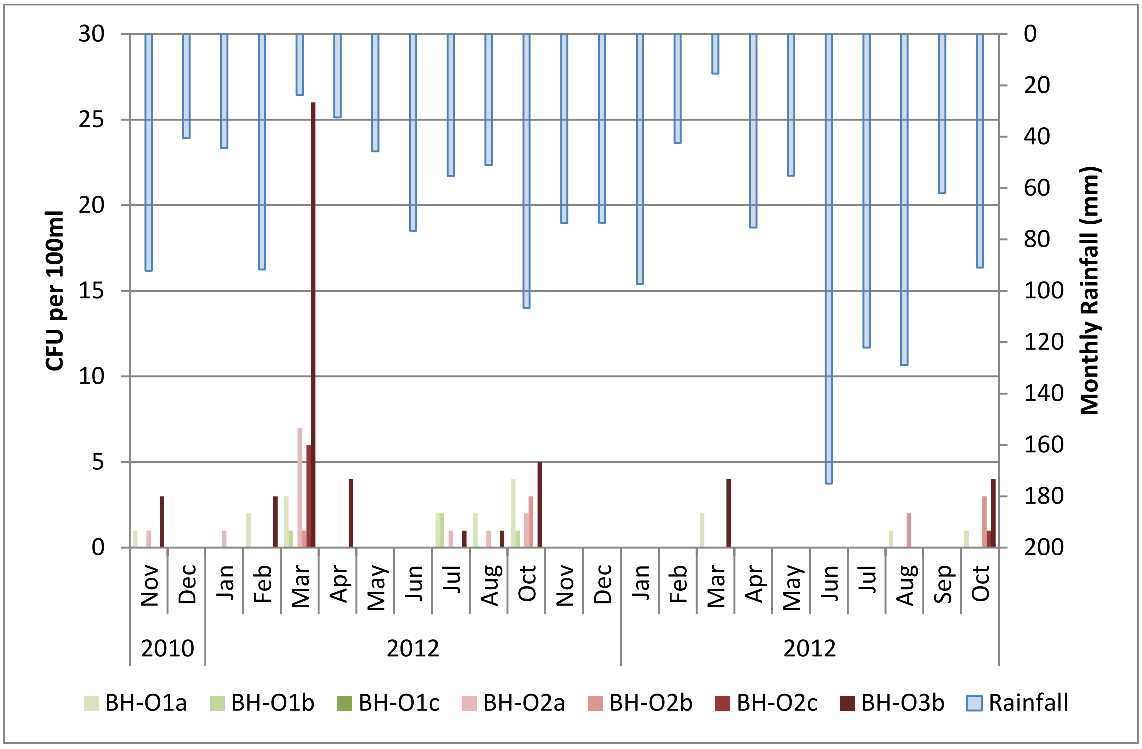


Figure 5.34 E-coli levels at Rhode, Co. Offaly

A significant difference between sample means at BH-O1 with depth into the aquifer (see Table 5.39) was found but not for BH-O2. Statistical analysis comparing upstream and downstream *E-coli* numbers gave a significant result at horizon 3 as shown in Table 5.40 below. It can be seen that mean *E-coli* numbers are higher at BH-O2c than at BH-O1c and numbers at BH-3b were in turn higher than at BH-3b indicating that overall *E-coli* numbers were increasing downstream of the study development.

Table 5.39 ANOVA output for E-coli at BH-O1 at Rhode, Co. Offaly

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-O1a	23	18	0.782609	1.359684
BH-O1b	23	4	0.173913	0.241107
BH-O1c	23	0	0	0

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	7.768116	2	3.884058	7.279012	0.001391	3.135918
Within Groups	35.21739	66	0.533597			
Total	42.98551	68				

Table 5.40 ANOVA output for E-coli in horizon 3 at Rhode, Co. Offaly

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-O1c	23	0	0	0
BH-O2c	23	7	0.304348	1.58498
BH-O3b	23	51	2.217391	29.81423

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	66.46377	2	33.23188	3.175101	0.048243	3.135918
Within Groups	690.7826	66	10.4664			
Total	757.2464	68				

As a further check a two sample t-test was carried out between BH-O1c and BH-O2c and this test did not yield a significant result (t-stat: -1 < t-crit: 2.07) as expected as mean *E-coli* numbers were 0.304 CFU/100ml at BH-O2c and 2.217 CFU/100ml at BH-O3b. As discussed above it can be seen in Figure 5.34 that the mean numbers for BH-O3b is being inflated by a small number of months that had very high numbers; given the inconsistent nature of these peaks it is likely they are of agricultural origin and not due to the fact that the sampling point is downstream of the cluster of OSWTS's.

Enterococci

The occurrences of *Enterococci* at Rhode, Co. Offaly over the monitoring period are shown in Figure 5.35 below, with average values for the study given in Table 5.41. *Enterococci* occurrences were either very low or absent altogether during the course of the study with the exception of a number of peaks at Bh-O3b as was the case with *E-coli* described above.

Table 5.41 Average Enterococci concentrations at Rhode, Co. Offaly (all values in cfu/100ml)

Horizon	BH-O1	BH-O2	BH-O3
Horizon a	0.73	1.47	-
Horizon b	0.08	0.26	-
Horizon c	0	0.34	4.65

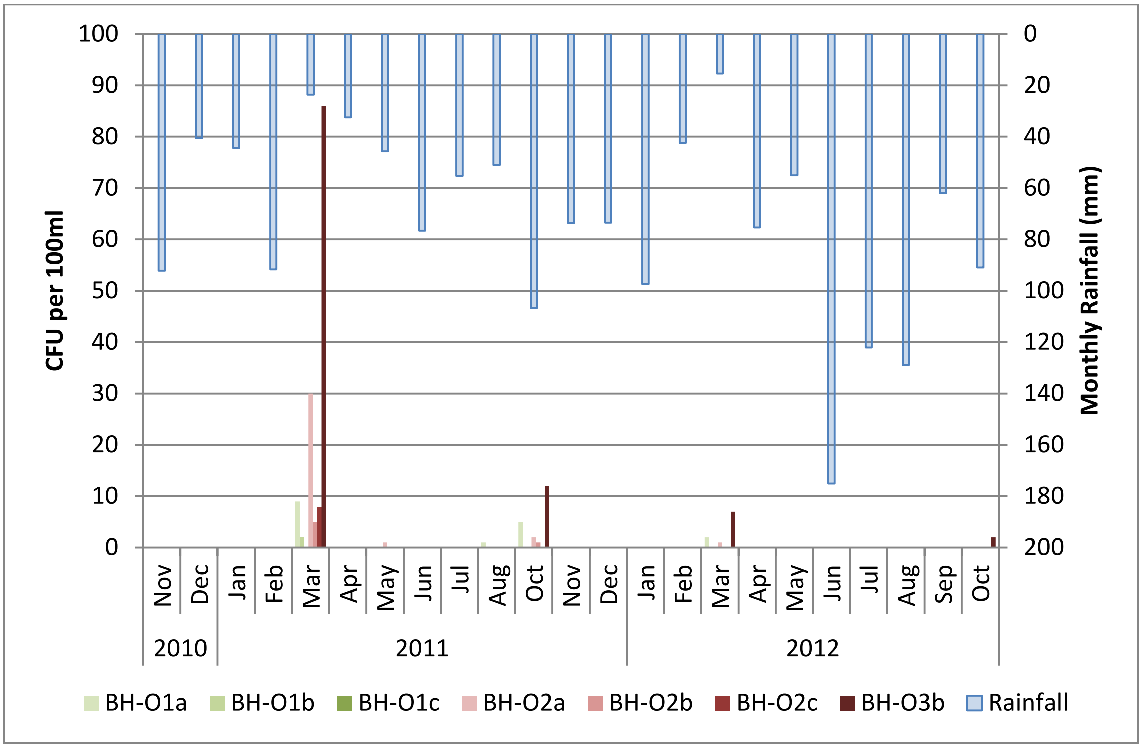


Figure 5.35 *Enterococci* levels at Rhode, Co. Offaly

No significant difference in sample means between sampling horizons at each of the borehole locations was found. However, a p-value of 0.096 was calculated for the ANOVA analysis for BH-O1 which would be significant at a less conservative confidence level. As with previous water quality parameters mean E*nterococci* numbers can be seen to be decreasing with depth into the aquifer. The statistical analysis comparing upstream and downstream *Enterococci* numbers did not give statistically significant results at any of the 3 horizons. However, as was seen with the similar analysis for *E-coli,* mean *E*nterococci numbers at BH-3b were higher than BH-2c and BH-O1c.

Rainfall and Bacterial Analysis

Given that both the *E-coli* and *Enterococci* groups of bacteria have been chosen as indicators of faecal contamination, a more detailed analysis of these indicators was undertaken to better understand their relationship with the preceding quantities of rainfall. This is similar to a technique carried out previously by Hynds et al. (2012) whereby rainfall was summed for the preceding 24, 48, and120 hour along with the 30 day rainfall quantity (monthly). A relationship (if one exists) is then established between rainfall and the presence or absence of the indicator bacteria groups. Given that the previous study carried out by Hynds et al. (2012) related mainly to high and extreme areas of the country, additional rainfall totals for the 21, 45 and 60 day intervals were included in the analysis for the current study to allow for the moderate vulnerability site as recharge times will be longer given the thickness of the subsoil. A regression analysis using the rainfall intervals described above was carried out with both *Enterococci* and *E-coli* and the results are shown in Table 5.42 and Table 5.43. It can be seen that the 21, 30 and 45 day rainfall quantities are all significant for enterococci numbers with the 30 day rainfall giving a highly significant p-value of 0.002

Table 5.42 Regression analysis for enterococci and varying rainfall intervals (Moderate vulnerability)

	Coeff	*SE Coeff*	*t Stat*	*P-value*
Intercept	1.013114225	1.79629216	0.564003	0.573578
24hr	0.288936734	0.329426565	0.87709	0.381813
48hr	-0.322137709	0.278251171	-1.15772	0.248781
120hr	0.149867906	0.163864337	0.914585	0.361849
21d	-0.117284774	0.05691077	-2.06085	0.04101
30d	0.157983433	0.05054256	3.12575	0.002123
45d	-0.088048389	0.041106198	-2.14197	0.033778
60d	0.021335433	0.029700044	0.718364	0.473629

Again the 30 and 45 day rainfall intervals are significant for *E-coli* numbers with the 30 day rainfall giving a highly significant p-value of 0.002. Given that both indicator bacteria groups show a significant relationship with 30 day rainfall, it can be concluded that following intense rainfall events at this moderate vulnerability site these bacteria groups take approximately 30 days to migrate down to the groundwater table.

Table 5.43 Regression analysis for E-coli and varying rainfall intervals (Moderate vulnerability)

	Coeff	*SECoeff*	*t Stat*	*P-value*
Intercept	0.239797	0.56907757	0.421378721	0.674069945
24hr	0.108631	0.104364576	1.040884278	0.299571274
48hr	-0.06553	0.088151863	-0.743420276	0.458367329
120hr	0.034359	0.051913336	0.661852602	0.509061288
21d	-0.03419	0.018029719	-1.896571581	0.05976926
30d	0.050271	0.016012227	3.139549682	0.002031033
45d	-0.03159	0.013022723	-2.425771186	0.016439449
60d	0.010276	0.009409176	1.092162761	0.276478345

For bacteria to enter the aquifer from agriculture or OSWTS's they must first migrate with the percolating effluent through the subsoil and thus the travel time through the subsoil to the water table is the key determining factor. During periods of low rainfall percolating water will travel at low velocities and may take a tortuous route and thus bacteria attenuation via processes such as adsorption and/or die-off is likely. However during or following periods of high or extreme rainfall the subsoil may approach saturated conditions (particularly during the winter months when antecedent soil moisture is high) and the bacteria may travel more directly downwards to the water table. Mulqueen and Rodgers (2001) have provided an estimate for converting from T-values (which were measured during this study) to field saturated hydraulic conductivity K_{fs} for the subsoil.

$$K_s = \frac{4.2}{t_m} \text{ [m/d]}$$

where: t_m is the T-value for the soil.

It is therefore possible to estimate the travel time through the subsoil, given saturated conditions, and then compare this travel time to the daily rainfall quantities for the days leading up to the sampling event as outlined above. The measured T-value for the subsoil at Rhode, Co. Offaly was 34.5mins/25mm which would then indicate a K_{fs} value for the subsoil of 0.12 m/d. In addition the K_s for the subsoil was calculated using the Hydrus unsaturated modelling software at between 0.087 and 0.129 m/d (see Chapter 7 for details).

During drilling the subsoil in the vicinity of the cluster of OSWTS's was found to be 7 m deep and given an average percolation trench invert depth of 0.9 m; the available depth of subsoil from the base of the percolation trenches to the water table is 6.1 m yielding a travel time of 46 - 50 days. This is higher than the 30 day interval which was found to be highly significant above, however the 45 day interval was also found to be significant and therefore the two methods of analysis compare reasonably well. Both analysis indicate that following an intense rainfall event it takes between 30 and 50 days for bacteria to enter the groundwater system. Nevecherya et al. (2005) have estimated that both the *E-coli* and *Enterococci* bacteria groups can survive for between 150 and 400 days in both saturated subsoil and groundwater depending on the ambient temperatures and therefore the analysis above can be considered consistent with the literature. More research would be needed in this area in the context of Irish soils; however the exercise does highlight that intense rainfall events are acting to move contaminants more quickly down into the aquifers below whether from agriculture or OSWTS.

5.4.3 High Vulnerability Site – Carrigeen, Co. Kilkenny

Nitrogen

As with the previous two study sites, nitrite and ammonium values at Carrigeen Co. Kilkenny (high vulnerability) were predominately below the laboratory limits of detection (<0.02 mg-N/l NO_2; <0.05 mg-N/l NH_4). Nitrate values ranged from 0.2 to 13.4 mg N/l during the sampling period and average values for the study period are given in Table 5.44 below. Figure 5.36 below shows average nitrate concentrations by quarter with upstream monitoring wells shown in green shading and downstream piezometers shown in purple shadings. Nitrate concentrations were frequently above 8 mg-N/l at BH-K1a and B-K1b during the study and following consultation with the owners of these two potable wells these values were attributed to slurry spreading in the fields upstream of these wells (within 20 m). As discussed later in this section this slurry spreading is also thought to have caused elevated bacteria levels present in these wells.

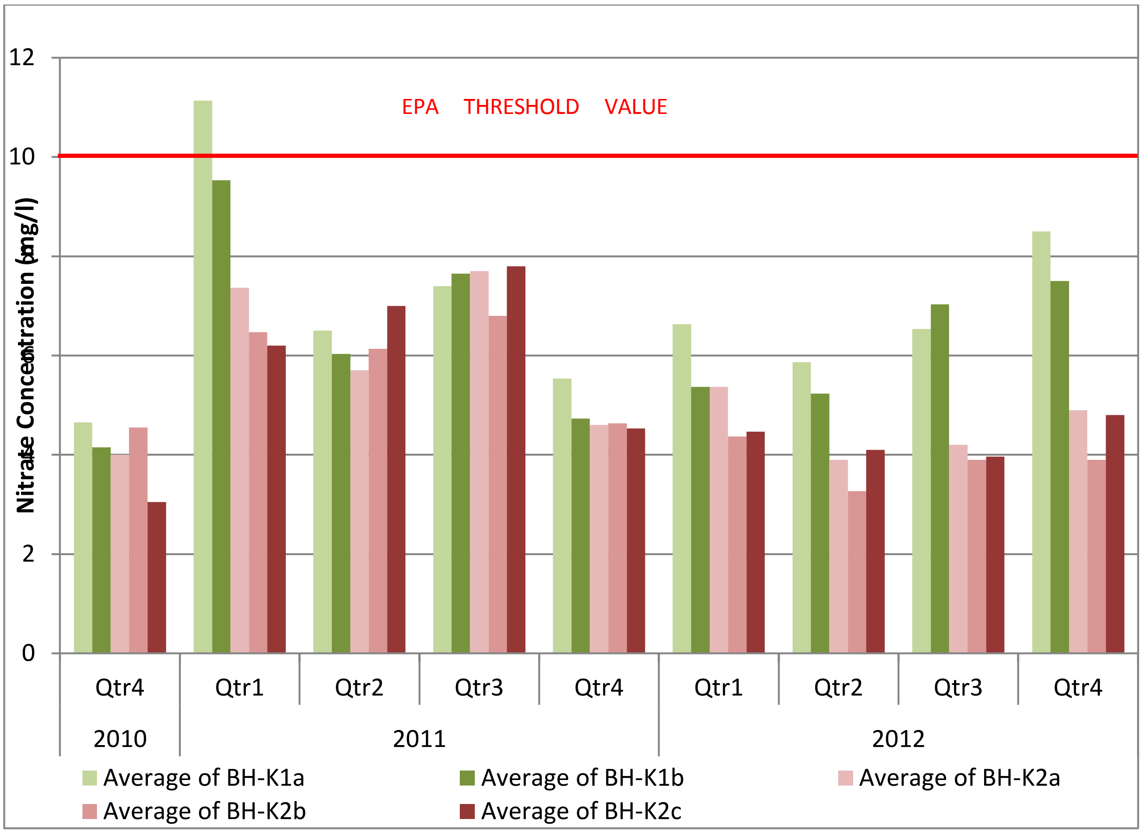


Figure 5.36 Seasonal Nitrate levels at Carrigeen, Co. Kilkenny

An additional time series plot of nitrate concentrations with rainfall quantities (Figure 5.37 below) suggests a relationship between rainfall and observed nitrate concentrations indicating that higher rainfall causes "flushing" of nutrients into the aquifer. However the Pearson correlation coefficients only yielded values in the range of 0.04 – 0.25 indicating that only a very weak correlation actually exists.

Table 5.44 Average Nitrate concentrations at Carrigeen, Co. Kilkenny (all values in mg/l)

Horizon	BH-K1	BH-K2
Horizon a	-	4.42
Horizon b	-	4.90
Horizon c	6.30	5.1

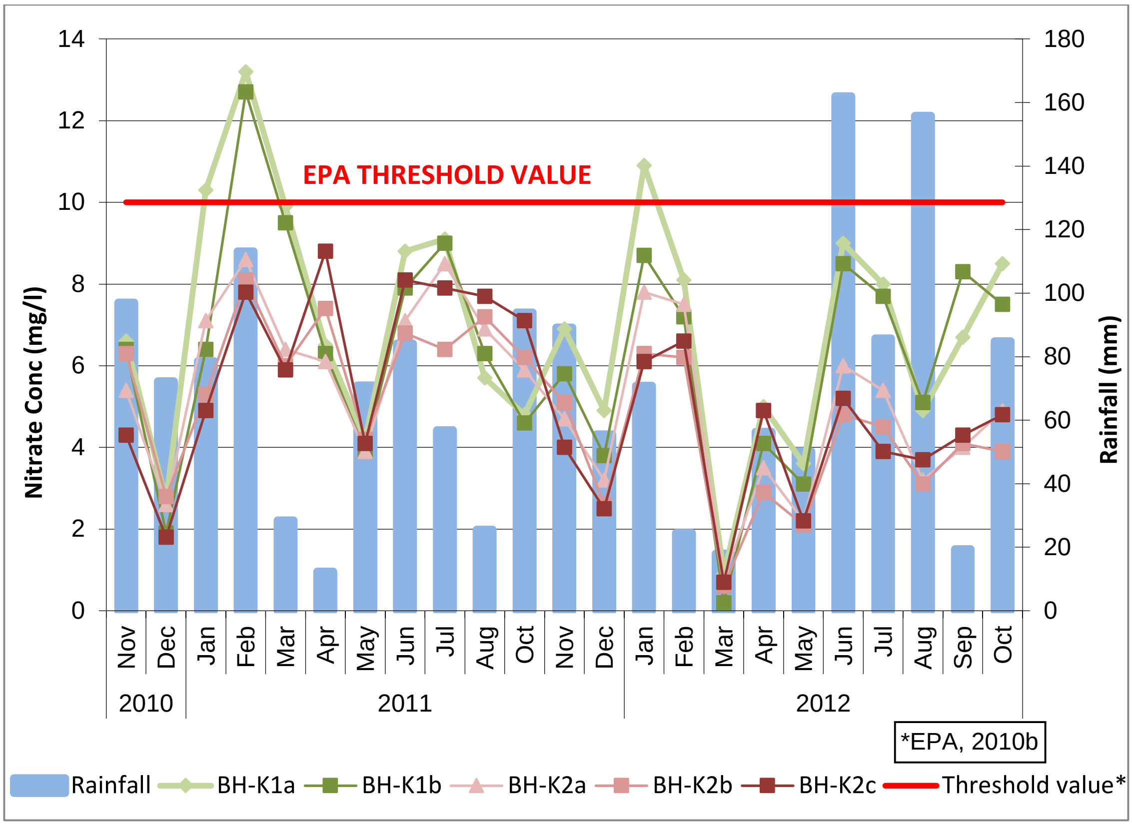


Figure 5.37 Nitrate concentrations with rainfall at Carrigeen, Co. Kilkenny

The impact of this slurry spreading was discussed with the land owner and the well owners during the study period and a greater set-back distance was maintained during the second year of monitoring (2012) which resulted in reduced nitrate concentrations as can be seen in Figure 5.36. When sample means were compared for BH-K2 at the different horizon depths the test produced a statistically insignificant result indicating that mean nitrate concentrations do not vary in the long run with depth into the aquifer. The test could not be carried out at the upstream monitoring locations as only single horizons were sampled. Statistical tests comparing upstream and downstream mean nitrate concentrations yielded an ANOVA p-value of 0.065 which is close to being significant at the 0.05 significance level – see Table 5.45 below. However, mean nitrate concentrations can be seen to decrease downstream of the study site falling from 6.92 mg N/l upstream to 5.1 mg N/l downstream. It is concluded therefore that the cluster development is not adversely affecting nitrate concentrations in the surrounding aquifer.

Table 5.45 ANOVA output for nitrate in horizon 3 at Carrigeen, Co. Kilkenny

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-K1a	23	159.2	6.921739	8.358142
BH-K1b	23	144.9	6.3	7.641818
BH-K2c	23	117.3	5.1	4.770909

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	39.44725	2	19.72362	2.848743	0.065061	3.135918
Within Groups	456.9591	66	6.923623			
Total	496.4064	68				

<u>Phosphorus</u>

Concentrations of total dissolved phosphorus recorded at Carrigeen, Co. Kilkenny over time are shown in Figure 5.38 below with average concentrations for the study period given in Table 5.46. Values are higher in winter/spring period compared to the summer/autumn period which is consistent with recorded nitrate trends. A comparison using ANOVA for BH-K2 at the different horizon depths the test produced an insignificant result indicating that mean phosphorus concentrations did not vary over time with depth into the aquifer. A comparison of upstream and downstream phosphorus concentrations using ANOVA yielded a p-value of 0.801 which is not significant. Mean phosphorus concentrations were therefore similar upstream and downstream of the cluster development indicating again that the cluster does not appear to be negatively impacting on the aquifer.

Table 5.46 Average Phosphorus concentrations at Carrigeen, Co. Kilkenny (all values in mg/l)

Horizon	BH-K1	BH-K2
Horizon a	-	0.055
Horizon b	-	0.063
Horizon c	0.064	0.066

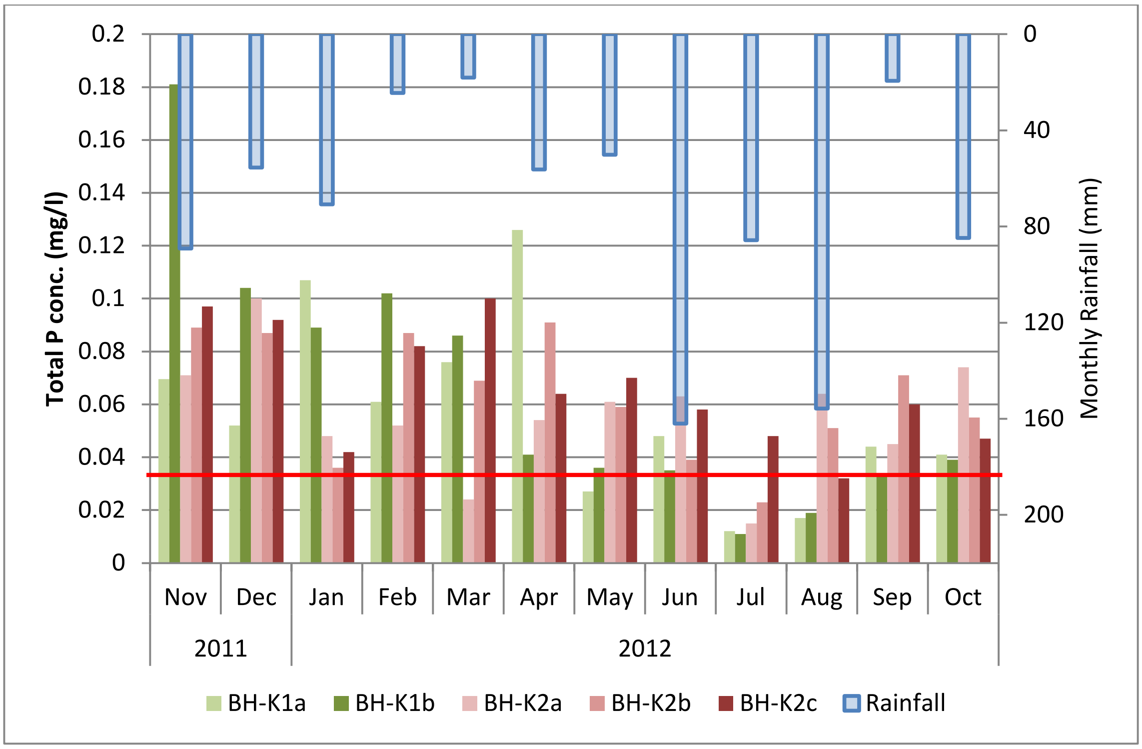


Figure 5.38 Total Phosphorus concentrations with rainfall at Carrigeen, Co. Kilkenny

Chloride (and Bromide)

Chloride values observed at Carrigeen, Co. Kilkenny during this study are given in Figure 5.39 below, with average values over the study period given in Table 5.47. Values ranged from 7.9 to 86 mg/l during the study period. Figure 5.39 illustrates a seasonal trend with higher chloride concentrations during spring to early summer periods. Statistical tests between means at the different horizon depths did not yield a significant result and neither did the comparison between upstream and downstream chloride concentrations with a p-value of 0.763. It is concluded therefore that mean chloride concentrations are not higher downstream of the study development. This backs up the trends seen with other indicator parameters at this site that the cluster development does not appear to be impacting negatively on the aquifer.

Table 5.47 Average Chloride concentrations at Carrigeen, Co. Kilkenny (all values in mg/l)

Horizon	BH-K1	BH-K2
Horizon a	-	26
Horizon b	-	25
Horizon c	29	25

Chloride/bromide ratios for Carrigeen are shown in Table 5.49. Elevated Cl/Br ratios at BH-K2 observed in October 2012 indicate possible anthropogenic influences. Table 5.48 below summarises all of the associated water quality parameters for October 2012 to put these results in context. The implication of this summary of the results is that the cluster of OSWTS's may be having a negative impact on water quality at this location, however given that OSWTS's tend to produce a constant loading output throughout the year and given that at most other times of the year the water quality is generally within the thresholds, it is difficult draw any definite conclusions. As was seen at the *MODERATE* vulnerability site, there is also a rise in the ratios from August to October.

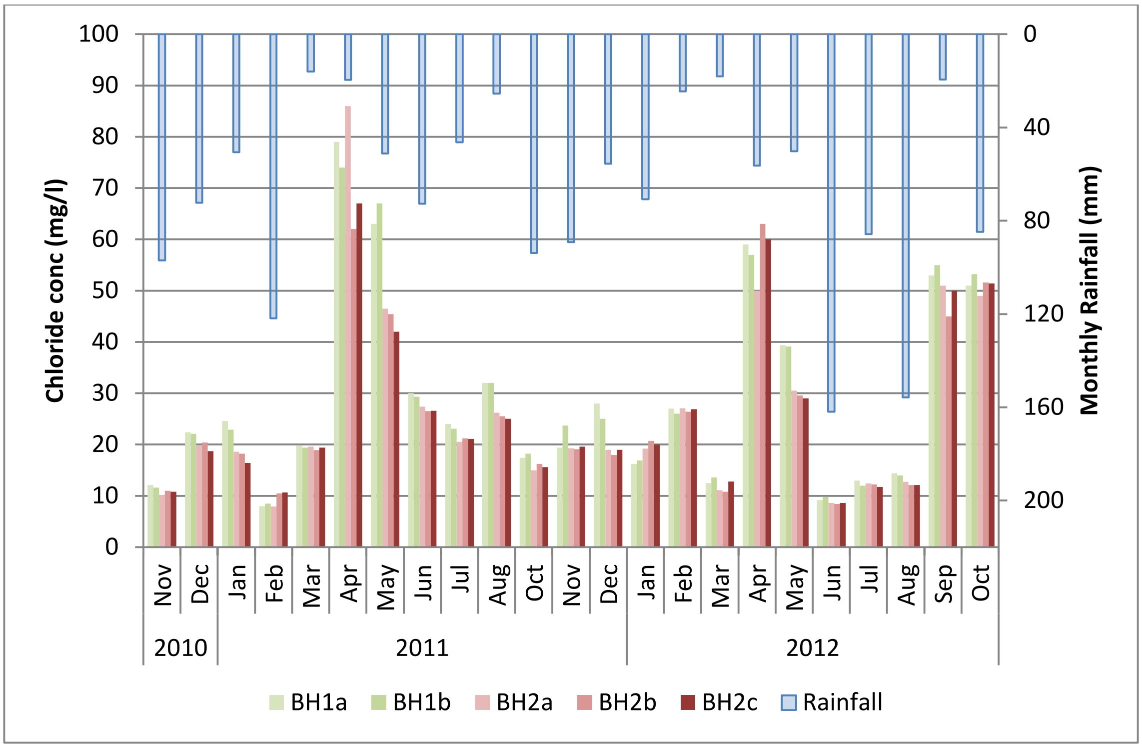


Figure 5.39 Chloride concentrations with rainfall at Carrigeen, Co. Kilkenny

Table 5.48 Summary of water quality results at Carrigeen for October 2012

	EC	pH	Nitrate	Total P	*E-coli*	Enterococci	Bromide	Chloride
BH-2a	477	7.89	4.9	0.074	4	1	0.149	49
BH-2b	396	7.78	3.9	0.055	1	0	0.096	51.6
BH-2c	391	7.67	4.8	0.047	0	3	0.087	51.4

Table 5.49 Bromide ratios for Carrigeen, Co. Kilkenny

Date	Sample	Bromide (mg/l)	Cl/Br Mass Ratio
Aug-12	BH-K1a	0.125	115
	BH-K1b	0.111	126
	BH-K2a	0.104	122
	BH-K2b	0.087	139
	BH-K2c	0.063	192
Sep-12	BH-K1a	0.164	323
	BH-K1b	0.171	322
	BH-K2a	0.124	412
	BH-K2b	0.119	377
	BH-K2c	0.128	391
Oct-12	BH-K1a	0.160	319
	BH-K1b	0.124	429
	BH-K2a	0.149	328
	BH-K2b	0.096	532
	BH-K2c	0.087	583

E-coli

E. coli occurrences at Carrigeen, Co. Kilkenny were more frequent and reached higher peaks than at either of the two previous study locations. Figure 5.40 below illustrates the numbers of colony forming units that were measured during the study period with average values over the study period given in Table 5.50. Peaks in *E-coli* numbers can be seen from December to March and also during July and October. The relationship between *E-coli* numbers and rainfall quantities is discussed further below.

Table 5.50 Average E-coli concentrations at Carrigeen, Co. Kilkenny (all values in cfu/100ml)

Horizon	BH-K1	BH-K2
Horizon a	-	1.39
Horizon b	-	2.04
Horizon c	2.21	1.04

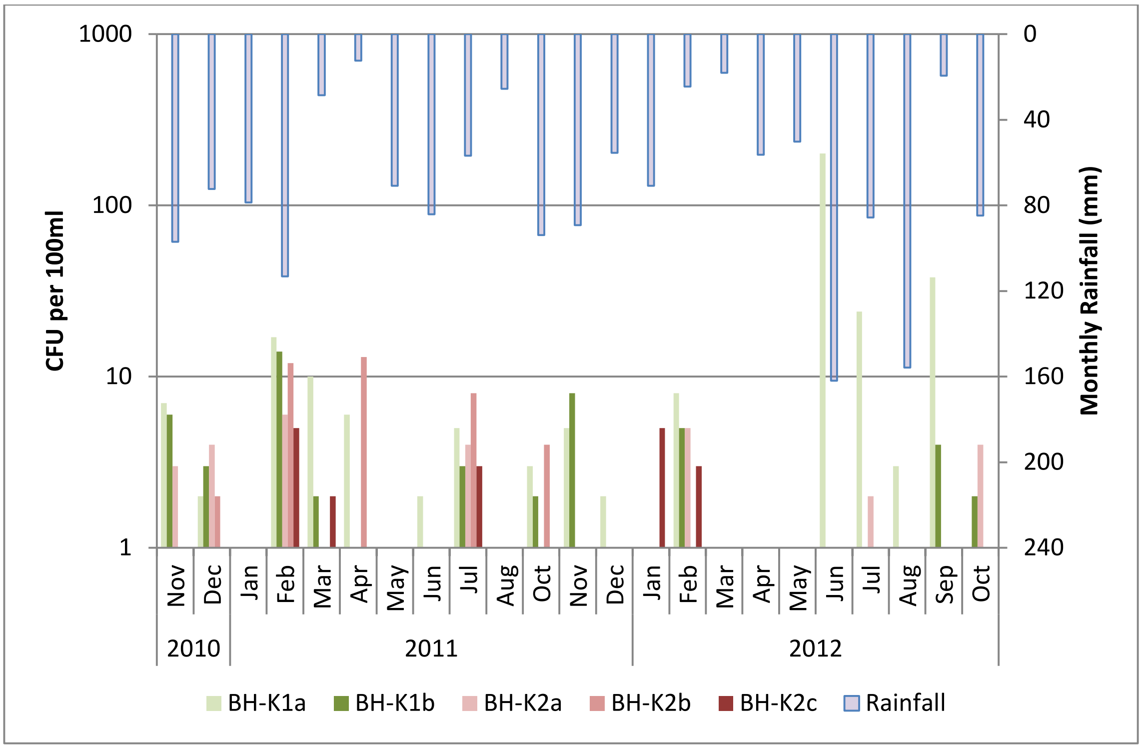


Figure 5.40 E-coli levels with rainfall at Carrigeen, Co. Kilkenny

A statistical test between mean *E-coli* numbers and sampling horizons at BH-K2 yielded a statistically insignificant result. ANOVA analysis used to compare mean *E-coli* numbers upstream and downstream of the study development also yielded a statistically insignificant result. It can be seen however that upstream (BH-K1b) mean *E-coli* numbers were higher at 2.043 CFU/100ml than downstream *E-coli* numbers at 1.043 CFU/100ml. This would indicate again that the cluster development is not adversely affecting groundwater quality in the area with respect to *E-coli* numbers present.

<u>Enterococci</u>

Enterococci occurrences at Carrigeen, Co. Kilkenny were again more frequent and reached higher peaks than the two previous study locations. Figure 5.41 displays the numbers of colony forming units that were measured during the study period together with rainfall and average values over the study period are given in Table 5.51. As was seen with *E-coli* above, peaks *E-coli* numbers can be seen from December to March and also during July and October. There is no trend between measured *Enterococci* numbers and rainfall quantities with Pearson correlation coefficients of 0.18 – 0.37. Analysis between mean *Enterococci* numbers and sampling horizons at BH-K2 yielded a statistically insignificant result. ANOVA analysis was used to compare mean *Enterococci* numbers upstream and downstream of the study development and the output of this analysis is shown in Table 5.52 below. The p-value was 0.118 indicating that the test could yield a significant result if a less conservative significance level were chosen. However, when examining the mean numbers of *Enterococci* for the upstream and downstream monitoring locations it can be seen that the result is only close to being significant due to the fact that downstream mean *Enterococci* numbers at 1.043 CFU/100ml were considerably lower than mean upstream numbers at 14.565 CFU/100ml indicating that, as for the *E-coli* results, mean enterococci numbers were lower downstream of the study development.

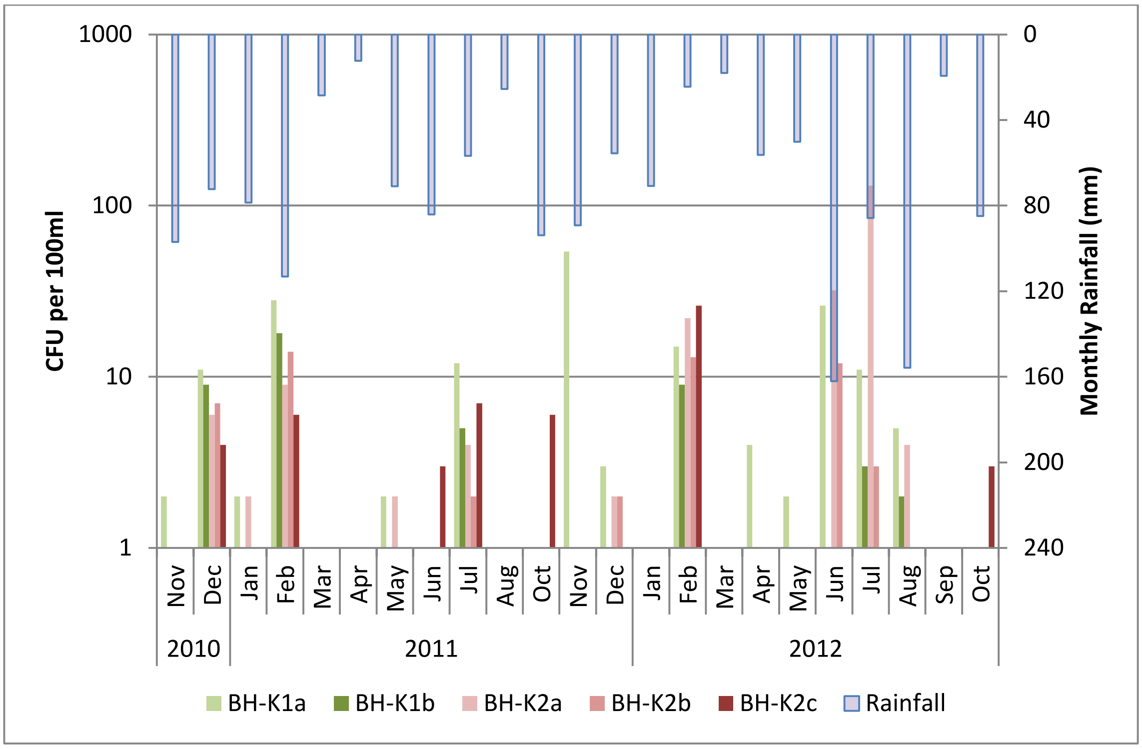


Figure 5.41 *Enterococci* levels at Carrigeen, Co. Kilkenny

Table 5.51 Average Enterococci concentrations at Carrigeen, Co. Kilkenny (all values in cfu/100ml)

Horizon	BH-K1	BH-K2
Horizon a	-	9.52
Horizon b	-	2.52
Horizon c	7.73	2.62

Table 5.52 ANOVA output for enterococci in horizon 3 at Carrigeen, Co. Kilkenny

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-K1a	23	335	14.56522	1734.621
BH-K1b	23	53	2.304348	11.31225
BH-K2c	23	24	1.043478	2.407115

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	2566.464	2	1283.232	2.201915	0.118651	3.135918
Within Groups	38463.48	66	582.78			
Total	41029.94	68				

Rainfall and Bacterial Analysis

Regression analysis using the rainfall intervals outlined previously gave a significant result for the 24hour rainfall interval for *E-coli* with a highly significant p-value of 0.007 (see Table 5.53), however no significant result was found for the *Enterococci* analysis.

Table 5.53 Regression analysis for E-coli and varying rainfall intervals (High vulnerability)

	Coeff	*SE Coeff*	*t Stat*	*P-value*
Intercept	-0.69655	3.880741333	-0.179489811	0.857886354
24hr	3.375116	1.239071607	2.723907368	0.007515751
48hr	0.560726	0.457777176	1.224888142	0.223257836
120hr	-0.73173	0.500714595	-1.461378352	0.146788519
21d	-0.02321	0.104175643	-0.22278936	0.824116402
30d	0.068834	0.084394872	0.815622837	0.416494613

The measured T-value for the subsoil at Carrigeen, Co. Kilkenny was 20mins/25mm which would then indicate a K_{fs} value for the subsoil of 0.21 m/d. The HYDRUS software calculated a lower K_s value for the subsoil of 0.14 m/d. During drilling the subsoil in the vicinity of the cluster of OSWTS's was found to be 5 m deep indicating an available depth of subsoil to the water table of 4.1 m and thus a travel time of c.19days for pollutants to enter groundwater. This is somewhat at odds with the above analysis indicating that the previous 24hours of rainfall is closely related to whether bacteria were found in the groundwater. It is possible that preferential flow paths have developed in the subsoil allowing contaminants to enter the groundwater. It is also possible that a more complicated subsoil/groundwater time lagged relationship is involved at this location and this will be investigated further in Chapter 7 during the unsaturated zone modelling. However, given that the site is deemed high vulnerability and given the steep gradients in the area it is likely that extreme 24hour rainfall events do impact on bacteria numbers in the aquifer.

5.4.4 Extreme Vulnerability Site – Faha, Co. Limerick

Nitrogen

Nitrate concentrations recorded during the study period ranged from 0.2 – 9.8 mg-N/l. Figure 5.42 below shows the average nitrate values observed in each quarter of the sampling period and the trend of higher values occurring during the winter months (Q4) which is consistent with previous studies where nitrogen in groundwater increases during winter months. As OSWTS's can be considered a constant year round load it is not likely that these peaks are being caused by the cluster development but rather by agricultural practices in the area particularly considering that this is an extreme vulnerability site and there should therefore be less lag time for pollutants entering the aquifer and less attenuation of any on-site pollutants before getting to groundwater. A plot of nitrate concentrations with rainfall is given in Figure 5.43 below. Average nitrate concentrations for the sampling period are given in Table 5.54.

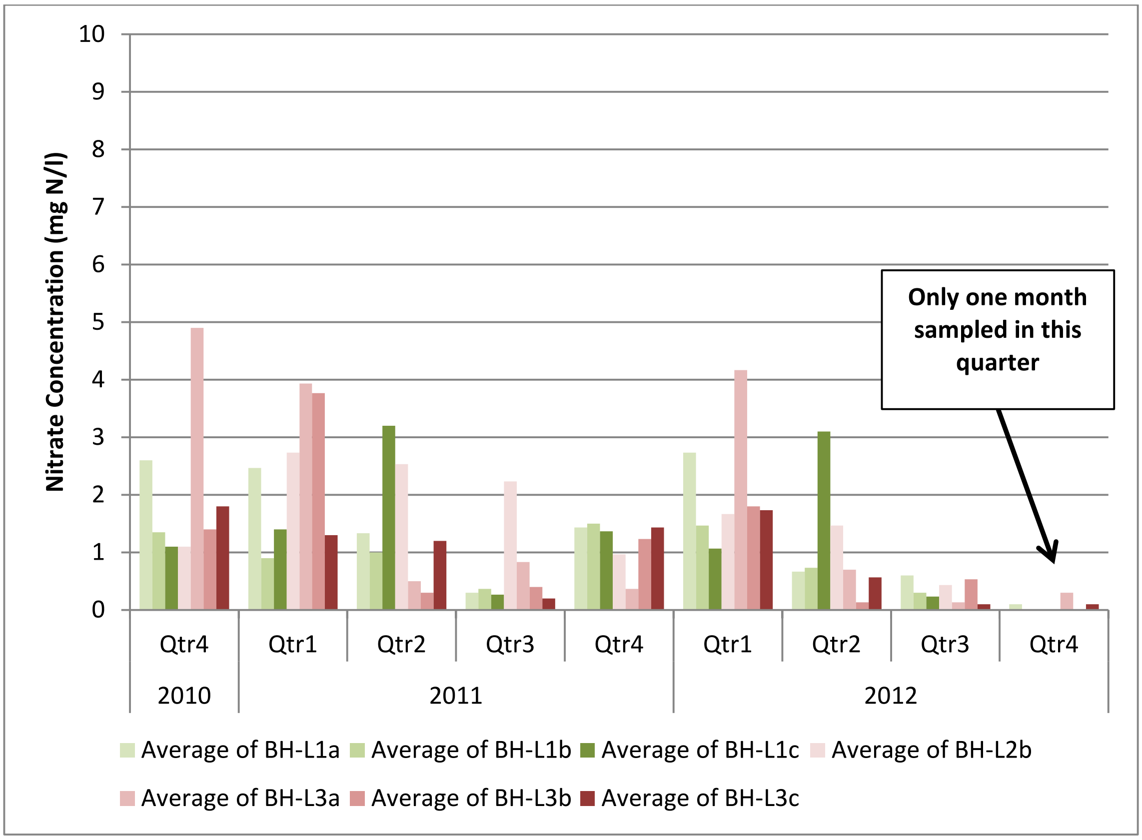


Figure 5.42 Seasonal nitrate concentrations at Faha, Co. Limerick

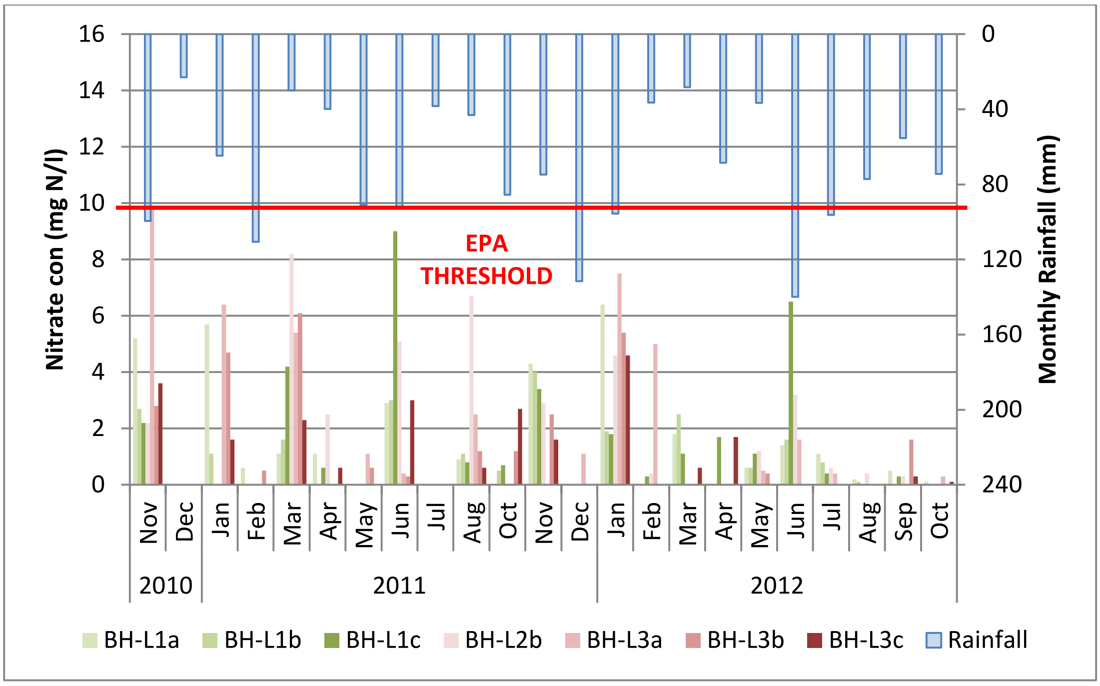


Figure 5.43 Nitrate concentrations with rainfall at Faha, Co. Limerick

Nitrite concentrations at Faha, Co, Limerick (extreme vulnerability) varied from 0.02 – 0.37 mg/l. Given that the EPA drinking water standard is 0.5 mg/l (as NO_2) it is clear that all of the recorded values are below this limit. It is notable however, that this is the only location where values of nitrite of any magnitude significantly above the detection limit were observed; although values higher than the detection limit only occurred on limited number of months during the study period. Elevated levels of nitrite in a groundwater sample indicate incomplete nitrification (in the subsoil) which might be either as a result of the particular geological conditions in the area (such as the subsoil properties) and/or relatively close pollution from a pathway perspective. This indicates that the pollutants are migrating into the groundwater aquifer very quickly which would be expected for at this location which is an extreme vulnerability site. A number of values were above the EPA/GSI trigger value of 0.1 mg/l as shown in Figure 5.44 below. Again, as for the nitrate concentrations, a clear trend of values occurring in the months November – March over two yearly periods can be seen.

Ammonium concentrations were generally below the detection limit during the study - however a number of months did record values above the limit. Ammonium concentrations over the study period together with rainfall are shown in Figure 5.45 below. No values exceeded the EPA limit of 0.3 mg N/l (as NH_4) however a number of values did exceed the EPA/GSI trigger value of 0.15 mg N/l. there is a weak indication that the peaks in recorded ammonium values occurred during months which had high rainfall quantities, however the seasonal trend seen with nitrite and nitrate is not as evident for ammonium.

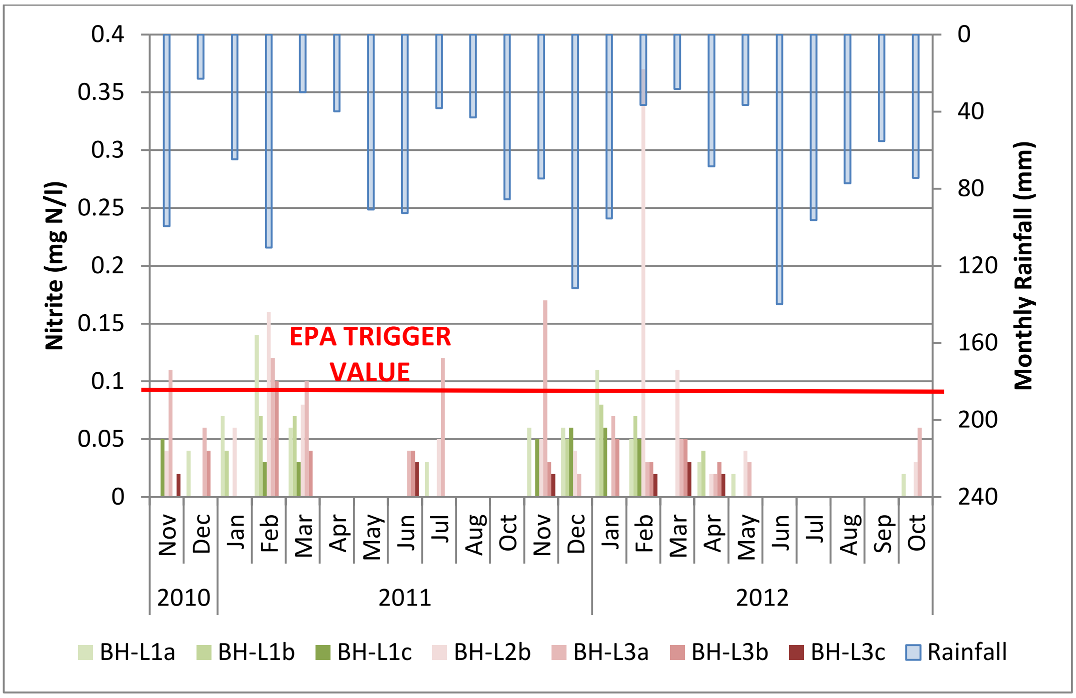


Figure 5.44 Nitrite concentrations with rainfall at Faha, Co. Limerick

Pearson's correlation analysis was carried out between nitrate, nitrite and ammonium concentrations as nitrite is an intermediary form of nitrogen in the process of NH_4 being converted to NO_3. A weak to moderate correlation was found between NO_2 and NH_4 with over 50% of the sampling horizons having a correlation coefficient of 0.4 or higher. Approximately 20% of sampling horizons had a Pearson correlation coefficient of greater than 0.5. These weak to moderate correlations between different forms of nitrogen indicates that at during times of the year when values of NH_4 and NO_2 were recorded there was either a large pollutant load added at the surface (such as slurry spreading or cattle grazing) or the pathway to groundwater was shortened in duration due to high rainfall. Only a very weak correlation between these values and monthly rainfall was indicated from the analysis and this is likely due to the fact that these higher values were likely caused by short intense rainfall events. Rainfall for the previous 24, 48, 120 hours periods together with the previous 21 and 30 day periods were summed and a regression analysis was carried out on these rainfall datasets with the various forms of nitrogen, as discussed previously for the microbial analysis. For both nitrite and ammonium highly significant p-values of 0.036 and 0.0004 were found through this analysis for the preceding 21day rainfall period indicating that the precious 21days rainfall dictated whether nitrite or ammonium would be found during this study. A similar result was not found for nitrates with no significant relationship between rainfall and nitrate found during the study period.

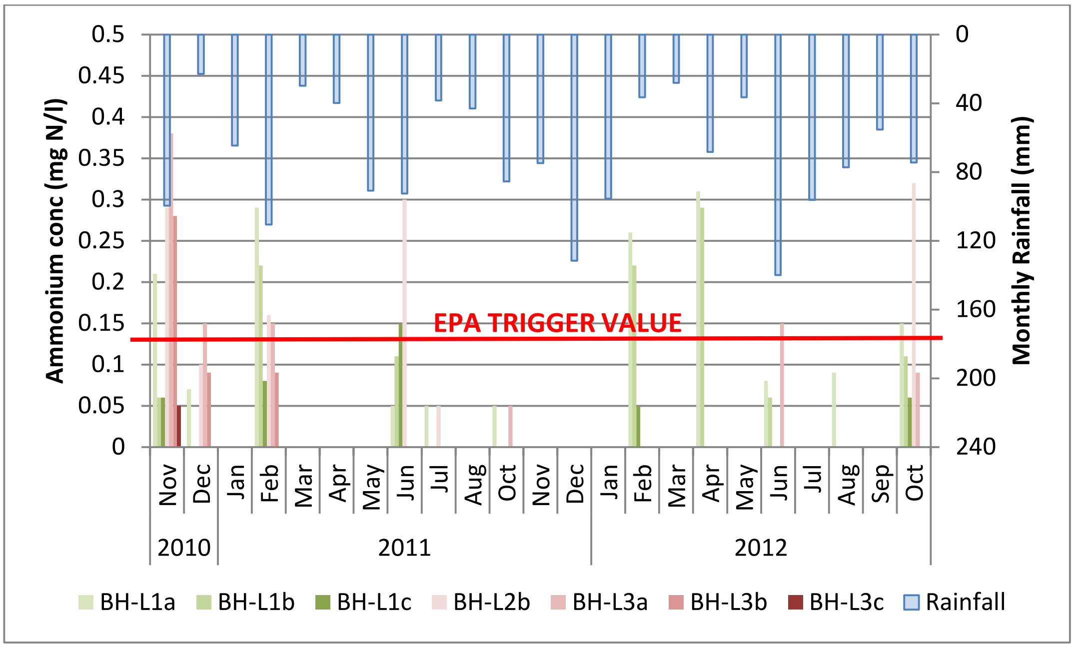


Figure 5.45 Ammonium concentrations with rainfall at Faha, Co. Limerick

Comparisons between mean concentrations for each of the forms of nitrogen in each of the sampling horizons at each borehole were carried out. A statistically insignificant result was found for BH-L1 for nitrite however a significant difference in sample means was found at BH-L3 for nitrite with a p-value of 0.00105 as shown in Table 5.55 below. The reason for these significant results can be seen in the summary table as mean nitrite concentrations can be seen to decrease with depth into the aquifer with BH-L3a having a mean nitrite concentration of 0.043 mg N/l and BH-L3c having a mean nitrite concentration of 0.006 mg N/l. a similar reduction in average nitrite concentrations was observed at BH-L1 with mean values reducing with depth from 0.03 mg N/l at BH-L1a to 0.014 mg N/l at BH-L1c however the p-value at BH-L1 was not significant at 0.216.

Table 5.54 Average Nitrate concentrations at Faha, Co. Limerick (all values in mg/l)

Horizon	BH-L1	BH-L2	BH-L3
Horizon a	1.47	-	1.82
Horizon b	0.93	-	1.18
Horizon c	1.48	1.66	1.01

Table 5.55 ANOVA output for nitrite at BH-L3 at Faha, Co. Limerick

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-L3a	23	1	0.043478	0.002506
BH-L3b	23	0.41	0.017826	0.000691
BH-L3c	23	0.14	0.006087	0.000116

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.01682	2	0.00841	7.618212	0.001055	3.135918
Within Groups	0.072861	66	0.001104			
Total	0.089681	68				

Similar results were found at BH-L1 and BH-L3 for mean ammonium concentrations with ANOVA p-values of 0.090 and 0.112 respectively, which would be significant at a less conservative significance level. Mean ammonium concentrations could be seen to reduce with depth into the aquifer at both locations as seen with mean nitrite concentrations. This is to be expected as indicating the process of nitrification with depth as described in detail in Chapter 2. Nitrates did not show significant differences in mean concentrations with depth into the aquifer when similar ANOVA tests were carried out with p-values of 0.534 at BH-L1 and 0.401 at BH-L3. This indicates that the values recorded during this study do not suggest any significant change in nitrate concentrations with depth in the aquifer which is somewhat surprising given that the ammonium and nitrite results suggest that nitrification was occurring with depth into the aquifer. Hence, this may also suggest that some denitrification of the nitrates was also occurring in parallel to the nitrification thereby acting to mute any increase in nitrate.

Similar statistical analysis between upstream and downstream mean concentrations of each of these forms of nitrogen at horizons 1 and 2 did not show any significant difference in mean nitrite concentrations; however the ANOVA test for horizon 3 resulted in a p-value of 0.021 which is significant as shown in Table 5.56 below. A further analysis was carried out which consisted of a two sample t-test comparing sample means for BH-L1 and BH-L2 only at horizon 3. This test produced a statistically insignificant result with a calculated test t-statistic of -1.76 which is a less extreme value than the critical value of ±2.059. It can be concluded therefore that mean nitrite concentrations at BH-L3c are significantly different from those at BH-L1c and BH-L2b and this was the reason for the ANOVA p-value of 0.021. However, it cannot be concluded that mean nitrite concentrations were higher downstream of the cluster development than upstream.

Table 5.56 ANOVA output for nitrite at BH-L3 at Faha, Co. Limerick

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-L1c	23	0.33	0.014348	0.000535
BH-L2b	23	1.05	0.045652	0.006717
BH-L3c	23	0.14	0.006087	0.000116

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	0.020038	2	0.010019	4.079779	0.021352	3.135918
Within Groups	0.162078	66	0.002456			
Total	0.182116	68				

Statistical comparisons of mean upstream and downstream ammonium concentrations resulted in very similar outcomes as those encountered for nitrite. Horizons 1 and 2 were not significantly different from each other however horizon 3 had an ANOVA p-value of 0.031. As with the nitrite analysis above, a further two sample t-test indicated that BH-L1c and BH-L2b were not significantly different from each other and again it must be concluded that downstream mean ammonium concentrations are not significantly different from those upstream of the study cluster development.

Statistical tests comparing upstream and downstream mean nitrate concentrations resulted in statistically insignificant results for all of the sampling horizons. It is therefore concluded that as with both ammonium and nitrite, mean nitrate concentrations downstream of the study cluster development are not significantly different from each other. Overall nitrogen concentrations downstream of the study site do not appear to be affected by the presence of the cluster of OSWTS's.

Phosphorus

At Faha, Co. Limerick, total dissolved phosphorus concentrations generally were in the range 0.025 – 0.1 mg-P/l with the majority monthly samples above the threshold value of 0.035 mg/l as shown in Figure 5.46 below. There appears to be a weak trend of concentrations of total dissolved phosphorus increasing with lower rainfall and reducing during periods of high rainfall and the Pearson correlation coefficients confirmed this yielding values of between -0.29 and -0.72, suggesting a moderately negative correlation.

Comparisons between mean concentrations with depth at each of the boreholes resulted in statistically insignificant outcomes indicating that phosphorus concentrations did not vary significantly with depth. Equally, no significant results were found between upstream and downstream mean phosphorus concentrations across the 3 sampling horizons. The cluster of OSWTS's does therefore not seem to be adversely affecting phosphorus concentrations in the surrounding aquifer.

Table 5.57 Average Phosphorus concentrations at Faha, Co. Limerick (all values in mg/l)

Horizon	BH-L1	BH-L2	BH-L3
Horizon a	0.079	-	0.078
Horizon b	0.078	-	0.071
Horizon c	0.075	0.054	0.069

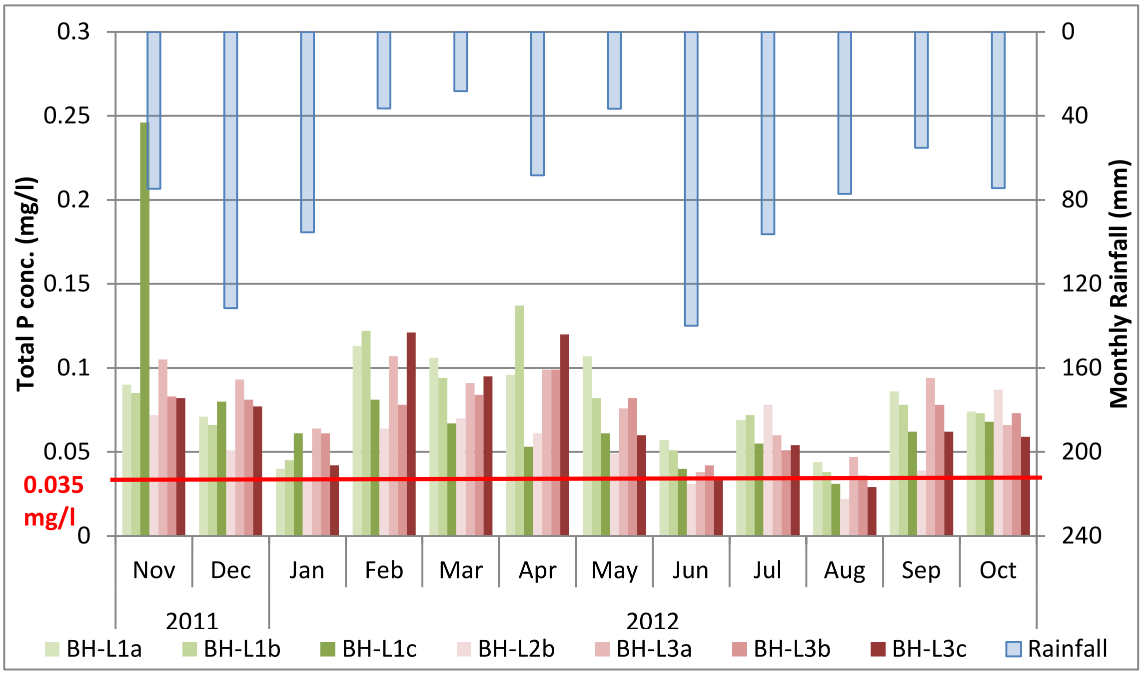


Figure 5.46 Total Phosphorus concentrations with rainfall at Faha, Co. Limerick

Chloride

Chloride concentrations over the period of the study together with rainfall are shown in Figure 5.47 below, with average concentrations for the study period given in Table 5.58. A seasonal trend can be seen however the Pearson correlation coefficients were in the range of 0.2 – 0.3 indicating only a weak correlation of chloride with rainfall. Tests between means at the different horizon depths did not yield a significant result at any of the borehole locations and neither when comparing upstream and downstream chloride concentrations. It is concluded therefore that mean chloride concentrations were not higher downstream of the study development.

Table 5.58 Average Chloride concentrations at Faha, Co. Limerick (all values in mg/l)

Horizon	BH-L1	BH-L2	BH-L3
Horizon a	33	-	35
Horizon b	33	-	34
Horizon c	32	30	32

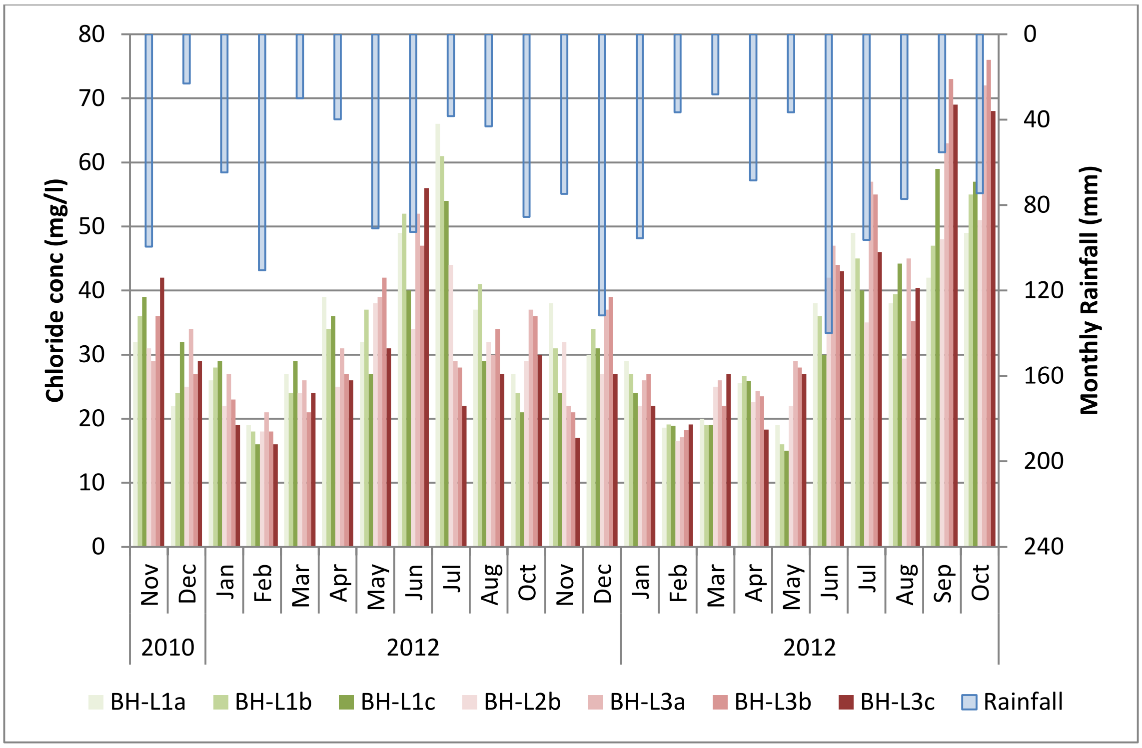


Figure 5.47 Chloride concentrations with rainfall at Faha, Co. Limerick

Chloride/bromide ratios for Faha are shown in Table 5.59 below. No elevated Cl/Br ratios were observed during the study. As was seen at the *MODERATE* and *HIGH* vulnerability sites there is a rise in the ratios from August to September.

Table 5.59 Bromide ratios for Faha Co. Limerick

Date	Sample	Bromide (mg/l)	Cl/Br Mass Ratio
Aug-12	BH-L3a	0.304	148
	BH-L3b	0.269	131
	BH-L3c	0.251	161
	BH-L1a	0.332	114
	BH-L1b	0.319	139
	BH-L1c	0.217	135
Sep-12	BH-L3a	0.507	83
	BH-L3b	0.307	153
	BH-L3c	0.249	237
	BH-L1a	0.335	188
	BH-L1b	0.418	175
	BH-L1c	0.414	167

E-coli

Occurrences of *E-coli* bacteria at Faha, Co. Limerick, shown in Figure 5.48 below, indicate clear peaks during two distinct periods of the year; March – April and August – October. Peaks in average values, ranging from 10 – 60 cfu/100ml, can be seen during these two periods of the year. Given the temporal nature of the peaks and the likelihood that contaminants from OSWTS to provide a more constant load to groundwater it will be difficult to attribute these peaks directly to OSWTS's in the area. However, although the loading arising from OSWTS's is likely to be fairly constant, the pathway and corresponding attenuation processes en route to the groundwater will be changing temporally. Soil moisture is higher during winter months therefore creating higher recharge rates during intense rainfall events (thus "pushing" the pollutants faster through the subsoil). In addition soil temperatures are lower during winter months, which again acts to dampen out attenuation processes. Overall it is likely that these peaks are due to agriculture, although there is not enough evidence to completely rule out OSWTS's. Table 5.60 below gives average *E-coli* values during the study period.

Table 5.60 Average E-coli concentrations at Faha, Co. Limerick (all values in cfu/100ml)

Horizon	BH-L1	BH-L2	BH-L3
Horizon a	1.82	-	1.81
Horizon b	1.86	-	16.60
Horizon c	16.17	2.47	35.60

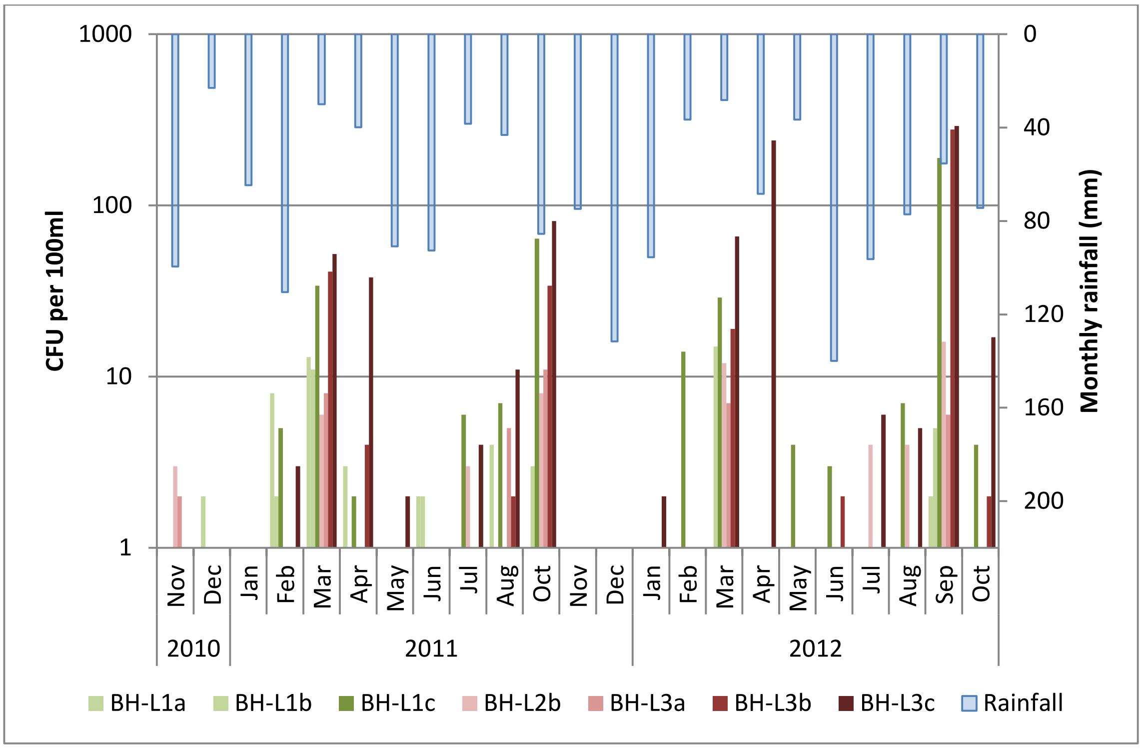


Figure 5.48 E-coli levels at Faha, Co. Limerick

The test of statistical significance between means at the different horizon depths did not yield a significant result at any of the borehole locations. However, at BH-L1 the ANOVA tests gave a p-value of 0.066 which would be significant at a less conservative significance level. The ANOVA output, shown in Table 5.61 below, indicates that mean E-coli numbers are increasing with depth into the aquifer which is the opposite of what was found at the high vulnerability site. Similar increases in mean E-coli numbers with depth into the aquifer can be seen at BH-L3 however again the ANOVA test did not result in a significant difference (p = 0.125) mainly due to the large variances that are seen at all of the sampling horizons. Comparing between upstream and downstream E-coli numbers were also carried out at the 3 sampling horizons. These tests did not give any significant results. The ANOVA p-value for horizon 3 was 0.086 as shown in Table 5.63 below. BH-L1a has mean

E-coli numbers of 16.17; BH-L2b has mean E-coli numbers of 2.47 and BH-L3c has mean E-coli numbers of 35.61. If a two sample t-test is used to compare only BH-L1c and BH-L2b a significant results is obtained with BH-L1 having a higher mean. Given that BH-L2 is the first downstream borehole located some 200 m closer to the cluster of OSWTS's than BH-L3, it must be concluded that overall downstream E-coli numbers are not higher downstream due to the cluster development. It is possible however that BH-L2 may in fact not be located in the plume of pollutants arising from the cluster of OSWTS's (if one exists) and in this case the plume might be "by-passing" BH-L2. This will be studied in more detail using tracking processes such as MODPATH in the groundwater modelling discussed in Chapter 8.

Table 5.61 ANOVA output for E-coli at BH-L1 at Faha, Co. Limerick

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-L1a	23	42	1.826087	9.059289
BH-L1b	23	43	1.869565	14.3004
BH-L1c	23	372	16.17391	1645.514

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	3146.986	2	1573.493	2.828542	0.066282	3.135918
Within Groups	36715.22	66	556.2912			
Total	39862.2	68				

Table 5.62 ANOVA output for E-coli at BH-L3 at Faha, Co. Limerick

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-L3a	23	42	1.826087	10.24111
BH-L3b	23	382	16.6087	3371.158
BH-L3c	23	819	35.6087	5849.431

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	13192.72	2	6596.362	2.143804	0.1253	3.135918
Within Groups	203078.3	66	3076.943			
Total	216271	68				

Table 5.63 ANOVA output for E-coli in horizon 3 at Faha, Co. Limerick

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-L1c	23	372	16.17391	1645.514
BH-L2b	23	57	2.478261	18.62451
BH-L3c	23	819	35.6087	5849.431

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	12748.96	2	6374.478	2.545186	0.086138	3.135918
Within Groups	165298.5	66	2504.523			
Total	178047.5	68				

Enterococci

Occurrences of Enterococci bacteria at Faha, Co. Limerick over the study period are shown in Figure 5.49 below with average values for the study given in Table 5.64. Again, peaks during two distinct periods of the year; March – April and August – October can again be seen as was the case for E-coli above. There does not appear to be any clear trend between the peaks and rainfall quantities and this is confirmed with Pearson coefficients in the range of 0.18 – 0.33.

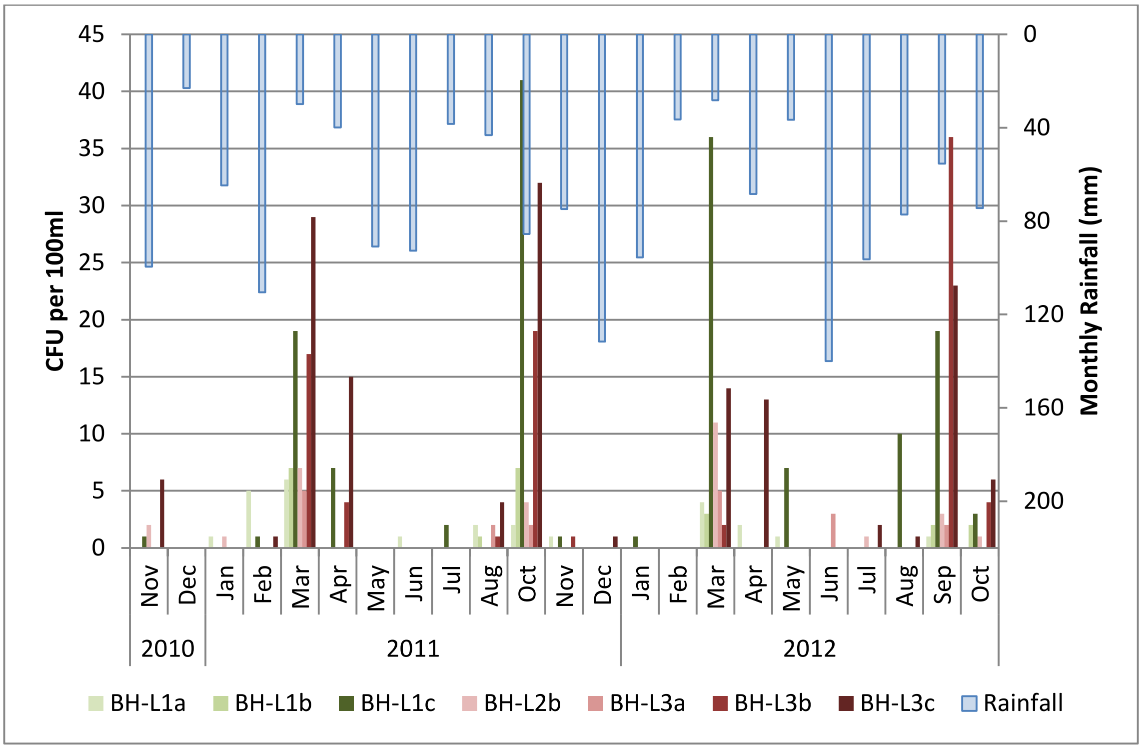


Figure 5.49 *Enterococci* levels at Faha, Co. Limerick

Table 5.64 Average Enterococci concentrations at Faha, Co. Limerick (all values in cfu/100ml)

Horizon	BH-L1	BH-L2	BH-L3
Horizon a	1.13	-	0.82
Horizon b	0.95	-	3.65
Horizon c	6.43	1.34	6.39

Analysis between means at the different horizon depths gave significant results at both BH-L1 and BH-L3 as shown in Table 5.65 and Table 5.66 below. Again Enterococci numbers tended to increase with depth into the aquifer which is similar to the findings for E-coli but at odds with what was found at the high vulnerability site.

Table 5.65 ANOVA output for *Enterococci* at BH-L1 at Faha, Co. Limerick

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-L1a	23	26	1.130435	2.936759
BH-L1b	23	22	0.956522	4.316206
BH-L1c	23	148	6.434783	134.6206

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	446.029	2	223.0145	4.715774	0.012181	3.135918
Within Groups	3121.217	66	47.29117			
Total	3567.246	68				

Table 5.66 ANOVA output for *Enterococci* at BH-L3 at Faha, Co. Limerick

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-L3a	23	19	0.826087	2.513834
BH-L3b	23	84	3.652174	76.23715
BH-L3c	23	147	6.391304	97.24901

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	356.2029	2	178.1014	3.03582	0.054793	3.135918
Within Groups	3872	66	58.66667			
Total	4228.203	68				

Statistical tests comparing upstream and downstream *Enterococci* numbers were also carried out at the 3 sampling horizons which did not give any significant results. The ANOVA p-value for horizon 3 was 0.089 as shown in Table 5.67 below. This is again similar to what was calculated for *E-coli* using the same tests. However BH-L1a has mean *Enterococci* numbers of 6.43; BH-L2b has mean *Enterococci* numbers of 1.3 and BH-L3c has mean *Enterococci* numbers of 6.39. If a two sample t-test is used to compare only BH-L1c and BH-L2b a significant results is obtained with BH-L1 having a higher mean. Again as was seen with results for *E-coli* it must be concluded that overall downstream *E-coli* numbers are not higher downstream due to the cluster development.

Table 5.67 ANOVA output for Enterococci in horizon 3 at Faha, Co. Limerick

SUMMARY

Groups	*Count*	*Sum*	*Average*	*Variance*
BH-L1c	23	148	6.434783	134.6206
BH-L2b	23	30	1.304348	7.403162
BH-L3c	23	147	6.391304	97.24901

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Between Groups	400.2029	2	200.1014	2.508871	0.089093	3.135918
Within Groups	5264	66	79.75758			
Total	5664.203	68				

<u>Rainfall and Bacterial Analysis</u>

Regression analysis using summed daily rainfall intervals gave a significant result for the 24 and 48 hour rainfall periods for both *Enterococci* and *E-coli* with highly significant p-values of 0.0001 and 0.006 for the 48 hour intervals (see Table 5.68 and Table 5.69).

Table 5.68 Regression analysis for *Enterococci* and varying rainfall intervals (Extreme vulnerability)

	Coeff	*SE Coeff*	*t Stat*	*P-value*
Intercept	4.043633074	1.423718509	2.840191	0.005114
24hr	-0.551882077	0.237248556	-2.32618	0.021305
48hr	0.727557453	0.189913373	3.830996	0.000185
120hr	-0.074720894	0.13689219	-0.54584	0.585962

21day	-0.018137443	0.042018908	-0.43165	0.666596
30day	-0.030117464	0.038301612	-0.78632	0.432878

Table 5.69 Regression analysis for E-coli and varying rainfall intervals (Extreme vulnerability)

	Coeff	*SE Coeff*	*t Stat*	*P-value*
Intercept	17.37239964	8.488527786	2.046573927	0.042389
24hr	-3.736025346	1.414528887	-2.641179958	0.009108
48hr	3.111534977	1.132305955	2.747963094	0.006708
120hr	-0.239385662	0.816181814	-0.293299432	0.769686
21day	0.197463982	0.250526118	0.788197187	0.431785
30day	-0.30931287	0.228362765	-1.354480319	0.177554

The measured P-value at Faha, Co. Limerick was 39 mins/25mm however the equation by Mulqueen and Rodgers (2001) can only be used converting T-values to K_s and therefore cannot be applied here. The Hydrus software calculated a K_s value of 0.11 m/d for the subsoil. As discussed previously it would appear from a visual survey of the area that the majority of sites in the area have raised their percolation areas in order to accommodate OSWTS's and drainage thus the average depth of subsoil between the bottom of percolation trenches and the water table in the area cannot be estimated accurately. However, for the bacteria to reach the water table in 48 hours the K_s value calculated by Hydrus would indicate 0.3 m would be required for this travel time which is unlikely to be the case. However, given that this is an extreme vulnerability area with shallow bedrock present, a 48 hour response time to rainfall events would seem appropriate for "pushing" bacteria that may have been migrating slowly through the subsoil prior to the event.

5.4.5 Extreme Vulnerability Karst Site – Toonagh, Co. Clare

Tracer Study

Before any study could commence at Toonagh Co. Clare it was first required to establish whether the upstream sink and downstream spring was hydraulically connected via a karst conduit network system in the area (thought to be the Toreen east stream re-emerging) and also whether the treatment plant at Toonagh was also connected to this system which emerges at the Kilcurrish spring. A number of attempts were made to confirm the link between the upstream sink and the Kilcurrish spring using fluorescein as a tracer. It was estimated that the time of travel was 6 – 12 hours and therefore 50 ml and 150 ml were added to the Torren east stream (see Figure 5.50) prior to it entering the underground system on two separate occasions. Concentrations of fluorescein were recorded every 30 minutes downstream using a handheld meter over two 12 hours periods but the tracer was not observed downstream on either occasion.

Figure 5.50 Torren east stream upstream of the sink following the injection of fluorescein tracer - May 2012

Following these two unsuccessful attempts, consultation was taken with available GSI and Dr David Drew a local expert in karst hydrology. It then established that a previous tracer test had been carried out at this location and a connection had been established. The time of travel had been between 24 and 36 hours which is much greater than had been initially estimated. Given that this connection had already been established and was entered into the GSI records for the area it was decided not to attempt any further studies on this connection and attention was turned to the connection between the treatment plant at Toonagh. Given that the estimated time of travel was up to 40 hours it was decided to use Rhodamine WT as the tracer for this second study the probe available could be used in conjunction with a field datalogger. The probe was calibrated in the laboratory and deployed at the downstream spring together with a power supply and the datalogger. A charcoal bag was also deployed at the Kilcurrish spring in case of any malfunction with the logging equipment. Initially the estimated calculation of dilution in the aquifer determined that 150 ml of Rhodamine dye should be injected at the treatment works at Toonagh. The dye was added to the outflow tank of the treatment works. The initial attempt was unsuccessful with no concentrations measured downstream using either the probe or the charcoal bag. A second attempt was undertaken on the 27/6/2012 this time with 1000 ml of the dye being injected. This attempt was successful and Rhodamine was detected downstream using both recording methods. The breakthrough curve obtained from the datalogger is shown in Figure 5.51.

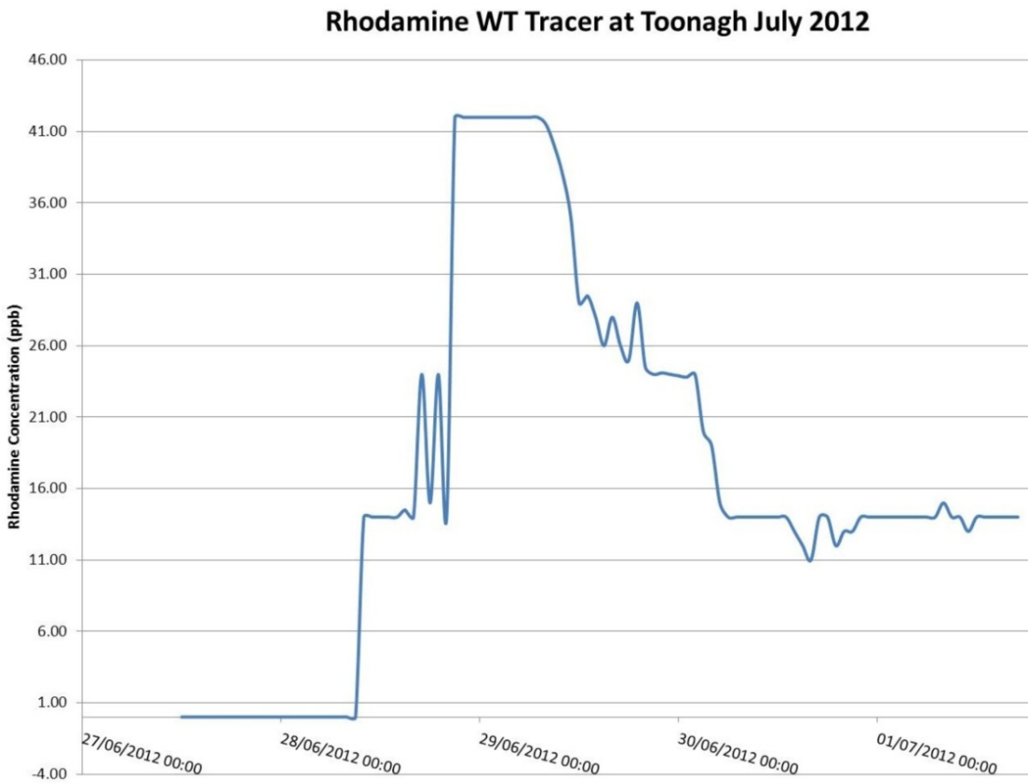


Figure 5.51 Breakthrough curves obtained from Rhodamine Tracer Test undertaken In July 2012

The maximum concentration detected downstream was 41 ppb. The breakthrough curve does not follow the usual shape of those obtained for similar studies and this is likely due to the location at which the dye was added. The dye was added to the outflow holding tank of the treatment works at Toonagh which would have slowly released the dye into the aquifer at a steady rate over an extended period of time (as opposed to the more usual spiked input used for a tracer study). A charcoal bag that was deployed at the downstream monitoring location also provided a qualitative positive result for the Rhodamine dye. A charcoal bag was also deployed upstream of the dye injection point at the sink to provide a ‘control’. Both charcoal bags following soaking in the eluting solution are shown in Figure 5.52 below.

Figure 5.52 Charcoal Bags located downstream (left) and upstream (right) showing tracer removed by eluting solution

Nitrogen

Nitrite and ammonium concentrations were negligible and below detection limits during the study period. Nitrate concentrations at Toonagh, Co. Clare ranged from 0.4 – 2.8 mg-N/l over the sampling period and are shown in Figure 5.53 below. There appears to be a weak trend of higher rainfall quantities and higher nitrate concentrations but again the sampling period is not of sufficient duration to make a definitive conclusion. Whilst mean nitrate concentrations were higher downstream the test did not find a significant difference between the two means (see Table 5.70 below). It is not clear if the observed increases could be specifically attributable to the treatment plant at Toonagh, however it is likely that it having some impact on overall nitrate concentrations in the aquifer given that the tracer test proved a connection between the conduit and the treatment plant.

Table 5.70 T-test output for the parameter Nitrate at Toonagh, Co. Clare (t-Test: Two-Sample Assuming Unequal Variances)

	Upstream	*Downstream*
Mean	0.8	1.344444444
Variance	0.2975	0.517777778
Observations	9	9
Hypothesized Mean Difference	0	
df	15	
t Stat	-1.808931089	

P(T<=t) one-tail	0.045273193
t Critical one-tail	1.753050356
P(T<=t) two-tail	0.090546387
t Critical two-tail	2.131449546

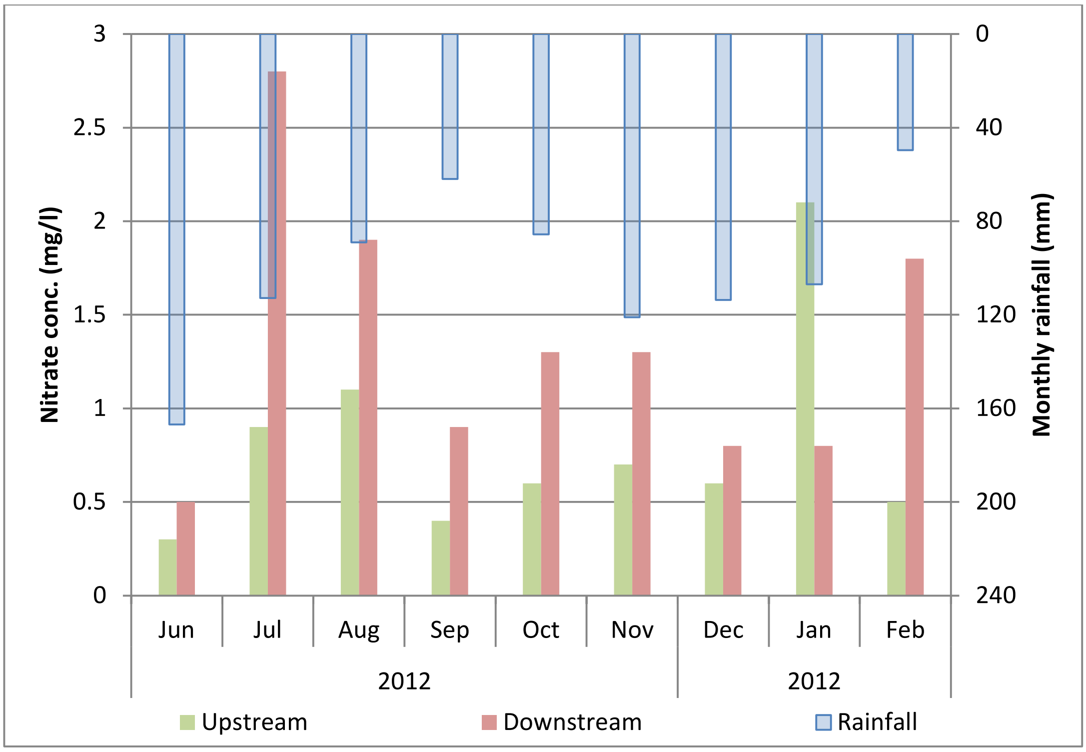


Figure 5.53 Nitrate concentrations with rainfall at Toonagh, Co. Clare

An 8 hour study of water chemistry parameters was undertaken on the 18th of January 2013 in order to investigate whether any diurnal pattern in nutrients or bacteria could be discerned. Samples were taken at intervals over the course of the day and analysed in the laboratory. Unfortunately the chosen day to undertake the study followed a localised 36 hour period of extreme intensity rainfall and the stream was flooded both upstream and downstream of the treatment plant. Nitrate concentration variation over the day is shown in Figure 5.54. Upstream concentrations were significantly higher at the beginning of the day which was likely to be associated with agricultural surface runoff being swept into the stream during the intense rainfall particularly in the upstream Namurian shale bedrock area. GIS information was used in Chapter 6 to delineate the extent of this allogenic catchment (water from a different bedrock source) and it was found to contribute a significant proportion of overall flow in the Toreen east stream – more details are given in Chapter 6. Over the course of the day upstream and downstream concentrations dropped and by the end of the sampling period the downstream concentrations were higher than upstream concentrations – a combination of the time lag in water travelling through the karst bedrock system and the more 'normal' baseflow water quality conditions returning to the system.

In general the indications are that overall water quality in the area with respect to nitrogen is good with low concentrations monitored over the entire sampling period. Given the fact that the treatment plant is connected directly into the karst conduit system, it does not appear to be having a very significant impact on overall nitrogen concentrations in the aquifer. The 8 hour study does highlight that the system is highly vulnerable in terms of susceptibility to pollutants being washed into the system during intense rainfall and the travel time between the sink and spring during these high flow events. And whilst this is not causing significant issues in terms of nitrogen it may cause hazards in terms of pathogens which will be discussed later.

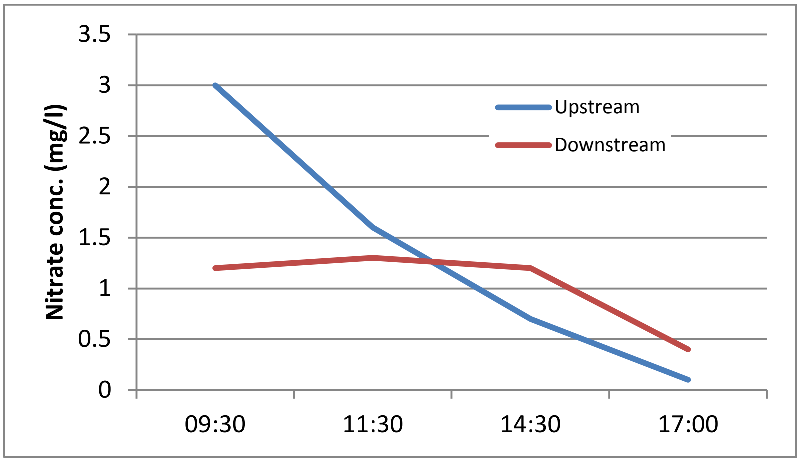


Figure 5.54 Nitrate concentrations over an 8 hour period at Toonagh, Co. Clare on 18/01/13

Phosphorus

Concentrations of total dissolved phosphorus at Toonagh, Co. over the duration of the monitoring period are shown in Figure 5.55 below. Upstream concentrations can be seen to be higher than downstream concentrations and this is confirmed by the significant result of the two sample t-test shown in Table 5.71 below.

Table 5.71 T-test output for the parameter Nitrate at Toonagh, Co. Clare (t-Test: Two-Sample Assuming Unequal Variances)

	Upstream	*Downstream*
Mean	0.086222222	0.029777778
Variance	0.000346694	9.54444E-05
Observations	9	9

Hypothesized Mean	
Difference	0
df	12
t Stat	8.053100145
P(T<=t) one-tail	1.75672E-06
t Critical one-tail	1.782287556
P(T<=t) two-tail	3.51345E-06
t Critical two-tail	2.17881283

Again this is likely due to surface runoff influencing the upstream concentrations with downstream concentrations being diluted due to mixing with groundwater in the karst bedrock aquifer system. The differences in phosphorus concentrations can also be attributed to the varying bedrock types. The upstream area will comprise of water derived from the Namuruan Shale whereas the downstream area comprises of water from Limestone bedrock origins. It is possible therefore that the trends seen can be explained from the geological conditions in the two separate catchments. Downstream phosphorus concentrations tended to generally be below the limit of 0.035 mg-P/l but did stray above this limit on a number of occasions over the monitoring period.

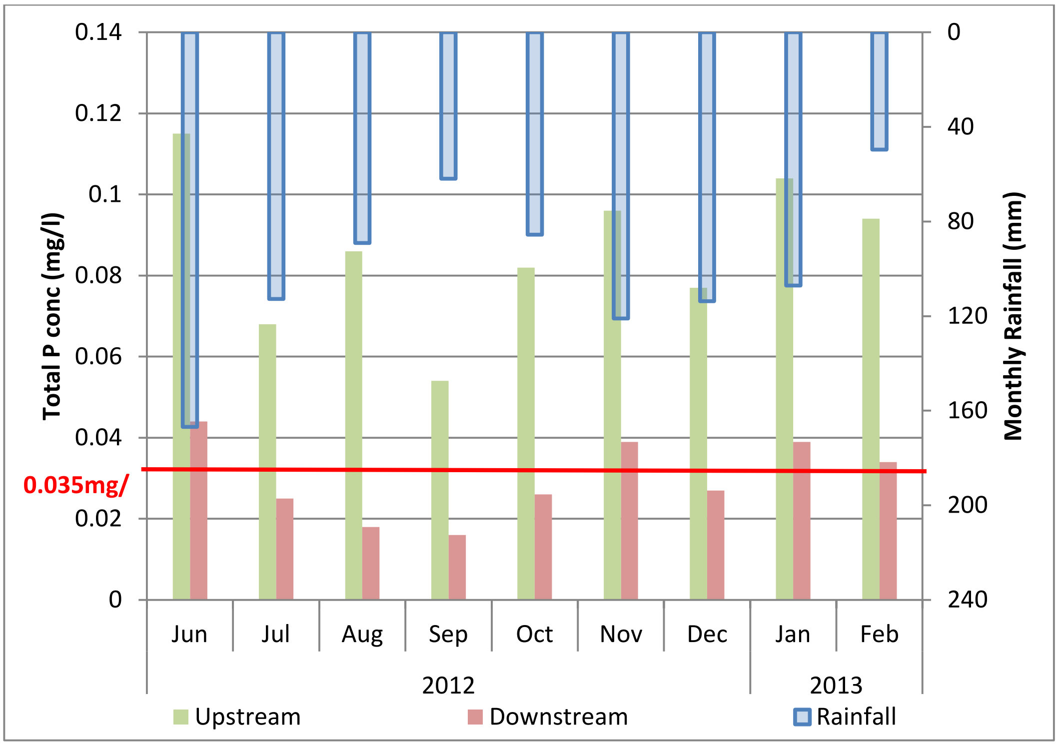


Figure 5.55 Total Phosphorus concentrations with rainfall at Toonagh, Co. Clare

Ortho-phosphate concentrations were measured on samples over an eight hour period on the 18th of January 2013 – see Figure 5.56. Ortho-phosphate and not total dissolved phosphorus was measured as it was expected that concentrations would be above the ascorbic acid test method upper limit of c.1mg-P/l For this reason it was not possible to determine an accurate phosphate concentration for the final downstream sample as it was below the test method lower limit of 0.1 mg-P/l PO_4. However, a trend of lowering phosphate concentrations over the day can be observed as the intense rainfall flood water levels dissipated.

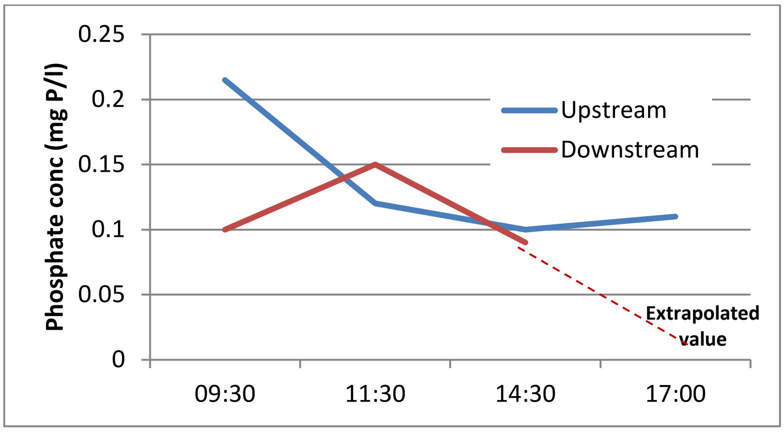


Figure 5.56 Ortho-phosphate concentrations over an 8 hour period at Toonagh, Co. Clare on 18/01/13

<u>Chloride</u>

At Toonagh, Co. Clare chloride values, given in Figure 5.57 below, ranged from 19 to 47 mg/l. A two sample t-test was carried out to compare upstream and downstream mean values and the test produced a statistically insignificant output.

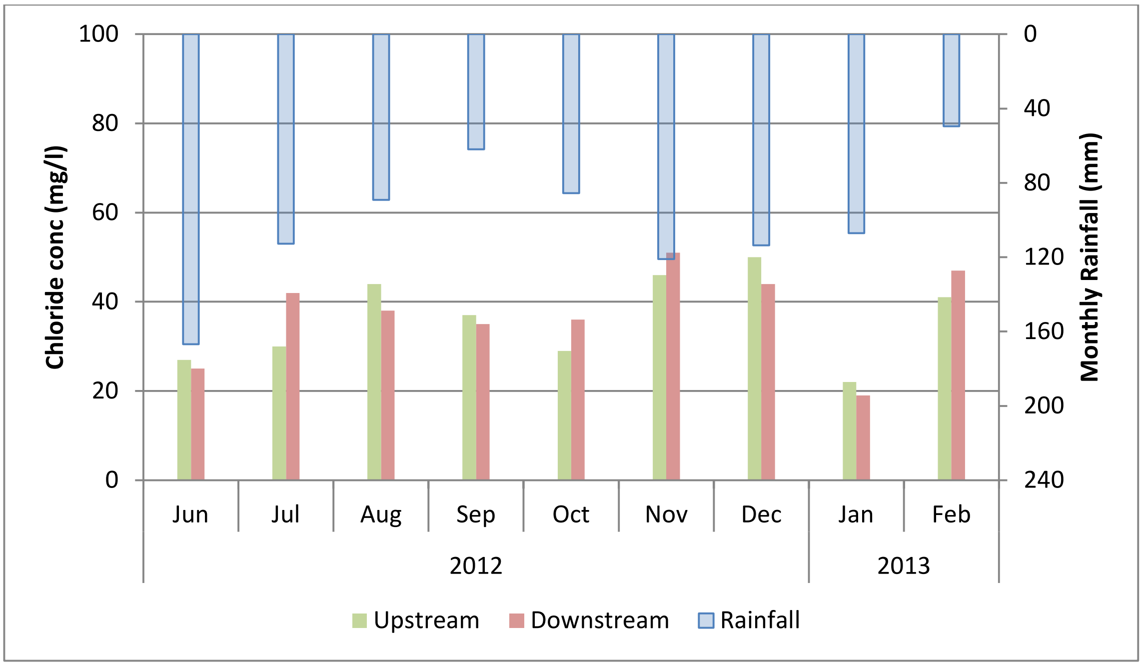


Figure 5.57 Chloride concentrations with rainfall at Toonagh, Co. Clare

All chloride/bromide ratios for Toonagh were inside the 400 – 1100 range (see Table 5.72). Given that the calculated ratios for Toonagh are all consistently well above the threshold there is strong evidence to suggest anthropogenic influences on the water quality. However, given that the ratios are higher upstream than downstream of the treatment plant, the elevated ratios cannot be attributed to the treatment plant at Toonagh but must be caused by some other source of pollution upstream of the sink.

Table 5.72 Chloride/Bromide ratios observed during the project at a number of monitoring locations

Date	Sample	Bromide (mg/l)	Cl/Br Mass Ratio
Nov-12	Toonagh U/s	0.053	863
	Toonagh D/s	0.064	797
Dec-12	Toonagh U/s	0.081	617
	Toonagh D/s	0.090	491

E-coli* and *Enterococci

Measured occurrences of both groups of bacteria monitored during this study at Toongah are given in Figure 5.58 below. Values recorded in January 2013 (which went up to 1100 CFU per 100ml upstream and downstream) during a period when the entire groundwater system in the area was completely flooded for a number of days due to very intense rainfall.

It is suspected that agricultural effluent was being washed into the system was mainly responsible for the huge peaks in observed values during the sampling visit in January 2013. This event does show how sensitive the groundwater quality is to heavy rainfall in such a kart network. For the other months of the year upstream *E-coli* and *Enterococci* numbers were higher for all months than the observed downstream values. A two sample t-test indicated that whilst the downstream sample mean is higher, there was no significant difference between the two means.

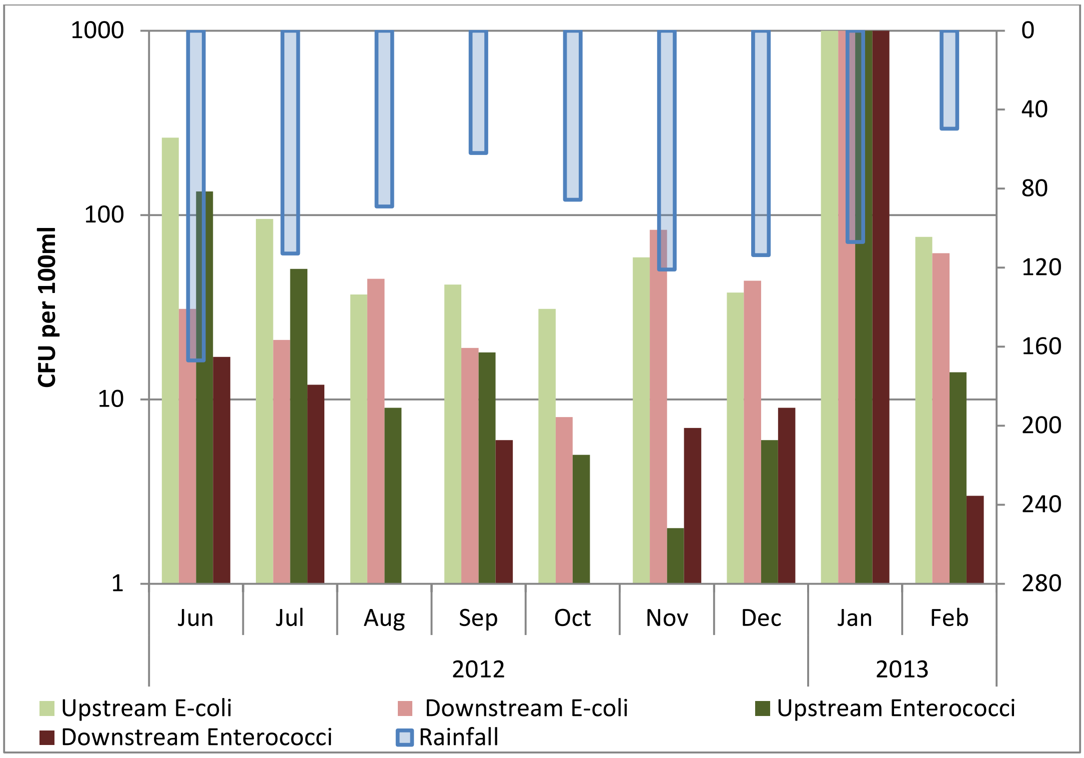


Figure 5.58 E-coli and *Enterococci* levels at Toonagh, Co. Clare

A two sample t-test carried out between upstream and downstream enterococci numbers also yielded a statistically insignificant result. It is therefore concluded that whilst the treatment plant at Toonagh is connected directly to the karst groundwater system, no discernible increases in bacteria downstream can be attributed to it.

<u>Treatment Plant Loading</u>

Samples were taken from the treatment plant at Toonagh in order to access the possible impacts of its direct connection to the groundwater system. A summary of the main water quality indicator parameters found is given below:

COD 60- 85mg/l

N

O_2 0.95 mg-N/l

PO_4 4.63 mg-

P/l

N

H_4 2.24 mg-N/l

NO_3 7.3 mg-

N/l

Cl /

Br 219 /

0.32mg/l

(Cl/

Br 684)

E-coli 2 x10^5 CFU/100ml (MPN) *Enterococci* 2 x10^4 CFU/100ml

Given the details above it can be seen that the treatment plant is only partially nitrifying the effluent water. In addition the microbial results appear higher than what would be expected and this is likely due to the age and condition of the treatment plant. Assuming a daily flow of 150 l/d/PE and given that the plant is approximately 60PE the treatment plant is therefore producing approximately 9 m^3/day. Given the low flow measured in the Toreen East stream was approximately 10 l/s (864m^3/day) upstream and 14 l/s (1210m^3/day) it can be seen that even during low flow conditions the treatment plant is only contributing c.1% of total daily discharge. With this level of dilution all of the water quality parameters listed above would not be significant enough to have any measurable impact on the groundwater system with the exception of the indicator bacteria which are high enough to causes water quality issues even with the expected levels of dilution involved. However, as no evidence of elevated *E-coli* or *Enterococci* numbers was seen when comparing upstream and downstream flows, it is likely that further bacterial attenuation is occurring between the holding tank at the plant and the Kilcurrish spring through natural die-off as well as possibly some other processes such as sorption and predation.

5.5 OSWTS Density, Aquifer Vulnerability and Groundwater Quality

The overall aim of this project is to compare the impact of cluster systems between varying groundwater vulnerabilities and the resulting observed groundwater quality parameters. The previous sections compared each of the study sites statistically between upstream and downstream monitoring points in order to assess the downstream impact on groundwater quality due to cluster developments with OSWTS's. This section will now use statistical tests to compare between each of the study sites for each of the monitored chemical and bacterial indicator parameters. This will be achieved using two factor ANOVA with replication. The data setup for this analysis will treat the replication as the four groundwater vulnerability ratings and each of the parameters will then be assessed separately. The second factor in these ANOVA tests will be the upstream or downstream borehole locations and the test will also check for interaction between the two factors (i.e. between the borehole locations and the vulnerability) in order to see if the two vary together. Random variation between borehole sampling visits at each study site will be treated as error.

5.5.1 Nitrates

Results of two-factor ANOVA analysis of nitrate concentrations between the four study sites are shown in Table 5.73 below. It can be seen that there is a highly significant result for both borehole location and between the different site vulnerabilities. An examination of the means for each of the different vulnerability ratings shows an increase in nitrate concentrations from the *LOW* to *HIGH* vulnerability areas with the exception of the *Extreme* vulnerability site which appears to behave differently (due presumably to the complicated groundwater flow regime in the area which is karstic limestone with strong tidal influences). In addition, the high recharge rates (given in the GSI database) in the area likely combines to flush the aquifer on very frequent intervals thus muting the effects of any pollutants entering the system. Given the analysis already completed at each of the sites individually, it is likely that these nitrate increases with vulnerability are a result of increased pollutants getting down into groundwater mainly from agriculture but also with some contribution from OSWWTs given the shallow subsoils.

Table 5.73 ANOVA results for statistical comparison between study sites for the groundwater indicator nitrates (all values in mg N/l)

Anova: Two-Factor With Replication — Between Study Sites

	Site Vulnerability				
SUMMARY	Low	Moderate	High	Extreme	Total
Nitrates Upstream					
Count	23	23	23	23	92
Sum	1.3	87.3	159.2	34.1	281.9
Average	0.056522	3.7956522	6.921739	1.482609	3.064130435
Variance	0.073478	2.9749802	8.358142	5.32332	10.85946847
Nitrates Downstream one					
Count	23	23	23	23	92
Sum	1	65.6	144.9	38.3	249.8
Average	0.043478	2.8521739	6.3	1.665217	2.715217391
Variance	0.02166	2.4680632	7.641818	5.79419	9.185699952
Nitrate Downstream two					
Count	23	23	23	23	92
Sum	1.8	75	117.3	23.3	217.4
Average	0.078261	3.2608696	5.1	1.013043	2.363043478
Variance	0.070878	3.2097628	4.770909	1.876643	6.277300526
Total					
Count	69	69	69	69	
Sum	4.1	227.9	421.4	95.7	
Average	0.05942	3.3028986	6.107246	1.386957	
Variance	0.053917	2.9508738	7.300094	4.280563	

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Borehole location	22.61022	2	11.30511	3.185747	0.042941737	3.029985
Vulnerability	1426.144	3	475.3815	133.9611	9.30154E-53	2.638795
Interaction	32.35587	6	5.392645	1.519632	0.171818826	2.133009
Error	936.8443	264	3.548653			
Total	2417.955	275				

5.5.2 Phosphorus

Results of two-factor ANOVA analysis of phosphorus concentrations between the four study sites are shown in Table 5.74 below

Table 5.74 ANOVA results for statistical comparison between study sites for the groundwater indicator Total phosphorus (all values in mg/l P)

Anova: Two-Factor With Replication Between Study Sites

	Site Vulnerability				
SUMMARY	Low	Moderate	High	Extreme	Total
Total P Upstream					
Count	12	12	12	12	48
Sum	0.691111	0.629	0.776	0.905	3.001111
Average	0.057593	0.052417	0.064667	0.075417	0.062523
Variance	0.000506	0.000255	0.002401	0.003093	0.00154
Total P Downstream one					
Count	12	12	12	12	48
Sum	0.77753	0.644	0.792	0.657	2.87053
Average	0.064794	0.053667	0.066	0.05475	0.059803
Variance	0.003708	0.000345	0.000508	0.000422	0.001199
Total P Downstream two					
Count	12	12	12	12	48
Sum	0.41248	0.653	0.792	0.834	2.69148
Average	0.034373	0.054417	0.066	0.0695	0.056073
Variance	0.000745	0.000674	0.000508	0.000936	0.000862
Total					
Count	36	36	36	36	
Sum	1.881121	1.926	2.36	2.396	
Average	0.052253	0.0535	0.065556	0.066556	
Variance	0.001732	0.000401	0.001074	0.001476	

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Borehole location	0.001007	2	0.000503	0.42844	0.652427	3.064761
Vulnerability	0.006299	3	0.0021	1.786854	0.152775	2.673218
Interaction	0.007816	6	0.001303	1.108717	0.360678	2.167953
Error	0.155097	132	0.001175			

Total	0.170219	143

The analysis indicates that there are no significant differences between mean phosphorus concentrations across the four study sites. The outcome of this test further supports the arguments made previously that the cluster developments are not adversely affecting phosphorus concentrations in the surrounding aquifers. It is notable however, that mean phosphorus concentrations across the four study sites were 0.06 mg-P/l which is above the 0.035 mg/l P EPA threshold value. However, even given that phosphorus concentrations appear to be somewhat elevated at all of the study sites, there is no statistical evidence to attribute this directly to the clusters of OSWTS's at each of the locations. The X-ray diffraction analysis indicated similar mineralogy at all of the study sites, all of which were described as various forms of till. Given that the attenuation of phosphorous is linked to the mineralogy of the soil it is natural then that the phosphorous results are therefore similar at the four study areas.

5.5.3 E-coli **and** *Enterococci*

Results of two-factor ANOVA analysis of E-coli and Enterococci concentrations between the four study sites are shown in Table 5.75 and Table 5.76 below. It can be seen that there is a highly statistically significant difference between mean E-coli and enterococci numbers at each of the four study sites across the differing vulnerabilities. Mean E-coli numbers can be seen to increase from zero at the LOW and MODERATE vulnerability sites to 2.3 and 16.17 CFU/100ml at the HIGH and EXTREME vulnerability sites. Mean Enterococci numbers can be seen to increase from zero at the LOW and MODERATE vulnerability sites to 2.2 and 6.4 CFU/100ml at the HIGH and EXTREME vulnerability sites. However, given that at both the HIGH and EXTREME vulnerability study areas upstream mean E-coli and Enterococci numbers were higher than those downstream of the study developments, there is no statistical evidence to suggest that the clusters of OSWTS's are contributing to the elevated bacteria at these two study sites.

Table 5.75 ANOVA results for statistical comparison between study sites for the groundwater indicator bacteria E-coli (all values in CFU/100ml)

Anova: Two-Factor With Replication — Between Study Sites

	Site Vulnerability				
SUMMARY	Low	Moderate	High	Extreme	Total
E-coli Upstream					
Count	23	23	23	23	92
Sum	1	0	53	372	426
Average	0.043478	0	2.304348	16.17391	4.630435
Variance	0.043478	0	11.31225	1645.514	446.3454
E-coli Downstream one					
Count	23	23	23	23	92
Sum	0	7	24	57	88
Average	0	0.304348	1.043478	2.478261	0.956522
Variance	0	1.58498	2.407115	18.62451	6.393693
E-coli Downstream two					
Count	23	23	23	23	92
Sum	0	51	24	819	894
Average	0	2.217391	1.043478	35.6087	9.717391
Variance	0	29.81423	2.407115	5849.431	1648.469
Total					
Count	69	69	69	69	
Sum	1	58	101	1248	
Average	0.014493	0.84058	1.463768	18.08696	
Variance	0.014493	11.13598	5.575874	2618.345	

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Borehole location	3561.246	2	1780.623	2.82596	0.061046	3.029985
Vulnerability	15586.29	3	5195.43	8.245473	2.9E-05	2.638795
Interaction	9278.58	6	1546.43	2.454281	0.025114	2.133009
Error	166345	264	630.0949			
Total	194771.2	275				

Table 5.76 ANOVA results for statistical comparison between study sites for the groundwater indicator bacteria Enterococci (all values in CFU/100ml)

Anova: Two-Factor With Replication   Between Study Sites

	Site Vulnerability				
SUMMARY	Low	Moderate	High	Extreme	Total
Enterococci Upstream					
Count	23	23	23	23	92
Sum	0	0	51	148	199
Average	0	0	2.217391	6.434783	2.163043
Variance	0	0	18.90514	134.6206	44.094
Enterococci Downstream one					
Count	23	23	23	23	92
Sum	2	8	61	30	101
Average	0.086957	0.347826	2.652174	1.304348	1.097826
Variance	0.173913	2.782609	30.6917	7.403162	10.94637
Enterococci Downstream two					
Count	23	23	23	23	92
Sum	0	107	61	147	315
Average	0	4.652174	2.652174	6.391304	3.423913
Variance	0	322.5099	30.6917	97.24901	114.6205
Total					
Count	69	69	69	69	
Sum	2	115	173	325	
Average	0.028986	1.666667	2.507246	4.710145	
Variance	0.057971	109.7843	26.01833	83.2971	

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Borehole location	249.4783	2	124.7391	2.320628	0.100212	3.029985
Vulnerability	785.8949	3	261.965	4.873558	0.002575	2.638795
Interaction	462.6377	6	77.10628	1.434474	0.201572	2.133009
Error	14190.61	264	53.75231			
Total	15688.62	275				

5.5.4 Chloride

Results of two-factor ANOVA analysis of mean chloride concentrations between the four study sites are shown in Table 5.77 below. Whilst the test does indicate a significant difference in concentrations between the four vulnerabilities, the mean values indicates that they are in fact all quite similar and range from 14 – 32 mg/l which can be considered background. It is concluded therefore that the four study sites do not show any evidence of elevated chloride concentrations due to the clusters of OSWTS's.

Table 5.77 ANOVA results for statistical comparison between study sites for the groundwater indicator chloride (all values in mg/l)

Anova: Two-Factor With Replication — Between study Sites

	Site Vulnerability				
SUMMARY	Low	Moderate	High	Extreme	Total
Chlorides Upstream					
Count	23	23	23	23	92
Sum	341.2	712.9	673.4	740	2467.5
Average	14.83478	30.99565	29.27826	32.17391	26.82065
Variance	170.6224	248.4059	363.3954	154.9075	276.0926
Chlorides Downstream one					
Count	23	23	23	23	92
Sum	497.9	669.4	594.4	694.5	2456.2
Average	21.64783	29.10435	25.84348	30.19565	26.69783
Variance	306.4481	345.5959	277.8062	86.93134	257.0015
Chlorides Downstream two					
Count	23	23	23	23	92
Sum	406.4	1102.4	594.4	745.8	2849
Average	17.66957	47.93043	25.84348	32.42609	30.96739
Variance	202.4968	1116.847	277.8062	233.7493	567.229
Total					
Count	69	69	69	69	
Sum	1245.5	2484.7	1862.2	2180.3	
Average	18.05072	36.01014	26.98841	31.59855	
Variance	227.7837	626.2062	299.9863	154.876	

ANOVA

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*	*F crit*
Borehole location	1086.818	2	543.409	1.722824	0.180563	3.029985
Vulnerability	12214.27	3	4071.425	12.90804	6.79E-08	2.638795
Interaction	4644.881	6	774.1469	2.454355	0.02511	2.133009
Error	83270.26	264	315.4176			
Total	101216.2	275				

5.5.5 Regression Analysis for Density Impact Risk

Each of the water main quality parameters discussed in the previous section were used as outputs in multiple linear regression models. The regression analysis used a number of predictors that were common to the four vulnerability sites such as subsoil depth, vulnerability category, portion of the year (quarters) and treatment system densities. The purpose of this regression analysis was to identify if the treatment system density would emerge as a statistically significant factor with respect to groundwater quality. Vulnerabilities and quarters in the year were entered into the regression as indicator variables represented by either a 1 or a 0 and due to redundancy one of the four categories was removed from each regression which was expected. These indicator variables were therefore entered in an order that would ensure that the ones of interest would not be removed by Minitab. During the regressions it became apparent that a number of the predictor variables were highly correlated with each other and for this reason the regressions tended to 'lose' one or more additional indicator variables which were removed by the software. Again it was ensured that the variables of greatest interest were not amongst those removed.

Table 5.78 Summary outputs of density regression analysis

Predictor	*Coeff*	*SE Coeff*	*t-stat*	*P-value*	*Predictor*	*Coeff*	*SE Coeff*	*t-stat*	*P-value*
		Enterococci					*Chloride*		
Density	*0.045*	*3.611*	*0.01*	*0.99*	Density	-17.95	11.31	-1.59	0.116
Subsoil	-0.042	0.09529	-0.44	0.657	V4	14.26	12.18	1.17	0.245
V4	2.676	3.89	0.69	0.494	V3	16.11	16.01	1.01	0.317
V3	3.285	5.112	0.64	0.522	Subsoil	0.1661	0.2985	0.56	0.579
Qtr4	-2.405	1.274	-1.89	0.063	Qtr4	9.819	3.992	2.46	0.016
Qtr3	-3.196	1.338	-2.39	0.019	Qtr3	4.717	4.191	1.13	0.264
Qtr2	-3.627	1.338	-2.71	0.008	Qtr2	6.579	4.191	1.57	0.12
		E-coli					*Total P*		
Density	*2.27*	*13.39*	*0.17*	*0.866*	Density	-2.649	1.254	-2.11	0.038
Subsoil	-0.154	0.3534	-0.43	0.665	V4	0.639	1.384	0.46	0.646
V4	12.41	14.43	0.86	0.392	V3	1.686	1.817	0.93	0.356
V3	*2.55*	*18.96*	*0.13*	*0.893*	Subsoil	0.0014	0.03106	0.04	0.964
Qtr4	-5.859	4.58	-1.28	0.204	Qtr4	-0.279	0.3724	-0.75	0.455
Qtr1	-3.272	4.962	-0.66	0.511	Qtr3	-0.308	0.4014	-0.77	0.445
Qtr3	*2.129*	*4.754*	*0.45*	*0.656*	Qtr2	-0.121	0.4056	-0.3	0.765
		Nitrate							
Density	-4.025	1.011	-3.98	0					
Qtr4	-0.843	0.3569	-2.36	0.021					
Qtr3	-0.686	0.3747	-1.83	0.071					
Qtr2	-0.277	0.3747	-0.74	0.462					
V4	2.081	1.089	1.91	0.06					
V3	8.448	1.431	5.9	0					
Subsoil	-0.001	0.02668	-0.05	0.961					

Summaries of the regression analysis are given in Table 5.78 and it can be seen that the density variable was significant for two of the five main indicator parameters monitored during this study; nitrate and phosphorus. The variables representing which quarter during the year, is significant for three of the indicator parameters with vulnerability significant for one of the five parameters.

Various regressions were carried out using different combinations of the chosen predictors and some clear patterns emerged. For the all of the parameter regressions, (with the exception of phosphorus) involving only quarters as the predictors, Qtr3 (autumn) and Qtr4 (winter) were significant indicating that seasonal patterns are common to the four study areas. It is notable that for all of these regression models R^2 of between 0.1 and 0.3 were achieved indicating that the regressions fitted were statistically weak and the model did not fit the data 'well'. This is to be expected as only four study sites were available leading to a very 'fragile' model that is very susceptible to data points that can have high leverage.

6 GEOLOGY AND CONCEPTUAL MODEL

6.1 Introduction

There were a number of stages involved in developing conceptual models for the five study areas and this included a detailed desk study and also an analysis of both the site visit data (water levels etc.) and fieldwork data (borehole logs, pumping tests etc.). The desk study was conducted from available Geological Survey data and reports where the subsoil and bedrock geologies were compiled from the original 6" field sheets.

Each of the study areas have been described in detail in Chapter 3 with respect to topography, surface hydrology and land use. This chapter provide more specific detail on bedrock geology, subsoils (Quaternary) geology, soils and depth-to-bedrock. The study areas have been delineated into their estimated groundwater catchments which are then described in detail. The study areas are also described in relation to aquifer characteristics and groundwater recharge. Finally, all of these details will be assessed leading to the formulation of a conceptual model for groundwater flow in the proximity of the five study areas.

The formulation of any conceptual model will be limited by the availability of relevant and useful data, however it is hoped the models generally reflect the situation at the five study locations in order to aid in the development of an understanding of groundwater flows in these areas. The conceptual models for groundwater flow at the five study areas arrived at here serve as the basis for the development of the groundwater flow models as detailed in Chapter 8.

6.2 Catchment Delineation

In order to build a finite difference model it is first necessary to define the extent of the study catchment area. The initial first estimate of the catchment area did not need to be completely accurate as it was refined later in the model development process. It has been identified however, that the catchment extents used in a finite difference model such as MODFLOW should be larger than the actual catchment in order to allow for expected lower accuracies at the extremities of the model grid (Environmental Simulations, 2011).

6.2.1 Catchment Delineation using Arc Hydro Tools

Given the highly spatial nature of the parameters involved in constructing a groundwater model, the ArcGIS software by ESRI is a very a useful application to both manage and process spatial data contained within Geographic Information Systems (GIS). Many tools have been developed which work within ArcGIS and allow processing of large quantities of spatial data within a common user interface. Arc Hydro Tools is one of these applications developed to be used within ArcGIS and allows for surface and groundwater geoprocessing at a catchment scale. The Arc Hydro Toolbox enables the delineation of drainage patterns within a catchment, which can then be used to determine contributing areas as well as the overall catchment boundary or watershed.

The Arc Hydro Toolbox 2.0 was used within ArcGIS 9.3 to develop an initial estimate of the catchment extents for three of the five study areas. This method was not used for the Naul study site as a detailed study had already been carried out in that area by the GSI (Hunter Williams et al., 2005) and that study will therefore be used to develop the initial catchment boundary for that study site – see Section 8.3.2. As the study area at Toonagh is karst with known major conduit flow features it was not modelled using MODFLOW, however Arc Hydro Tools was used to estimate catchment areas in a slightly difference context for this study site. This will be described separately in Section 8.3.3.

There are a number of steps involved in delineating a catchment using Arc Hydro Tools and these will be only briefly summarised. The majority of operations used within Arc Hydro Tools require the Spatial Analyst toolbox available as an add-on for ArcGIS. The data required to carry out this exercise include:

- A Digital Elevation Model (DEM) for the area
- A polygon shapefile of streams and rivers
- A raster or vector file containing Ordnance Survey Data for the area.

The Arc Hydro Toolbox was used to generate a number of interim data sets that were then combined and used to graphically represent the drainage patterns of the catchment. The majority of the operations performed were raster analysis and these operations created interim data sets on; flow direction, flow accumulation, stream definition, stream segmentation and catchment delineation. Once these raster analysis operations were completed, the data were then used to develop a graphical representation of catchment boundaries and drainage lines from selected points in the data. Figure 6.1 – Figure 6.2 illustrate some of the processes involved in delineating the initial catchment boundaries using Arc Hydro Tools.

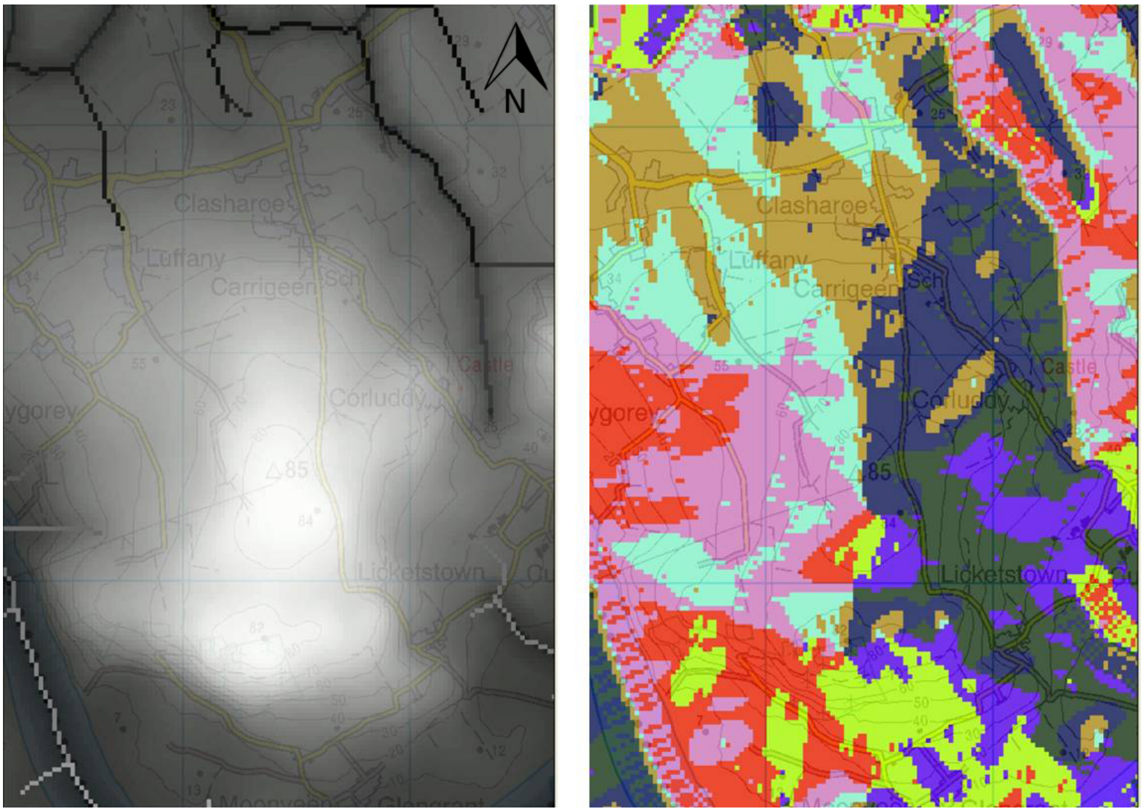


Figure 6.1 (a) DEM agreement with calculated stream links (b) Calculated flow direction for study area at Carrigeen, Co. Kilkenny (not to scale)

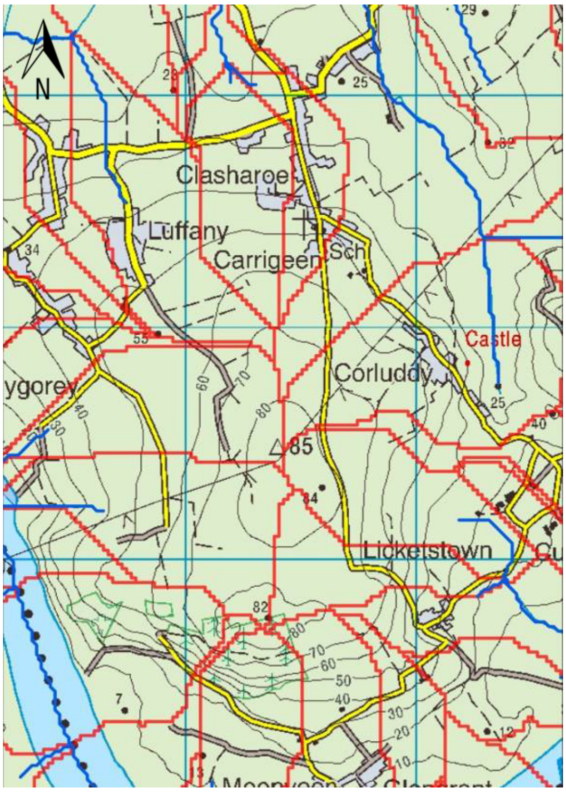


Figure 6.2 Calculated drainage lines with small scale catchments for study area at Carrigeen, Co. Kilkenny (not to scale)

The approximate catchment areas for the study sites as calculated using Arc Hydro Tools are summarised in Table 6.1 and the associated catchment boundaries are illustrated graphically in Figure 6.3 – Figure 6.4.

Table 6.1 Catchment Areas as calculated using Arc Hydro Tools

Study Area	Initial Catchment Area (km^2)
Rhode, Co. Offaly	3.262
Carrigeen Co. Kilkenny	0.637
Faha, Co. Limerick	0.945

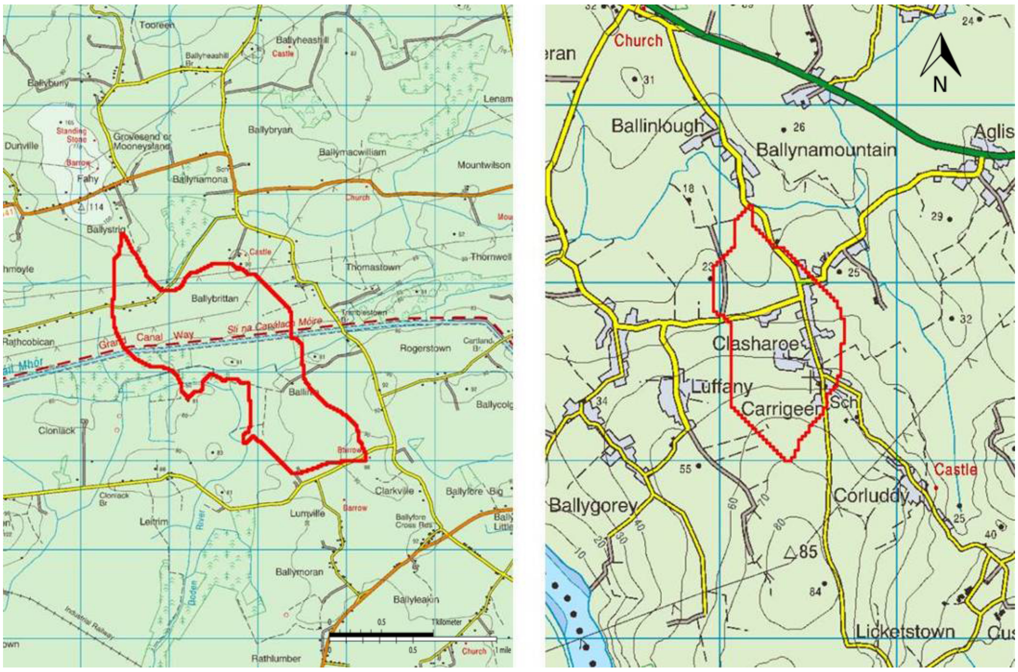


Figure 6.3 (a) Catchment Estimate for Offaly Study Area (b) Catchment Estimate for Kilkenny Study Area

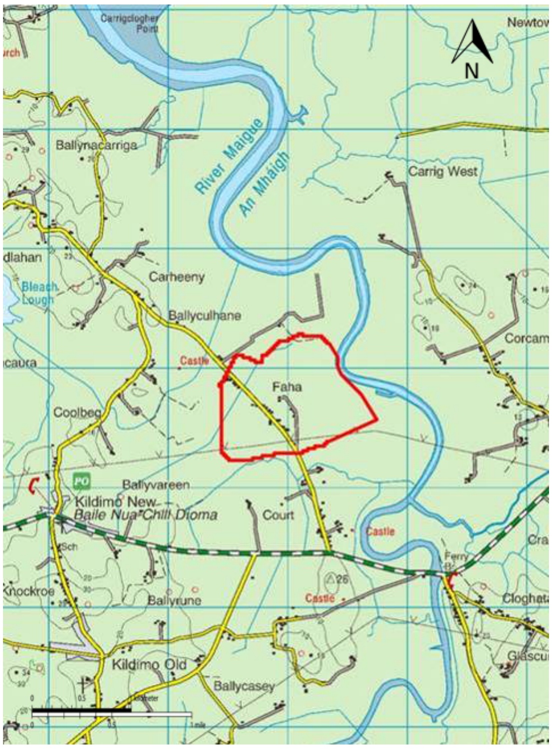


Figure 6.4 Catchment Estimate for (a) Limerick Study Area (b) Clare Study Area

These initial estimates of the catchment boundaries assume that groundwater flow will be based generally on the surface water catchments. In reality groundwater catchments may in fact be larger or oriented in different flow directions due to aquifer properties such as hydraulic permeability and porosity or the presence of aquitards. For this reason these initial estimates for catchment boundaries were then increased to incorporate obvious groundwater divides in order to model an area that would not under-estimate the catchment. For the study areas both at Kilkenny and Limerick it was possible to use linear features such as surface water bodies and topographical divides to better estimate a larger catchment for modelling purposes.

The catchment area for modelling groundwater at the site in Carrigeen, Co Kilkenny was increased to include the path of the local stream along the eastern perimeter as far as its intersection with the Dungooly stream. The catchment boundary then follows the path of the Dungooly stream as far as the catchment outlet to the west.

For the study area at Faha, Co. Limerick the catchment area was increasing along the length of the River Magure upstream as far as Ferrybridge and downstream as far as the projected catchment outlet. To the west the catchment was increased by an area large enough to most likely include any possible unpredicted groundwater divide that is not obvious from surface features.

The catchment boundary for groundwater modelling for the site at Rhode, Co. Offaly was not straightforward. Due to the complex nature of the surface water and topographic divides in the area it is possible that groundwater could be moving in three separate directions; however based on the outcome of the catchment delineation from ArcHydro tools it would seem likely that groundwater is moving to the south and is in the catchment of the Doden River. It was therefore necessary to first model the study site at Rhode for a much larger area incorporating the three possible catchment outlets. It would then become clear which direction groundwater beneath the study site is moving and the catchment area for the groundwater model can be reduced and refined. The initial larger catchment for modelling with the three possible outlets is shown in Figure 6.5 below.

The initial catchment areas for groundwater modelling is summarised in Table 6.2 below.

Table 6.2 Catchment Areas for Groundwater Modelling

Study Area	Initial Catchment Area (km^2)
Rhode, Co. Offaly	25.01
Carrigeen Co. Kilkenny	2.23
Faha, Co. Limerick	3.45

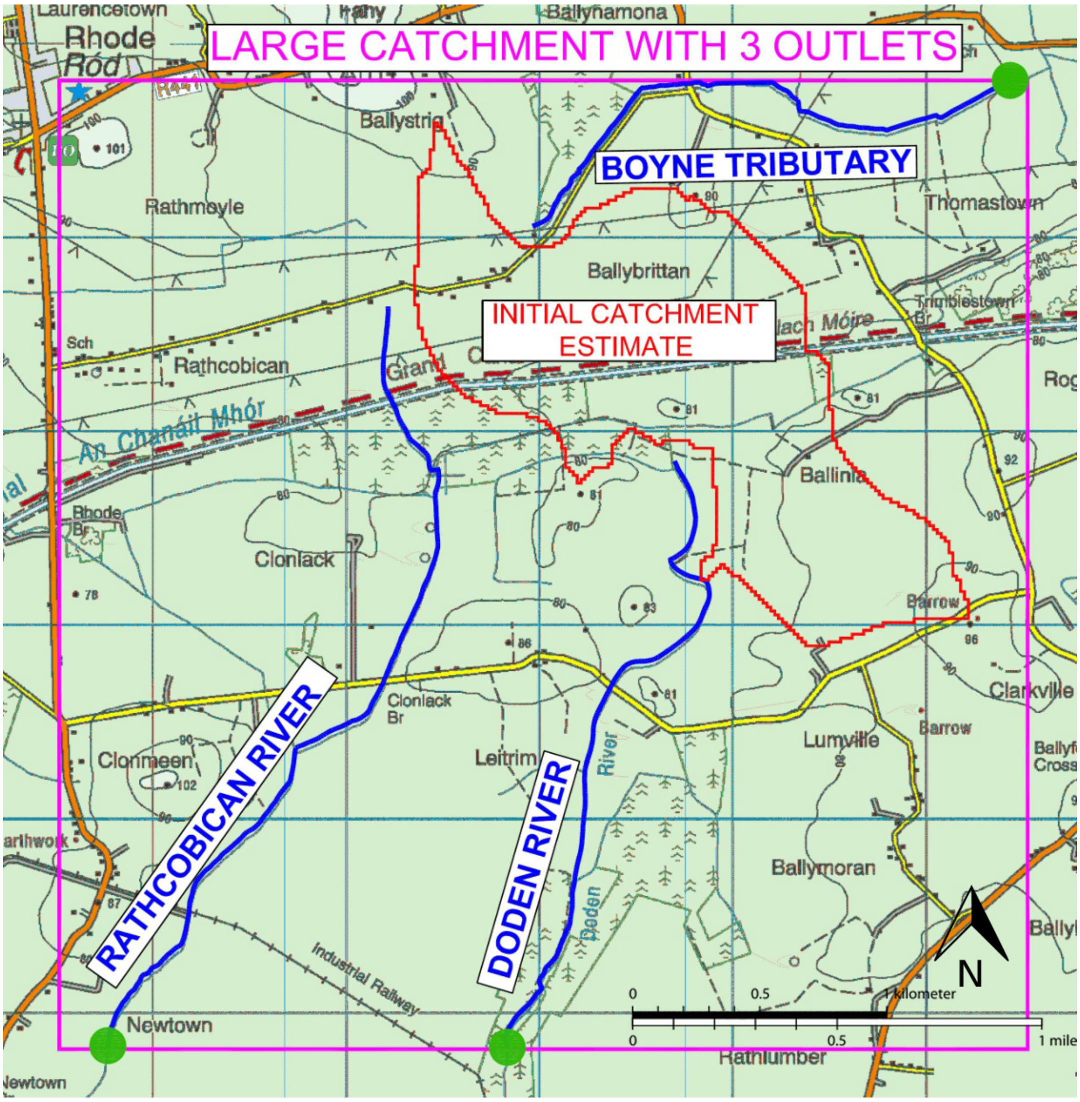


Figure 6.5 Catchment at Rhode for larger area incorporating three possible catchment outlets

6.2.2 Catchment Delineation for Site at the Naul, Co. Dublin

The initial estimate of the groundwater catchment boundary for the Naul study area was based upon a study that was carried out by the GSI (2005) in which a much larger catchment was modelled to identify the outer and inner zones of protection for the adjacent Bog of the Ring groundwater abstraction scheme.

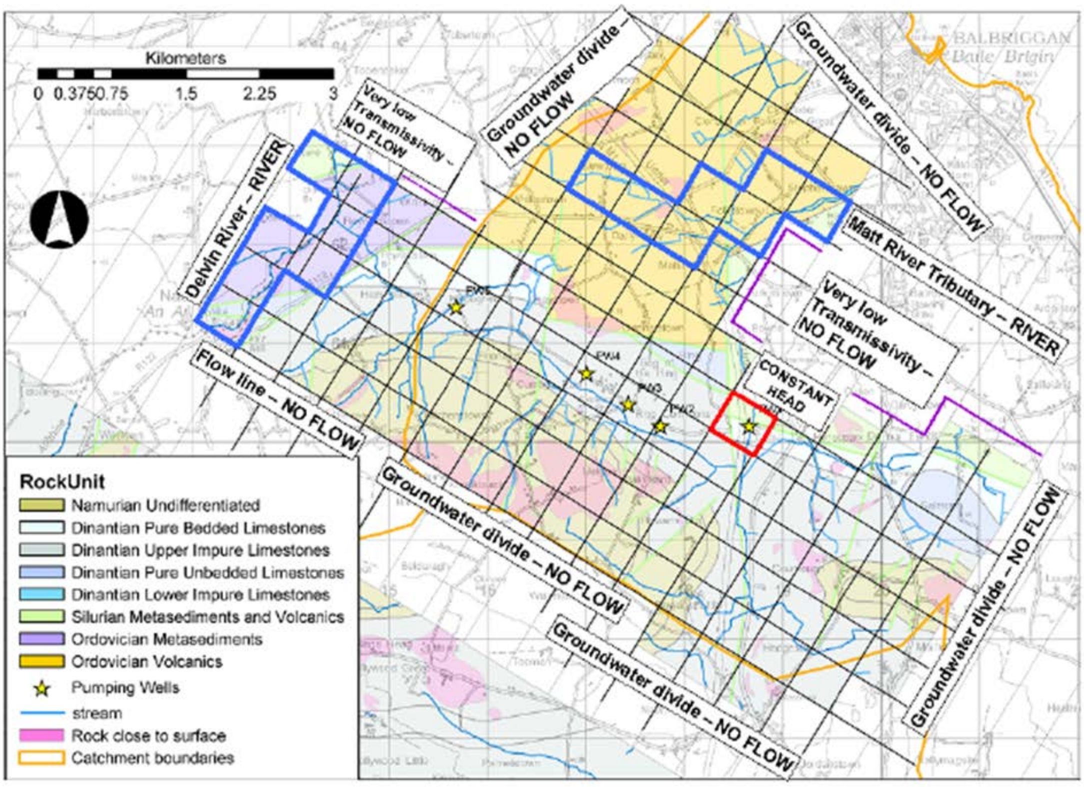


Figure 6.6 Catchment Boundary for Bog of the Ring Groundwater protection Scheme Model (GSI, 2005)

The initial estimate of the catchment boundary to be used for this study was based on the groundwater divides and flow directions identified during the GSI study (GSI, 2005) and the associated groundwater contours, augmented with data collected during this current study. Figure 6.6 above illustrates the Bog of the Ring study model catchment boundary. Figure 6.7 shows the localised catchment boundary that will be used as an initial estimate for this study. This boundary was constructed using the groundwater divide to the south-west, the Delvin River to the north-west and a local stream to complete the boundary extents. The initial catchment boundary for this study is estimated from this exercise to be 2.394 km^2.

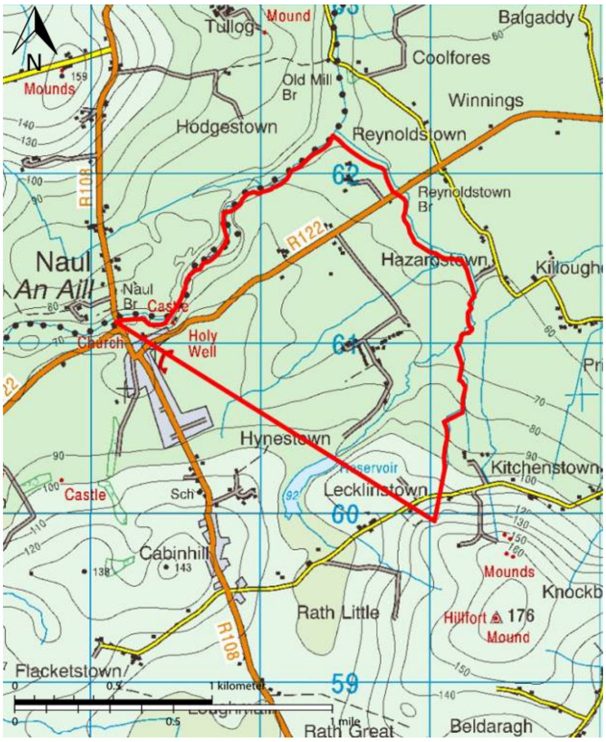


Figure 6.7 Catchment Estimate for Naul Study Area

6.2.3 Catchment Delineation for Site at the Toonagh, Co. Clare

As the area at Toonagh is underlain by karst with groundwater flow suspected to be dominated by conduit flows, in addition to groundwater and surface water interaction and interchange at various locations, it was not considered sensible to use ArcHydro Tools to quantify the catchment which could vary hugely from surface topography. The Toreen east stream disappears into the sink at the interface between the siltstone/sandstone and cherty limestone formation (Gull Island) and the Ailwee member limestone formation as shown in Figure 6.8 below.

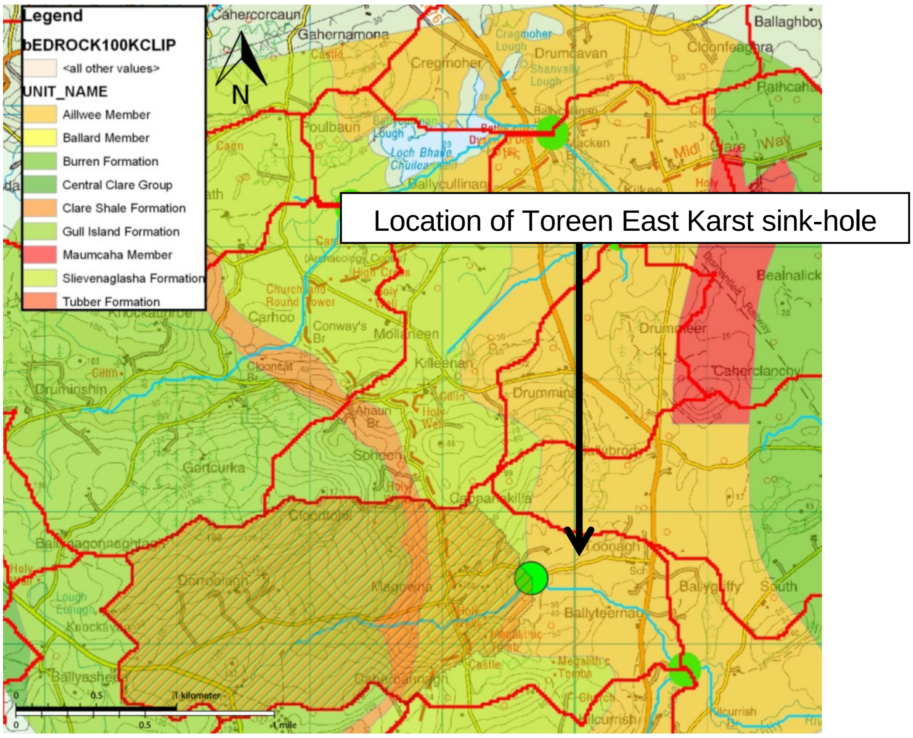


Figure 6.8 Rock formations and surface water features at Toonagh, Co. Clare

Arc Hydro Tools was used however, to delineate catchment boundaries for the non-karst areas which should have given a reliable estimate for those areas that follow topography. Therefore a drainage point was added at the location where the Toreen East stream goes underground and the contributing upstream surface water catchment was calculated using Arc Hydro Tools. This yielded a contributing area of 4.08 km^2 as shown in the hatched area in Figure 6.9 below.

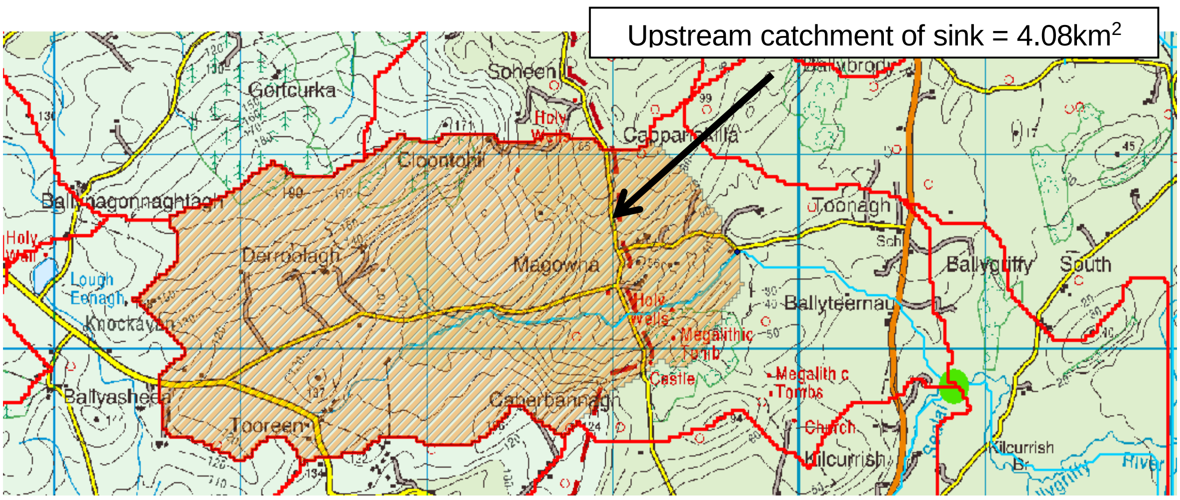


Figure 6.9 Upstream Catchment at Toonagh, Co. Clare (not to scale)

A very simple calculation using average effective rainfall for the area, estimated runoff using recharge coefficient mapping and downstream flow in the Toreen East stream as observed during this study is given below.

R_e = 650 mm (assumed effective rainfall)
$R_{c(avg)}$ = 0.65 (averaged from GSI maps in area)

➔ Average Catchment Runoff = 227.5 mm/year

Average downstream yearly discharge = 100 l/s (at the Kilcurrish spring)
Assume baseflow = 10 l/s (summer gauging – dry conditions)

➔ Average discharge from runoff = 90 l/s

Average yearly discharge = 90 l/s = 2.84×10^6 m^3 per year

Catchment size = yearly discharge / yearly rainfall = 2.84×10^6 / $227.5/10^3$ ~ 12 km^2

This demonstrates that even though the upstream surface water catchment is approximately 4 km^2, a far larger area is in fact draining to the Kilcurrish spring. A connection has been established between the upstream sink-hole and the Kilcurrish spring (see Chapter 5); therefore it must be assumed that the conduit system is joined by another larger flow path in the area of Toonagh. Based on the results of the tracer study carried out as part of this study whereby a karst connection was established between the treatment plant at Toonagh and the Kilcurrish spring, it would seem likely that surface/ground water is moving into this system from the north or north-west.

6.3 Aquifer Characteristics and Recharge

6.3.1 Aquifer Characteristics

Site at Naul, Co. Dublin

An extensive study of groundwater quantity, flow and direction was undertaken adjacent to the study area at the Naul by the GSI in 2005, in order to develop inner and outer zones of protection for the local Bog of the Ring groundwater supply (Hunter Williams et al., 2005). This GSI study involved the collation of field data from a number of monitoring and production wells leading to the production of a groundwater model to predict the impact of groundwater abstractions at the Bog of the Ring. As the area of interest for this current study is inside the area modelled as part of the GSI study, a significant quantity of relevant data is available for groundwater elevations, flow direction and aquifer hydraulic properties. This data has been assessed and supplemented with knowledge and experience gained during this study in order to form input for aquifer characteristics for use in the development of a conceptual model. There are four main rock units in the area as illustrated in Figure 6.10 below.

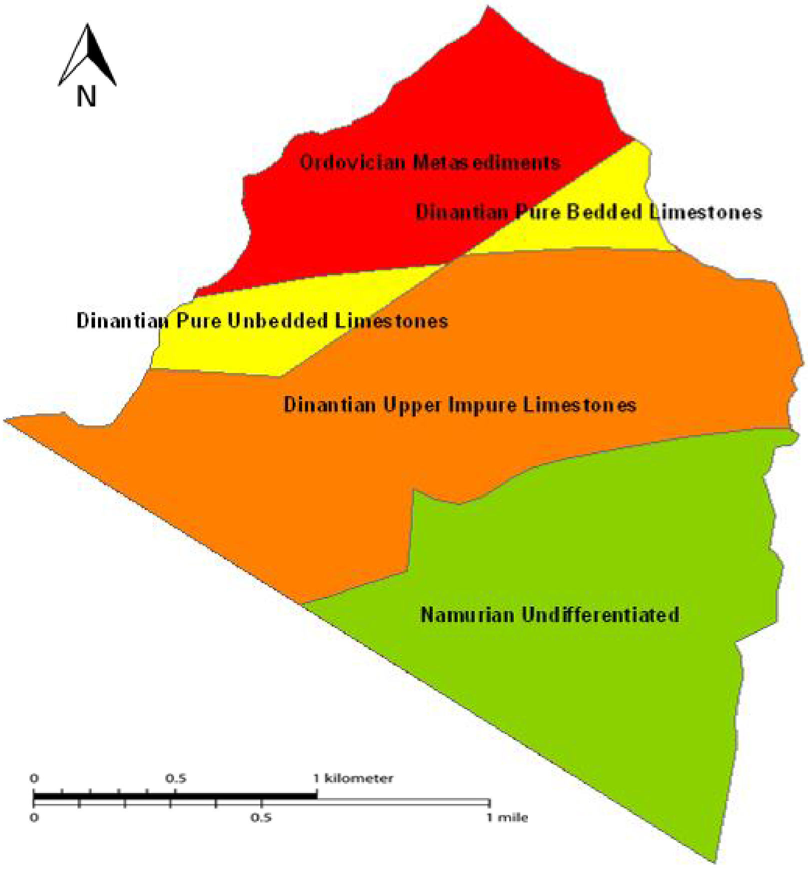


Figure 6.10 Generalised Bedrock map for the study catchment area at the site at the Naul, Co. Dublin

The individual rock formations within these rock units have been described in Section 6.2 previously. Aquifer characteristics for these four rock units were estimated using the GSI study values as a starting point initially to set-up and run the model and these values are shown in Table 6.3 below. The values set out in Table 6.3 are similar to values estimated during slug tests undertaken during the present study (See Chapter 5). Porosity was taken to be 1% for the Namurian rocks and 2% for all other rock units.

Table 6.3 Summary of aquifer properties used in GSI Bog of the Ring groundwater study (GSI, 2005)

Aquifer unit	Sub-area	Horizontal permeability (m/d) [1]	Transmissivity (m^2/d) [2]
Impure Limestones	Central zone	9	630
	Peripheral zone	5	350
	Eastern zone	0.7	49
	Matt River/ M1 zone	12	840
Ordovician Volcanics	Main zone	0.6	24
	Hill zone	0.25	10
Lower Palaeozoic Metasediments	N/A	0.25	7.5
Namurian Mudstones and Sandstones [2]	Knockbrack Hill	0.0625–0.125	3.1-4.6
	Eastern zone	0.0313-0.0625	2-3.75

Site at Carrigeen, Co. Kilkenny

The site at Carrigeen in Co. Kilkenny contains three different rock units as shown in Figure 6.11 below. Estimates for hydraulic conductivity for these rock units have been made using information obtained from the GSI and EPA database on Irish rock unit permeability's which is being compiled by Tobin's Consulting Engineers (still in draft format) and from slug tests undertaken as part of this study. Whilst slug tests undertaken in the Devonian rock unit yielded a reasonable value when compared with other values for similar rock units, the slug test attempted in the Dinatian rock unit had to be abandoned as outlined in Chapter 5. Groundwater flow is assumed to follow topography and move from south-east to north-west with a relatively steep gradient. Porosity is assumed to be 2% for all rock units in this area. Table 6.4 below summarises rock permeability's for the study area.

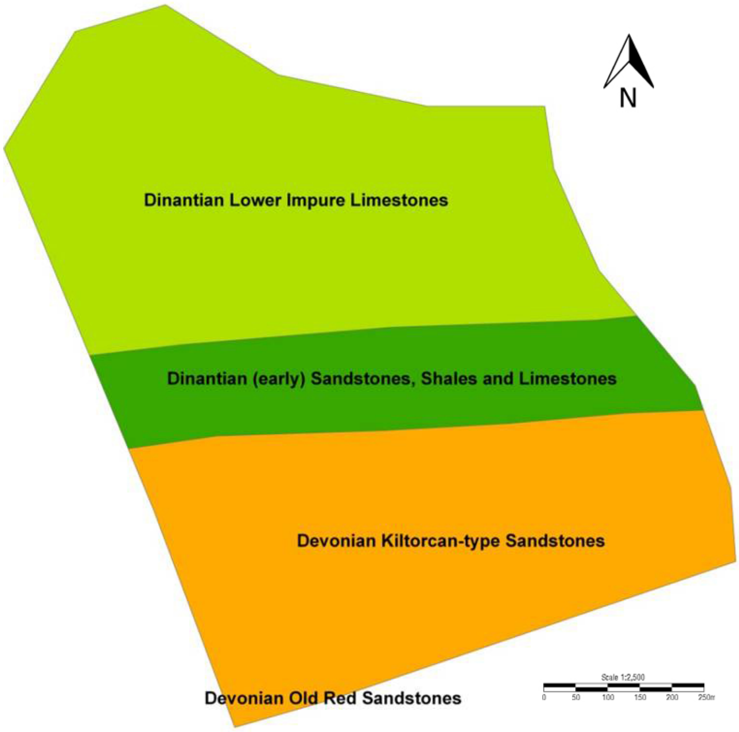


Figure 6.11 Generalised Bedrock map for the study catchment area at the site at Carrigeen, Co. Kilkenny

Table 6.4 Summary of aquifer properties (Hegarthy, 2002; GSI, 2007)

Aquifer Unit	Horizontal Permeability (m/day)
Devonian Sandstone	2 - 80
Dinantian Sandstone, Shale, Limestone	1 – 20
Dinantian Lower Impure Limestone	0.1 – 2

Site at Rhode, Co. Offaly

The entire initial larger catchment at Rhode is underlain by Dinantian pure bedded limestone and makes modelling more straightforward. A groundwater protection scheme report was developed by the GSI for the local Toberdaly public water supply and contains relevant hydraulic property data for the aquifer unit in the area (Hudson, 1996). Hudson assumed that the limestone in this area has permeability (K) of 5 – 10m/d and an effective porosity of 2%. This is consistent with slug tests that were carried out as part of this study and also with the GSI/EPA draft database and these values will serve as initial estimates for setting up the groundwater model.

Site at Faha, Co. Limerick

The entire study area at Faha is underlain by Dinantian Pure Unbedded Limestone. This limestone is said to be karstified in this area but no evidence of this was found during drilling as discussed in Chapter 4. In addition no karst features are noted in the area on the available GSI maps and none were observed during the many site visits during this study. It is therefore assumed that the limestone in this area is not significantly karstified and will be modelled as a typical bedrock aquifer. A groundwater protection scheme report was developed by the GSI for the local Croom water supply and contains estimates for aquifer properties in the area (Deakin, 1995). The groundwater abstraction borehole at Croom is located in the same Waulsortian limestone rock unit as the study location although it is located approximately 13 km to the south. This GSI study contained results of a pumping test carried out on the groundwater abstraction borehole with transmissivities ranging from 95 to 145 m^2/d. This was based on an open length of 28.34 m and thus the horizontal permeability was in the range or 3 – 6 m/d. Slug tests undertaken as part of this current study yielded an estimate of horizontal permeability in the range of 8 -15 m/d. Hence, a value of 10 m/d was used initially to set-up the model. Effective porosity of 2% will again be assumed for this rock unit.

6.3.2 Groundwater Recharge

In order to develop an understanding of the quantity and movement of groundwater beneath the five study site locations it was first necessary to quantify recharge - i.e. the amount of water replenishing a groundwater flow system. There are a number of methods that can be used to estimate the quantity of recharge which is the main driving force behind groundwater movement and so it is important that its estimation is realistic and as accurate as possible. For the purposes of this study recharge was estimated on an annual average basis and is assumed to consist of rainfall as the input less outputs which are assumed to be evapotranspiration and runoff:

Recharge (R) = Annual Average Rainfall – Annual Evapotranspiration (A.E.) – Runoff

Data were obtained from Met Éireann from the closest rainfall monitoring stations to each of the study locations – see Chapter 3 for details. In most cases the closest Met Éireann monitoring station was not a synoptic or climatological stations but recorded daily rainfall only. It was therefore necessary to attempt to relate data from the closest synoptic station and make a good estimation of the corresponding vales in the area of interest. Data were compared using the Mintab statistical software package and a good correlation was found in all cases between the closest rainfall station and the closest synoptic station (as shown in Table 6.5). It was then possible to estimate both average annual rainfall and annual evapotranspiration for the study locations, also using information from local groundwater protection scheme reports, as shown in Table 6.6.

Table 6.5 Rainfall and Climatological Stations Used for Weather Data Acquisition

Rainfall Station	Synoptic Station	Correlation Factor
Patrickswell	Shannon Airport	0.729
Naul	Dublin Airport	0.832
Derrygreenagh/Edenderry	Mullingar/Oakpark	0.776
Waterford/Adamstown	Oakpark/Johnstown Castle	0.790

Table 6.6 Rainfall and Climatological Stations Used for Weather Data Acquisition

Rainfall Station	Annual Average Rainfall (mm/year)	Annual Average Evapotranspiration (mm/year)
Patrickswell	870	490
Naul	840	445
Derrygreenagh	944	440
Waterford	1015	530

Potential Recharge was calculated by subtracting Annual Evapotranspiration from Annual Average Rainfall. This gives an estimation of the quantity of rainfall that is available for infiltration into the subsoil which eventually makes its way to the groundwater system. However, the quantity of rainfall that will actually make it to groundwater will be significantly less that that available as Potential Recharge. Actual Recharge is the estimated amount of water that will infiltrate to groundwater. Recharge quantities will vary depending on a number of factors including subsoil permeability and depth to bedrock. Therefore recharge is likely to be greater in areas overlain by higher permeability subsoils and shallower depths to bedrock. Similarly an area with low permeability subsoils and deep bedrock will have low recharge. In order to account for these spatial variations in subsoil permeability and depth to bedrock a national recharge map has been developed by the GSI. This map applies recharge coefficients to different areas of the country. These recharge coefficients can be applied to estimates of potential recharge in order to arrive at the vales of actual recharge to groundwater. The national recharge map for the country is shown in Figure 6.12 below.

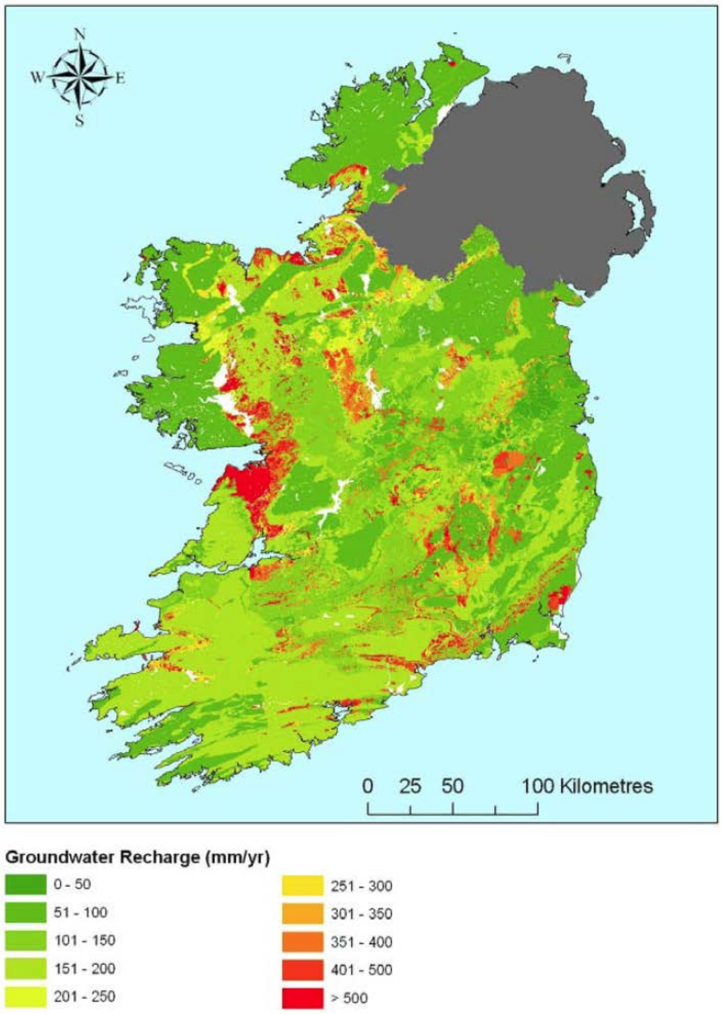


Figure 6.12 National Recharge Map of Ireland (ERBD, 2007)

A proportion of the country is however, underlain by bedrock which has only a limited ability to accept infiltration water as recharge, due to its characteristics and poorly fractured nature. In these areas it has been decided that a recharge cap will be applied. For example a recharge cap of 100 mm/year may be applied to certain bedrock units - i.e. 100 mm/year is deemed as the maximum annual recharge that can be accepted by that aquifer. The method used to determine recharge and the corresponding recharge coefficients was similar to that outlined by Mistear et al. (2009).

For each of the study locations the GSI national recharge map was consulted and the associated recharge coefficients were extracted – in most cases each of the study areas contained a number of different recharge zones due to the varying recharge coefficients contained within them. Each of the areas was checked to ensure that a recharge cap was not applied to any of the bedrock units. Estimated Actual Recharge could then be calculated for each of the subdivided recharge zones by applying the recharge coefficients (as outlined above) to the appropriate values of potential recharge. Figure 6.13 – Figure 6.14 illustrate the different recharge zones at the study locations.

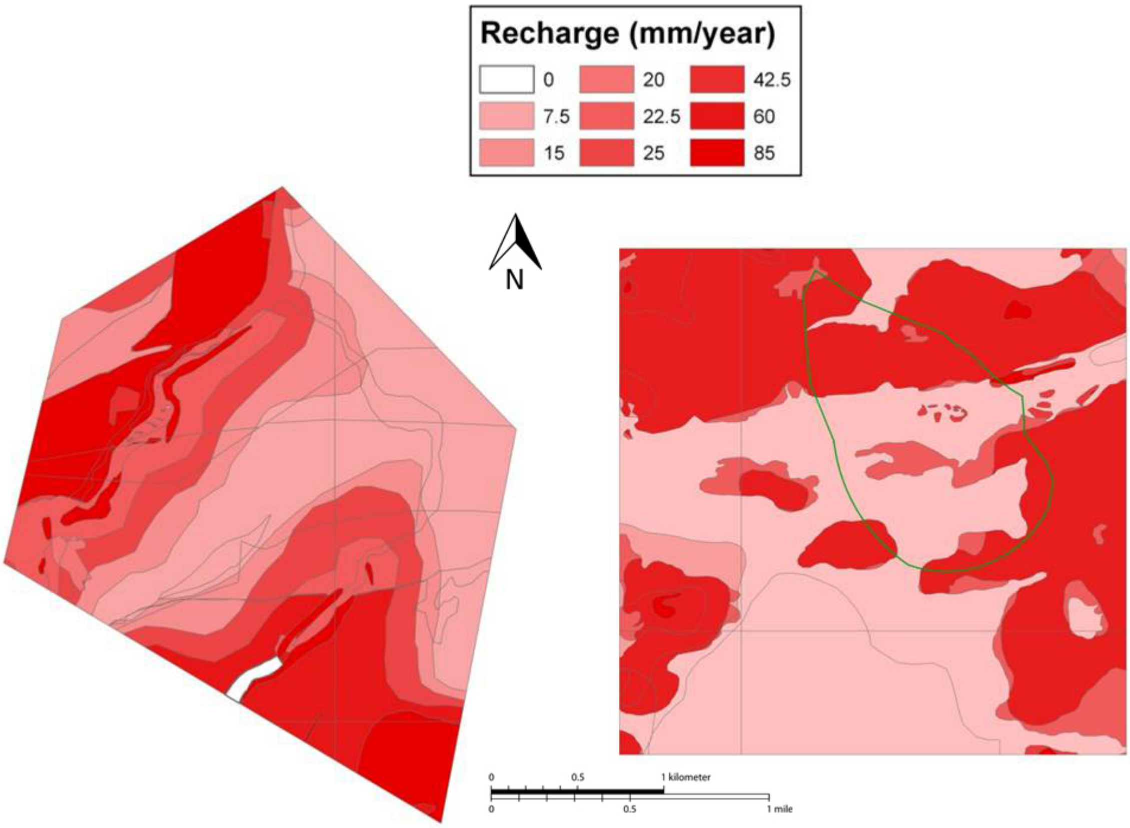


Figure 6.13 Recharge Map of (a) Study Area at The Naul, Co Dublin (b) Study Area at Rhode, Co. Offaly

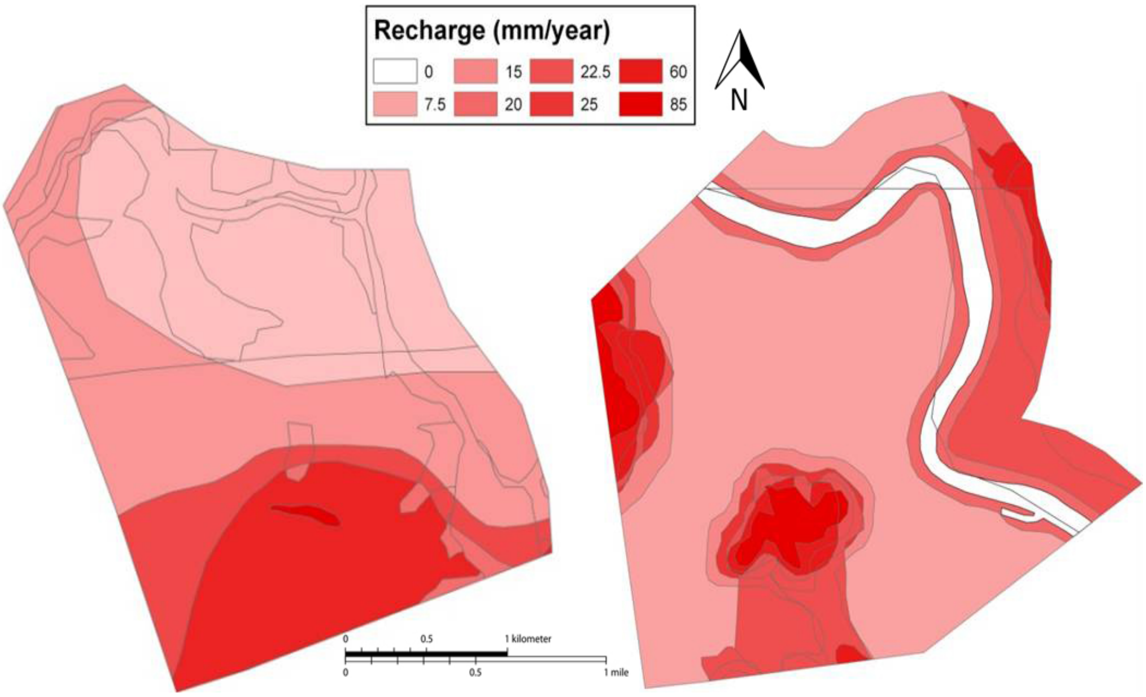


Figure 6.14 Recharge Map of (a) Study Area at Faha, Co. Limerick (b) Study Area at Carrigeen, Co. Kilkenny

6.4 Conceptual Model - Summary

Site at the Naul, Co. Dublin

Groundwater at the Naul is assumed to be moving from south-east to north-west. The catchment is defined by the Delvin River and a local stream and is assumed to discharge to the River Delvin at a catchment outfall to the north-west. The groundwater gradient is quite steep reflecting topography. The aquifer permeability's range from 9 m/d to 0.06 m/d based on the specific rock unit. Recharge is quite low in the area reflecting the subsoil properties. Recharge is estimated to be about 7% of potential recharge for the majority of the catchment however some areas having higher proportions particularly to the south near Knockbrack hill.

Site at Rhode, Co. Offaly

Groundwater in Rhode could be moving in one of three possible directions as the study site is located in the centre of a groundwater divide. A larger area must therefore be modelled initially in order to refine the exact catchment boundary. An initial estimate, based on topography only, predicts that groundwater is moving to the south and outfalling to the

Doden River. Groundwater gradients in the area are very flat and this reflects the generally flat nature of the surface topography. The area is underlain by one single rock unit and its permeability is estimated to be in the range of 5 – 10 m/d. Recharge in the area is generally very low at 4%, given the low permeability tills present, but rising to 60% in areas of higher ground where subsoils are thinner.

Site at Carrigeen, Co. Kilkenny

A steep groundwater gradient exists at Carrigeen with groundwater assumed to be moving from south to north and outfalling to the Dungooly stream at the bottom of the catchment. The area is underlain by highly permeable sandstones and less permeable limestone and shales. The permeability ranges from 0.1 – 80 m/d. Recharge in the area ranges from 8% in the lower areas and increase to 85% at the top of the catchment where bedrock is shallow and exposed in places.

Site at Faha, Co. Limerick

Groundwater in Faha is assumed to being moving from south to north outfalling to the River Maigue. The groundwater gradient is quite shallow given the low-lying nature of the surrounding land. The limestone bedrock in the area is assumed not to be Karstified and its permeability is estimated to be 10 m/d. Recharge in the area is low due to the presence of alluvium deposits but is much higher to the south and west where bedrock is exposed in places.

7 UNSATURATED ZONE MODEL AND RESULTS

7.1 Introduction

This Chapter will present the methodology and results of numerical modelling used to simulate water and solute transport in the unsaturated (vadose) zone. The goal of this process was to predict the contaminant loading and concentrations discharging to the groundwater table from the unsaturated zone due to the various types of OSWTS's and the associated subsoil discharge for use as inputs to the groundwater solute transport model described in Chapter 8. Simulations were carried out using both the HYDRUS 1D and 2D/3D packages where appropriate. HYDRUS 1D was used to simulate the water content and pressure heads in the soil profile using available precipitation and evapotranspiration data and then used as initialisation input into the 2D/3D package. The HYDRUS 2D/3D software package was used in order to calibrate for soil hydraulic properties based on field work carried out during this study. The 2D/3D package was then used to model groundwater flow and contaminant transport. A brief background to the software is given followed by descriptions of the main flow and solute transport parameters used. Results of each of the simulations are then given followed by a parameter sensitivity analysis.

7.2 Model Setup

Water flow was simulated using the HYDRUS suite of software packages allowing simulations in 1, 2 and 3 dimensions. HYDRUS simulates variably saturated transient water content and volumetric flux using a numerical solution to the Richards' equation; a background to the equations and input terms involved have been described in detail in Chapter 2 (Simunek et al, 2008).

The model requires a number of inputs including soil hydraulic properties, solute transport parameters and a set of initial and boundary conditions, some of which may be time-variable. In order to simplify the model and reduce the time required for model convergence it was decided to break the modelling process into three separate processes which would allow the model to be built in a stepwise process thus reducing the possibility for error and models crashing. The three model steps and the package used are given below.

1. Determine soil hydraulic parameters – 2D inverse model
2. Carry out initialisation with respect to pressure heads – 1D direct model
3. Determine contaminant loading and concentrations at the water table – 2D direct model with water and solute transport

<u>2D Inverse Model Summary</u>

As outlined in Chapter 5, in-situ falling head percolation tests were carried out at each of the site locations and soil samples were also recovered from each of the sites (at one or more depths). Particle size distribution analysis was carried out on the soil samples (see Chapter 5) and this data together with the head/time data from the falling head tests were used to calibrate HYDRUS for soil hydraulic parameters using the inverse solution code contained within HYDRUS 2D/3D. The inverse solution was implemented using the Levenberg Marquardt optimisation module contained within the HYDRUS software. The inverse method is based upon the minimisation of a suitable objective function, which expresses the differences between measured and simulated values. The software seeks to minimise the Sum of Squares Residuals (SSQR) over the input dataset within a prescribed maximum number of iterations. Quality in parameter estimation is generally assessed using two indicators; the coefficient of determination (r^2) and SSQ. Full details of the equations and routine followed for the inverse solution procedure within HYDRUS are given by Simunek (1999).

A specially written piece of code by the software's author Jirka Simunek was implemented within the HYDRUS 2D/3D software in order to accurately simulate the falling head test data. The standard software assumes either Type 1 (Dirichlet) ***Pressure head*** or Type 3 (Neumann) ***Flux*** boundary conditions but neither could accurately represent a falling pressure head boundary condition. A summary of the additional ***Well*** boundary condition code that had to be written by the software's author Jiri Simunek is given below:

"The well boundary condition is implemented in the following way. A user specifies a seepage face boundary condition on the boundary representing the well wall, well radius r_w, and the initial position of the water level in the well h_w. HYDRUS then evaluates the following mass balance equation to determine the position of the water level in the well:

$$\pi_w^2 \frac{dh_w}{dt} = Q_{in}(t) - Q_p(t)$$

which in its finite difference discretization is given by:

$$\pi_w^2 \frac{h_w^{j+1} - h_w^j}{\Delta t} = Q_{in} - Q_p$$

$$h_w^{j+1} = h_w^j + \frac{\Delta t}{\pi\, r_w^2}(Q_{in} - Q_p)$$

Where: r_w is the well radius; h_w is the water level in the well; Q_{in} is the water inflow into the well from the soil profile across the well wall (or its screened part); Q_p is the pumping rate; Dt is the time step, and h_w^j and h_w^{j+1} are water levels in the well at the previous and current time levels.

Parts of boundary below and above the water level in the well are then assigned the (time-variable) pressure head (Dirichlet) and seepage face boundary conditions, respectively. HYDRUS then calculates which part of the seepage face boundary is active (with prescribed zero pressure head) and which is inactive (with prescribed zero flux). HYDRUS also calculates and reports separately fluxes across these two parts of the boundary representing a well. HYDRUS does not report in the output the position of the water level in the well, h_w. A user needs to specify an observation node at the bottom of the well to obtain this information."

The resulting calibrated soil hydraulic parameters are given in Table 7.1 below.

Table 7.1 Soil Hydraulic parameters for the different vulnerability sites calibrated using the inverse solution with HYDRUS 2D/3D

Location	Depth	Soil Texture Class	ρ_p	θ_r	θ_s	K_s	α	n	m
Naul, Co. Dublin	1.2m	Clay LOAM	1.42	0.0701	0.4033	0.0084	0.0193	1.3658	0.5
Rhode, Co. Offaly	0.6m	Sandy SILT	1.31	0.0708	0.4047	0.0061	0.0189	1.3671	0.5
-	1.1m	Sandy SILT/CLAY	1.52	0.0595	0.3872	0.0090	0.0259	1.3551	0.5
Carrigeen, Co. Kilkenny	1.2m	Sandy SILT/CLAY	1.49	0.0726	0.4086	0.0100	0.0179	1.3708	0.5
Faha, Co. Limerick	0.55m	SILT	1.4	0.0897	0.4626	0.0080	0.0122	1.3876	0.5
-	0.9m	Sandy CLAY	1..4	0.0813	0.4343	0.0043	0.0137	1.3872	0.5

α, fitting parameter that is related to the air entry pressure value (cm^{-1}); K_s, saturated soil hydraulic conductivity (cm h^{-1}; m, dimensionless soil moisture retention function = 1 – (1/n); n, dimensionless fitting parameter related to the pore size distribution; ρ_p, soil bulk density (g cm^{-3}); θ_r, residual soil moisture content (cm^{-3} cm^{-3}); θ_s, saturated soil moisture content (cm^{-3} cm^{-3})

1D Direct Model Summary

HYDRUS 1D was used (with the soil properties determined previously) in order to determine initialisation pressure heads for the contaminant transport and water flow 2D models. Whilst it would be possible to start the 2D models with arbitrary initial pressures head profiles (with the resulting soil water contents), it is more robust to input initialisation pressures heads prior to calibrating a more detailed model. Consequently the 1D version of the software was used for this initial part of the model setup to reduce the complexity of the model and process times required for convergence. The 1D model allowed for water flow over an extended period of time and utilised precipitation and evapotranspiration data that was available from the closet Met Eireann rainfall or synoptic station (see Chapter 4 for details) over a 3 year period. A general outline of the boundary conditions for this model setup is given in Figure

7.1. Water flow included a sink term to account for water lost to plant roots. The sink term (S) is calculated using the approach introduced by Feddes et al. (1978) whereby the potential transpiration rate is distributed over the root zone using a stress response function that accounts for water and osmotic stresses (Feddes et al., 1978; van Genuchten, 1987; Simunek and Hopmans, 2009). The function requires five variables that describe the dependence of the extraction of water from the soil on pressure head. The values for these parameters were set using the database contained within HYDRUS and crop cover was taken to be grass with a root zone extending 30 cm into the soil which was found to be suitable by Beggs et al (2011).

The simulation was represented by a 100 cm wide cross-section through the soil profile extending to the depth of the observed water table at each of the four site location. The profile was descretized into a number of nodes of 2 cm density at the top of the soil profile widening to 10 cm at the bottom of the profile as shown in Figure 7.1. The initial condition for pressure head was set at 100 cm throughout the profile. The top of the profile was applied with an ***Atmospheric with Surface Runoff*** boundary condition. The base of the profile was set to the ***Free Drainage*** boundary condition. The model included 1095 (3 years) time variable boundary conditions for the atmospheric boundary condition which included values for precipitation and potential evapotranspiration. The model was run for duration of 1095 days with minimum time steps of $1x10^{-5}$ days and maximum time steps of $1x10^{-3}$ days. The water flow model was van Genuchten-Mualem with no hysteresis (default settings). The soil profile was broken into a number of layers based on the profile encountered during field work at each of the sites as outlined in Chapter 4. Each of these layers contained materials with properties defined as per the calibration in the 2D inverse calibration described previously (see Table 7.1). The resulting soil pressure head profiles for the four sites as entered into the 2D model for contaminant transport are shown in Figure 1.2.

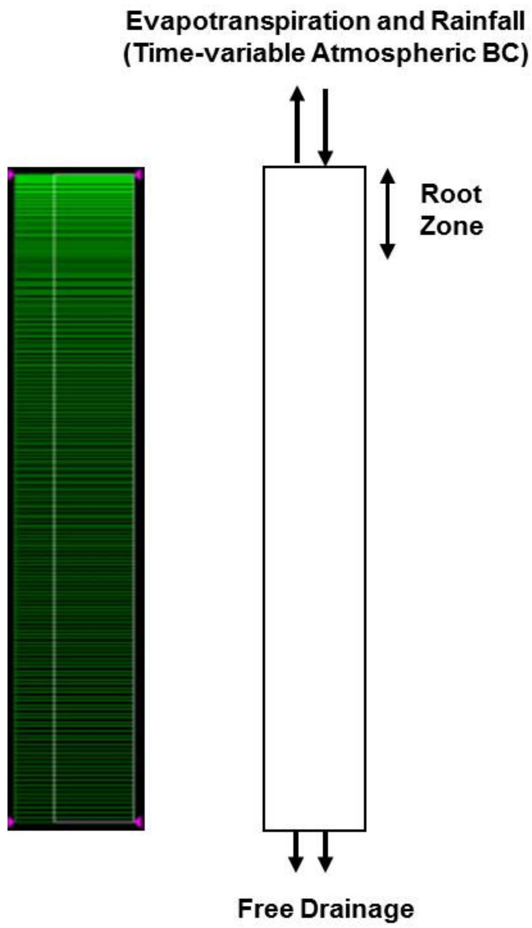


Figure 7.1 1D model profile showing nodal discretization and density (left) and boundary conditions (right)

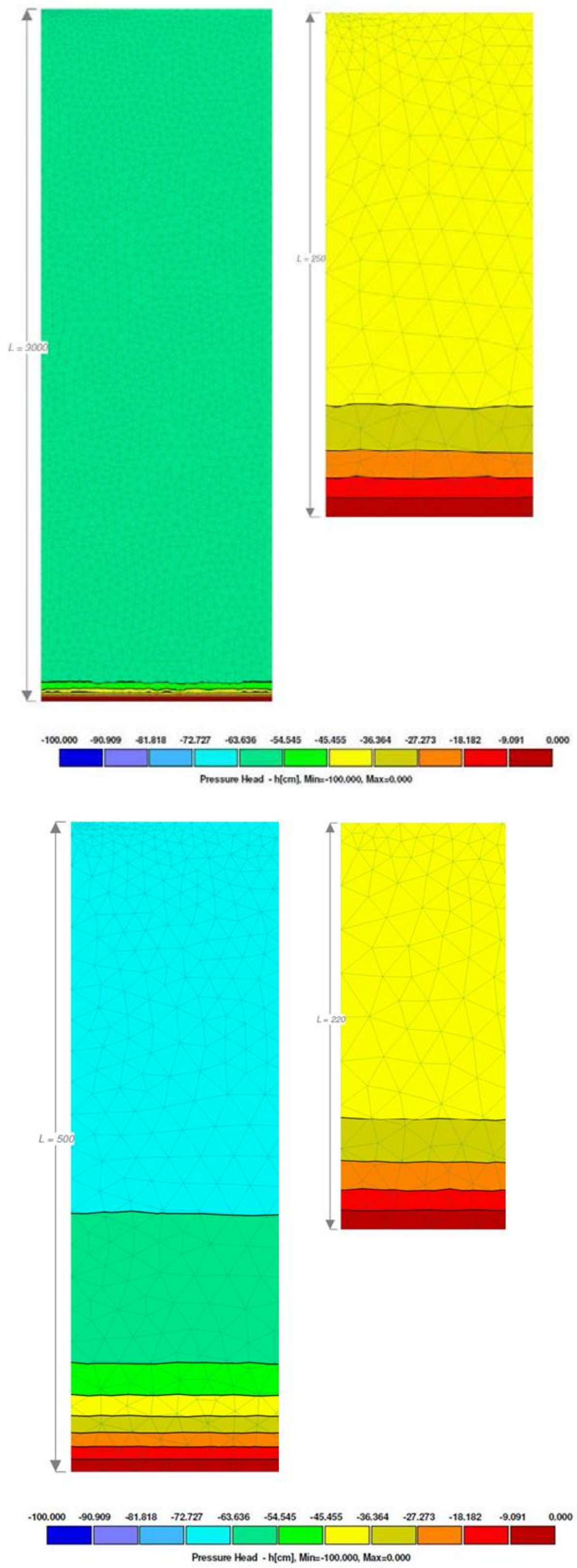


Figure 7.2 2D model pressure head profiles as entered based on the 1D model outputs for initialisation heads. (Top left = *LOW* vulnerability site; top right = *MODERATE* vulnerability site; bottom left = *HIGH* vulnerability site; bottom right = *EXTREME* vulnerability site)

2D direct model with water and solute transport

The Fickian-based convection-dispersion equation was used to model nitrogen and phosphorus transport. An additional attachment-detachment module contained within HYDRUS was also used for modelling the transport of bacteria. Solute transport regimes also incorporate provision for linear equilibrium adsorption, zero order production and first-order reduction for contaminant transport (Simunek, 1999). The governing equations were solved using Galerkin-type linear finite-element technique applied to a network of triangular elements referred to as a ‘mesh’. This finite element mesh was generated for each cross section (see Figure 7.3) using MESHGEN-2D which is a subroutine of HYDRUS (Simunek, 1999). MESHGEN-2D is a mesh generating code which designs boundary curves of computational domains for numerical modelling in Continuum Mechanics (Hassan et al., 2008). Finite element techniques provide solutions for irregular and time-dependent boundaries. The software also incorporates finite differences using Crank-Nicholson Scheme which minimise errors when integrating over small time increments.

The model domain was setup using the dimensions of a typical soil infiltration trench that would be included in an OSWTS that is disposing water to the subsoil. Two model scenarios were modelled for all of the study sites. The first was where a percolation area would be constructed as per the EPA Code of Practise consisting of 4 trenches of specific dimensions. In order to simplify the model, the trench and surrounding subsoil were assumed to be symmetrical about the centre line of the trench and thus only half of the domain geometry needed to be simulated. In order to allow for the possible interaction between two neighbouring trenches two trenches were modelled about their centre point with the prescribed distance between their centres as shown in Figure 7.3. The trench was assumed to be 450 mm in width (225 mm simulated) and the bottom boundary of the trench was taken to be 0.9 m below ground level. The vertical plane was extended down as far as the water table and this dimension varied for each study site. The second scenario modelled took account for the fact that in Ireland it has been observed that poor building practices have led to instances whereby only a single percolation trench has been installed instead of the required array of four parallel trenches. For this model scenario the full loading rate that would be divided between the four trenches is assumed to all be discharged to a single trench of the same dimensions as those given above.

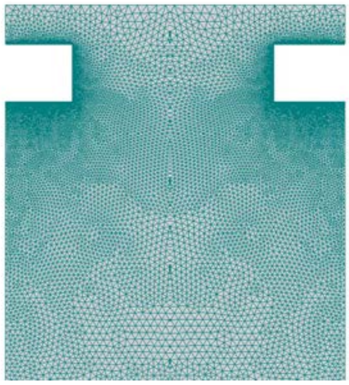


Figure 7.3 Finite element mesh for the 2D HYDRUS model (exaggerated scale)

The finite element mesh was setup with a targeted element size of 5 cm surrounding the trench with a coarser mesh size of 25 cm for the rest of the domain thus allowing for a finer and more accurate result close to the infiltrating water surface. The right-hand boundary was also set as ***No Flux*** with the bottom boundary set as ***Free Drainage***. The top boundary condition was ***Atmospheric with Surface Runoff***. The base of the trench was set to ***Variable Flux*** to represent the expected flow rate which will be described later. For solute transport the third type (Neumann) flux boundary condition was used and all of the boundary conditions used for these model setups are shown in Figure 7.4.

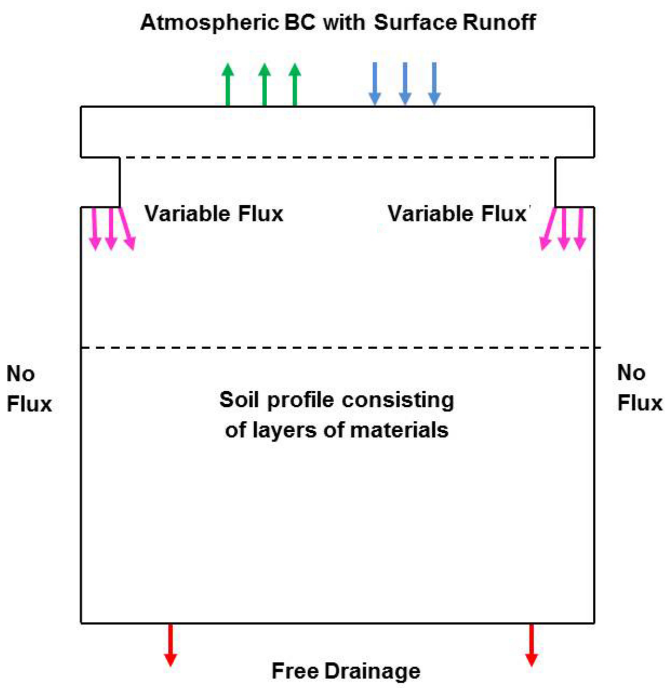


Figure 7.4 Boundary conditions for the 2D HYDRUS model

7.3 Solute Transport Theory and Parameters

For the solute transport modelling nitrogen and phosphorus were considered. Bacteria were also incorporated into the solute transport modelling using a specific attachment-detachment regime contained within the HYDRUS software. Transport parameters had to be set for each of these contaminants and the expected loading rates had to be considered in order to set the model inputs. Each of the solute parameters is described below. Models were run for two scenarios incorporating expected concentrations and loading rates for both Septic Tank Effluent (STE) and Secondary treated Effluent (SE).

7.3.1 OSWTS Hydraulic Loading rate

Accurate loading rates for the various contaminants entering the vadose zone from both Septic Tank Effluent (STE) and Secondary treated Effluent (SE) were available from two previous studies in Ireland by Gill et al. (2005; 2009). These studies included both the concentrations of contaminates and the daily discharges. The daily discharges varied from 70 – 130 l per percolation trench per day however for these simulations the figure from the EPA Code of Practice has been used which is 150 l per head per day and a conservative estimate of occupancy of 4 persons has been used. This discharge rate was then used to calculate the variable flux entering the soil from the base of the percolation trench based on the area assuming a 20 m trench length and a standard 450 mm trench width as shown in Figure 7.5. The calculated flux was therefore 1.667 cm/day. For SE it was assumed that the effluent only infiltrated along one third of the trench length due to the limited formation of the biomat layer. The formation of the biomat has been shown to be much muted in these conditions (i.e. for SE wastewater) due to much lower organic effluent strength (cf STE) as outlined by (Gill et al., 2005; 2009). In these instances the loading rate was increased to 5.005 cm/day and the area of recharge in MODFLOW was reduced accordingly.

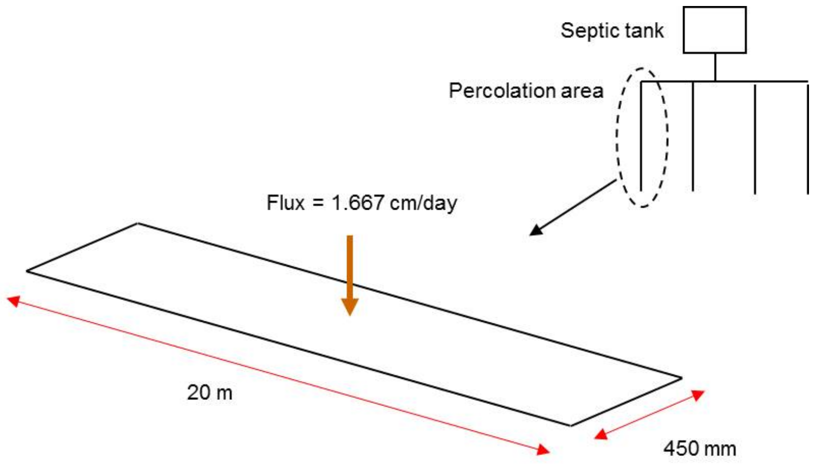


Figure 7.5 Conceptual loading rate from base of a percolation trench incorporating a standard 4 trench design and for STE wastewater

7.3.2 Nitrogen

Nitrogen was modelled as a solute chain reaction for three of its occurring forms as discussed and summarised in Chapters 4 and 5. The processes whereby nitrogen is transformed between its various forms both in the OSWTS and in the subsoil was discussed in detail in Chapter 2. Nitrogen occurs mainly in the form of organic-N and ammonium (NH_4) in Septic Tank Effluent (STE). These forms undergo a series of reactions in aerobic conditions of the unsaturated subsoil converting to NO_3 through nitrification and then N_2 or N_2O by denitrification processes. In general the nitrogen in Secondary Treated Effluent (SE) from on-site treatment processes will have already undergone nitrification and therefore tends to have far lower amounts of NH_4 and higher quantities of NO_3 when discharged to the percolation area. For simplicity the influence of fast reacting intermediate or minor reaction products such as NO_2 and N_2O were lumped in with the major compounds and this approach is widely used when modelling nitrogen in the vadose zone (Hanson et al., 2006; Hassan et al., 2008; Beggs et al., 2011). The solute chain reaction for nitrogen assumed for the model is therefore given by:

$$NH_4 \longrightarrow NO_3 \longrightarrow \begin{matrix} N_2 \\ N_2O \end{matrix}$$

Reaction rates for both nitrification and denitrification were assumed to be first-order as given below (Wagenet et al., 1977; Hanson et al, 2006):

$$\frac{\partial[NH_4^+]}{\partial t} = -\lambda[NH_4^+]$$

$$\frac{\partial[NH_3^-]}{\partial t} = \lambda[NH_4^+] - \mu[NO_3^-]$$

where: λ and μ are the nitrification and denitrification rate coefficients [T^{-1}] and t is time.

The solute transport equations contained within HYDRUS incorporate the effects of first order degradation independent of other solutes and also allows for first order decay/production reactions through coupling between solutes involved in the sequential first order chain. Many studies have been carried out previously using HYDRUS to simulate the nitrogen chain reaction described above. Nitrate and nitrogen were assumed to be present in the dissolved and gaseous phases only and therefore no adsorption to the solid phase was simulated (K_d = 0 cm^3 g^{-1}). Ammonium was assumed to adsorb to the solid phase using a distribution coefficient of 3.5 cm^3 g^{-1} with similar values having been reported by Hanson et al. (2006), Filipovic et al. (2012), Ramos et al, (2011) and Liu et al. (2012). Nitrification from ammonium to nitrate was simulated using a degradation rate (λ) as discussed above. A nitrification rate of 0.2 d^{-1} has been used in many previous studies and was based mainly on work by Hanson (2006, 2008). Beggs et al. (2011) used a degradation rate of 0.72 d^{-1} for a similar study however this was to account for microbial acclimation in the area of the study which had higher soil temperatures which are favourable for nitrifying bacteria. Data were available from previous vadose zone field studies where ammonium and nitrate rates were monitored at decreasing depths in the subsoil beneath the percolation trenches using lysimeters (Gill et al. 2005, 2009). This data was used empirically and with the HYDRUS software to determine a nitrification rate that is appropriate for Irish soil conditions which can be very different from those encountered in the Hanson (2006; 2008) studies. A summary of the data used to fit a nitrification rate is given in Table 7.2. A average first order degradation coefficient of 1.96 d^{-1} was found, however this was based on 4 values for sites that had similar subsoils to those in this study and 3 from sites that had a very different lithology (Sites A1, A2 and B) and therefore when averaging it was decided to apply a higher weighting to similar sites and a lower weighting to sites that had a very different lithology. The weighted averaging gave a nitrification rate of 0.7 d^{-1} and this value was used in this study.

Table 7.2 Summary of data fitted to obtain nitrification rate for simulations

	NH_4 conc. (mg-N/l)						
Site ID	Site 1	Site 2	Site 3	Site 4	Site A1	Site A2	Site B
Influent (0 m depth)	53.0	20.5	41.7	6.5	75	58	19.2
0.35 m depth[1]	9.9	5.8	5.1	3.8	10.2	2.7	0.8
0.65 m depth	5.1	1.5	7.2	3	4.9	2.5	0.4
0.95 m depth	5.8	-	3.8	2.1	1.4	1.2	0.6
Effluent Treatment	STE	SE	STE	SE	STE	SE	SE
Ks (m/d)	0.28	0.14	0.13	0.08	1.14	1.14	0.93
Fitted λ (d^{-1})	0.61	0.57	0.30	0.09	4.68	4.36	3.11
Average λ (d^{-1})	1.96						
Average λ (d^{-1}) with weight factors applied			0.7				

It was noted by Gill et al. (2005, 2009) that dentrification does not take place as readily in subsoils receiving SE due to the limited availability of organics. For this reason it was decided that two denitrfication rates should be used in the simulations with separate values for STE and SE. The first order degradation coefficient for denitrification of NO_3 to N_2 and N_2O was attempted to be fitted in a similar manner as for nitrification (described above) and the data used in this process is given in Table 7.3. This fitting process resulted in higher values than previously reported with denitrification rates μ = 0.53 and 0.296 d^{-1} for STE and SE respectively. The value used frequently in previous studies and noted in the literature is μ = 0.002235 d^{-1} which is orders of magnitude smaller than those obtained from the available Irish data. It must be assumed that beyond the initial first meter of the vadose zone the nitrification rate decreases dramatically with the limited availability of organic matter. In addition, the values calculated assumed saturated conditions (and therefore short travel times) which would not be the case in practice and this could explain their overestimation. The values calculated could not therefore be used as they would cause all of the NO_3 concentration to disappear within a few centimetres of subsoil which is not accurate. Similar studies have often omitted denitrification from the simulations due to the large sensitivity of denitrification rates on model results and the little information available in the literature on the sensitivity of denitrification rates as a function of soil moisture ((Hanson et al, 2006; Pang et al, 2006). However, it was decided to include denitrification in the simulations as it is an integral part of the nitrogen chain reaction and reasonable results have been achieved by Beggs et al. (2011) and Hassan et al. (2008). Due to the poor outcome of attempts to fit the Irish data and estimate reasonable values for both STE and SE the dentrifification rate of 0.002235 was used for STE and a value half that magnitude was used for SE due to the limited availability of organics in SE effluent which was based upon a comparison of the two values obtained from the fit above.

Table 7.3 Summary of data fitted attempting to fit a denitrification rate

	Total N at each depth (mg-N/l)						
Site ID	Site 1	Site 1	Site 1	Site 1	Site 1	Site 1	Site 1
Influent (0 m depth)	54.5	64.0	48.4	58.7	76.3	58.5	74.0
0.35 m depth[1]	14.5	59.2	31.3	53.7	37.6	39.6	72.7
0.65 m depth	12.8	57.9	30.1	50.5	36.7	34.2	57.6
0.95 m depth	12.0	-	11.7	46.9	31.6	27.4	57.6
Effluent Treatment	STE	SE	STE	SE	STE	SE	SE
Ks (m/d)	0.28	0.14	0.13	0.08	1.14	1.14	0.93
Fitted μ (d^{-1})	0.41	64.0	48.4	58.7	76.3	58.5	74.0
Average μ (d^{-1}) STE	0.53		Average μ (d^{-1}) SE		0.296		

A study has been carried out by Beltman et al. (1993) whereby the ionic concentration of rainwater was analysed over three periods in the west of Ireland. Rainwater was found to contain relatively significant concentrations of the contaminants of interest in this study and they were therefore included in the simulations and the nitrogen concentrations used in the simulations are given in Table 7.4 below.

Table 7.4 Nitrogen loading rates at base of percolation trenches used in the simulations (Gill et al. 2005, 2009)

Source	NO_3 conc. (mg-N/l)	NH_4 conc. (mg/l)
STE	1	74
SE	35.3	20.5
Rainwater	0.97	0.37

7.3.3 Phosphorus

Phosphorus has been found to be largely attenuated within the first metre of subsoil beneath the percolation trenches in the previous Irish field studies. The processes by which phosphorus is removed have been discussed in detail in Chapter 2. HYDRUS accounts for relatively complex processes of adsorption and cation exchange by means of empirical linear or nonlinear adsorption isotherms. The isotherm is governed by the distribution coefficient of the solute species K_d. Hanson et al. (2006) reported that there is a relatively large uncertainty concerning the value of the distribution coefficient (K_d) for phosphorus in the subsoil suggesting a range of values from 19 cm^3 g^{-1} – 185 cm^3 g^{-1}. As with the nitrogen specific parameters, available data were used to estimate a K_d value that was more appropriate to Irish subsoils for phosphorus. For all of the study sites examined in data obtained from Gill et al. (2005, 2009) PO_4 concentrations had reduced to an average of 1 mg-P/l regardless of the soil hydraulic properties. Therefore a simple trial and error HYDRUS calibration was undertaken with a single soil profile of unit depth in order to estimate a distribution coefficient that would match this value. A K_d value of 50 cm^3 g^{-1} was found to give results that gave a good match and therefore this value was used in these simulations. Concentrations of phosphorus in the effluent at the base of the percolation trenches are given in Table 7.5.

Table 7.5 Phosphorus loading rates at base of percolation trenches used in the simulations (Gill et al. 2005, 2009)

Source	PO_4 conc. (mg-P/l)
STE	16.6
SE	33.6
Rainwater	0.21

7.3.4 Bacteria

As with phosphorus, very high removal rates up to 5 log (99.999%) of bacteria have been found within the first meter of subsoil beneath the percolation trenches in previous Irish studies. The processes by which bacteria are removed have been discussed in detail in Chapter 2 and consist of filtration, straining, attachment and colony die off. Previous studies using HYDRUS to model the movement of bacteria in the vadose zone have focused on *E-coli* and usually involve comparisons with very controlled laboratory experiments. HYDRUS has been used in previous studies to successfully simulate the movement of *E-coli* and other bacteria and virus species through unsaturated porous media (Pang et al., 2006; Gargiulo et al., 2007; Jiang et al., 2010). No record was found in the literature of HYDRUS being used to model the transport of *Enterococci* in the vadose zone.

In order to accurately model the transport of bacteria in unsaturated conditions, the concept of attachment/detachment must be considered and accounted for within the solute transport reaction coefficient. HYDRUS can also be modified to include a mobile-immobile contaminant transport model (MIM). MIM is a two region mobile-immobile model which assumes that contaminant transport is limited to the mobile water region and that water in the immobile water region is stagnant, with a first-order diffusive exchange process between the two regions Due to the complex nature of these processes a large number of input parameters are required to set up and calibrate a model using these processes. Jiang et al. (2010) found that the attachment coefficient to the solid phase (K_a), the air-water interface attachment coefficient (K_{aa}) and the inactivation rate (μ) can be successfully lumped together to form a lumped total removal rate (λ). Pang et al. (2006) used a “lumped” first order degradation coefficient to simulate *E-coli* movement in the vadose zone with the effects of bacteria die-off and all other removal mechanisms lumped into this one parameter. There is a danger of introducing error into the model using lumped values, however given the complexity of processes involved for bacteria in the vadose zone it is considered reasonable to take this approach particularly if reasonable calibration results can be achieved against the field data. The lumped removal rate was therefore used for HYDRUS simulations involving bacteria in this study.

Jiang et al. (2010) used inverse HYDRUS models and laboratory data to optimise transport parameters for *Fecal Coliforms* (*E-coli*) and computed values for λ of 0.3 – 0.753 day^{-1}. Pang et al. (2006) used HYDRUS, which was calibrated using a sensitivity analysis, to model *Fecal Coliform* removal in an unsaturated gravel media and found that a value for λ of 3 day^{-1} gave satisfactory results. As a starting point a values for λ of 1 day^{-1} was used in the HYDRUS model and this was adjusted in order to match the data from previous studies beneath percolation trenches in Irish subsoils. A value of 0.7 day^{-1} was found to give the closest match to the previous field study data.

As for both nitrogen and phosphorus, previous studies had monitored the occurrences of *E-coli* in the percolating water at varying depths below the percolation trenches in Irish subsoils and this provided inputs for the numbers of bacteria colonies for the simulations. *Enterococci* occurrences had not been monitored in the vadose zone and no similar literature for Irish subsoils was available. Due to the unavailability of data for Enterococci it was assumed that these bacteria would behave similarly to *E-coli* and that their concentrations in the effluent would also be similar. Therefore for all simulations "Bacteria" will refer to both the *E-coli* and *Enterococ*ci concentrations. Concentrations of both bacteria species have therefore been taken from average values observed in previous Irish studies and are given in Table 7.6.

Table 7.6 Bacteria loading rates at base of percolation trenches used in the simulations (Gill et al. 2005, 2009)

Source	***E-coli*** **(cfu/100ml)**
STE	7.44×10^5
SE	2.5×10^4

7.4 Model Results

Initial HYDRUS models were run in 1D using 3 years of rainfall and evapotranspiration data that were available at each of the site locations in order to gain an understanding of the water content and head profiles in the soil profiles for use as initial conditions for the 2D models (i.e. model initialisation). 2D transient models were than set up as described earlier with these initial head profiles and the solute transport parameters outlined previously. As each of the sites have been in operation for extended periods of time, the 2D models were setup for a simulation period of 10 years (3650 days) in order to access the long term contaminant loads to the water table. Effective rainfall was input at the ground surface using annual average values (cm/day) calculated using the appropriate recharge coefficient and values for annual average evapotranspiration – this technique will be discussed in more detail in Chapter 8. Results of the model runs for each of the chosen contaminants are given below.

7.4.1 Nitrogen

The HYDRUS model was setup for two species of nitrogen (NH_4 and NO_3) and the simulation was run with both of these solutes. Whilst a denitrification sink term in both the liquid and gas phases were incorporated for NO_3 reaction parameters, the resulting N_2 and N_2O concentrations were not considered. As per previous field studies (see section 7.3.2) the HYDRUS simulations (for all four soil types and model setups) that NH_4 concentrations did not migrate further than 1.15 m below the base of the percolation trenches being completely removed mainly through nitrification to NO_3. Nitrate concentrations migrated significantly downwards and formed plumes beneath the percolation trenches after 2 years of the simulations for both STE and SE with differing maximum concentrations. The initial migration of the nitrate plume happened relatively quickly and the next 2 - 4 years of the simulation only saw a maximum of another meter of migration downwards with steady-state conditions occurring after approximately 4 years at all of the sites. Results of the resulting NO_3 concentrations at the water table interface are given in Table 7.7 below. Values are highest at the *MODERATE* and *EXTREME* vulnerability sites and this is due to the water table being present at a shallower depth with less unsaturated subsoil available between the base of the trench and the water table. As the water table at the *LOW* vulnerability site is at an average depth of 23 m below ground level, the simulations did not predict any NO_3 or NH_4 concentrations reaching that depth – even when only a single trench with a higher loading rate was considered. In order to investigate if the plume would continue to migrate downwards the model durations were all extended to 15 years (5475 days), however as steady state conditions had been achieved the plumes did not migrate any further into the subsoil. The ammonia and nitrate plumes at each of the sites after the initial model run duration of 3650 days are shown in Figure 7.6 – Figure 7.13.

Table 7.7 HYDRUS simulated average nitrate concentrations at the water table interface after 3650 days (10 years) for STE and SE sources

Location	4 Trenches		Single Trench	
	STE conc. (mg-N/l)	SE (mg-N/l) conc.	STE conc. (mg-N/l)	SE (mg-N/l) conc.
Naul – Low Vulnerability	0	0	0	0
Rhode – Moderate Vulnerability	32	24	42.1	28.7
Carrigeen – High Vulnerability	16	9.2	38.1	22.4
Faha – Extreme Vulnerability	34.6*	26.3*	44.3*	31.2*

**Assumed 1.2m of unsaturated subsoil beneath base of trench*

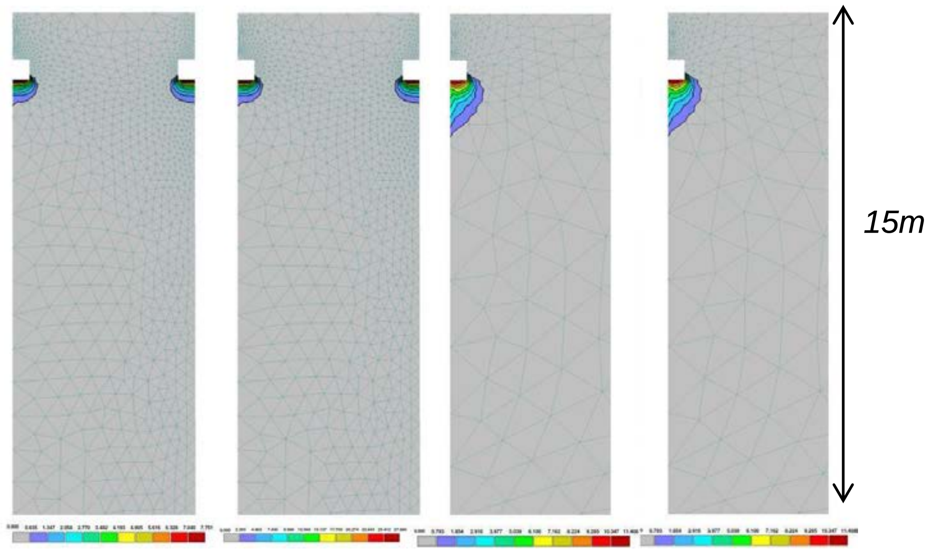


Figure 7.6 HYDRUS simulation output for Ammonia at the *LOW* vulnerability site (concentrations shown are in mg-P/l) (Left to right; 1 = STE; 2 = SE; 3 = single trench receiving higher loading rate STE, 4 = single trench with higher loading rate SE)

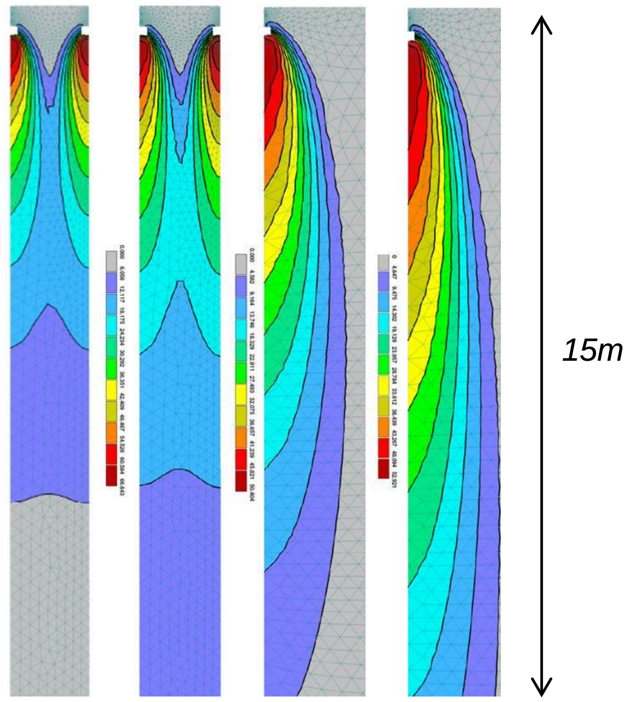


Figure 7.7 HYDRUS output for Nitrate at the *LOW* vulnerability site (concentrations shown are in mg-N/l) (Left to right; 1 = STE; 2 = SE; 3 = single trench receiving higher loading rate STE, 4 = single trench with higher loading rate SE)

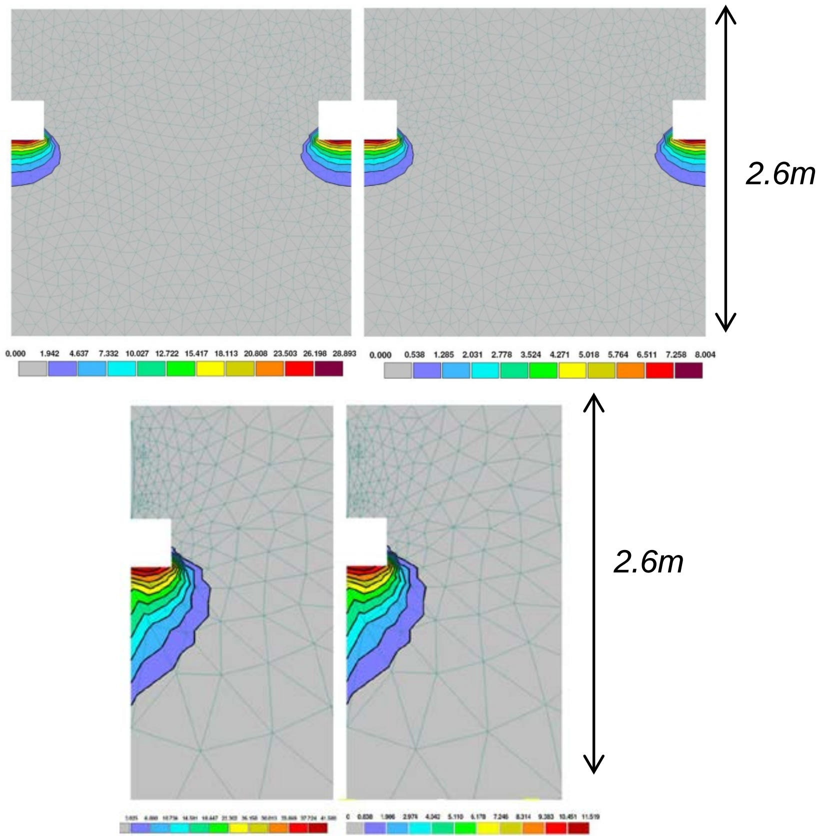


Figure 7.8 HYDRUS output for Ammonia at the *MODERATE* vulnerability site (concentrations shown are in mg-N/l) (top left = STE; top right = SE; bottom left = Single trench STE; bottom right = single trench SE)

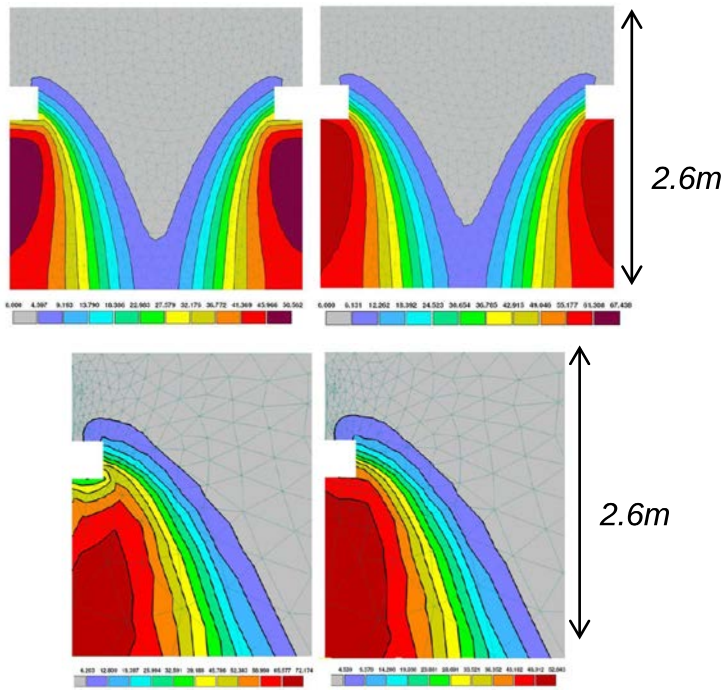


Figure 7.9 HYDRUS output for Nitrate at the *MODERATE* vulnerability site (concentrations shown are in mg-N/l) (top left = STE; top right = SE; bottom left = single trench STE; bottom right = single trench SE)

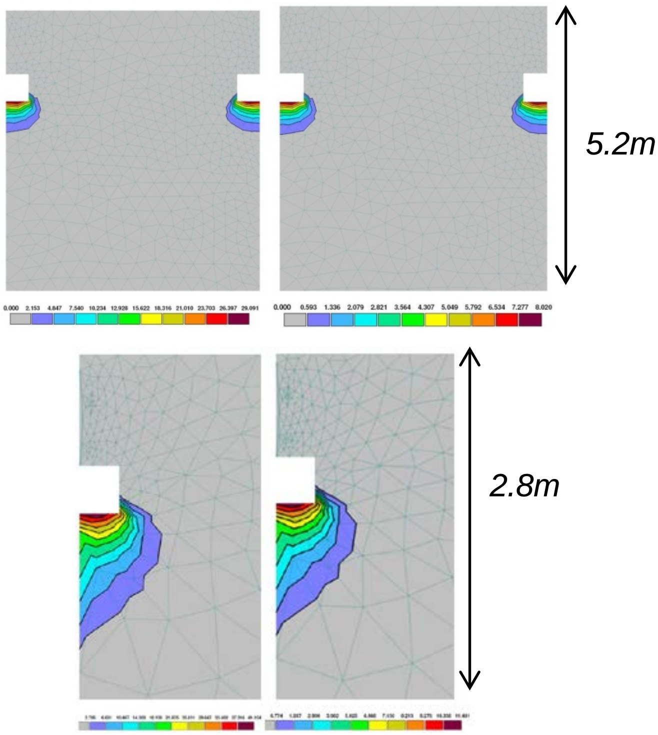


Figure 7.10 HYDRUS output for Ammonia at the *HIGH* vulnerability site (concentrations shown are in mg-N/l) (top left = STE; top right = SE; bottom left = single trench STE; bottom right = single trench SE)

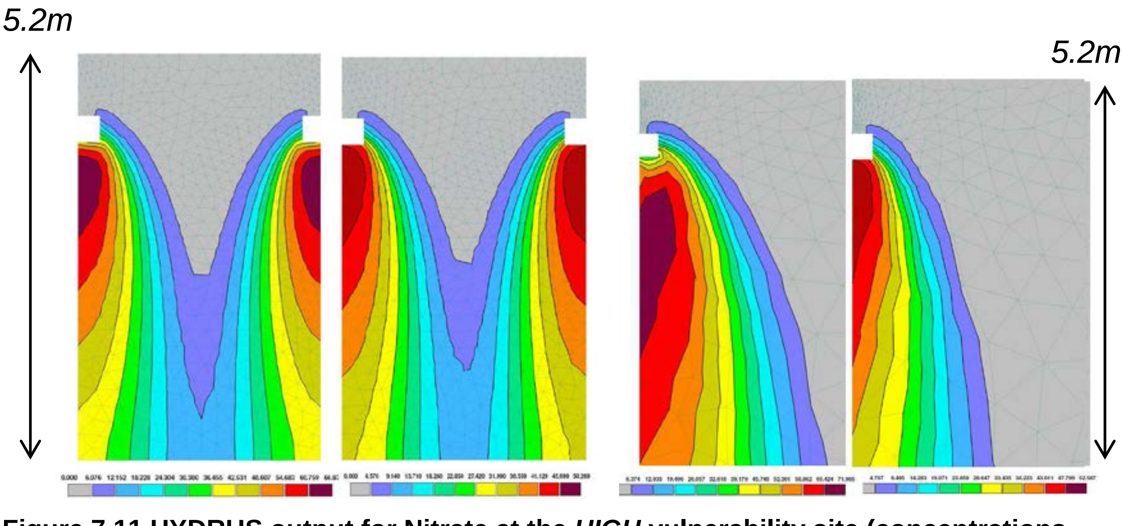


Figure 7.11 HYDRUS output for Nitrate at the *HIGH* vulnerability site (concentrations shown are in mg-N/l) (Left to right; 1 = STE; 2 = SE; 3 = single trench SE; 4 = single trench STE)

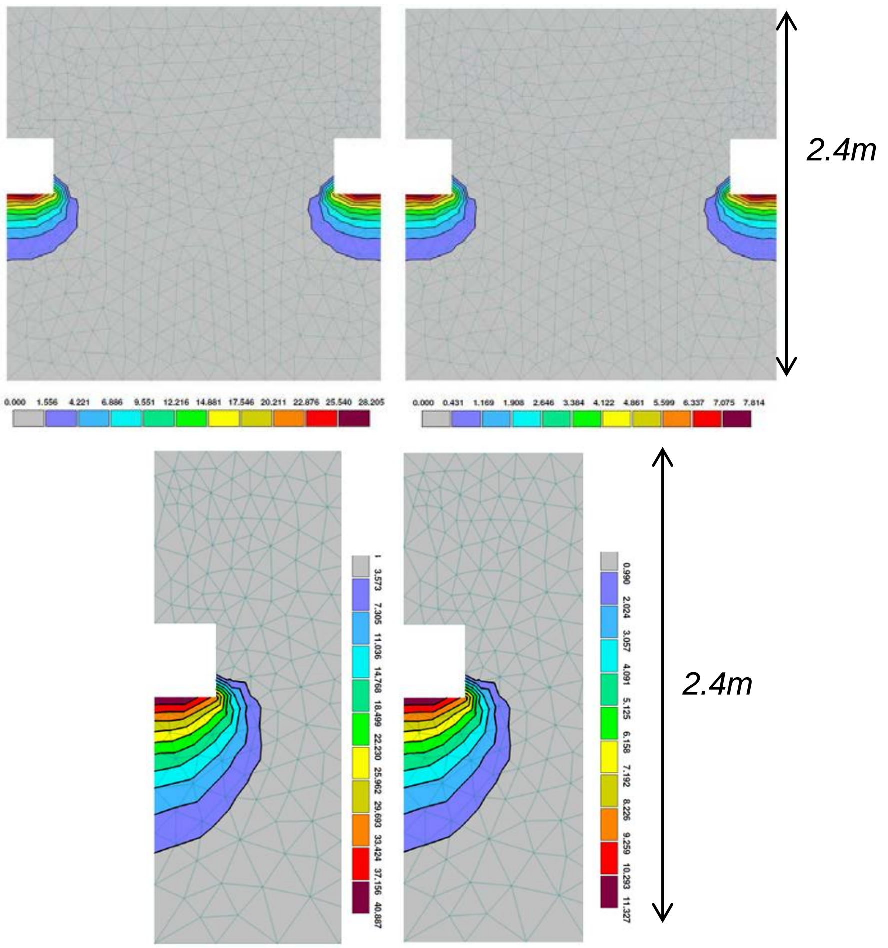


Figure 7.12 HYDRUS output for Ammonia at the *EXTREME* vulnerability site (concentrations shown are in mg-N/l) (top left = STE; top right = SE; bottom left = single trench STE; bottom right = single trench SE)

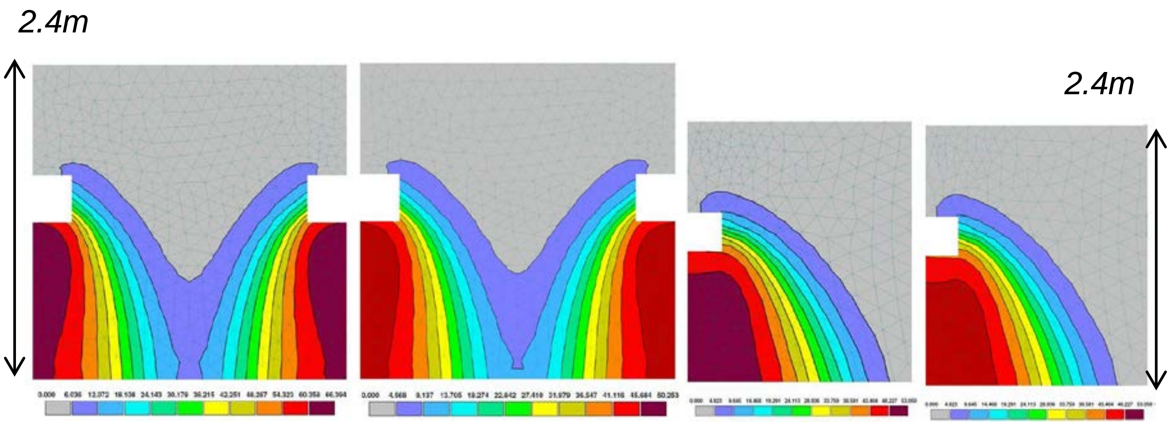


Figure 7.13 HYDRUS output for Nitrate at the *EXTREME* vulnerability site (concentrations shown are in mg-N/l) (top left = STE; top right = SE; bottom left = single trench STE; bottom right = single trench SE)

Calculation of the groundwater table flux

The purpose of these HYDRUS simulations was to provide inputs to the groundwater models (Chapter 8) and therefore the results had to be interpreted in a manner that allows an accurate representation to be made as inputs to the MODEFLOW and MT3D software. This was achieved for all of the contaminant species though cross-sections inserted at the groundwater interface. The cross-sections allowed both the concentrations (mg/l and CFU/l) and water velocities (cm/day) to be determined across the plume cross-section and therefore the input water flux was determined. Sample output from HYDRUS at one of these cross-sections is given in Figure 7.14. This method was used for nitrogen, phosphorus and bacteria water and concentration flux calculations. There were two flux areas modelled in HYDRUS – a cross-section between two trenches inside the percolation area allowing for interaction between the two plumes from neighbouring percolation trenches and also a cross-section of a trench at the outside of the percolation area where no interaction with an adjacent trench plume could take place. In practise the flux would vary across a section of the percolation area as shown in Figure 7.14 below, however for ease of input into MODFLOW MT3D an average flux was taken across a whole section of the percolation area and applied across the entire input area in MT3D – this is illustrated Section A-A. The process of calculating these fluxes is illustrated in Figure 7.14 below.

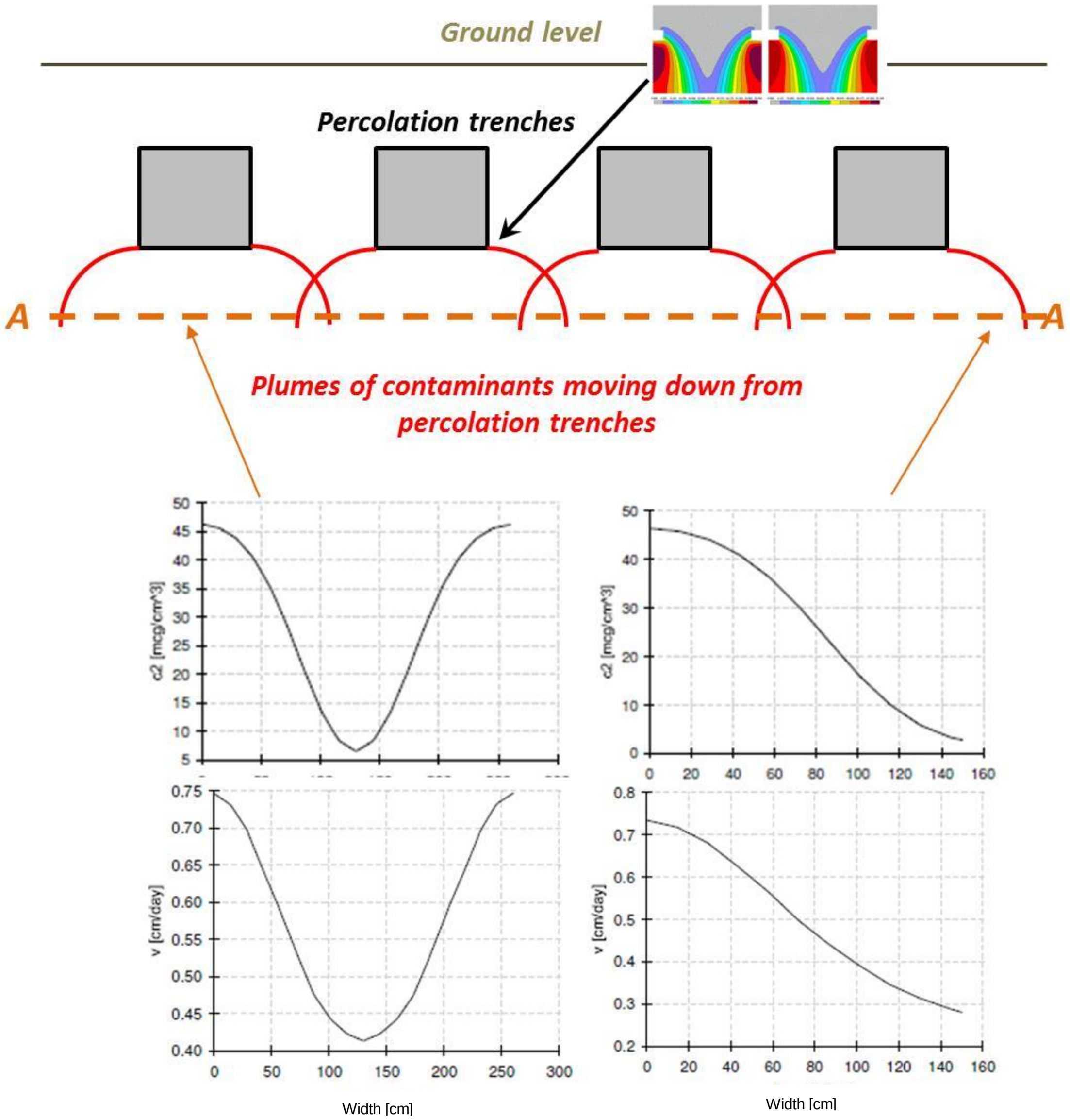


Figure 7.14 Output of HYDRUS simulated NO_3 concentrations for STE percolation water at the *MODERATE* vulnerability site (Offaly) at time = 2920 days (8 years), showing graphs of concentration and water velocity at the cross-section interface for both the inside "combined plume" and outside "single plume" scenarios. The average flux was calculated taking the average across the section marked A-A above.

7.4.2 Phosphorus

HYDRUS simulations resulted in average phosphorus concentrations at the water table interface as given in Table 7.8. The results of the HYDRUS simulations for the four study sites agreed well with data from previous Irish studies – it can be seen that the phosphorus concentrations are considerably higher if only a single percolation trench is simulated. The downward migration of the phosphorus plumes are shown in Figure 7.15 – Figure 7.18. As for the nitrogen results, concentrations at the water table interface were highest at the *MODERATE* and *EXTREME* vulnerability sites due to the shallower water table present at these locations.

Table 7.8 HYDRUS simulated average phosphorus concentrations at the water table interface after 3650 days (10 years) for STE and SE sources

Location	4 Trenches		Single Trench	
	STE conc. (mg-P/l)	SE (mg-P/l) conc.	STE conc. (mg-P/l)	SE (mg-P/l) conc.
Naul – Low Vulnerability	0	0	0	0
Rhode – Moderate Vulnerability	0.24	0.44	0.74	1.12
Carrigeen – High Vulnerability	0.008	0.003	0.11	0.19
Faha – Extreme Vulnerability	0.46*	0.97*	0.87*	1.31*

**Assumed 1.2m of unsaturated subsoil beneath base of trench*

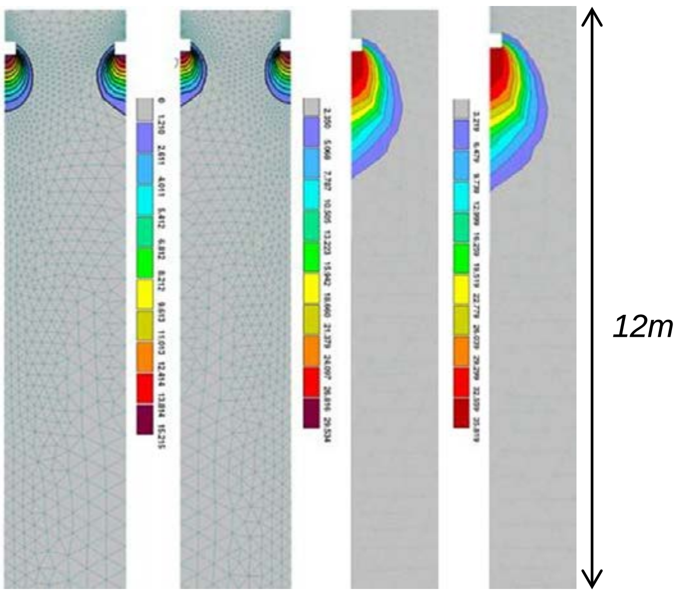


Figure 7.15 HYDRUS output for Phosphorus at the *LOW* vulnerability site (concentrations shown are in mg-P/l) (left to right; 1 = STE; 2 = SE; 3 = single trench receiving all the loading for STE, 4 = single trench receiving all the loading for SE)

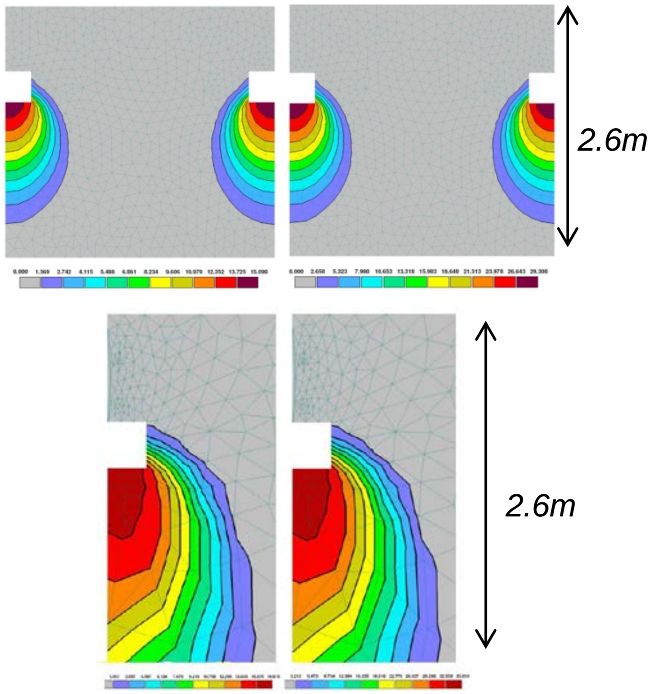


Figure 7.16 HYDRUS output for Phosphorus at the *MODERATE* vulnerability site (concentrations shown are in mg-P/l) (top left = STE; top right = SE; bottom left = single trench STE; bottom right = single trench SE)

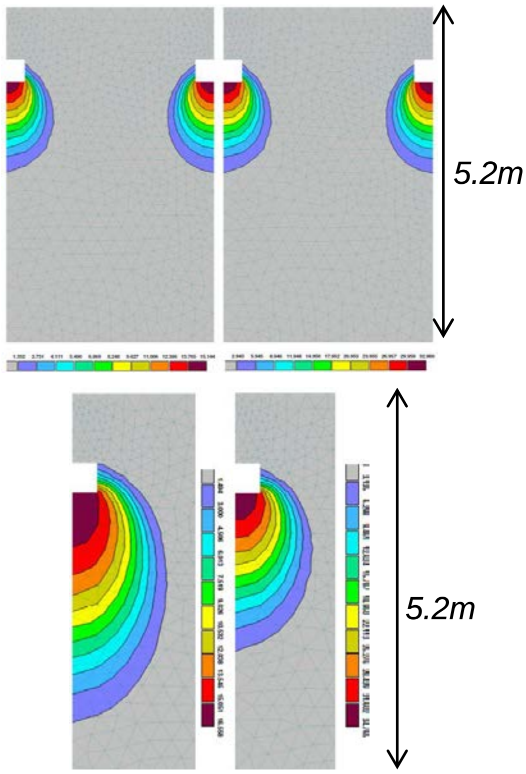


Figure 7.17 HYDRUS output for Phosphorus at the *HIGH* vulnerability site (concentrations shown are in mg-N/l) (top left = STE; top right = SE; bottom left = single trench STE; bottom right = single trench SE)

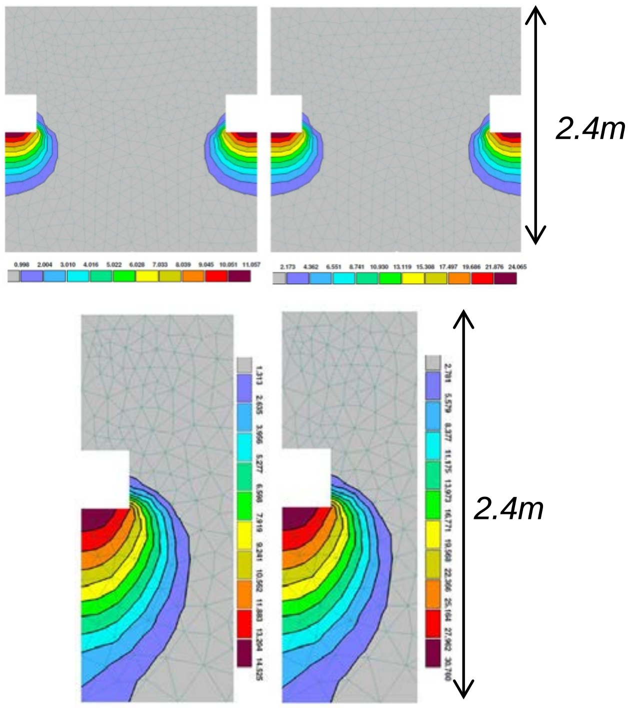


Figure 7.18 HYDRUS output for Phosphorus at the *EXTREME* vulnerability site (concentrations shown are in mg-N/l) (top left = STE; top right = SE; bottom left = single trench STE; bottom right = single trench SE)

7.4.3 Bacteria

Results of the HYDRUS simulations for both *E-coli* and *Enterococci* (hereafter jointly referred to as Bacteria) are given in Table 7.9. HYDRUS simulated that the vast majority of both bacteria groups were removed within the first metre of subsoil with very low numbers of either group reaching the water table. It must be noted that due to the limited information available on the transport of *Enterococci* in the unsaturated zone (and in groundwater) the values from the literature for *E-coli* have been used in all cases. This may in fact not be the case and the numbers of *Enterococci* entering groundwater of OSWTS's in the area may in fact be higher than those simulated and given in Table 7.9. In the absence of other input data this was the only method available however. The results of the HYDRUS simulations did however provide a good agreement with data from previous Irish studies for *E-coli*. HYDRUS outputs for bacteria are given in Figure 7.19 – Figure 7.21 below.

Table 7.9 HYDRUS simulated average bacteria concentrations at the water table interface after 3650 days (10 years)

Location	4 Trenches conc. (CFU/l)

	Single Trench conc. (CFU/l)	STE	SE
Naul – Low Vulnerability	0	0	0
Rhode – Moderate Vulnerability	38	1	0
Carrigeen – High Vulnerability	0	0	0
Faha – Extreme Vulnerability	422	40	11

**Assumed 1.2m of unsaturated subsoil beneath base of trench*

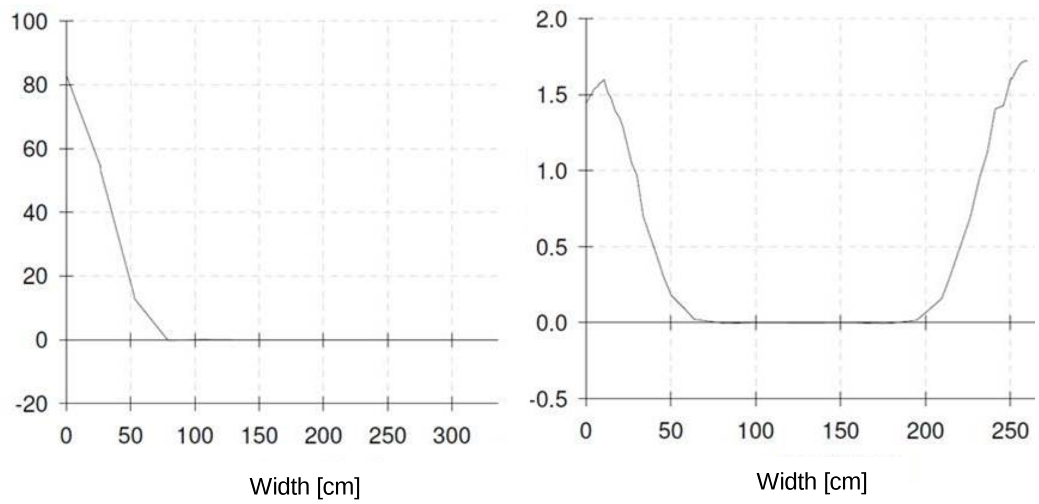


Figure 7.19 HYDRUS output for Bacteria at the *MODERATE* vulnerability site (concentrations shown are CFU/l) (left = single trench receiving all of the loading; right = two trenches)

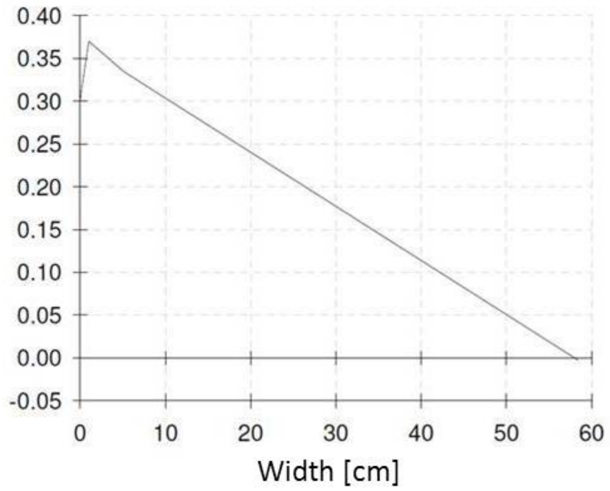


Figure 7.20 HYDRUS output for Bacteria at the *HIGH* vulnerability site (concentrations shown are CFU/l) (only single trench output shown; 4 trench output at water table was zero)

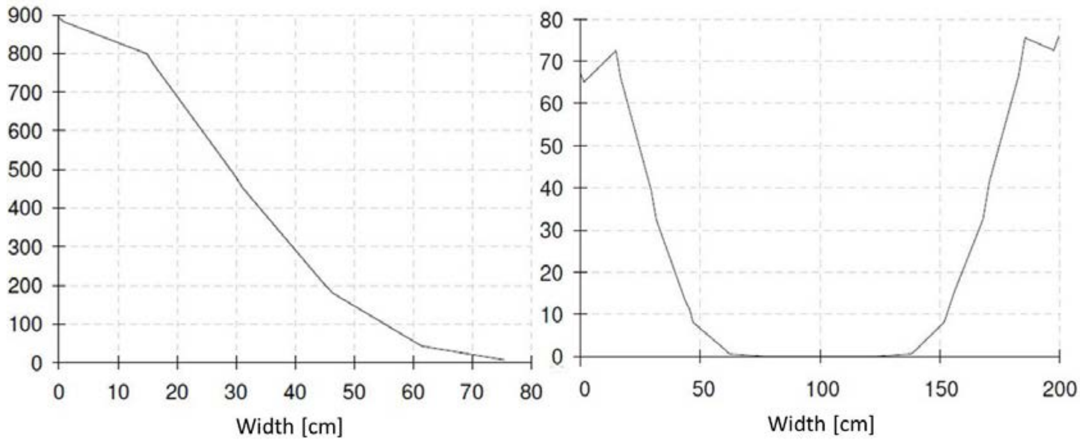


Figure 7.21 HYDRUS output for Bacteria at the *EXTREME* vulnerability site (concentrations shown are CFU/l) (left = single trench; right = two trenches)

7.5 Model Parameter Sensitivity Analysis

A sensitivity analysis of a number of the parameters used in the simulations detailed above was carried out in order to assess the robustness of the model. Parameters considered within this analysis were the nitrification (λ) and denitrification (μ) rate coefficients, the bacteria removal rate (λ) and the loading rate of the incoming percolation water. As discussed in Section 7.3.2 , an exercise was undertaken with Irish field study data to estimate specific nitrification and denitrification rates for Irish subsoils. This process achieved limited success with a nitrification rate obtained for use within the simulations however the process did not produce a satisfactory denitrification rate due to some of the limitations discussed previously. During the sensitivity analysis the rates that were used in the simulations were varied by ±20% to investigate the effects on the model outputs. The effects of varying these rates can be seen for the *EXTREME* vulnerability site can be seen in Figure 7.22 below. The denitrification rate was kept fixed at μ = 0.002235 whilst varying the nitrification rate and similarly the nitrification rate was kept fixed at λ = 0.7 whilst varying the denitrification rates. All other parameters were held fixed during the sensitivity analysis other than the parameter being varied. The sensitivity analyses were carried out for the *EXTREME* vulnerability site which represents a worst case scenario.

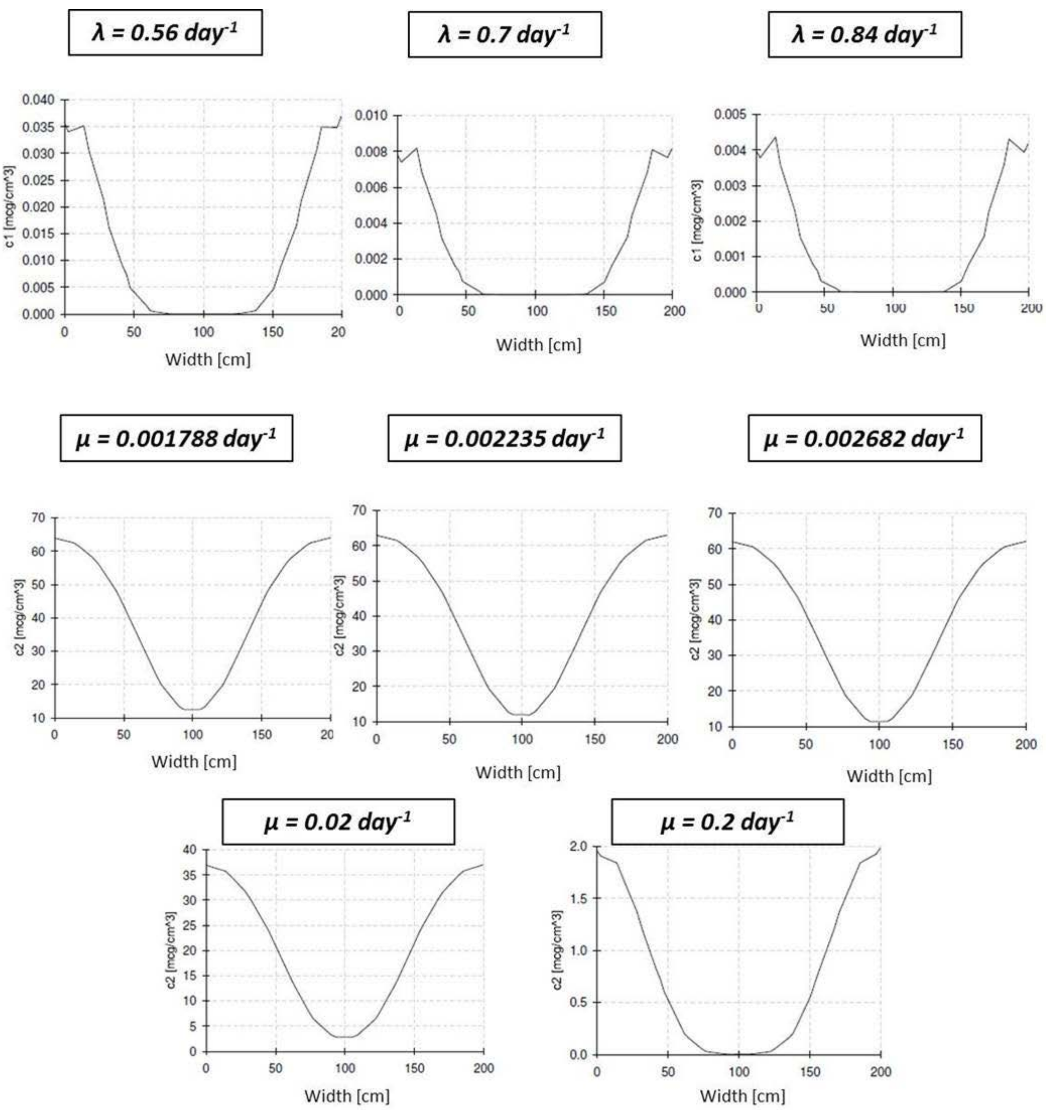


Figure 7.22 HYDRUS output for sensitivity analysis changing denitrification and nitrification rates by ±20% at the *EXTREME* vulnerability site. Also shown is the effect of changing denitrification by one and two orders of magnitude

The sensitivity analysis demonstrated that changing the nitrification rate by +20% did not have a significant effect on the model output with concentrations of ammonia at the water table only decreasing by 2 µg-N/l – which is negligible. However, when the nitrification rate was reduced by 20% the effect was to increase ammonia concentrations at the water table by 27 µg-N/l, although this concentration is still very low. Increasing and decreasing the dentrification rate by 20% had almost no impact on nitrate concentrations at the water table with concentrations varying by ±3 mg-N/l from a base of 62 mg-N/l – a small proportion of the overall load. Additional model runs with denitrification rates increased in orders of magnitude are also shown in Figure 7.22 above. It can been seen that when a nitrification rate of 0.02 day^{-1} was used maximum concentrations at the water table fell from 62 mg-N/l to 37 mg-N/l and when this rate was increased further to 0.2 day^{-1} this fell from 62 mg-N/l to 1.9 mg-N/l. These additional model runs were carried out to demonstrate some of the difficulties that were encountered in Section 7.3.2 previously and show that denitrification rates can only fall into a narrow band of magnitude in order to match the site data from previous Irish studies.

The effects of varying the bacterial "lumped" first order removal rate are shown in Figure 7.23 below; the rate was varied from 0.6 – 0.8 day^{-1}. It can be seen that when the removal rate was reduced to 0.6 day^{-1} maximum bacteria concentrations at the water table on the Extreme vulnerability site increased from 81 to 155 CFU/l; almost twice the concentration for the 0.7 day^{-1} rate that was used in the simulations. Similarly increasing the removal rate from 0.7 to 0.8 day^{-1} reduced maximum concentrations to 39 CFU//. It can therefore be seen that this removal rate is highly sensitive to small changes and it is therefore vital that this rate accurately represents what occurs in the subsoil. Given that the rate used in the simulations was calibrated based on the results of previous Irish field studies the results of these simulations would appear to be representative particularly for the purposes of this study.

There has been much debate in Ireland over recent years as to an appropriate loading rate per head per day to assume for the design of OSWTS. The rate was as high as 220 l/head/day but was reduced to 150 l/head/day in the recent EPA CoP (EPA, 2009). Research by Gill et al. (2005; 2009) indicated that this loading rate could be as low as 100 l/head/day and therefore five different loading rates were selected and simulated for the sensitivity analysis. The loading rates simulated were 100, 120, 140, 180 and 220 l/head/day along with the figure that was used in the models of 150 l/head/day. The results of this analysis are given in Figure 7.24 – Figure 7.25 below.

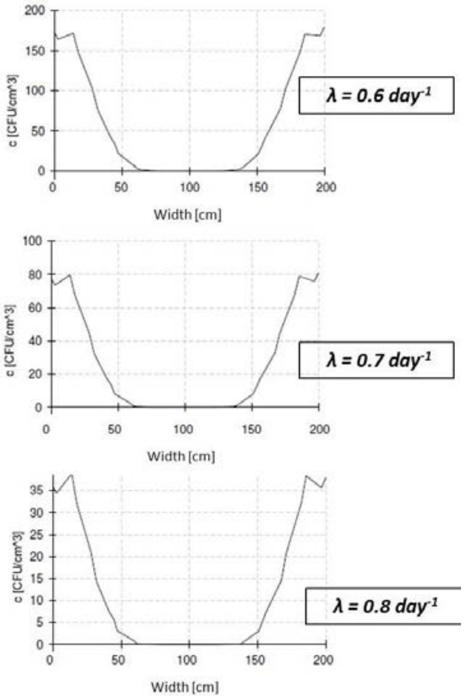


Figure 7.23 HYDRUS output for sensitivity analysis changing bacteria removal rate at the EXTREME vulnerability site from 0.6 – 0.8 day^{-1}

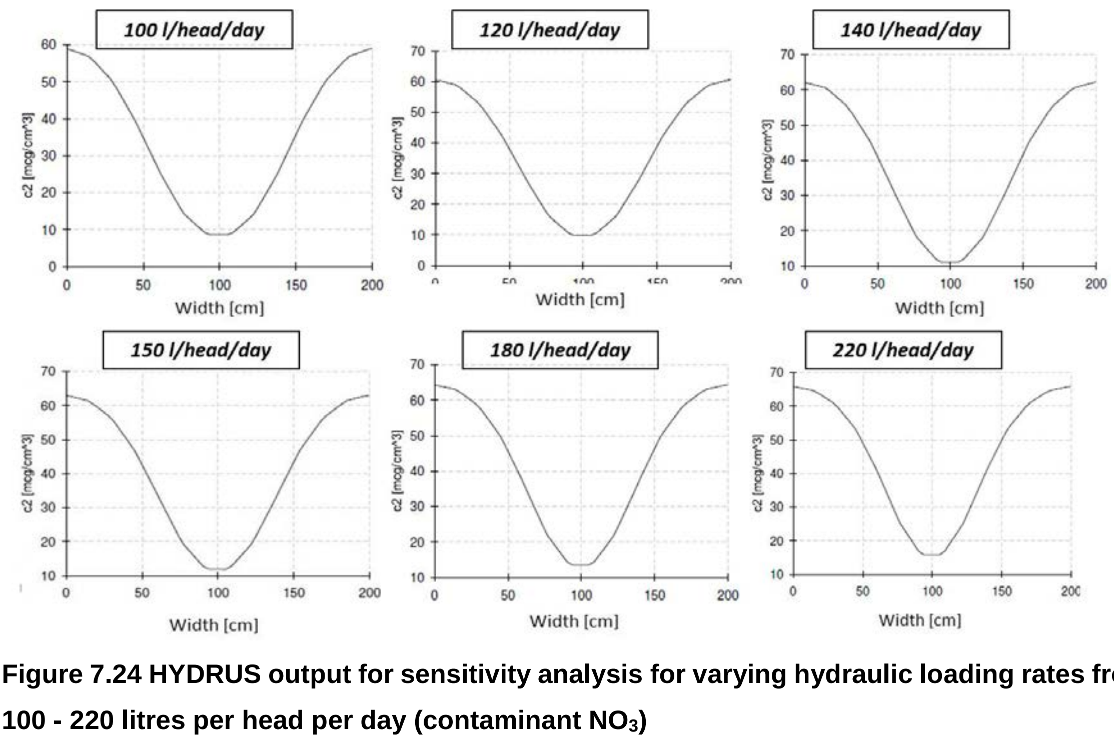


Figure 7.24 HYDRUS output for sensitivity analysis for varying hydraulic loading rates from 100 - 220 litres per head per day (contaminant NO_3)

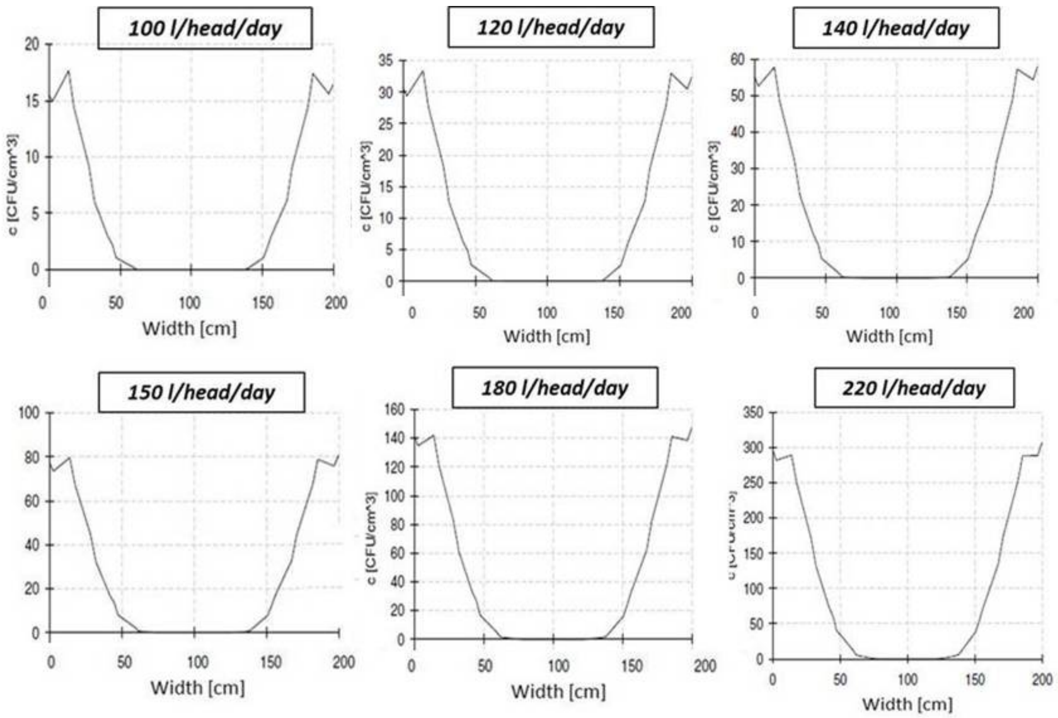


Figure 7.25 HYDRUS output for sensitivity analysis for varying hydraulic loading rates from 100 - 220 litres per head per day (contaminant bacteria)

It can be seen from Figure 7.24 that changing the loading rate has very little impact on the maximum NO_3 concentrations, with a loading rate change from 100 – 220 l/head/day only resulting in an increase in nitrate concentration at the water table of 7 mg-N/l. However, when the loading rates are varied for bacteria it can be seen that this has a very significant impact on concentrations at the water table as illustrated in Figure 7.25. Increasing the loading rate from 100 – 220 l/head/day results in a change in maximum bacteria levels from 18 CFU/l to 290 CFU/l. It can been seen in Chapter 8 that the concentration of bacteria entering the aquifer has a very significant impact on the development of bacteria plumes and therefore this is of great interest. It was appropriate for this study to use the EPA recommended loading rate of 150 l/head/day; however, given the results detailed above, it is possible that the actual contaminant load to groundwater is much lower than had been assumed previously if the hydraulic loading rate is in fact more in the region of 100 l/head/day. In addition water saving devices are increasingly being incorporated into new builds and modern appliances now use much less water (EPA, 2010) and therefore it is more likely that flow rates will decrease in the future.

8 GROUNDWATER MODEL AND RESULTS

8.1 Introduction

A conceptual model of the four study catchments was developed as described in detail in Chapter 6. In this chapter numerical groundwater flow regimes have been developed for each of the study areas using fully distributed, finite difference modelling software. The development of these groundwater flow models will give a good approximation of groundwater flow at the study sites and these models will then be used to predict contaminant levels arising in the groundwater bodies due to decentralised treatment systems in the study areas. These models can then be calibrated using the data collected as part of this study (Chapter 6). Various scenarios can then be simulated involving varying type and density of decentralised treatment systems in order to predict the effects that they can have on groundwater quality.

In order to develop a distributed finite difference model for the study catchments, it is first necessary to determine the quantity of data required to effectively set-up and calibrate the model. The model set up will require information on the spatial distribution of the following parameters:

- Hydraulic conductivity/Aquifer storage properties
- Subsoil permeability
- Aquifer recharge
- Aquifer head data at discreet points in the study catchment

It can be very difficult to acquire these data without extensive and intrusive field work and it is often necessary to approximate many of these values based on previous studies and appropriate values from literature. A more detailed description of the approach used to acquire the necessary data to populate the finite difference model is given in Chapter 6.

Background to Model Software (MODFLOW)

Analytical solutions to the partial-differential equation (PDE) of groundwater flow are normally only possible for very simple problems and therefore various numerical methods to approximate solutions have been employed (Harbaugh, 2005). The two most commonly used numerical techniques to solve the groundwater flow equation (see Chapter 2) are finite difference and finite elements. The finite difference method replaces the continuous system given in equation [Eq. 2.16] with a finite set of discrete points in space and time with differing head values being calculated at these discrete points (Harbaugh, 2005). A system of simultaneous linear algebraic difference equations can then be used to approximate a solution of the partial-differential equation of flow. MODFLOW, one of the most widely used three-dimensional numerical groundwater flow models originally developed by the U.S. Geological Survey (McDonald and Harbaugh, 1988) employs the finite difference method as illustrated in Figure 8.1. It should be noted that surface runoff and unsaturated flow are not included in the MODFLOW groundwater flow model.

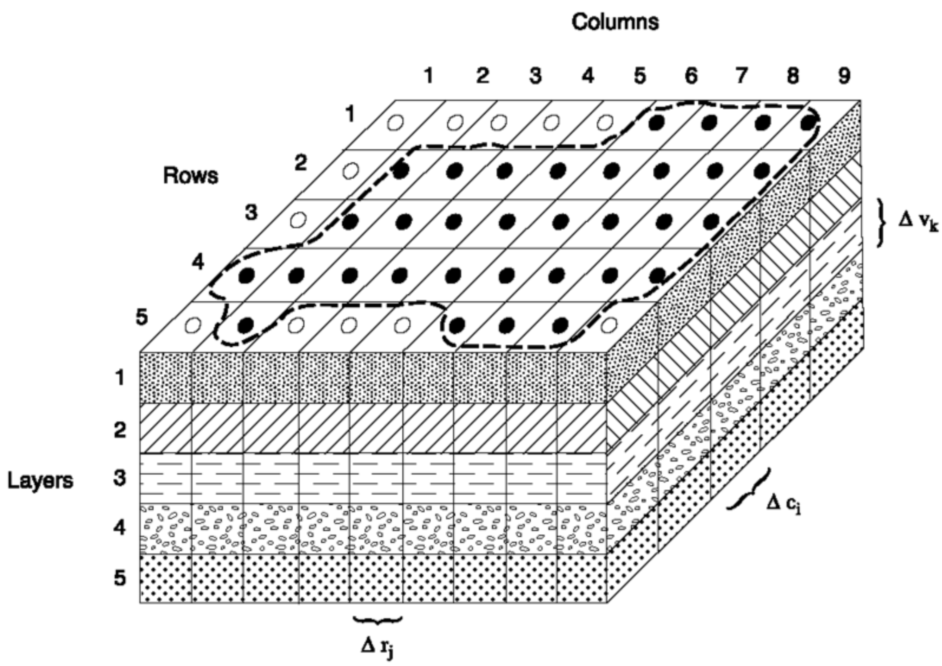


Figure 8.1 Three-dimensional finite difference grid used in MODFLOW

The ability to solve the partial-differential equation (PDE) of groundwater flow is given by the finite-difference equation developed by Harbaugh (2005) and expressed as [Eq. 8.1]:

$$\sum Q_i = SS \frac{\Delta h}{\Delta t} \Delta V$$

[Eq. 8.1]

Where:

Q_i is a flow rate into the cell (L^3T^{-1})

SS is Specific Storage - the volume of water that can be injected per unit volume of aquifer material per unit change in head (L^{-1})

ΔV is the volume of the cell (L^3)

ΔH is the change in head over a time interval of length Δt

For example, if two cells within an aquifer are considered as shown in Figure 8.2, then Darcy's law can be used to sum the flow into the cell i,j,k from cell i,j-1,k (McDonald and Harbaugh, 1988). If this is then combined with flow from the other five adjacent cells an estimate for the change in head in a single cell can be made for a given steady state set of parameters.

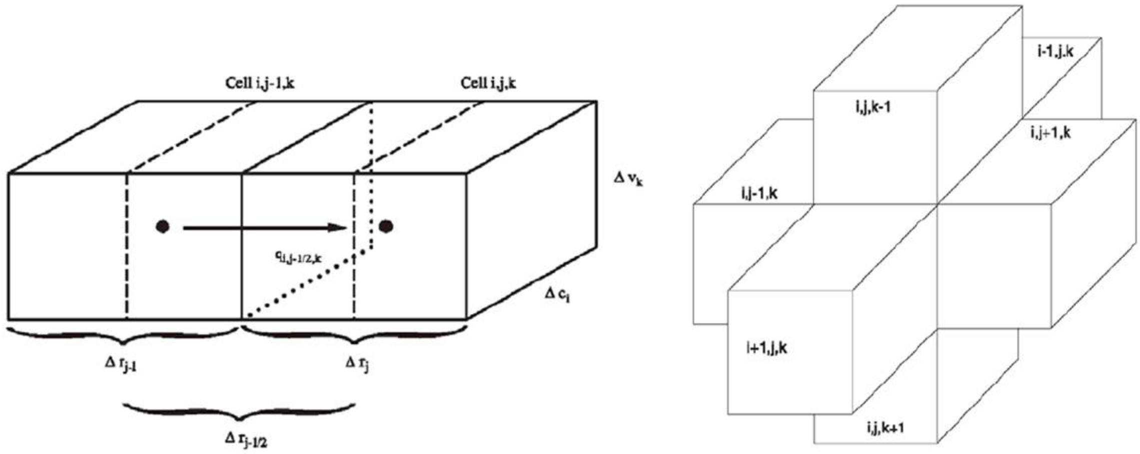


Figure 8.2 (a) Flow in two cells of a finite difference grid (b) the six adjacent cells to cell i,j,k hidden at the centre (Harbaugh, 2005)

In addition to calculating the head in each cell within the boundaries of a defined finite-differences grid, MODFLOW also contains packages that can handle flow to and from rivers, drains, wells and other defined boundary conditions (Environmental Simulations, 2011). As shown in Figure 8.1, the model grid is broken up into cells that can having varying properties and is also further broken up into layers which can represent different aquifer units and can be assigned varying values of storage and conductivity or other properties of interest (Environmental Simulations, 2011). MODPATH is a particle tracking post-processing package for MODFLOW which enables the user to track the travel of particles within the aquifer model (Harbaugh, 2005). Many of the groundwater modelling software packages available use parts or modified forms of the MODFLOW code and it can therefore be viewed as the 'industry standard' for groundwater flow modelling. Many companies have developed user interfaces for the ease of use of the MODFLOW packages which can usually run additional packages such as MODPATH and MT3D (Wang and Anderson, 1982). In order to implement the MODFLOW package in conjunction with these additional software packages it was necessary to utilise a software suite called Ground Water Vistas. Ground

Water Vistas contains a Graphical User Interface (GUI) that allows for simple implementation of all of these software packages and manages the associated files to avoid tedious and repetitive tasking during the modelling process.

8.2 Model Setup and Input Data Preparation

8.2.1 Input Data Preparation

The software package chosen to carry out the groundwater flow and contaminant transport modelling allows for the easy input of data through the use of spatially distributed shapefiles. Data were obtained as inputs for the groundwater modelling software from a number of sources including; the EPA, the GSI and the Ordnance Survey of Ireland (OSI). In addition, data were also acquired during the project itself through the review of Local Authority planning files and field measurements such as topographical surveys. Due to the many sources of data and the specific input requirements of Groundwater Vistas, a large amount of data manipulation was required in order to get all of the data into a usable and supported format. This section will give a brief explanation of the main data input preparations involved.

Topographic input files

The most suitable input for topographic information is a Surfer grid file; however topographic information was available in both a Digital Elevation Model (DEM) contained within a raster file (TIFF) and also in comma separated value text files (CSV) from the topographic surveys of the study areas and the stream and river beds. ArcGIS was used to first convert both data sources to the same file format (shapefiles) and then merge the two files together. The resultant file was then exported to a new format (CSV) which could be imported to Surfer 10. Surfer 10 was then used to sample this x,y,z data and fill a grid using Kriging interpolation at 20 m intervals. This interval was chosen due to the limitation of the DEM being at a resolution of 20 m as illustrated in Figure 8.3 (i.e. the DEM raster file contained a TIFF with an elevation at the centre of each 20 x 20 m cell). The resultant Surfer grid was then converted back to CSV format for editing in Excel – this will be described in more detail below under the depth-to-bedrock input file preparation heading.

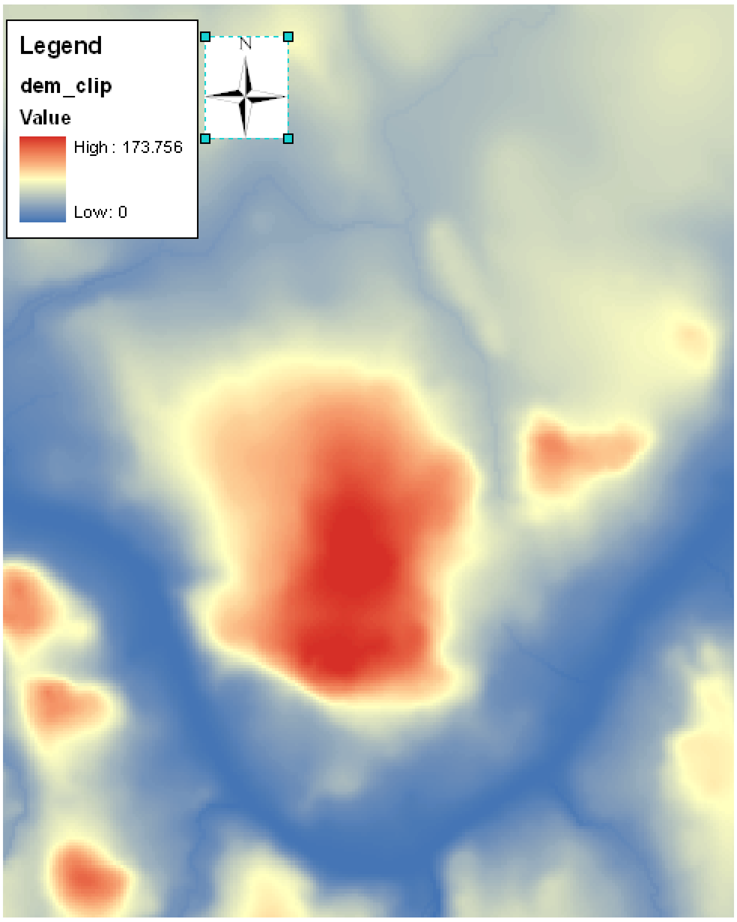


Figure 8.3 DEM Raster File of Site at Carrigeen, Co. Kilkenny

<u>River/stream input files</u>

A shapefile is required for the groundwater model containing information on rivers and streams. This shapefile must contain the path of the watercourse as a polyline. However, to improve the accuracy of the model this polyline should be broken up into as many segments as possible with each segment containing information on; water level at its start and end, river/streambed elevation, width of the water body and the hydraulic conductivity or permeability of the river/streambed. A shapefile was available containing the watercourse paths as polylines. This shapefile was converted to a format that could be imputed to AutoCAD (DWG) in order to break each of these polylines up into smaller segments based on the available elevation data from topographic surveys. Each of the smaller watercourse segments were then assigned an ID using AutoCAD and then the file was converted back to shapefile format and input into ArcGIS. The topographic information obtained from the survey of the river/streambeds was the loaded into ArcGIS having been assigned corresponding ID tags in excel. The two files were then merged using the spatial join command in ArcGIS - see in Figure 8.4.

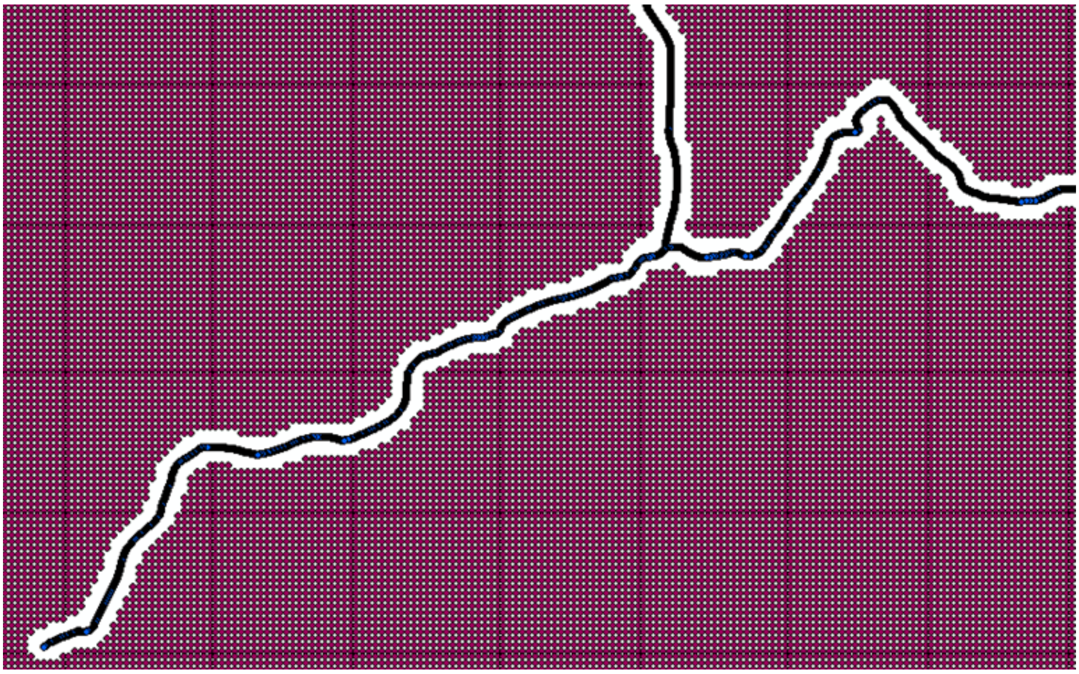


Figure 8.4 DEM File in Point (x,y,x) format with stream buffer removed and surveyed interpolated river points included at Carrigeen, Co. Kilkenny

Depth-to-bedrock input files

Depth-to-bedrock data is required in order to create the required layers inside the groundwater model. Depth-to-bedrock information is used in conjunction with the topographic information described above in order to create layer elevations. Having already created the topographic files the depth-to-bedrock files were created in a similar fashion, however the process was more complicated due to the format of the available data files. Depth-to-bedrock information was created in x,y,z file format (CSV) with the input data extracted from GSI borehole information and rockhead data acquired during the course of this study from both TCD drilled boreholes and information about existing local wells and boreholes. A generalised depth-to-bedrock file was available from the GSI; however this data was stored as a shapefile and was more intended as a visual aid rather than to be used for populating a digital elevation grid. In order to extract useful data from the GSI bedrock map, a number of file conversions and operations had to be performed. The data was first converted from a polyline shapefile to a raster TIFF file and then from a raster to a point file – see Figure 8.5 below. It is important to understand how useful information was stored within the original shapefile. Each of the polylines represents a border between two zones of bedrock depths, for example between an area with a depth to bedrock of less than 1 m (<1) and an area with a depth-to-bedrock or between 1 m and 3 m (1-3 m). In this context the polylines contained in the original GSI file represented a continuous "line" where the depth-to-bedrock was at one single value therefore for the previous example a polyline between the <1 m zone and the 1-3 m zone would represent a continuous line where depth-to-bedrock is exactly 1 m.

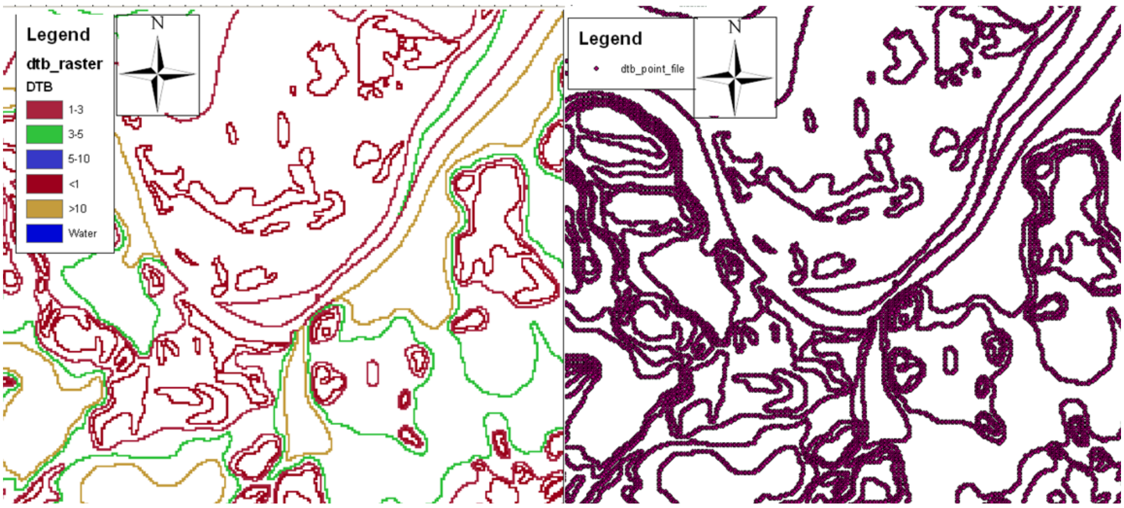


Figure 8.5 (a) Depth to Bedrock file in raster format (b) Depth to Bedrock file in point (x,y,z) – both at Carrigeen, Co. Kilkenny

Similarly this logic can be applied at all of the boundaries between the other depth-to-bedrock zones such as 3-5 m, 5-10 m and >10 m. Converting the shapefile to a raster, samples the data along each of the polylines and applies one value based on the adjacent depth-to-bedrock zone. Once this raster was then converted to a point file and x,y values were added, it was exported to Excel whereby the data was analysed based on its assigned bedrock zone and assigned a value of depth-to-bedrock. This table of x,y,z data was then merged with a previously created CSV file containing the extra depth-to-bedrock data and the resultant file was imported into Surfer 10; an example of a Surfer 10 projection can be seen in Figure 8.6. This x,y,z data was then used to fill a grid using Kriging interpolation at 20 m intervals in order to be consistent with the topographic information created previously.

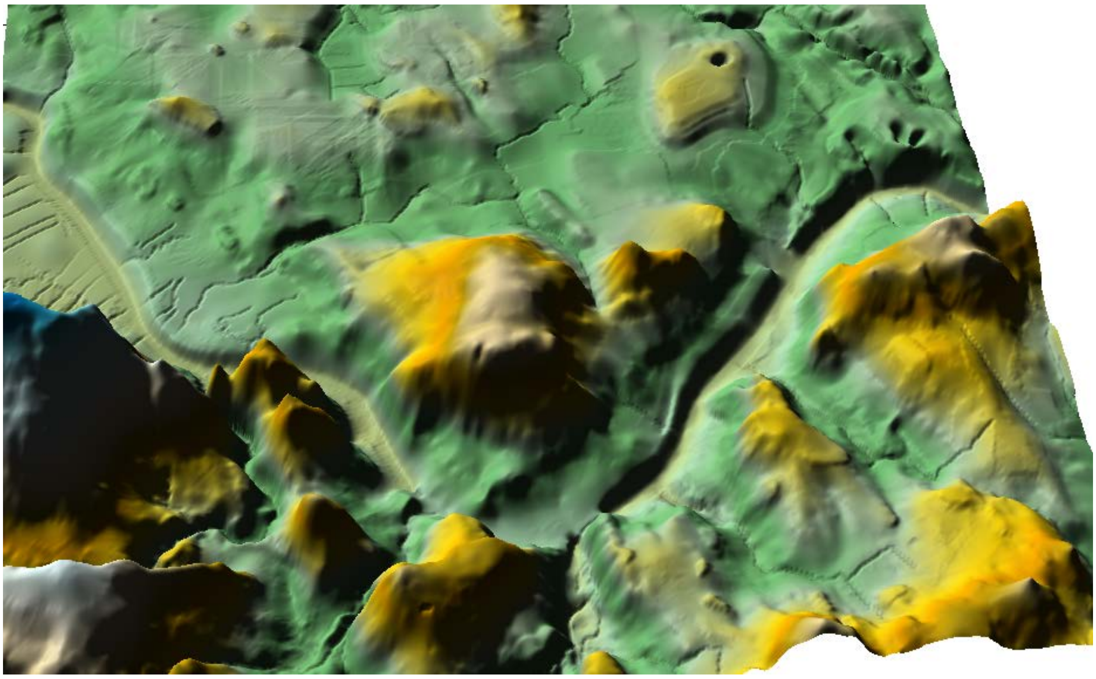


Figure 8.6 Surfer 3D Surface Projection of "Top of Bedrock" interpolated from depth-to-bedrock map combined with other available rockhead information at Carrigeen, Co. Kilkenny

Once the topographic and depth-to-bedrock files had been created, Excel was used to create offsets for the different layers that would be used in the groundwater model. Gridfiles were then created in Surfer 10 for each of the layer top and bottom elevations and these files were then ready for input into Groundwater Vistas. Other input files required to populate the groundwater model included:

- Catchment boundary shapefile
- Recharge zone shapefile (R)
- Hydraulic conductivity (K) shapefile
- Specific storage shapefile (S_y)
- Subsoil permeability shapefile

All of these shapefiles were created using ArcGIS using the available GIS shapefiles which were clipped to the study catchments extents.

8.2.2 Boundary Conditions and Model Properties

In order to populate a steady state groundwater model in Groundwater Vistas a number of properties have to be input including:

- Spatial elevations of bottom and top of each bedrock/aquifer layer
- Zones of hydraulic conductivity and the associated values
- Zones of recharge and the associated values
- Boundary conditions and target heads

For each of the study areas, with the exception of the karst site at Toonagh, Co. Clare, a groundwater model was set-up based upon the conceptual model as outlined previously in Chapter 6. Initially each model was set-up using a coarse 100 m x 100 m grid (see Figure 8.7) with one single aquifer layer extending 50 m below the bedrock surface. Once the model properties had been adjusted to give a "good fit" with the target heads available inside the model extents, then the model was redesigned at a more fine level with the grid cell size adjusted to 20 m x 20 m based upon the information available within the DEM. The aquifer was also broken up in a number of layers with the hydraulic conductivity decreasing with depth which is a better reflection on what most likely occurs. Cell size was refined in areas surrounding the treatment systems to 5m x 5m, which were incorporated into the model as point emitters, in order to give more precise results in these areas.

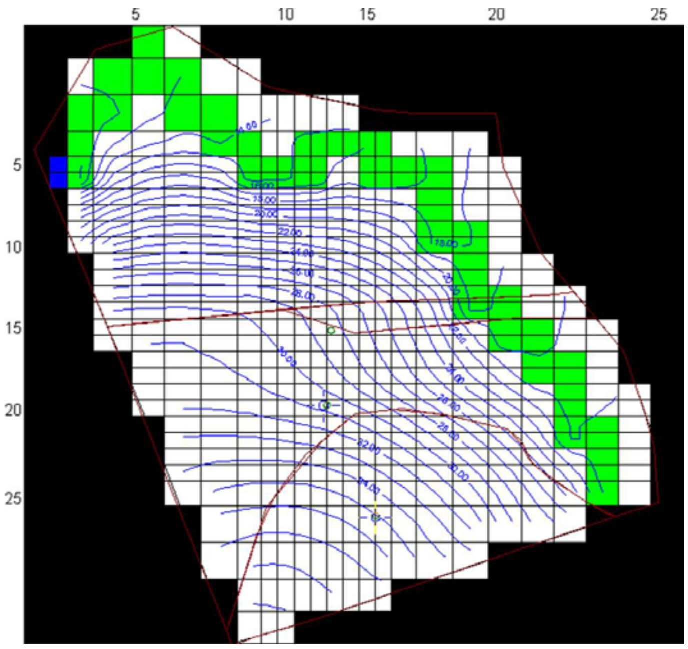


Figure 8.7 Model output grid with groundwater contours at Carrigeen, Co. Kilkenny

Additional model property details were required in order to run the groundwater models in transient conditions. These include:

- Time dependent boundary conditions
- Zones of specific storage and the associated values
- Time dependent target heads

Initial model properties were for each of the study sites were based upon the estimations outlined in Chapter 6. Values for hydraulic conductivity (K), recharge (R) and specific storage (S_y) were estimated based on all available information including data acquired during the course of this study and GSI reports and records. These initials estimates were refined during the initial model calibration and therefore subject to change and will not be repeated here (see Chapter 6 for full details).

Site at the Naul, Co. Dublin (Low Vulnerability)

Model boundary conditions for the Low vulnerability site at Naul, Co. Dublin were straightforward to implement due to the large amount of information available from previous GSI studies in the area (GSI, 2005). Boundary conditions used for the initial coarse model calibration are shown in Figure 8.8 below. The model outlet was set as a CONSTANT HEAD boundary on the river Delvin to the north and this is applicable only for steady state conditions.

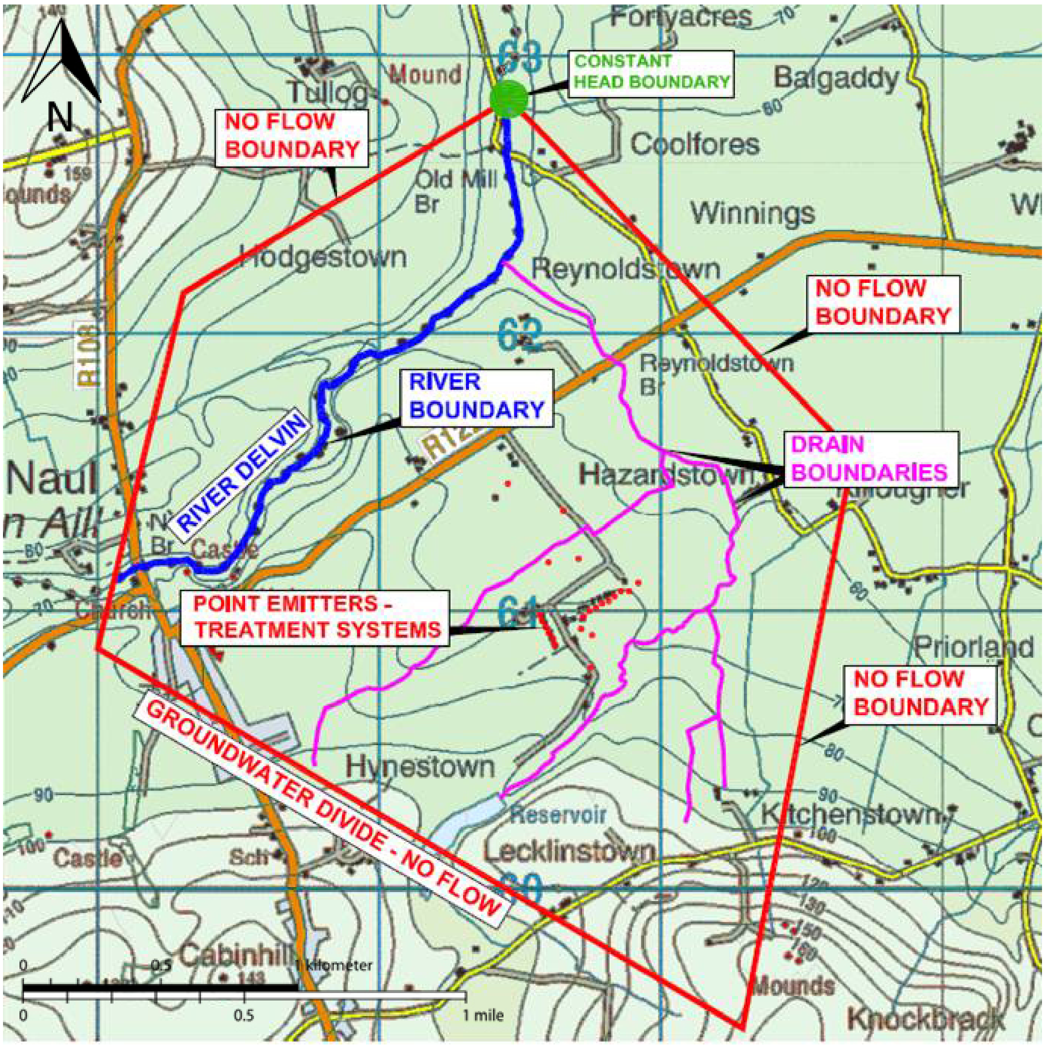


Figure 8.8 Boundary conditions at Naul, Co. Dublin

Site at Rhode, Co. Offaly (Moderate Vulnerability)

Boundary conditions for the Moderate vulnerability site were initially developed for a much larger groundwater catchment as shown in Figure 8.9 below. This was necessary as there were three possible catchments that the study area could be within and it was not clear which was most likely due to the area being located at a groundwater divide as discussed in Chapter 6. An initial model was therefore set up based on a much larger catchment with three CONSTANT HEAD outlets. Once the model had been roughly calibrated the MODFLOW package was then used to identify which of these catchments the study site fell within. Particles were deployed along the topographical divides on the three sections of the larger area and allowed to reach steady state conditions using the previously calculated model groundwater heads. The software traces the path that these particles will likely follow and thus the groundwater divides can be identified and the actual study area catchment can be estimated – which will be much smaller that this initial modelled area.

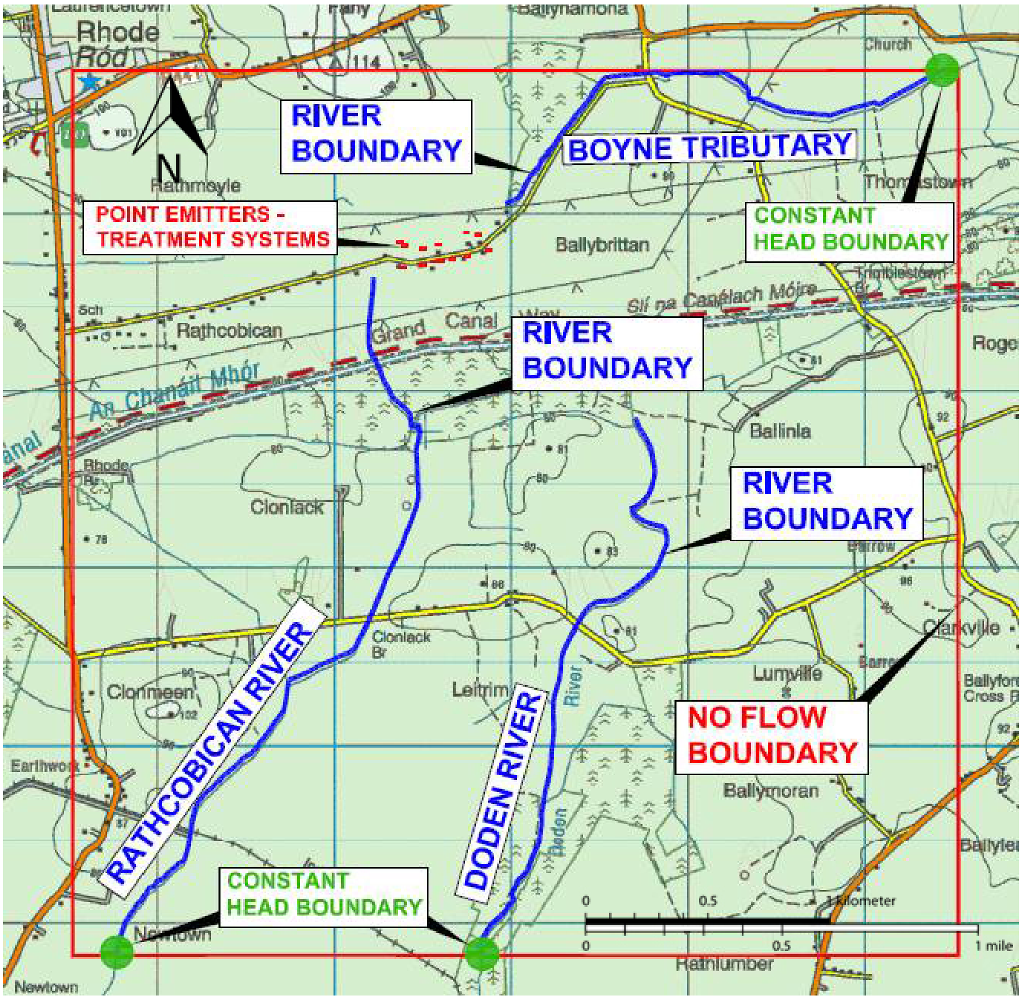


Figure 8.9 Large area boundary conditions at Rhode, Co. Offaly

The resulting MODPATH trace lines are shown in Figure 8.10 below. It can be seen that, as was earlier estimated, the study area is in the river Doden catchment and the catchment size for the groundwater model could be resized accordingly. The resulting catchment extent for the model together with the initial boundary conditions are shown in Figure 8.11 below. The outflow at the base of the model extents was set as a CONSTANT HEAD boundary condition with the model extents having the NO FLOW boundary condition and the river Doden having the RIVER boundary condition.

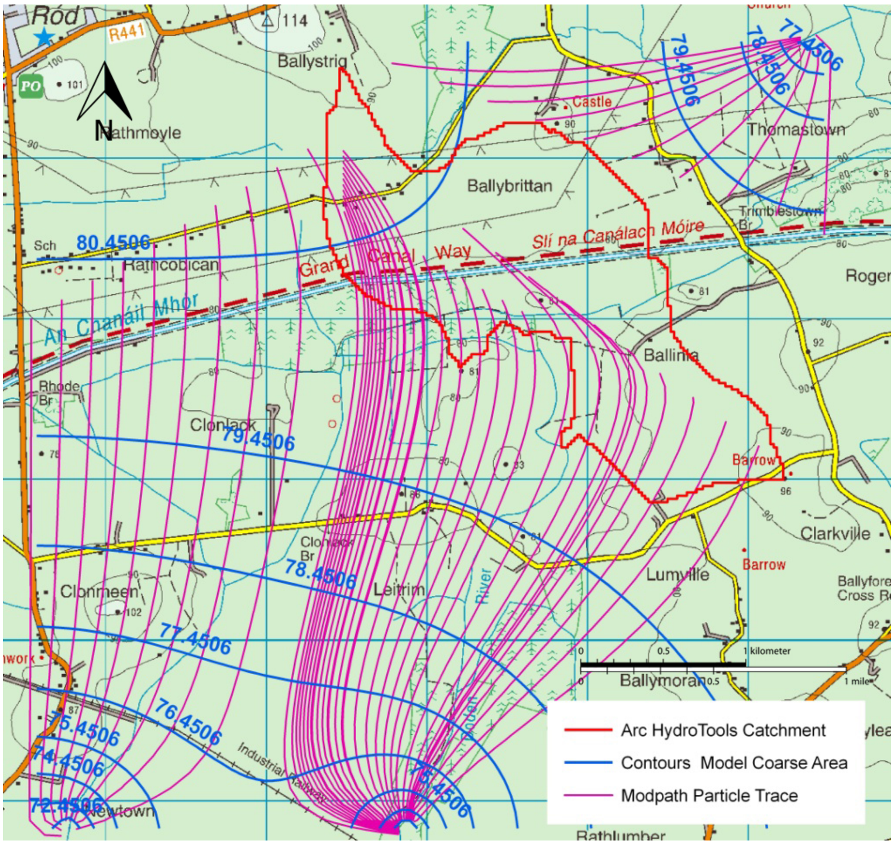


Figure 8.10 Results of particle trace using MODPATH at Rhode, Co. Offaly

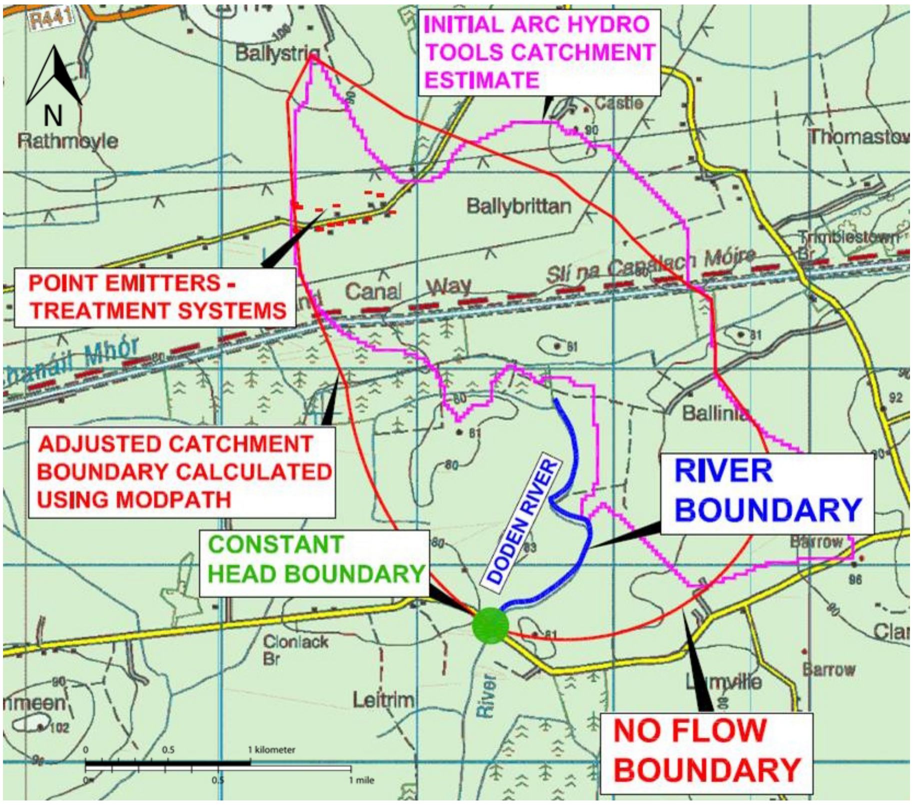


Figure 8.11 Refined catchment area calculated using MODPATH with initial boundary conditions at Rhode, Co. Offaly

Site at Carrigeen, Co. Kilkenny (High Vulnerability)

Initial boundary conditions for the High vulnerability study site at Carrigeen, Co. Kilkenny are shown in Figure 8.12 below. The outflow at the base of the model extents was set as a CONSTANT HEAD boundary condition with the groundwater divides? also set as NO FLOW boundary condition – this was based on the fact that the site is steeply declining to the north-west and it was deemed very unlikely that groundwater would be moving to any great extent in any other direction. The Dungooly stream was set as a RIVER boundary condition.

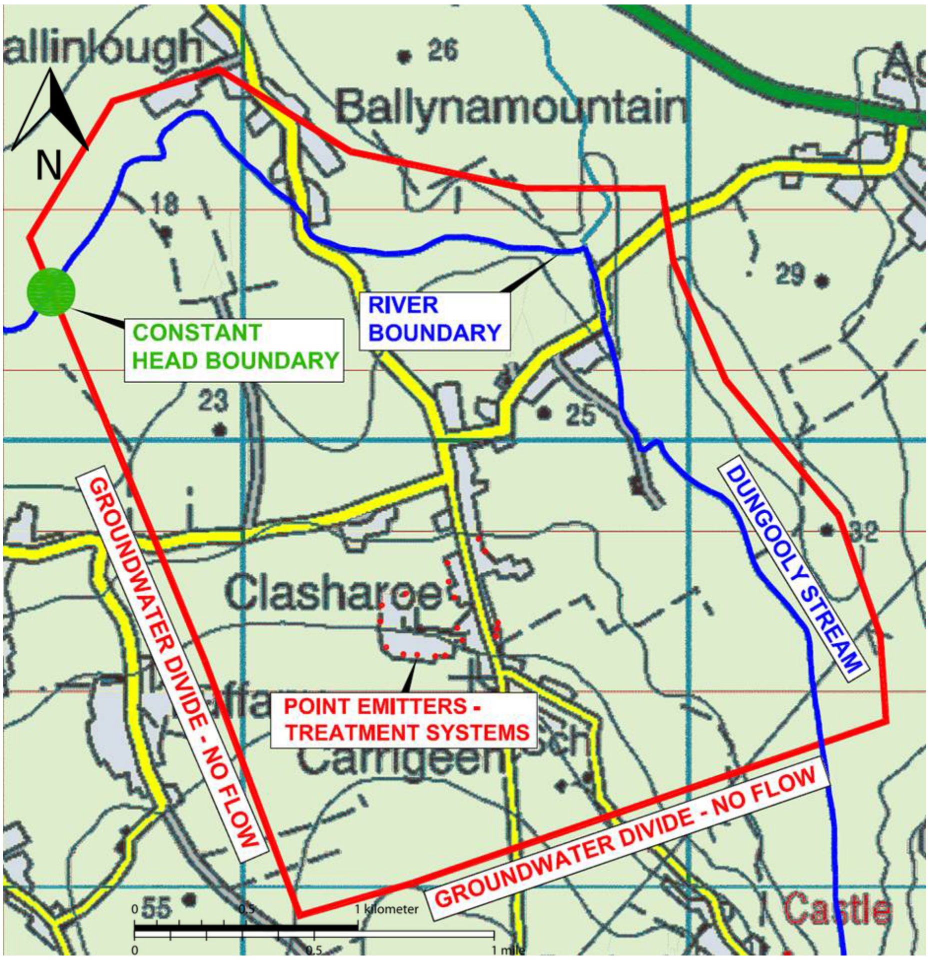


Figure 8.12 Boundary conditions at Carrigeen, Co. Kilkenny

Site at Faha, Co. Limerick (Extreme Vulnerability)

As discussed in Chapter 6, the area surrounding the study area at Faha, Co. Limerick forms a low plateau in the floodplains of the river Maigue with no clear groundwater topography. Ultimately, given the similar water levels and the close proximately of the river Maigue, it would appear that groundwater in the area is hydraulically connected to the river and for this reason a point sufficiently downstream from the study area was chosen as the outfall for the groundwater catchment - this reflects the results of the exercise carried out using ArcHydro Tools described in Chapter 6. A buffer of 100 m on the north-eastern side of the river was modelled to allow for any possible local error near the boundaries of the model. The model extents were set as NO FLOW boundaries with the river Maigure a RIVER boundary condition and the outflow point downstream set as a CONSTANT HEAD boundary condition – see Figure 8.13 for details.

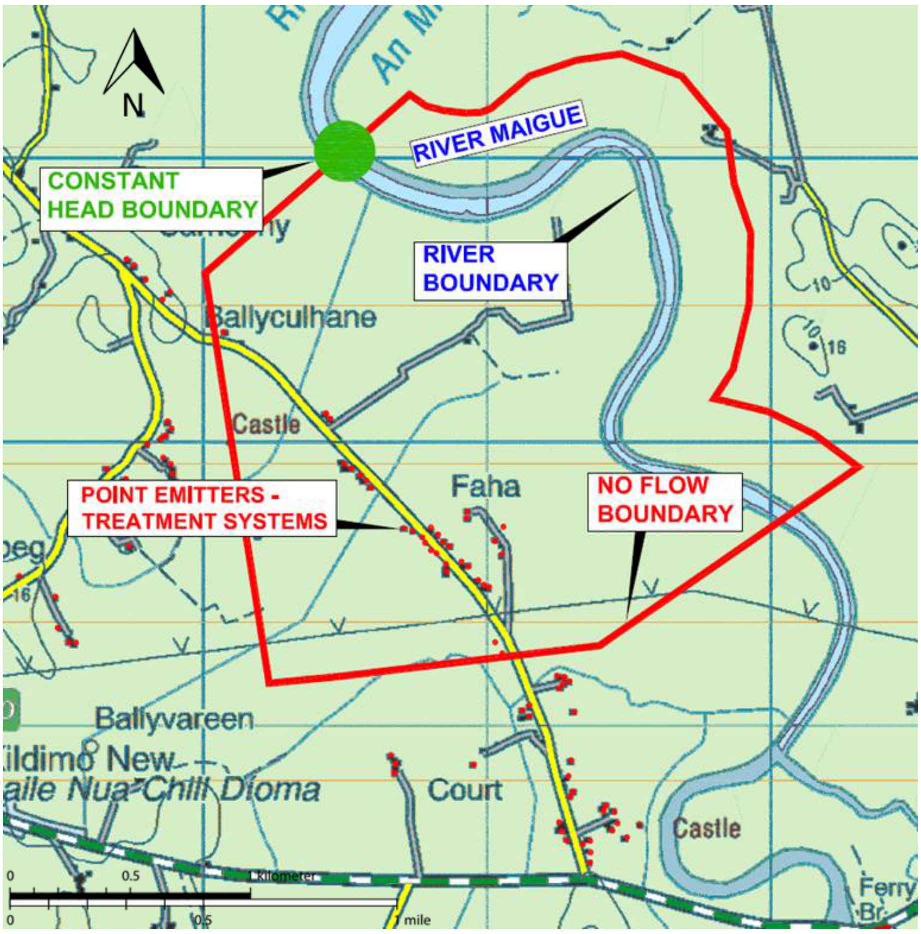


Figure 8.13 Boundary conditions at Faha, Co. Limerick

8.3 Model Execution and Calibration

8.3.1 Initial Steady State Model

The models were set up using the data input files that had been developed previously (see Section 8.2.1) and the boundary conditions as outlined in the previous section. Models were then executed using the MODFLOW package of Groundwater Vistas with the hydraulic conductivity and recharge values assigned as described in Chapter 6. Some of these parameters had to be adjusted in order to give a ‘better fit’ to the target head values across the catchment, which is discussed in detail in Section 8.1.4 during the sensitivity analysis. Results of the initial model run for Naul, Co. Dublin are shown in Figure 8.14 below.

Figure 8.14 Modelled groundwater contours at Naul, Co. Dublin

Groundwater elevations can be seen to reduce from 114 mOD at the south to 46 mOD to the north at the catchment outlet. A plot of modelled versus the three observed values (Figure 8.15) produced a very good agreement with an R^2 value of 99.997% with a corresponding root mean square (RMS) error of 0.003m.

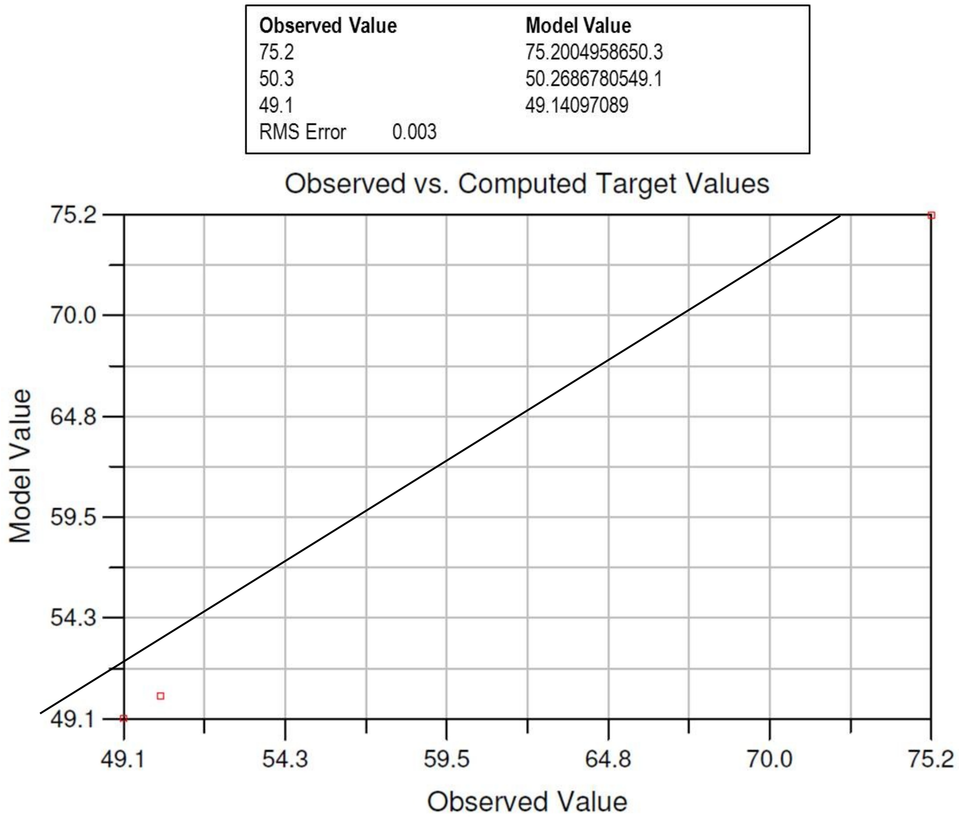


Figure 8.15 Model calibration statistics output showing target head agreement with fitted values at Naul, Co. Dublin

Modelled groundwater heads at Rhode, Co. Offaly can be seen in Figure 8.16 below. Heads reduce from 80.82 mOD at the top of the catchment to 75.53 mOD at the catchment outlet reflecting the shallow gradients in the area. A plot of modelled versus observed values produced a very good agreement with an R^2 value of 99.73% and a RMS error of 0.27 m for the three borehole heads.

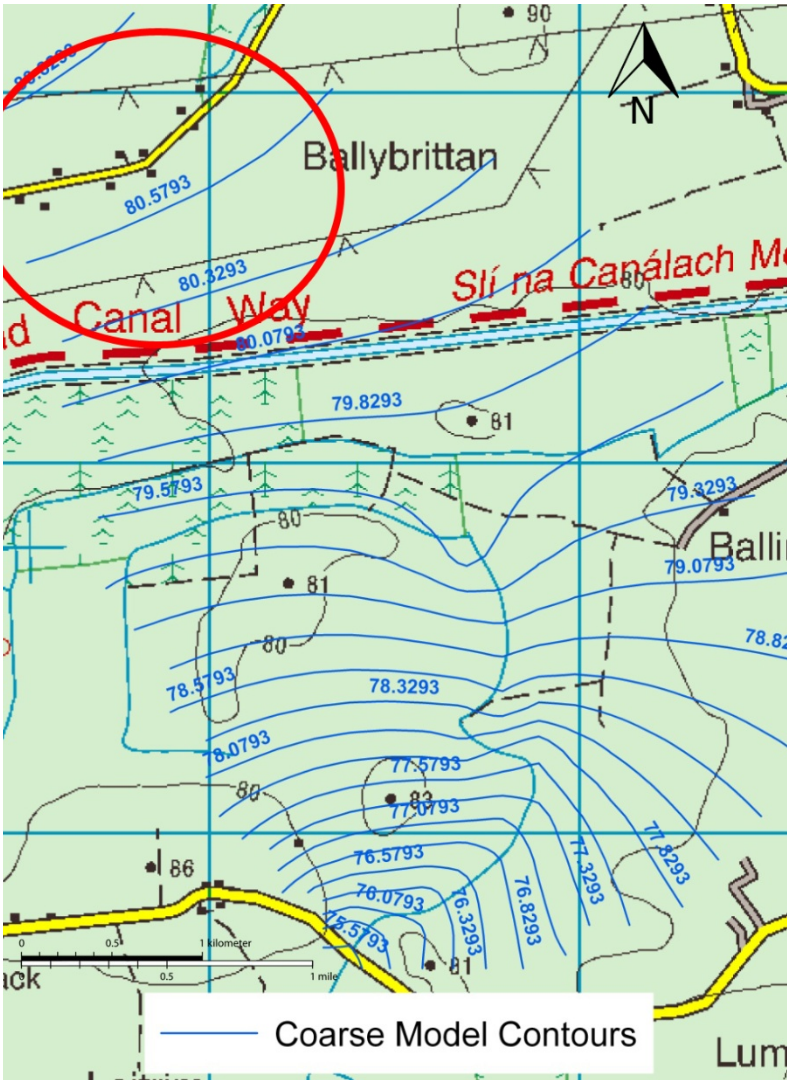


Figure 8.16 Modelled groundwater contours at Rhode, Co. Offaly

Modelled groundwater head values at Carrigeen, Co. Kilkenny fell from 39 mOD at the top of catchment to 12 mOD at the outlet of the catchment as shown in Figure 8.17 below. There is quite a steep transition between the top of the hill and the base of the hill with the topography levelling off to a very gentle gradient some 200 m past the study cluster development. Initially this was very difficult to model within MODFLOW with a twisted or distorted water table occurring due to this very extreme change in topography and bedrock geology. A ‘buffer zone’ of transition hydraulic conductivity was incorporated into the model and this had the desired result of correcting this distortion and provided a very good agreement between observed and fitted values with an R^2 of 99.9% and a RMS error of 0.001 m for the three borehole heads; however without this ‘fix’ the model RMS error was 0.54 m with an R^2 of 63.67 % and it is noted that this ‘buffer’ most likely does not exist and therefore serves only to help calibrate the model and may affect the model output in a negative manner.

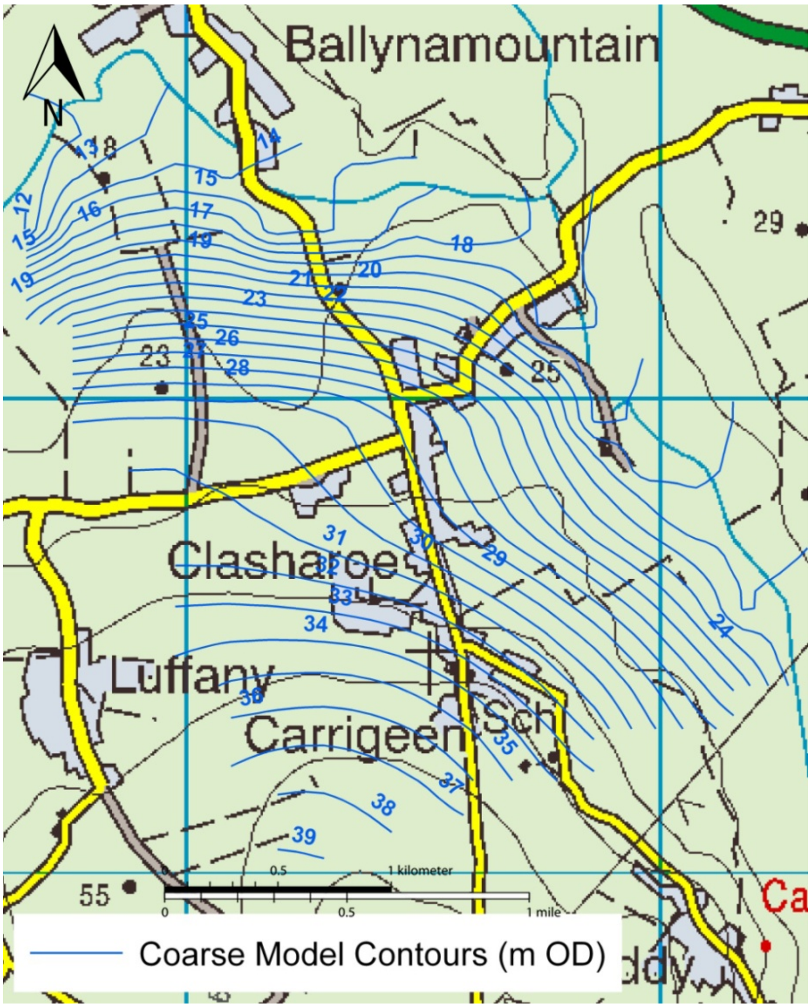


Figure 8.17 Modelled groundwater contours at Carrigeen, Co. Kilkenny

Groundwater heads fitted by the MODFLOW package at Faha, Co. Limerick are shown in Figure 8.18 below. Simulated heads fell from 1.2 mOD in the west to 0.16 mOd to the north-east with a very shallow groundwater gradient which reflects the flat nature of land in the area. A plot of observed versus fitted values yielded a R^2 value of 99.89% with a RMS error of 0.11 m for the three borehole heads.

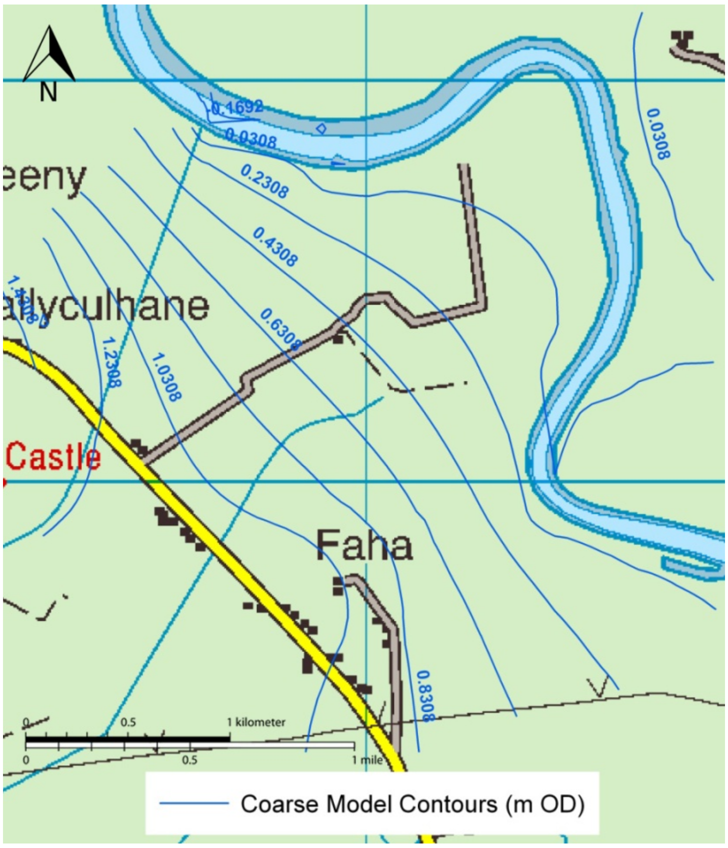


Figure 8.18 Modelled groundwater contours at Faha, Co. Limerick

8.3.2 Steady State Model with Contaminant Transport

Following the calibration of the groundwater models in steady state for water flow only, the MT3D contaminant package was run for the various contaminant packages again for steady state water flow conditions. It is not possible to run MT3D in steady state conditions and a stress period associated with the contaminants must be incorporated with the associated discretion of time steps. The model treats the water flow aspect as steady state and the contaminant transport package is then run over the specified stress period with the specified number of time steps. For the purposes of this steady state simulation a stress period of 365 days was used with daily time steps. The model must be propagated with a number of additional inputs and parameters specific to each contaminant. The values used for dispersivity in the aquifer are common to all of the contaminants. . However, no measured values for dispersivity were available for the associated aquifers in this project and therefore the values used will be taken from available literature. Schulze-Makuch (2005) conducted an extensive review of the available literature on values of longitudinal dispersivity (α_x) for a range of porous media from soils to bedrocks. During a sensitivity analysis of their groundwater model, Pang et al. (2006) found that the dispersivity values have only a limited impact on the model output when varied by large amounts. Values were therefore extracted from Schulze-Makuch (2005) and varied during the sensitivity analysis. A summary of the longitudinal dispersivity values used in this study is given in Table 8.1 below. As longitudinal dispersivity is scale dependant with values of α_x reducing as the scale reduces, values in the appropriate scale range have therefore been used. Transverse (α_y) and vertical dispersivity (α_z) have been assumed to be 10% of longitudinal dispersivity which is the general convention for groundwater simulations (Pang et al., 2006).

Table 8.1 Longitudinal dispersivities for the aquifer units

Location	Aquifer Type	Longitudinal Dispersivity (m)
Nual, Co. Dublin	Shale/Sandstone (Namurian)	3.2 – 30
Rhode, Co. Offaly	Limestone (Dinantian)	8 – 170
Carrigeen, Co. Kilkenny	Fractured Sandstone (Devonian)	6 – 110
Faha, Co. Limerick	Dolomitic Limestone (Dinantian)	15 - 170

All of the contaminants were added to the input using an area of recharge analogous with the area of the plume at the water table as determined from the HYDRUS modelling (see Chapter 7). The recharge flux was set to that predicted by HYDRUS (m/day) and the contaminant was added as a concentration in that incoming recharge water. As HYDRUS did not predict any contaminants entering the aquifer from the unsaturated zone at the *LOW* vulnerability site, this site will not be considered in these simulations. All of the contaminant transport models considered rainfall in steady state based on annual average values as described in Chapter 6.

8.3.2.1 Nitrogen

Nitrogen input values were taken from the HYDRUS simulation outputs at the groundwater table and are given in Chapter 7. For all simulations involving nitrogen it has been assumed to be only present in groundwater in the form of nitrate. Nitrate was treated as a conservative tracer with dissolved NO_3 tending to be non-reactive and mobile. The simulated steady state nitrate concentration plumes for each of the study areas can be seen in Figure 8.19 – Figure 8.21. At the *MODERATE* vulnerability site in Offaly it can be seen that the plume does not reach the first downstream monitoring borehole, BH-O2. The plume only extends a maximum of 140 m downstream of the cluster development. The maximum concentrations observed in the aquifer after the stress period is 6 mg-N/l, however this is localised around the emitting treatment system.

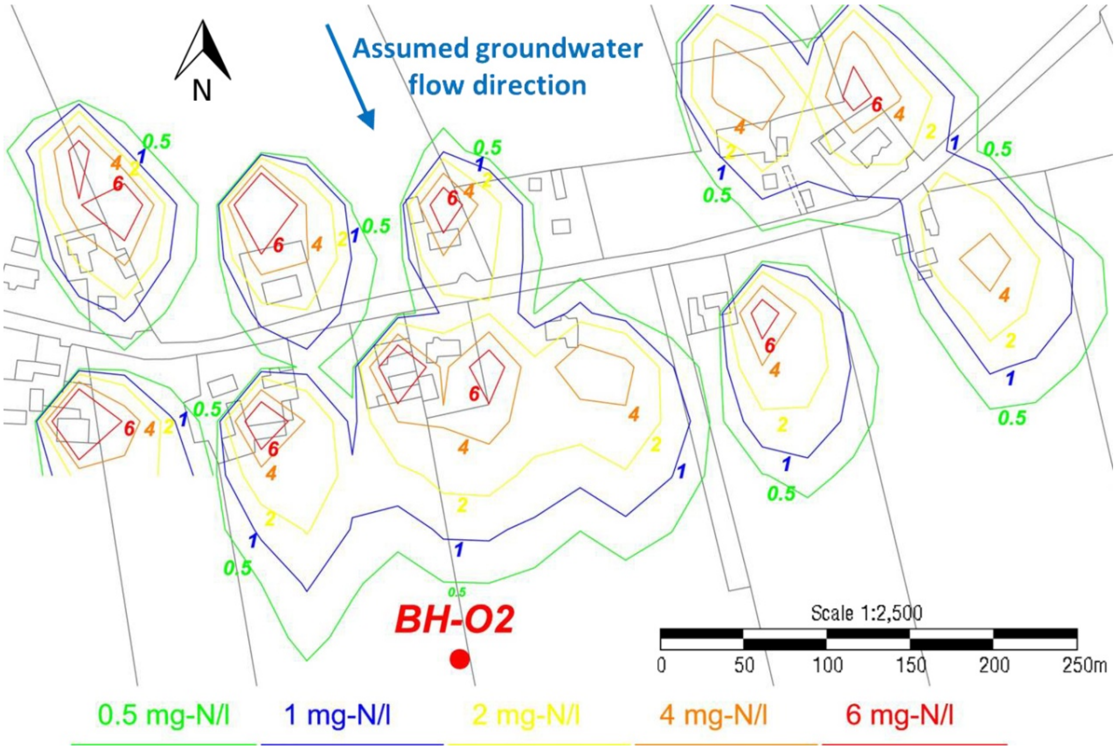


Figure 8.19 Steady State Nitrate plume from the cluster development at Rhode Co. Offaly

At the *HIGH* vulnerability site in Kilkenny there is more evidence of an overall "cluster plume" forming (see Figure 8.20) that was not as evident at the *MODERATE* vulnerability site. However, the concentrations of NO_3 in this plume are low and again the plume does not extend more than 180 m downstream of the cluster development. BH-K2 is however located within the nitrate plume however the concentration at that location is c. 0.8 mg-N/l. A plume of concentration 2 mg-N/l can be seen to extend up to 320 m to the east at the *EXTREME* vulnerability (see Figure 8.21). There is a stronger tendency for a combined "cluster plume" to form however beyond 350 m from the cluster concentrations in the plume have decreased to less than 0.5 mg-N/l. It can be seen that the plume migrates locally to the east in the vicinity of the cluster development and this would appear to be due to the presence of a rock outcrop located near BH-L2. It can therefore be seen that the location of the two downstream boreholes are outside the main "cluster plume".

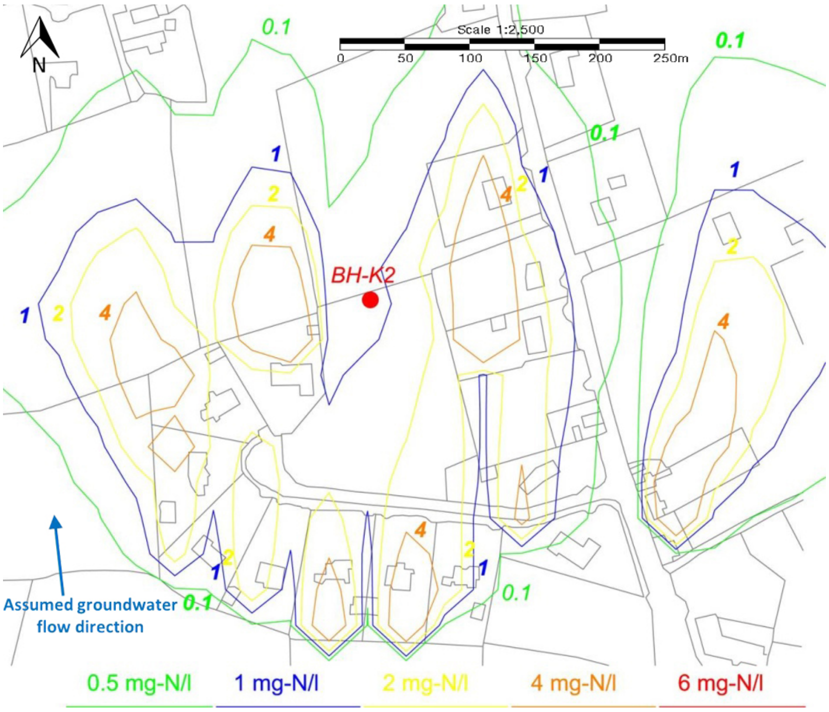


Figure 8.20 Steady State Nitrate plume from the cluster development at Carrigeen Co. Kilkenny

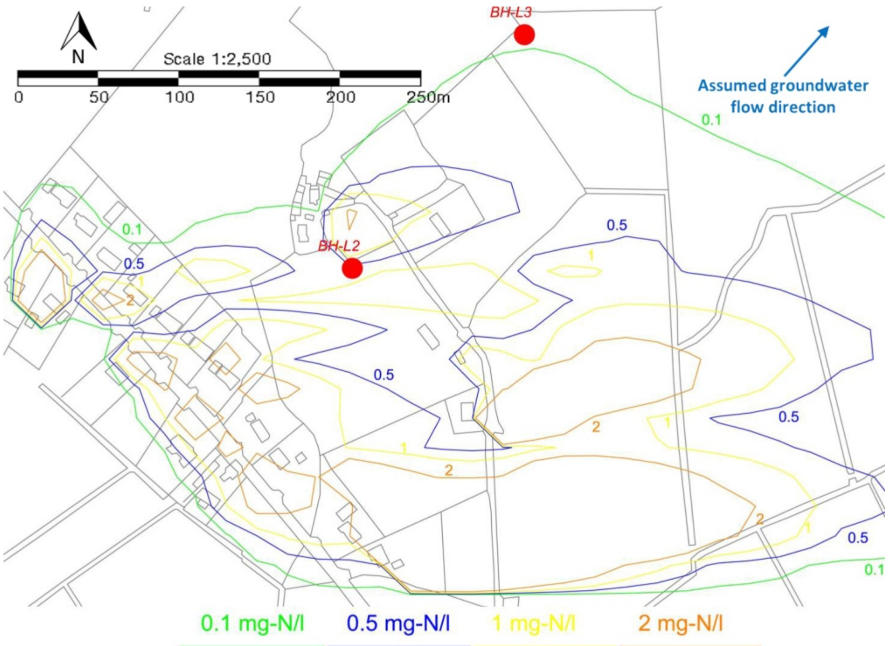


Figure 8.21 Steady State Nitrate plume from the cluster development at Faha Co. Limerick

It was not anticipated prior to drilling that any contaminant plume would migrate around the hill or rock outcrop and the downstream boreholes were located accordingly. It can be seen however, that whilst a plume of nitrate does develop due to the cluster development, the concentrations downstream do not pose any real risk to the aquifer particularly at the concentrations observed.

8.3.2.2 Phosphorus

Input values for Phosphorus were taken from the HYDRUS simulation outputs at the groundwater table (see Chapter 7 for details). As for the nitrate simulations, phosphorus was treated as a conservative tracer for all simulations. In practise this may not be true, however very little information is available in the literature on the behaviour of dissolved phosphorus in aquifers. Any degradation or removal in the aquifer that is not considered in these simulations would lead to a lower concentration downstream plume and so the model may overestimate but will not underestimate the downstream concentrations. The simulated steady state phosphorus concentration plumes for each of the study areas can be seen in Figure 8.22 – Figure 8.24.

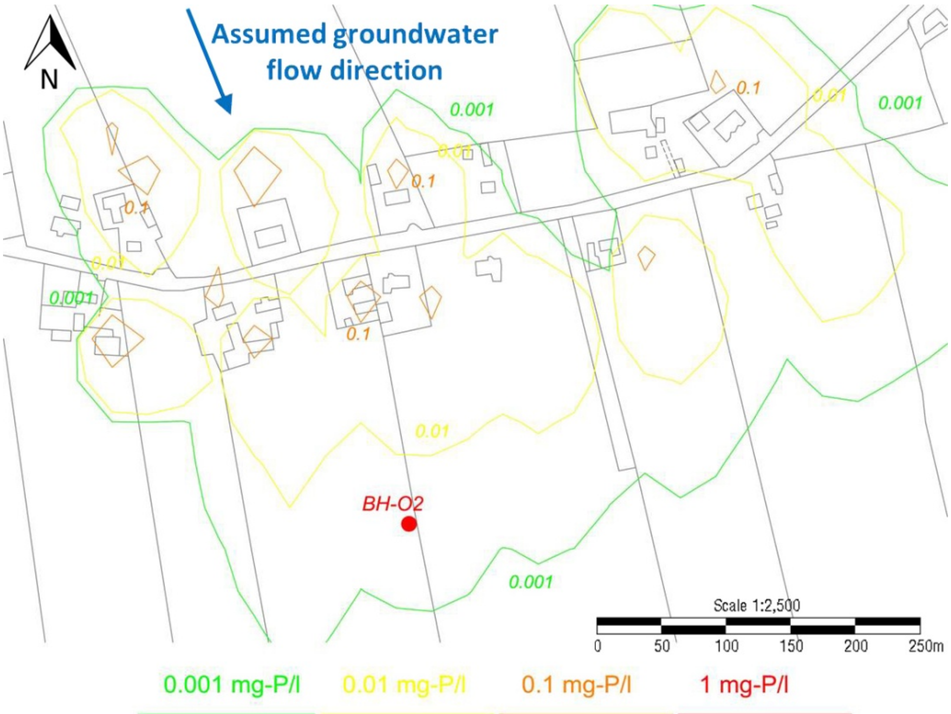


Figure 8.22 Steady State Phosphorus plume from the cluster development at Rhode Co. Offaly

At the *MODERATE* vulnerability site in Offaly the phosphorus plume reaches the first downstream monitoring borehole, BH-O2, at a concentration of 6 µg-P/l. A plume of concentration 0.01 mg-P/l can be seen to have extended outwards from the cluster development approximately 100 m downstream. As was seen with the simulations involving nitrate, localised plumes of higher concentration (0.1 mg-P/l) occur close to the individual percolation areas, however these dissipate within 10 – 30 m.

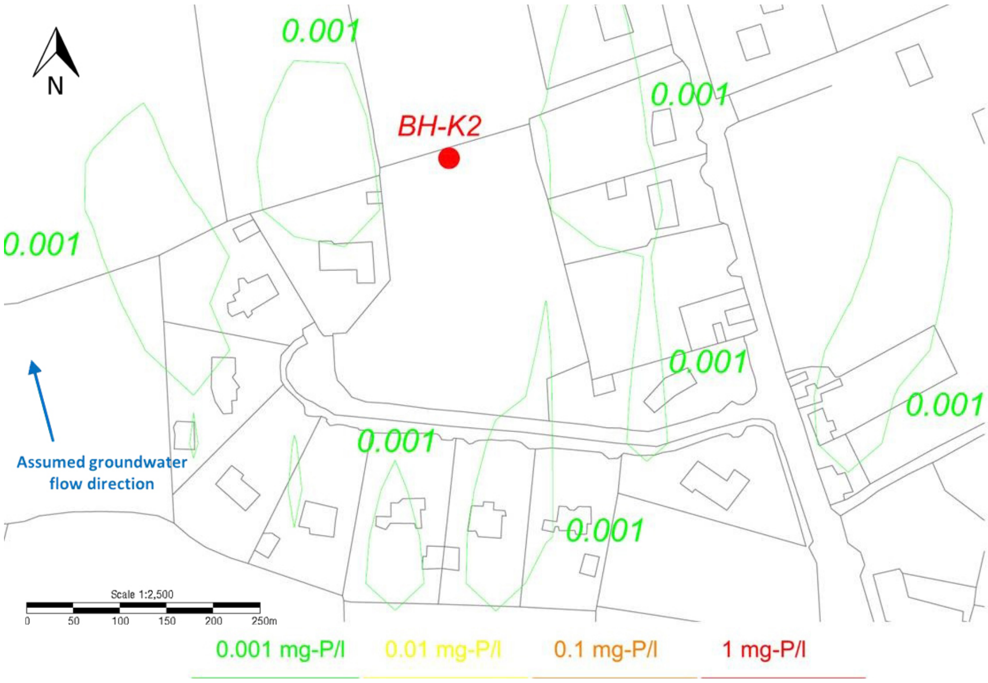


Figure 8.23 Steady State Phosphorus plume from the cluster development at Carrigeen Co. Kilkenny

Localised plumes of low concentration (0.001 mg-P/l) can be seen to occur at the *HIGH* vulnerability site in Kilkenny. None of these plumes extend to the location of BH-K2 the first downstream monitoring borehole. Simulated concentrations of phosphorus are lower here than at the *MODERATE* vulnerability site due to the presence of a slightly thicker unsaturated zone and given the high removal rates of phosphorus in the subsoil lower input concentrations enter the groundwater. Simulated plumes of phosphorus extend the furthest with relatively higher concentrations at the *EXTREME* vulnerability site in Limerick. A 0.1 mg-P/l contour extends some 260 m to the east of the cluster development. The first and second downstream monitoring boreholes are located within the contours of 0.01 mg-P/l. a contour of 0.001 mg-P/l can be seen to extend in a wide arc up of to 400 m downstream of the cluster development. The reasons for this extensive plume are likely due to the relatively high input concentrations coupled with the transmissive bedrock and the assumed high dispersivity characteristics. It must be noted however that even with the extensive plume that was simulated, the downstream values are still relatively low and the plume has reached negligible concentrations some distance before reaching the River Maigue.

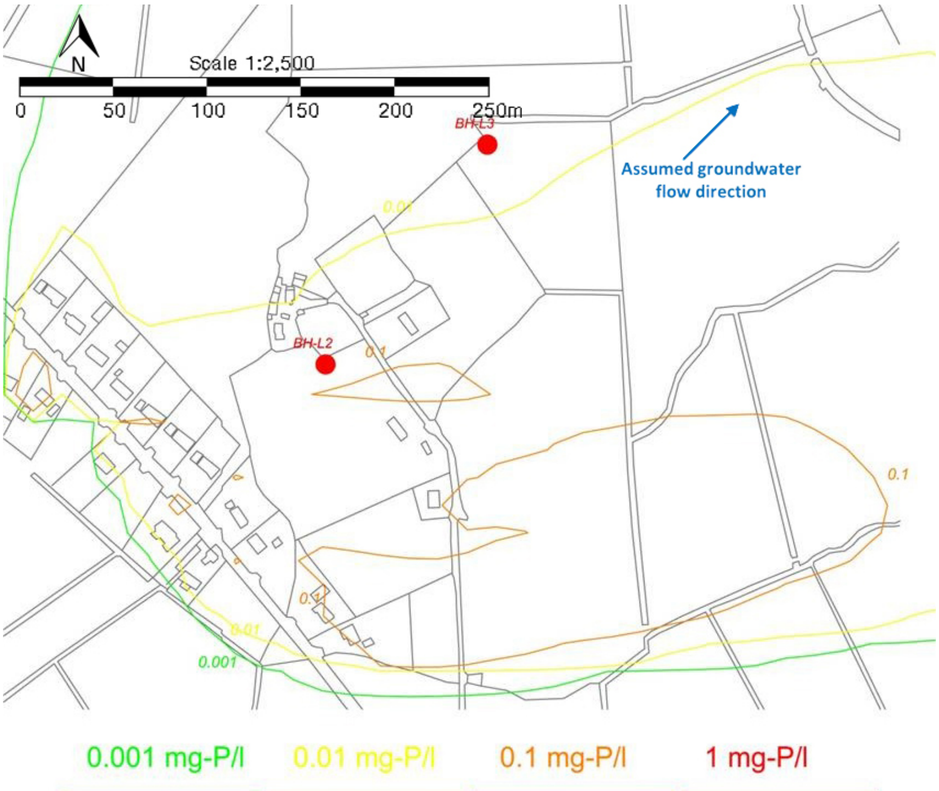


Figure 8.24 Steady State Phosphorus plume from the cluster development at Faha Co. Limerick

8.3.2.3 Bacteria

The transport of bacteria in the aquifer cannot be treated as a conservative tracer, as was the case used for both nitrate and phosphorus, due to the natural die-off that occurs over time. As described in Section 7.3.4, a lumped removal rate was used for bacteria which accounts for all of the removal methods that occur. Values for this removal rate (λ) for *E-coli* have been estimated at between 0.6 - 3 day^{-1} (Pang et al., 2006; Jiang et al., 2010). For the steady state model with a stress period of 365 days a removal rate at the lower end of the reported values of 0.6 day^{-1} was used as this was deemed to be a conservative approach. As outlined in Chapter 7 it can be assumed that some proportion of the percolation areas contained within the cluster of OSWTS's will not have been constructed correctly and instead a single percolation trench may have been incorporated with this trench receiving the entire daily loading rates. This scenario was modelled in HYDRUS and for these simulations it was assumed that 5% of the systems would have had these poorly constructed percolation systems incorporated. For all other systems either the STE or SE outputs as detailed in Chapter 7 were input where appropriate. MT3D did not predict any significant plume of bacteria downstream of the cluster developments at the *MODERATE* or *HIGH* vulnerability sites as shown in Figures 1.25 – 1.27 below. All of the bacteria were removed within 60 m of the cluster with most being removed within 30 m. Given that the lower end of the reported removal rates was used in these simulations it is considered reasonable to assume that only very localised plumes of bacteria exist in the close vicinity of the OSWTS's and that bacteria do not migrate downstream - again, this is in line with the results of the Pang et al. (2006) study. At the *EXTREME* vulnerability site however, a plume of concentration 1CFU/l does migrate almost 150 m downstream of the cluster development. A localised plume of 2 CFU/l also developed in the vicinity of the OSWTS that was assumed to have only a single percolation trench installed; a similar localised plume also developed at the *MODERATE* vulnerability site. The migration of this plume of bacteria is a cause for concern however considering that conservative removal and loading rates were assumed this low concentration must be viewed as a worst case scenario.

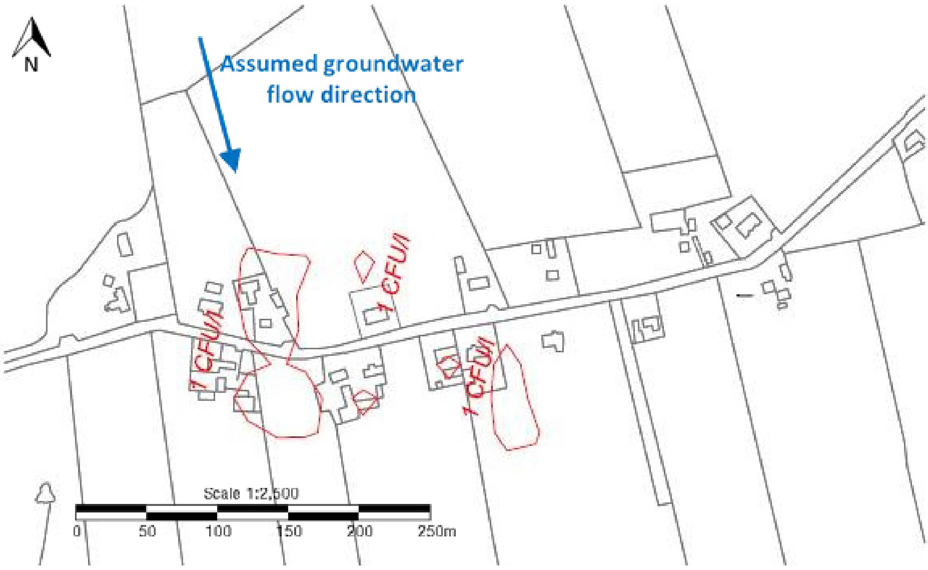


Figure 8.25 Steady State bacteria plume from the cluster development at Rhode, Co. offaly

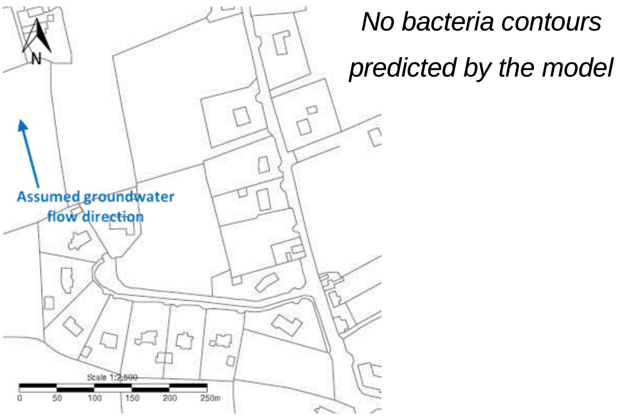


Figure 8.26 Steady State bacteria plume from the cluster development at Carrigeen, Co. Kilkenny

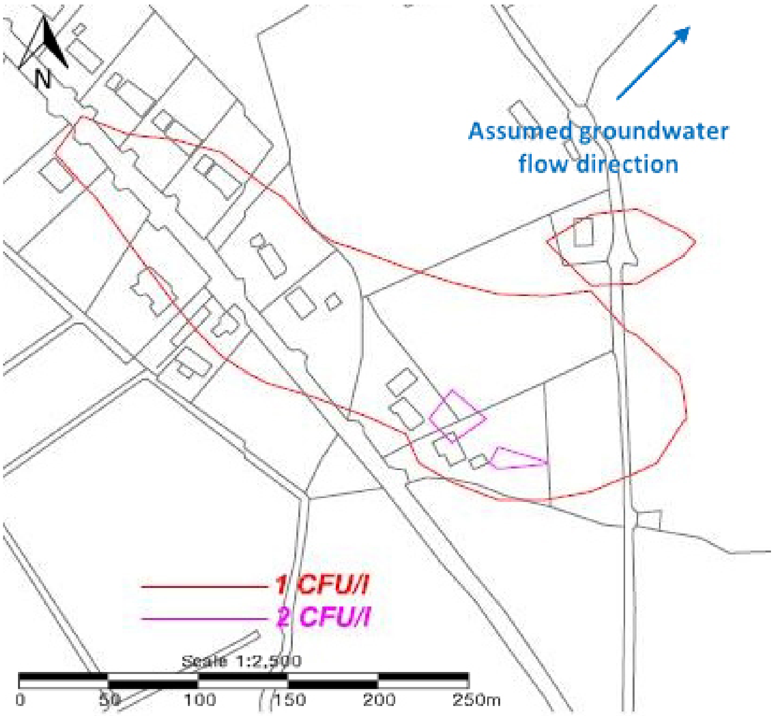


Figure 8.27 Steady State bacteria plume from the cluster development at Faha Co. Limerick

8.3.3 Transient Model with Contaminant Transport

Transient Model Setup

Transient modes in MODFLOW were setup following the satisfactory steady state simulations. In order to setup and run a transient model, additional input information was required as discussed in Section 8.1.2, the most important of which was the time series recharge and head information which drive the model. The way in which Groundwater Vistas allows input of transient data is in the form of stress periods with recharge multipliers for each stress period. The models were therefore setup in daily time steps or stress periods with the initial conditions of the first stress period being those from the initial steady state setup. Due to the long run time of the modelling software when incorporating MT3D, the transient model was run for 365 days with 365 stress periods. Following on from this the models were run for 1095 days (3 years) with 30 day time steps and stress periods to allow for a long term migration of any plumes that were developed. For the 365 day model setup recharge was set as the yearly recharge from 2011 and recharge multipliers for each of the 365 days were calculated based on total recharge divided by the calculated daily recharge amounts. As discussed in Chapter 6, porosity was assumed to be 2% for all of the aquifers during these simulations. Nitrate and phosphorus were again treated as conservative tracers with bacteria assumed to be subject to first order decay with an initial value of 1 d^{-1} estimated from the literature (Pang et al., 2006). Variable head data for the outlets to each of the catchments was available for the site at Faha, Co. Limeirck from an Office of Public Works (OPW) gauging station on the River Maigure – this was discussed in detail in Chapter 5. For the other two catchments modelled in MODFLOW, water level data was not immediately available at the catchment outlet. A process was undertaken that interpolated water levels in the watercourses at the catchment outlets based upon levels further downstream at the nearest gauging station. Whilst this method may not be entirely accurate, it is noted that water levels in the streams both at Carrigeen and Rhode were fairly static during the course of the study (from visual observations) and only varied by 5 – 20 cm following intense rainfall events. Given the size of the catchments modelled these small head variations in the stream outlets would not have a large effect in the overall groundwater contours and therefore this method was appropriate for this exercise.

Transient Model outputs

The 365 day model run with daily time steps and contaminant stress periods did not show any significant variation from the steady state model runs which is to be expected as the steady state models were based on annual recharge and contaminant loading and were run for a 365 stress period for MT3D. However, when the models were extended to 3 years duration the plumes did migrate further downstream before again appearing to reaching a "quasi semi-steady state" after 2 years with the plume not migrating further in the third year of the simulation. This is similar to the output of the HYDRUS models in Chapter 7 whereby the models reached steady state conditions after 2 years duration. At the *MODERATE* vulnerability site in Rhode the plume of nitrates extended to almost 500 m at 1 mg-N/l with the 0.01 mg-N/l contours extending up to 1 km downstream as shown in Figure 8.28. The plume for phosphorus did not extend further than 400 m downstream at 1 µg-P/l (i.e. negligible). Figure 8.29 shows the transient model output after 3 years at the *MODERATE* vulnerability site for bacteria. It can be seen that the localised plumes of bacteria have migrated slightly further downstream and have spread out marginally. Overall the levels of bacteria involved and the extent of the plume do not appear to be enough to cause any significant concern.

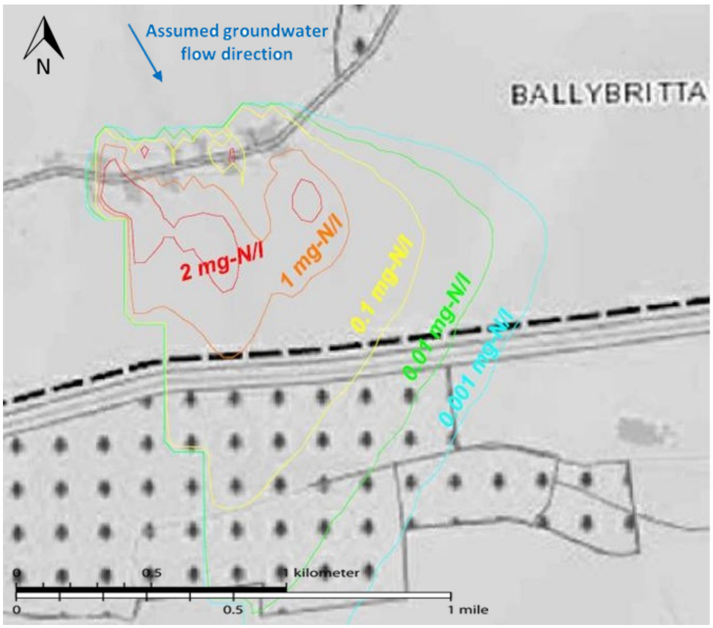


Figure 8.28 Transient model contours for nitrate at the *Moderate* vulnerability site after 3 years

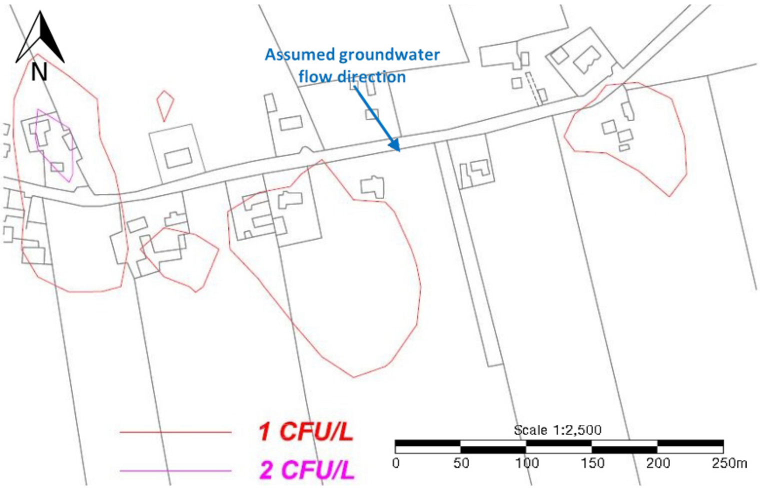


Figure 8.29 Transient model contours for bacteria at the Moderate vulnerability site after 3 years

At the *HIGH* vulnerability site in Carrigeen the plume of nitrates extended to almost 500 m at 1 mg-N/l with the 0.01 mg-N/l contours extending up to 1 km downstream as shown in Figure 8.30. The plume for phosphorus did not extend further than 400 m downstream at 1 µg-P/l. As was seen with bacteria in the steady state model no plume developed over the 3 year simulation period as shown in Figure 8.31 below.

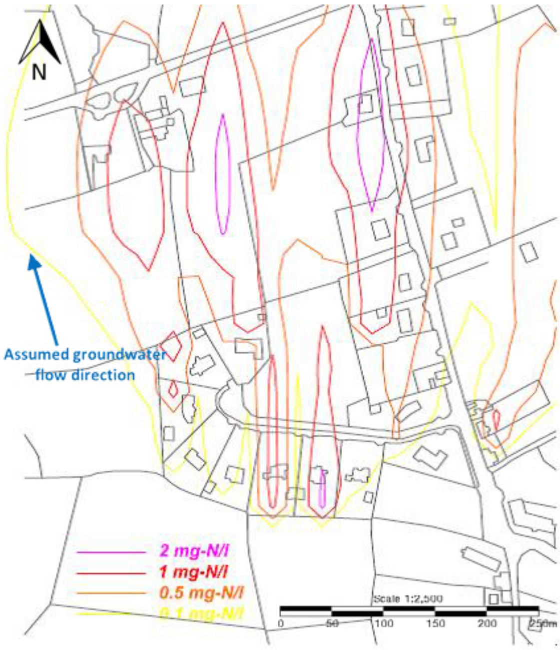
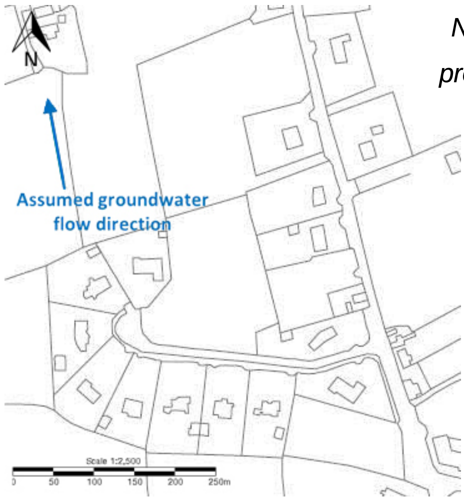


Figure 8.30 Transient model contours for nitrate at the *High* vulnerability site after 3 years



No bacteria contours predicted by the model

Figure 8.31 Transient model output for bacteria at the High vulnerability site after 3 years

At the *EXTREME* vulnerability site in Faha the plume of nitrates extended to almost 350 m at 2 mg-N/l with the 0.01 mg-N/l contours extending up to 800 km downstream as shown in. The plume for phosphorus did not extend further than 500 m downstream at 1 µg-P/l.

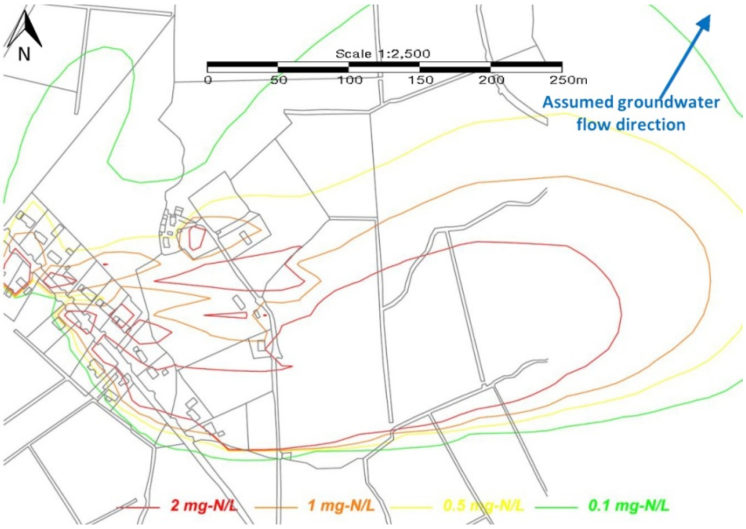


Figure 8.32 Transient model contours for nitrate at the Extreme vulnerability site after 3 years

The plume of bacteria at the EXTREME vulnerability site after the 3 year transient simulation is shown in Figure 8.33. a plume of 1 CFU/l extended up to 300m downstream of the cluster development with a slightly higher concentration plume arising from the OSWTS's assumed to be poorly constructed. It must be noted again that this plume is moving in the opposite direction to that which was assumed prior to drilling the downstream monitoring boreholes. As was seen with the other sites the levels of bacteria involved are very low and considering the very cautious and conservative approach taken throughout the setup of the models the simulated levels are not a cause for significant concern,

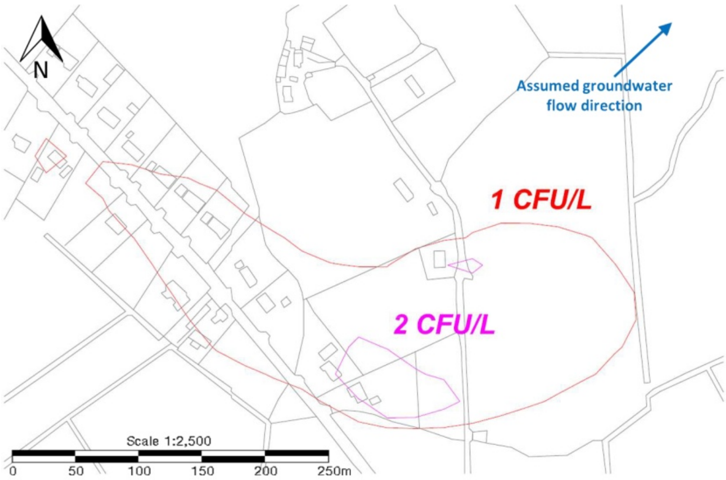


Figure 8.33 Transient model contours for bacteria at the Extreme vulnerability site after 3 years

8.4 Model Validation and Sensitivity Analysis

8.4.1 Model Validation

In order to validate the model it was decided to compare field observed values not used in the model calibration with modelled values at those locations. The averaged observed heads for the three monitoring boreholes were used to calibrate the steady state groundwater flow model. Over the monitoring period and during site visits once-off spot head measurements were taken at boreholes or private wells in the study catchment for this purpose. The locations of these spot measurements are shown in Figure 8.34.

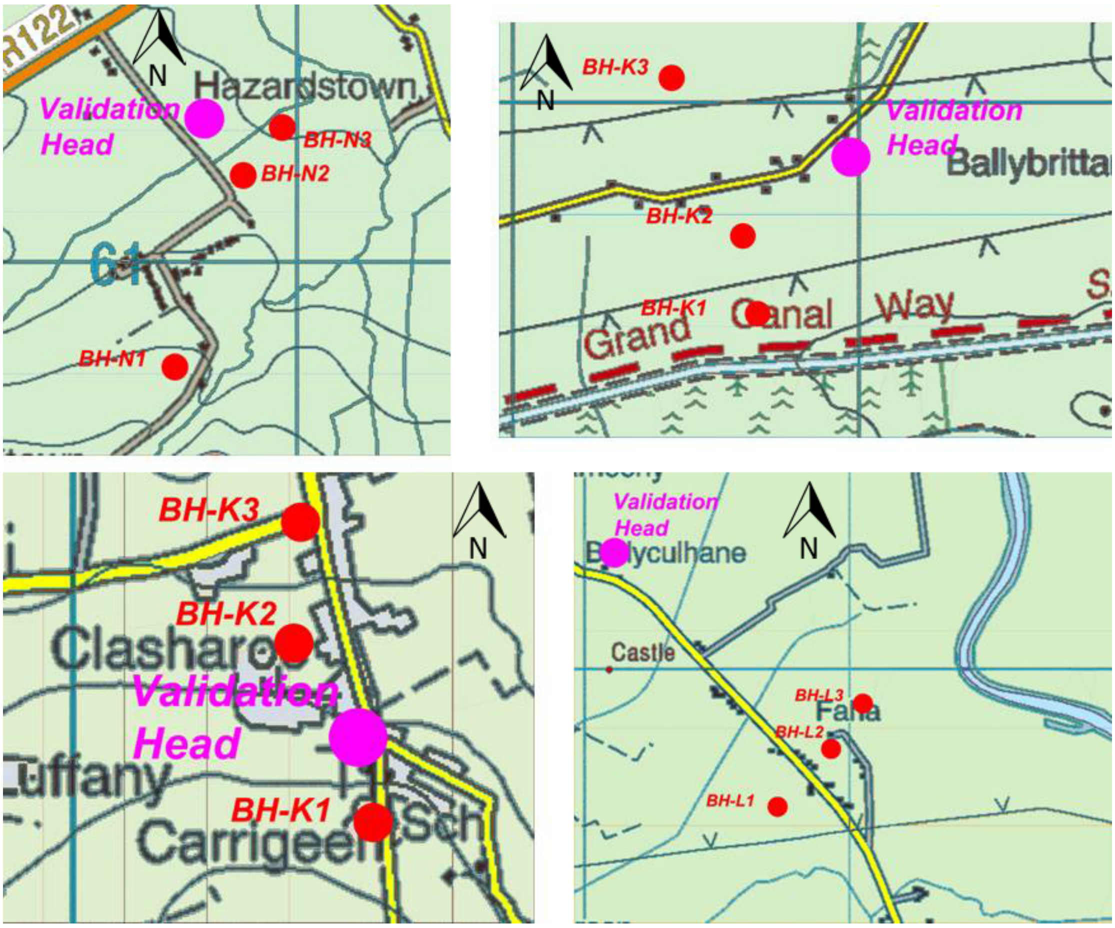


Figure 8.34 Locations of validation heads at the modelled locations (not to scale)

These spot measurements were compared against the modelled groundwater heads at the locations in order to validate the model. In nearly all cases it was not possible to access these locations on more than one occasion and therefore no time varying data is available at these locations. The steady state model was developed on averaged steady state head values and it is possible that a "like-for-like" comparison is therefore not being made, however this was the only available data to validate the model. A summary of the head for model validation are shown in Table 8.2 below.

Table 8.2 Validation of model head values

Location	Observed Value (mOD)	Simulated Value (mOD)
Naul, Co Dublin	46.59	48.18
Rhode, Co. Offaly	81.4	80.59
Carrigeen, Co. Kilkenny	34.4	34.28
Faha, Co. Limerick	1.38	1.26

It can be seen from Table 8.2 that the simulated values differ from the observed values from between 0.12 m – 1.59 m. Given that the model was calibrated on averaged data over a 24 month period and that the validation heads were once off spot measurements it is concluded that the models in general give an accurate reflection of groundwater head values across the catchment. It must be noted that the models take in large areas and that the error seen above is generally acceptable given the large areas being modelled.

8.4.2 Model Parameter Sensitivity Analysis

In order to assess the robustness of the model a number of parameters were varied within a reasonable range to identify the resulting effects. In order to properly calibrate the model in steady state conditions for water heads a parameter calibration model using PEST, a package contained with Groundwater vistas, was used which resulted in ideal values for hydraulic conductivity. As the model was finely calibrated for hydraulic conductivity to give the best fit and lowest RMS error for the target heads it was not be considered as a parameter to be subjected to the sensitivity analysis. The key parameters identified within the simulations for contaminant transport were longitudinal dispersivity, porosity (storage) and the lumped removal rate for bacteria λ. The longitudinal dispersivity values were estimated initially from the literature as given in Table 8.1, and the simulation had used mid-range values from therange of values cited for each of the bedrock unit.. The sensitivity analysis considered values within the entire range given in Table 8.1.

The results of the sensitivity analysis indicate that the model is very sensitive to dispersivity values. Decreasing the dispersivity values resulted in a plume of higher concentration contaminants migrating further downstream. Higher values of dispersivity resulted in much smaller plumes of contaminants in lower concentrations and the plume did not migrate very far downstream. For example, the sensitivity analysis for bacteria at the *EXTREME* vulnerability site is given in Figure 8.35 below and it can be seen that for a dispersivity of

15m the plume is 220m at its widest point and extends over 450m downstream; this compares to a 160m wide plume extending only 90m downstream for a dispersivity of 150m. Similar results were found both at the *MODERATE* and *HIGH* vulnerability sites.

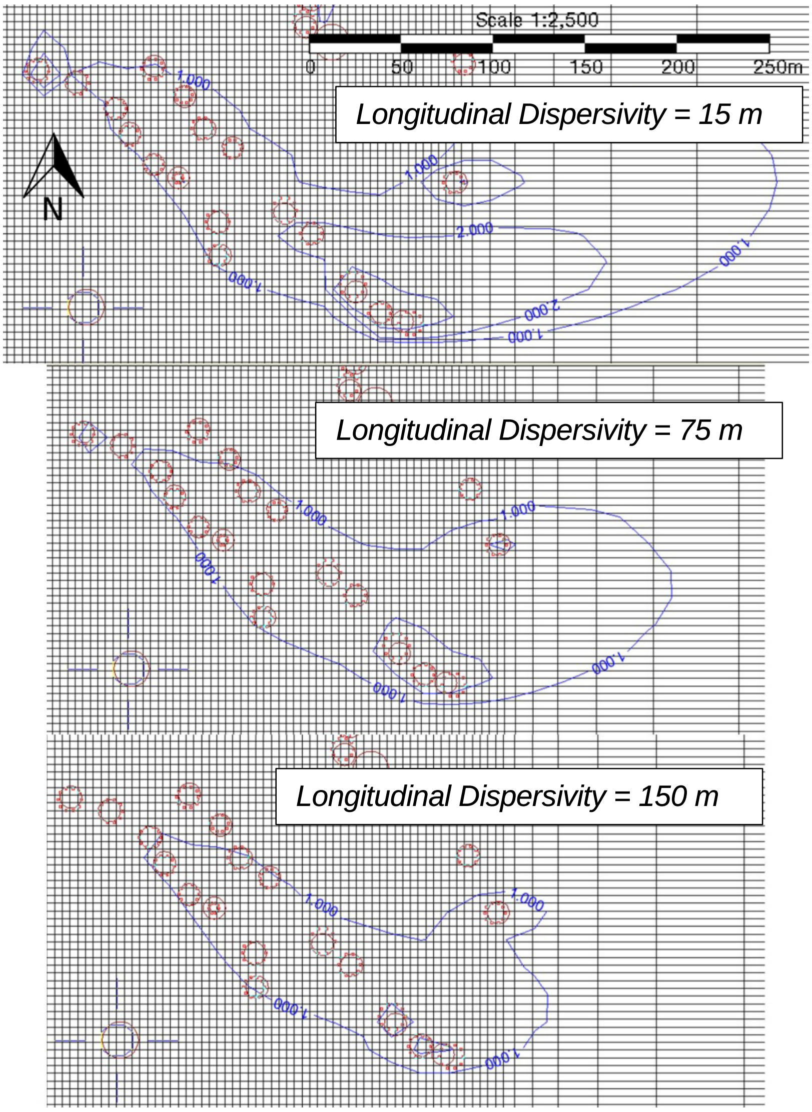


Figure 8.35 Effects of varying the longitudinal dispersivity at the *Extreme* vulnerability site in Limerick

It can therefore be seen that selecting an accurate value for dispersivity can have a large impact on the outcome of the simulations. As the only data available to chose such a parameter for the groundwater modelling in this project were the mid-range values from literature, (it is possible that this parameter has either been over or under estimated, although it should be noted that the values chosen did appear to give representative results when compared to the field monitoring results.

The porosity and the associated storage parameters for an aquifer were also shown to be highly important when considering contaminant transport. Porosity will also have an impact on water levels in a transient model due to the varying quantity of storage available for recharge. Again limited information was available on the specific aquifers in the study and a value of 2% porosity was assumed for all simulations as per GSI reports detailed in Chapter 6. The effects of changing porosity and the resulting storage parameters are illustrated in sample MODFLOW MT3D output from the EXTREME vulnerability site – see Figure 8.36 below.

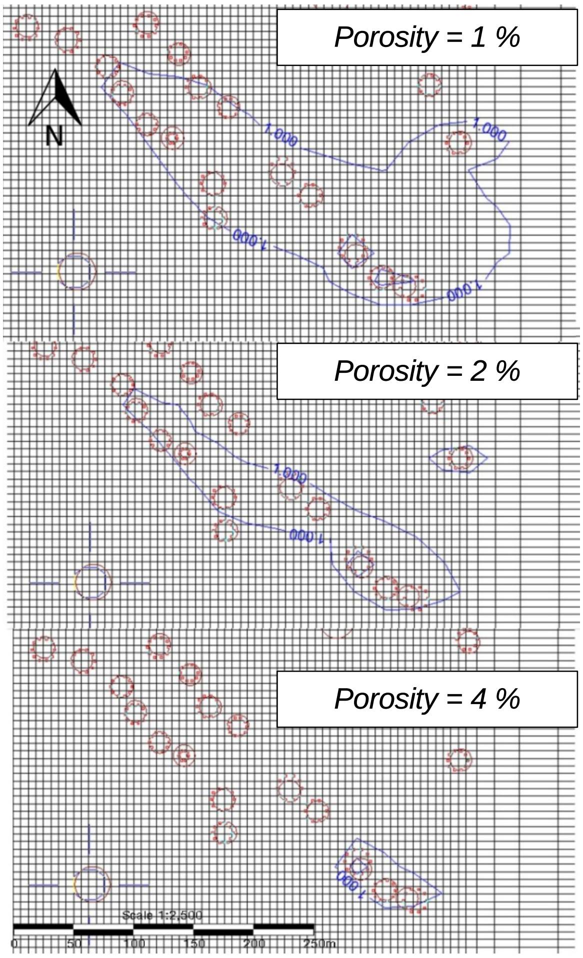


Figure 8.36 Effects of varying porosity values at the *Extreme* vulnerability site

As was seen with the results of the dispersivity sensitivity analysis, decreasing the porosity of the aquifer increases the concentrations of contaminants observed and also produces a wider and more extensive plume downstream of the cluster development as seen in Figure 8.36 above a porosity of 1% resulted in a plume that was 90m wide extending 150m downstream and this compares with a porosity of 4% which only produced an isolated plume to the north 40m in width extending no mnore than 30m downstream of the cluster development. This is to be expected as increasing the porosity increases the storage available within the aquifer and thus provides more water for dilution and dispersion of the contaminants. Given that Irish aquifers do not tend to have very large faults, with the exception of conduit flow in karst regions, the 2% porosity assumed in these simulations would appear reasonable (GSI, 2009).

The initial estimate of the lumped removal rate for bacteria was based on a study by Pang et al. (2006) which was similar to this current study. A value of 0.6 day^{-1} was found to give only very localised plumes of contaminants surrounding the cluster of OSWTS's with values reducing to zero within 100 m downstream. The removal rate was varied from 0.5 – 3 day^{-1} during the sensitivity analysis which showed very little effect on the outcome of the simulations; when changed from 0.5 – 3 day^{-1} the maximum plume concentration did not change and the plume extent reduced by 20m. What was found to have the biggest impact on the outcome for bacteria was in fact the concentration of the incoming bacteria and specifically what proportion of the existing OSWTS's were assumed to have poorly installed percolations areas with only a single trench receiving the entire daily loading rate. Initially for all of the simulations it was assumed that the treatment systems included at each of the cluster developments contained 5% of poorly installed percolation areas. The effects of increasing this to 10% and 20% of poorly installed treatment systems with only a single trench receiving the daily loading is shown in Figure 8.37. It can be seen that the plume of bacteria increases to levels that are of concern when more than 5% of poorly constructed treatment systems are included in the simulations. Increasing this proportion further to 20% results in the plume migrating further downstream with a large area now contaminated with bacteria which would be of great concern. Again it is very difficult to know exactly what the standard and condition of any existing percolation area is without extensive and intrusive site works. All of the simulations included as part of this study therefore included many assumptions in this regard – the sensitivity analysis detailed above for bacteria illustrates a potential weakness in the results of these simulations due to the limited data available.

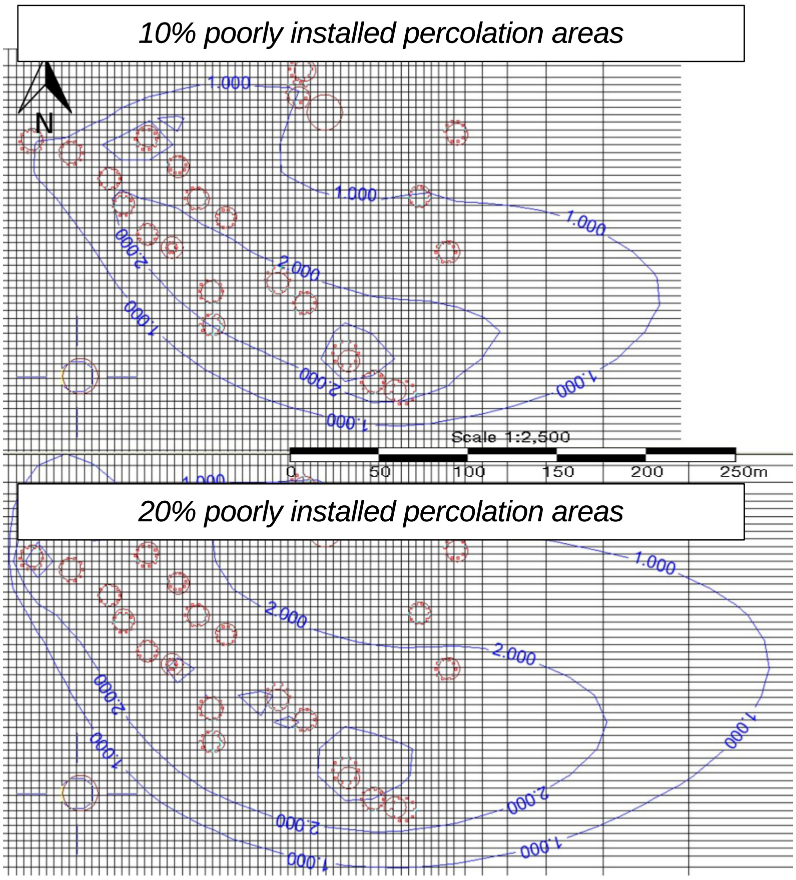


Figure 8.37 Effects of varying the proportion of poorly installed percolation areas at the Extreme vulnerability site

9 DISCUSSION

9.1 Introduction

This chapter will present a discussion of the results and outcomes of this project. A comparison will be made between the results from the field monitoring chemical and bacterial analysis and with those predicted using the groundwater modelling software. Following on from these comparisons, an attempt will be made to investigate the implications of the study on OSWTS's in Ireland into the future. The main aim of the project was to investigate what the impacts of the density of OSWTS's on groundwater quality are. In addition, suggestions are made as to appropriate guideline densities of these systems, given the groundwater vulnerability rating, for future planning and development purposes. This analysis has been achieved by using the groundwater models developed and described in Chapter 8 to simulate increases in the density of these systems and the associated impact on downstream water quality.

9.2 Comparison of Field and Model Results

Field monitoring results were given in Chapter 5 and the groundwater modelling outputs were given in Chapter 8. Hence, the outputs of these two sets of data are compared in order to investigate the implications of these analyses on groundwater quality and OSWTS density at the four vulnerability areas. Whilst average groundwater head values were used to calibrate the initial steady state groundwater models, it was decided that the observed chemical and bacterial data from the field monitoring would not be used as target values to calibrate the models for transport properties. The reason for this decision was due to the purpose of the model and also with the availability of input data. The model is being used primarily to quantify the effects that OSWTS's have on groundwater quality and does not consider other major contaminant inputs such as agriculture, whereas the observed field data contains contaminants from all inputs. Therefore if the model were calibrated to the field data it would be necessary to quantify all inputs into the catchment which is beyond the scope of this project.

Using only OSWTS data as contaminant inputs to the model, it was possible to see what ***additional*** loading is being caused by clusters of these systems in order to address the scope and objectives of this study. Given the above, this section will therefore compare the concentrations of the various contaminants at the monitoring boreholes and any concentrations over and beyond those predicted by the model will therefore be explained through either unaccounted for inputs into the models or through underestimations or errors in the model parameters.

9.2.1 *Low* Vulnerability Site

As was seen in Chapter 7, HYDRUS did not predict any of the OSWTS's loading reaching the water table even when the modelling period was extended to 15 years. This was the case for both the best case scenario where the percolation areas were built to the correct standard (4 trenches) and also allowing for a worst case scenario whereby all of the effluent was directed down only one percolation trench. The results of the unsaturated zone modelling match the field monitoring data very well for ammonium, nitrates and bacteria. Average values of these contaminants observed during the study were all zero as predicted by HYDRUS (MODFLOW contaminant transport models were not carried out). For phosphorus, HYDRUS again predicted that none of the OSWTS loading would reach the groundwater table however values observed in the aquifer during the field monitoring ranged between 0 and 0.223 mg-P/l with average values of 60 µg-P/l. Average values of phosphorus in the aquifer were therefore above the threshold value of 35 µg-P/l for the protection of surface waters. However, these values were observed both upstream and downstream of the cluster development and since there were no increased levels of any other contaminants associated with OSWTS's, it can only be concluded that the increased levels of phosphorus in the aquifer can be associated with another input. One possible explanation for this is that the phosphorus values observed may in fact be due partly to the natural hydro-chemistry of the groundwater. Groundwater hydrochemistry is dependent on the nature of the subsoils and rocks that the groundwater passes through and therefore in this area it is possible that the bedrock geology is contributing to the dissolved concentrations of phosphorus in the area. This has been discussed in some detail previously for a core group of substances (EPA, 2003) however phosphorus was not one of these. Bedrock in this *Low* vulnerability area is shaley limestone with the geology upstream of the site comprising of shales, thin sandstones/siltstones with occasional thin limestones (See Chapter 3 for details).

9.2.2 *Moderate* Vulnerability Site

The unsaturated zone models described in Chapter 7 predicted concentrations of nitrates, phosphorus and bacteria entering groundwater from the OSWTS's in varying concentrations depending on the treatment system type and whether the percolation area was constructed to the correct standard. One of the more difficult aspects of modelling the effects of the cluster system in the groundwater model was whether to assume that the percolation areas had been installed correctly and therefore which of the contaminant loading factors to apply. As described in Chapter 4, an attempt was made to assess what type of system was installed at each of the existing dwellings (at each of the site locations) and the likely condition of the associated percolation area and it was based upon this assessment that the model was setup for contaminant loading.

The main indication from the groundwater model with respect to nitrate concentrations, was that whist the cluster development was raising groundwater concentrations locally, the effects were dissipated within 100 – 200 m downstream of the development with the simulations predicting concentrations generally increasing by 0.5 – 1 mg-N/l. Given that the average nitrate concentrations observed during the field study ranged from 0 – 13.2 mg-N/l, it can be concluded that OSWTS's in the area are only contributing to a small proportion of overall loading in the area and it is likely that the concentrations observed are mainly arising from other sources with OSWTS's making some smaller contribution. This argument was further supported using statistical methods with no significant difference found between upstream and downstream mean concentrations over the course of the field study.

As was seen in the results for phosphorus at the *LOW* vulnerability site, concentrations both upstream and downstream of the study development exceeded the recommended limit of 35 µg-P/l for surface water protection. Statistical tests between upstream and downstream concentrations did not find any significant difference. The groundwater model indicated similar results as those for nitrates, with a plume of 1 µg-P/l extending approximately 250 m beyond the cluster development. Given that observed values in the area ranged between 20 – 55 µg-P/l, the cluster of OSWTS's would not appear to be contributing any significant portion of this loading and again this is what would have been expected at the outset.

The HYDRUS simulations predicted low levels of bacteria entering the groundwater system of between 1 – 38 CFU/l (0.1 – 3.8 CFU/100ml) again depending on the state and type of treatment system involved. Based upon these loading rates the groundwater model only predicted very localised plumes of bacteria and only in areas where it was assumed that a substandard percolation system had been installed. The numbers of both bacteria groups observed during the field monitoring were also very low with spikes of higher numbers observed during a limited number of months. It was seen again however that there was no significant difference between upstream and downstream bacteria numbers which would again indicate that any spikes observed were not due to the cluster of OSWTS's. Given that HYDRUS simulated only very localised plumes of bacteria forming (<40 m) around the OSWTS's, it is possible that the spikes that were observed during the study were due to agricultural practices in the fields where the boreholes were located, particularly following periods of heavy rainfall which would have driven the bacteria into the groundwater. This argument is supported due to the fact that the spikes were observed during the same periods that either animals would have been grazing or slurry would have been spread. The regression analysis of rainfall and bacteria indicated that the presence of bacteria was related to the rainfall in the preceding days/weeks and this would further support the logic given above. Overall it is concluded that whilst the cluster of OSWTS's in the area is contributing some proportion of contaminant loading to groundwater, the levels involved do not represent a significant threat to quality in the area at the current density.

9.2.3 *High* Vulnerability Site

HYDRUS predicted lower concentrations of nitrates, phosphorus and bacteria entering groundwater from the OSWTS's at the *HIGH* vulnerability site than at the *MODERATE* vulnerability site which was at first surprising; however, given that the water table is deeper at the *HIGH* vulnerability site there is a thicker layer of subsoil available in the vadose zone for treatment of contaminants. Groundwater vulnerability ratings in Ireland are based upon the thickness of subsoil above the bedrock and not above the water table. The cluster development at Carrigeen in Co. Kilkenny consisted mainly of a housing development that was built relatively recently when compared with the other sites and therefore more information was available on the location and type of treatment systems installed. Detailed plans were available from the local authority and these were used when setting up the groundwater model. It is notable that the development consisted mainly (>75%) of secondary treatment systems with low pressure soil infiltration systems.

Observed field monitoring results for nitrates ranged from 0.5 – 12 mg-N/l with upstream values higher than downstream values for the entire duration of the monitoring period and this was demonstrated through statistical tests on the data. The groundwater model simulated a plume of concentration between 0.5 – 1 mg-N/l reaching the downstream monitoring point due to the cluster of OSWTS's. It was concluded from observations and discussions with local homeowners during the entire period of the project that the upstream nitrate concentrations were being influenced from poor agricultural practices in adjacent fields, particularly slurry spreading. It has been reported that similar Devonian sandstones in Co. Kilkenny to those found at the study site contribute up to 3 mg-N/l to background nitrate concentrations (EPA, 2003). Given that the average downstream nitrate concentration observed at the *HIGH* vulnerability site was 5.1 mg-N/l the groundwater model would indicate that up to 20% of this can be attributed to the cluster development of OSWTS's at the first downstream borehole in the bedrock horizon However, similarly to what was seen at the *MODERATE* vulnerability site, these levels are not a cause for concern and as the statistical analysis showed that mean concentrations were higher upstream overall, indicating that the cluster development was not degrading downstream water quality.

The groundwater model predicted a plume of phosphorus reaching the downstream monitoring point of less than 1 µg-P/l. Observed phosphorus concentrations at the downstream borehole during the study were in excess of 22 µg-P/l with similar values observed upstream of the cluster development. Statistical tests comparing upstream and downstream concentrations did not find any significant differences indicating that the cluster is not contributing significantly to groundwater phosphorus concentrations in the area. Given that overall phosphorus concentrations in the area appear to be higher than what would be expected with most months sampled exceeding the limit of 35 µg-P/l it would appear that there is a significant source of phosphorus in the area. This source could, as discussed previously, be the bedrock aquifer or the subsoil itself, although it is not clear what is causing these phosphorus concentrations. Given the data collected and the outcome of the groundwater simulations it was concluded that the cluster development is not the source of these increased phosphorus concentrations.

Given the available depth of unsaturated subsoil at the *HIGH* vulnerability site, the HYDRUS simulations only predicted very low numbers of bacteria reaching groundwater and consequently the groundwater model did not predict any significant plumes of bacteria developing. This is at odds with what was found during the field monitoring where a number of months recorded spikes up between 1 – 22 CFU/100ml for both *E-coli* and *Enterococci.* It was seen during regression analysis that bacteria numbers were statistically highly related to intense rainfall in the previous 24 – 48 hour period. During such intense rainfall events when the subsoil is in a temporarily saturated state the travel times of bacteria to the groundwater table would have been significantly increased. It is also likely that the groundwater table would have been temporarily raised during these intense rainfall events particularly due to the high K_s value determined during field tests. The water table was observed to have fluctuated highly over the course of the study from 1.6 m BGL to 5.4 m BGL. These fluctuations may not have been accounted for accurately in the HYDRUS simulations. It is also possible that preferential flow paths exist in the subsoil that were not modelled explicitly; preferential flow paths being particularly prevalent in glacial tills due to the heterogeneous nature of such subsoil. The effects of preferential flow is accentuated however when the subsoil is shallow, as interconnectivity does not need to be extensive to provide a direct pathway to the water table. HYDRUS does have the option to simulate using a dual-porosity model with two zones of unequal porosity which could be calibrated to simulated preferential flow paths, however it was not possible to fully calibrate and parameterise this type of model during this present study. Given that upstream and downstream values of bacteria were significantly different with upstream values being higher on average it is difficult to determine the origin of the bacteria observed at the downstream monitoring point during this study. Overall it was concluded that whilst both phosphorus and bacteria concentrations are high in the area, it is not statistically reasonable to assume that the cluster of OSWTS's were the cause and thus specific conclusions are difficult to draw in this case.

9.2.4 *Extreme* Vulnerability Site

Nitrogen was observed in groundwater samples during the study as ammonium, nitrite and nitrate. Given that ammonium and nitrite have been seen to change into the form of nitrate very quickly in the subsoil (Gill et al., 2005; 2009) this indicates that when these forms of nitrogen were observed the source had to have been very close to where the sample was taken. Given that both the nitrite and ammonium time series data indicate distinct seasonal patterns with values being detected between December and April there are two possible explanations for these observed values; the seasonal pattern is due to agriculture with cattle grazing and local slurry pits the likely sources or the pattern exists due to higher effective rainfall during these winter/spring months "pushing" the contaminants through the subsoil quickly before the nitrification process had fully taken place. The regression analysis carried out validated the process seen previously whereby concentrations of both nitrite and ammonium were found to be significantly related to the preceding 21 days rainfall (in months that had an intense rainfall event in the previous 21 days). As discussed in Chapter 7, the transition of ammonia to nitrate was assumed to occur almost instantly and therefore NO_2 was not considered in the simulations and in addition HYDRUS did not predict any ammonia (as ammonium) reaching groundwater due to the high nitrification rates in the subsoil. It must be noted that all of these simulations assumed a minimum of 1.2 m of unsaturated subsoil beneath base of the percolation trenches and the water table. However, given that the water table is present at less than 1 m BGL it is possible, and likely that these raised percolation areas may not have been included during the installation of OSWTS's in the area. Equally, only between 0.5 – 1 m of unsaturated subsoil is available across the area modelled for the treatment of ammonium from agricultural effluent discharged at ground level. A cross-section of the ammonia concentration at 0.5m below the percolation trench from the HYDRUS models is shown in Figure 9.1 below. The model indicates an average ammonia concentration of 0.22 mg-N/l when only 0.5 m of subsoil is available to treat the incoming wastewater. This may explain the ammonium concentrations encountered during the study although it is still not evident whether it arises from human or agricultural wastewater. As neither nitrite nor ammonium were input into the groundwater models (based on the HYDRUS output) comparisons between field results and model output were not possible for these forms of nitrogen. Given that statistical tests comparing upstream and downstream nitrite and ammonium concentrations gave an insignificant result it could be concluded that agriculture was the likely source of these contaminants. However, there is the possibility (however unlikely) that a plume of high nitrite and ammonium concentrations did exist downstream of the study area and it was missed due to the incorrect locating of the downstream boreholes. The groundwater model did indicate that the plumes for nitrate and phosphorus were moving south-east around the topographical high adjacent to BH-L2 and this will be discussed further below.

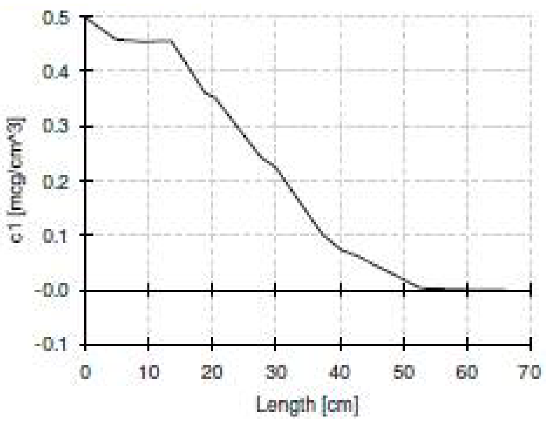


Figure 9.1 HYDRUS predicted ammonia at 0.5 m below wastewater input depth at Extreme vulnerability site

Nitrate concentrations recorded during the field study ranged from 0.2 – 9.8 mg-N/l with no statistical difference in means upstream and downstream concentrations. The groundwater model simulated a plume of nitrate concentrations of 0.1 – 0.5 reaching the downstream monitoring boreholes. It appears clear from the output of the MODFLOW MT3D model that the plume of contaminants is migrating around the outcrop of rock in the centre of the study area in a south-east direction. A plume with concentrations of up to 2 mg-N/l was seen to form up to 250 m downstream of the cluster development. If this is the case on the ground then the conclusions from the study at this site may not be definite, however even at levels of 2 mg/l the plume does not pose a major issue to groundwater quality in the area. A similar outcome was seen for phosphorus simulations with the plume migrating to the south-east. Even though statistical tests between upstream and downstream concentrations indicate that the cluster development is not having an impact on groundwater concentrations in the area, it is possible that had the boreholes been located where the plume is predicted by MT3D the outcome would have been different. However the plume has largely dissipated downstream (>500 m) of the cluster development, similar to what was seen for both the *MODERATE* and *HIGH* vulnerability sites – the cluster development is having an impact on immediately adjacent groundwater quality but the effects on local groundwater quality are highly muted.

HYDRUS did predict high concentrations of bacteria entering the groundwater system at the *EXTREME* vulnerability site, however the development of a plume of bacteria was not extensive as might have been assumed and the resulting downstream concentrations were very low. It was seen however, in Section 8.4.2 that the bacteria model was very sensitive to the input loading particularly to the scenario discussed in Chapter 7 whereby a percolation area was not constructed to the proper standard and all of the loading is discharged via a single percolation trench. Increasing the proportion of systems built incorrectly in this manner to 10 and 20% of the OSWTS's contained within the cluster development was seen to significantly increase the migration and concentration of the plume of bacteria downstream of the cluster development. For example, assuming 20% of systems incorrectly constructed results in a plume that extends up to 500 m downstream with a concentration of 2 CFU/l. Given that there is a possibility that many systems around the country may in fact be built to a substandard, this modelling suggest that there may be some cause for concern. However given the extremely low reported outbreaks of water related pathogen sickness in Ireland this would not appear to be the case. This is an issue that will receive more attention in the coming years with the introduction of the national OSWTS inspection plan. In line with this, it should be noted that when no substandard systems were included in the simulations, downstream plumes and concentrations of bacteria at all of the study site simulations were negligible.

9.3 *Extreme* Vulnerability Karst Site

At the outset it was hoped to establish a groundwater link between the treatment system at Toonagh, Co. Clare and the local karst conduit groundwater system. This was successfully achieved during this study. Given that therefore SE treated wastewater is being discharged directly into groundwater, it was then hoped to quantify what effect this was having on local water quality. Given the direct links and inter-changeability between surface and groundwater in the area it was difficult to make a definitive conclusion in this regard. It was noted during the study that neither phosphorus, nitrate nor bacteria were higher downstream of the treatment plant suggesting that a combination of dilution and/or treatment in the aquifer were successfully serving to assimilate input contaminants. During the study period it was observed that a high intensity rainfall event resulted in bacteria levels in the spring downstream to exceed 1000 CFU/100ml which poses a significant risk. It was not clear whether a portion of this was attributable to the treatment plant discharges as the upstream concentrations were also similarly high. The observed chloride/bromide ratios indicated that anthropogenic sources of contamination were likely to be occurring upstream in the catchment and this again serves to highlight the vulnerability of these types of groundwater systems to faecal contamination. Further studies would be required to fully understand the complicated processes occurring in this study area particularly with respect to the treatment plant at Toonagh.

9.4 Implications for OSWTS Density

The main aim of this study was to identify if the density of OSWTS's has an impact on groundwater quality. In general the results of both the modelling and the field monitoring do not suggest that the studied cluster developments are having an impact on groundwater quality at their current density of OSWTS. However, additional housing units were added to the existing houses with associated OSWTS's, it is possible that eventually a density would be reached whereby an undesirable impact on groundwater quality would occur. This section will build upon the models developed in Chapter 8 and investigate what (if any) is effect of increasing the density of systems for each of the groundwater vulnerability ratings. This exercise will not be carried out for the LOW vulnerability site due to the fact that HYDRUS did not predict any loading to groundwater from OSWTS and as the effects would be similar for all contaminants only nitrates will be considered. For this type of catchment the impacts of OSWTS's is more likely to be on surface water and this has been investigated previously by Hynds et al. (2012). For ease of reference the details for each of the study areas given previously in Chapter 4 have been repeated below in Table 9.1 – Table 9.2.

Table 9.1 Breakdown of treatment systems by study location

	Number of Systems Present					
	Septic Tank			Secondary Treatment		
Location	Pre 1991	EPA 2000	EPA 2009	Pre 1991	EPA 2000	EPA 2009
Naul, Co. Dublin	17	1	0	0	3	0
Rhode, Co. Offaly	10	0	0	0	1	0

Carrigeen, Co. Kilkenny	1	2	0	0	14	0
Faha, Co. Limerick	12	1	0	0	6	1
Toonagh, Co. Clare	2	0	0	*60 PE Secondary treatment plant*		

Table 9.2 Summary of OSWTS density at each study area

Site Location	Average Plot Size (hectare)	No. Units in Cluster Development	Approx. Area of Cluster (hectares)	Treatment System Density (Units/hectare)
The Naul, Co. Dublin	0.35	21	11.6	1.82
Rhode, Co. Offaly	0.30	11	10.6	1.04
Carrigeen, Co. Kilkenny	0.18	17	6.9	2.44
Faha, Co. Limerick	0.20	20	9.9	2.04

As was discussed in Chapter 2 and 3, cluster developments containing OSWTS's in rural Ireland tend to take the form of ribbon developments along local roadways. Figure 9.2 below gives an indication of the plot size required for a standard rural dwelling place with the associated treatment system and percolation area. The layout of the dwelling and OSWTS are constrained by the EPA CoP (EPA, 2009) as set out in Table 9.3 below. It can in Figure 9.2 below that increasing the density incrementally with additional units per hectare from 1 up to 6 units/hectare, does not pose issues for accommodating the constraints listed in Table 9.3 below. Beyond a density of 6 units/hectare it would become extremely difficult to accommodate the required constraints given the decreased plot size. This would indicate that an area of approximately 0.17 ha or 0.42 acres is the minimum plot size required to accommodate a dwelling with an OSWTS. It is noted that currently a number of local authorities apply a rule of thumb requiring a minimum plot size of 0.5 acres or 0.2 ha (i.e. 5 per hectare) as discussed in Chapter 2 – it would seem that this requirement is not unreasonable. For the exercises detailed below additional OSWTS's were added in a similar manner as they would be built in a ribbon type arrangement along the local roadway.

Table 9.3 Minimum separation distances for: septic tanks, intermittent filters, packaged systems, percolation areas and polishing filters (m) (taken from Table 6.1 in the EPA CoP, 2009)

Wells	-
Surface water soakaway	5
Watercourse/stream	10
Open drain	10
Heritage features, NHA/SAC3	-
Lake or foreshore	50
Any dwelling house	7 septic tank; 10 percolation area
Site boundary	3
Trees	3
Road	4
Slope break/cuts	4

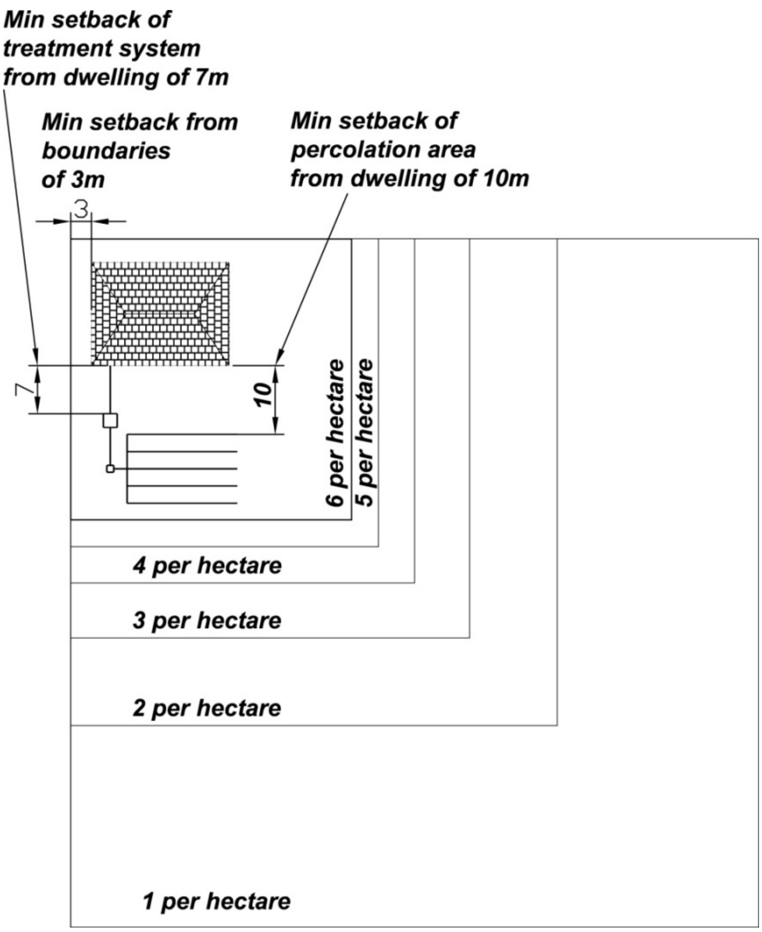


Figure 9.2 Sketch showing a range of plot sizes based on increasing OSWTS densities

9.4.1 OSWTS Density in *Moderate* Vulnerability Areas

This study area currently has a density of 1.04 units/hectare which is the lowest of all the developments monitored during this study. This density was increased to 2, 2.5 and 3.5 units/hectare in the MODFLOW MT3D model and the resulting concentration plumes for nitrate and bacteria are given in Figure 9.3 and Figure 9.4. It can be seen that whilst increasing the density has the effect of increasing the concentrations in the plume in a localised area surrounding the OSWTS's; the downstream concentrations are very similar for all scenarios modelled indicating that the effects of increasing densities of OSWTS's has a big influence at a micro-scale for the protection of drinking water abstractions however on a more local and regional scale the effects are greatly muted.

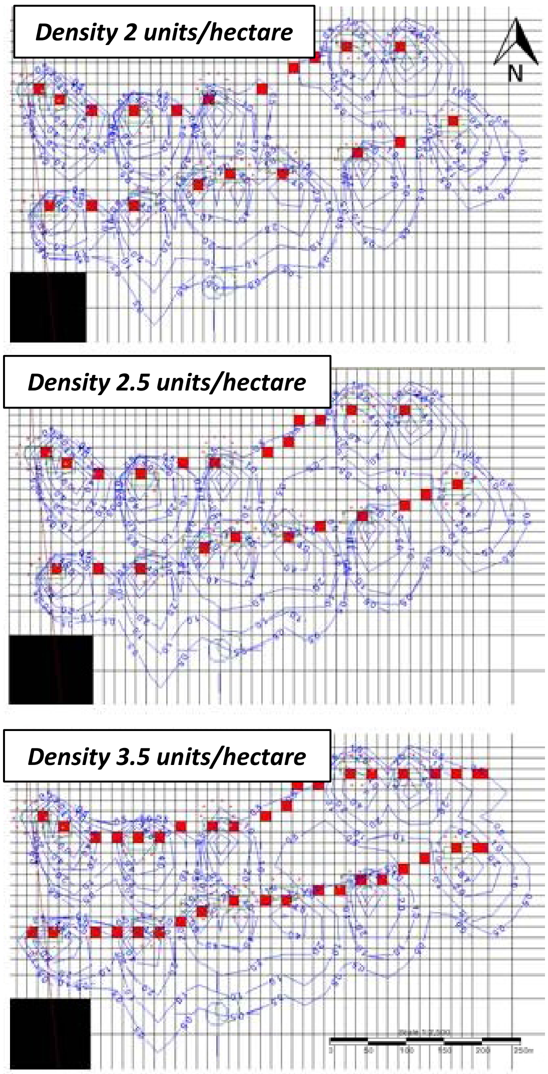


Figure 9.3 Nitrate plumes for increasing densities of OSWTS's at the Moderate vulnerability site

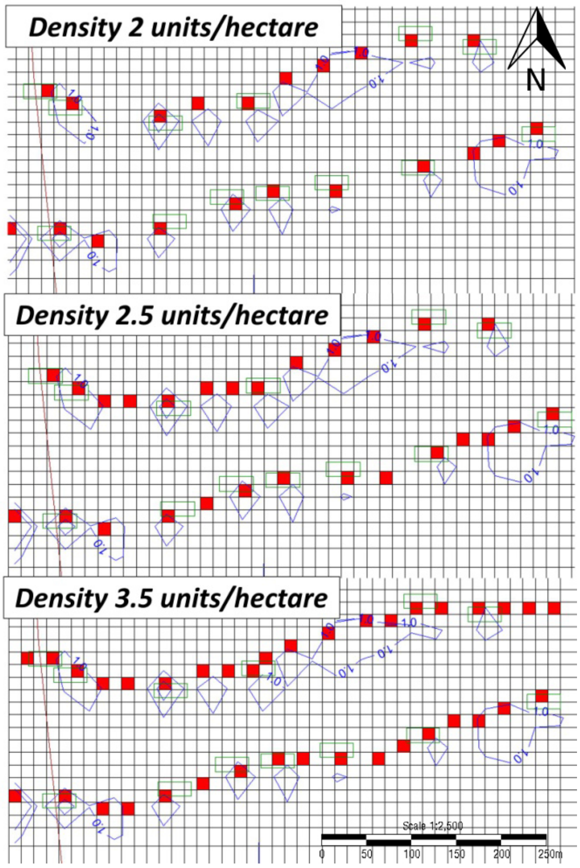


Figure 9.4 Bacteria plumes for increasing densities of OSWTS's at the *MODERATE* vulnerability site

9.4.2 OSWTS Density in *High* Vulnerability Areas

This study area currently has a density of 2.44 units/hectare. This density was increased to 3, 3.5 and 4 units/hectare in the MODFLOW MT3D model and the resulting concentration plumes for nitrate are given in Figure 9.5 below. As HYDRUS did not predict any bacteria loading to groundwater bacteria was not considered in these simulations. For nitrate it can be seen that a similar effect occurs as that which was seen at the *MODERATE* vulnerability site with localised concentrations surrounding the OSWTS's increasing significantly but further downstream the effects become muted quite quickly. The plume does appear to be migrating slowly downstream however, with downstream concentrations very similar for all scenarios modelled.

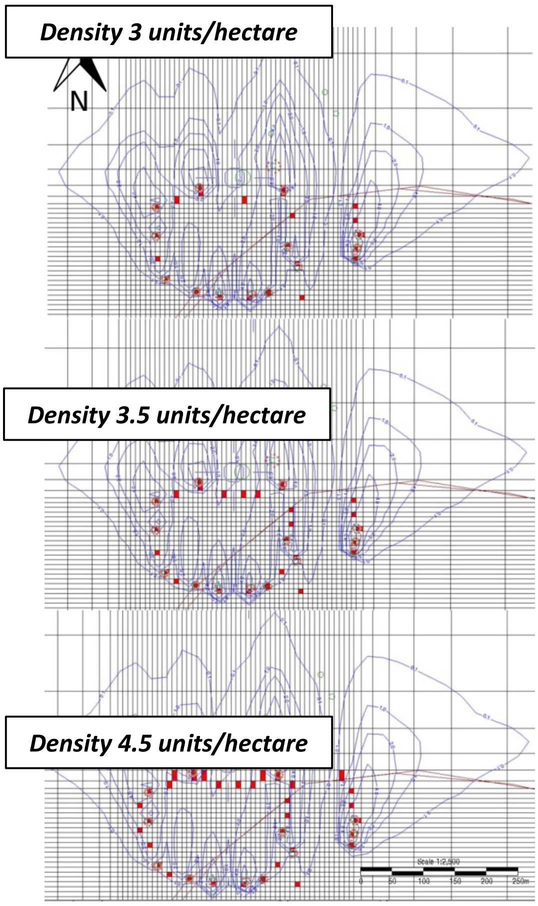


Figure 9.5 Nitrate plumes for increasing densities of OSWTS's at the High vulnerability site

9.4.3 OSWTS Density in *Extreme* Vulnerability Areas

This study area currently has a density of 2.04 units/hectare. This density was increased to 2.5, 3, 3.5 and 4 units/hectare in the MODFLOW MT3D model and the resulting concentration plumes for nitrate and bacteria are given in Figure 9.6 and Figure 9.7 below. Localised concentrations surrounding the OSWTS's can again be seen to increase with downstream effects muted for nitrate as seen previously. The plume does appear to be migrating in different directions and this is due mainly to the topography of the bedrock geology in the area. Given that additional OSWTS were added in the ribbon pattern along the local roadway it would appear that this is the reason the plume is spreading out to the north-east and the south-west. For bacteria the increasing the density to 2.5 units per hectare has the effect of creating a new concentration contour of 5 CFU/l which was not observed in the contaminant transport results in Chapter 8 – a concentration of 2 CFU/l was the highest observed previously. Further increases in the density to 3 and 3.5 units/hectare results in this 5 CFI/l contour migrating further downstream with steady state then reached and increasing the density further then does not cause it to migrate any further. This indicates that increasing the density in the area may have negative impacts on bacteria levels in groundwater in the area, however as was seen in the HYDRUS output in Chapter 7; were the new systems to have an increased unsaturated zone (through a mounded percolation area) or reduced hydraulic loading rates (through the use of increased percolation trenches) these effects could be mitigated with bacteria loading potentially fully removed before discharging to groundwater.

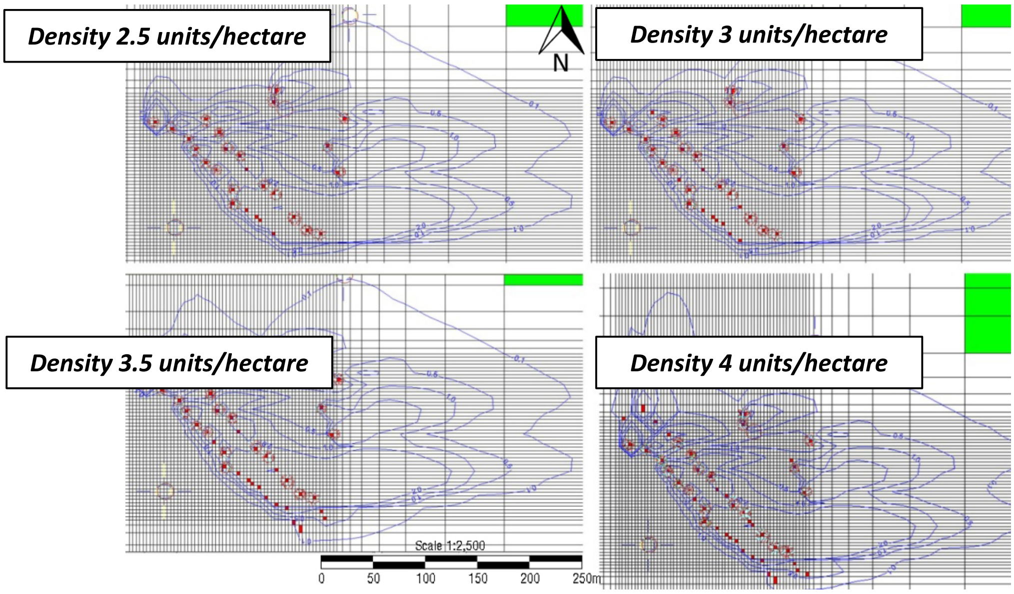


Figure 9.6 Nitrate plumes for increasing densities of OSWTS's at the Extreme vulnerability site

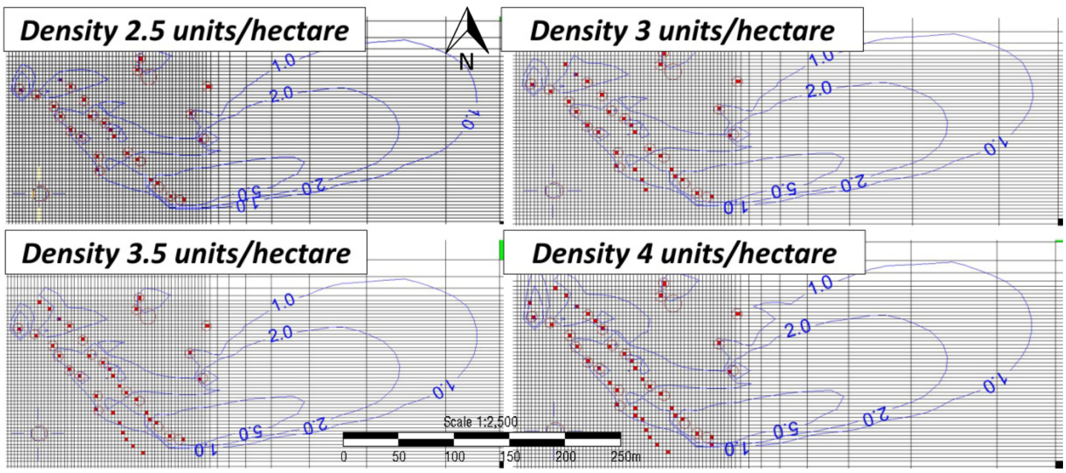


Figure 9.7 Bacteria plumes for increasing densities of OSWTS's at the Extreme vulnerability site

10 CONCLUSIONS AND RECOMMENDATIONS

10.1 Conclusions

Field Monitoring

- The field monitoring results indicate that the clusters of OSWTS's are not having a significant negative impact on groundwater quality in any of the different sites monitored across. Statistical analysis of the results showed that in nearly all scenarios mean concentrations of all parameters monitored were similar upstream and downstream of the clustered developments regardless of the groundwater vulnerability.

- It was found that bacterial spikes were recorded following intense rainfall events and regression analysis indicated that rainfall appears to be the main driver of bacteria concentrations in groundwater (although the source of such contamination could not be explicitly linked to the on-site systems)

Vadose Zone Models

- The vadose zone models confirmed that the thickness of unsaturated subsoil beneath the percolation area is key to the magnitude of the concentrations of contaminants entering groundwater

- An estimation of nitrification rates in Irish subsoils was made using previous field study data. The modelling showed that (as per the field studies) the vast majority of nitrogen in both STE and SE in the form of ammonia (and organic N) being nitrified to NO_3 within about 1 m of unsaturated subsoil

- An estimation of denitrification rates in Irish subsoils was also made using previous field study data for both STE and SE however this model calibration was not successful and instead values from literature were used

- The vadose zone modelling also confirmed that phosphorus loading to groundwater from OSWTS's is likely to be low even in varying subsoil scenarios, again as shown in previous Irish field studies and confirmed with the presented field monitoring results which did not show significant changes upstream and downstream of the study developments

- It was noted that unless an adequate percolation area is incorporated in the OSWTS construction, the soil will become overloaded and much higher contaminant loading will occur to groundwater

- The hydraulic loading rate per person per day was investigated and it was found that the actual rate will have a significant impact on the movement of bacteria into the subsoil

Groundwater Models

- The groundwater models indicated that plumes of contaminants from clusters of OSWTS's tend to be localised in nature with plumes of significant concentrations not spreading significantly further than 250 m downstream

- The more mobile contaminants (particularly NO_3) were seen to migrate up to 500 m downstream of the cluster developments when a long term transient model scenario was employed – however the input data for these models was limited

- Simulations of the effects of adding to the density of OSWTS's at each of the study areas all indicated the same findings. Whilst increasing the density of OSWTS's does increase the concentrations of pollutants within the localised plumes, the effects further downstream are significantly muted and have dissipated within 250 – 500 m downstream

Overall Impact of Density of OSWTS

- Taking all of the available information together, it is concluded that the density of clusters of OSWTS's does not appear to impact directly on groundwater quality under typical Irish hydrological and hydrogeological conditions

- It would appear that an appropriate density for OSWTS for future development should be based upon the plot size required to comfortably accommodate the recommendations for minimum setbacks distances in the EPA CoP (EPA, 20009), providing an adequate depth of unsaturated subsoil is vital that can take the expected hydraulic and organic loading rates. This would indicate that developments incorporating OSWTS's should not exceed 6 Units/Hectare

10.2 Recommendations for further research

During the course of this study a number of areas for further research were identified and these are summarised below:

Field Methods

- Development of laboratory analytic methods that allow the easy identification of the source of faecal contaminant (i.e. between human and agriculture). At the time of writing work is underway at a number of institutions on this very topic

- Quantifying the effects of poorly functioning OSWTS's on surface water (particularly in low permeability areas of the country) and therefore estimating the implications of remediation within the contexts of Irish commitment under the Water Framework Directive (WFD)

- Construct a database for Irish bedrock and subsoil specific hydraulic parameters to aid similar future research projects

Numerical Modelling

- Collate and organise a database of applicable hydraulic parameters from calibrated field studies for modellers in the Irish context

- A range of scenarios could be modelled in the vadose zone in order to establish a better understanding of the chemical and bacterial attenuation and treatment processes that occur specific to Irish subsoil conditions (with a particular emphasis on the typical high soil moisture and its implications).

- Develop accurate nitrification rates for Irish subsoils which would involve extensive vadose zone modelling at the micro scale. This research would have to pay particular attention to the processes that occur in the biomat zone which was not considered in detail in this current project

- Investigate whether long term attenuation rates of phosphorus (and other contaminants) decrease over time. This study assumed the high phosphorus attenuation rates observed in previous studies would continue into the future which may not be the case. Similar studies could also look at the fate of more synthetic contaminants in the subsoil (or groundwater) such as personal care products (PCPs) and other chemicals which may be known to be endocrine disrupting chemicals such as oestrogen

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APPENDICES

Appendix A – Borehole drilling logs

Appendix B – Site Characterisation Forms

Appendix C – Field Monitoring Results

Appendix D – XRD Plots

Appendix E – Salt Dilution Gauging Data and Plots

APPENDIX A

J S DRILLING
Geological & Environmental Drilling Contractors

MOBILE: 0877433451
FAX: 0567793887
E-MAIL: jim@jsdrilling.ie
WEBSITE: www.jsdrilling.ie
VAT. NO: IE 6431197 F

JS DRILLING LTD
20 THE BELFRY
CHAPEL LANE
THOMASTOWN
CO. KILKENNY

Soft Ground Boring: Daily Record

Location: Dublin	Site: Naul
Client: DCSEE Trinity	Borehole No: 1
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA: soft firm stiff / loose med.d. dense / colour / clayey silty sandy / fine med. coarse / soil name / with gravel, cobbles, sand bands, etc
0.0	Start of days boring — Time
1.1	Soft clays
28.5	Dense black boulder clays
40.0	Competent limestone bedrock
40.0	End of days boring — Time

SAMPLES

No	Type	Depth (m) From	Depth (m) To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	7.0	28.0			
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

CASING		
Size (mm)	From (m)	To (m)
220	0.0	4.5

Delays, bad access, pits, standing time, etc...

Installation 1 – base 40mbgl 3m slotted with geosock 37.5m plain.
Installation 2 – base 31mbgl 3m slotted with geosock 28.5m plain.
Finished with raised cover and concrete plinth.

DRILLING			
Diameter	From (m)	To (m)	Type
165mm	4.5	40.0	6" hammer

Client..................................

Driller............Colm O'Neill..................

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JS DRILLING LTD
20 THE BELFRY
CHAPEL LANE
THOMASTOWN
CO. KILKENNY

Soft Ground Boring: Daily Record

Location: Dublin	Site: Naul
Client: DCSEE Trinity	Borehole No: 2
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA						
	soft firm stiff	loose med.d. dense	colour	clayey silty sandy	fine med. coarse	soil name	with gravel, cobbles, sand bands, etc
0.0	Start of days boring						Time
1.8	Dry red clays						
31.5	Dense black boulder clays						
35.0	Competent limestone bedrock						
37.5	Fractured bedrock – falling in						
37.5	End of days boring						Time

SAMPLES											
No	Type	Depth (m) From	Depth (m) To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	31.5				
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

CASING		
Size (mm)	From (m)	To (m)
220	0.0	4.3

Delays, bad access, pits, standing time, etc...

Installation 1 – base 35mbgl 3m slotted with geosock 32.5m plain.
Installation 2 – base 31mbgl 3m slotted with geosock 28.5m plain.
Installation 3 – base 10mbgl 3m slotted with geosock 7.5m plain.
Finished with raised cover and concrete plinth.

DRILLING			
Diameter	From (m)	To (m)	Type
165mm	4.3	37.5	6" hammer

Client..................................

Driller............Colm O'Neill...................

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CO. KILKENNY

Soft Ground Boring: Daily Record	
Location: Dublin	Site: Naul
Client: DCSEE Trinity	Borehole No: 3
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA						
	soft firm stiff	loose med.d. dense	colour	clayey silty sandy	fine med. coarse	soil name	with gravel, cobbles, sand bands, etc
0.0	Start of days boring						Time
1.0	Clays and gravels						
12.0	Boulder clays						
18.0	Shales						
18.0	End of days boring						Time

SAMPLES												
No	Type	Depth (m)		U100 Blows	S.P.T / C.P.T (mm)						Remarks
		From	To		0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	12.0				
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

CASING		
Size (mm)	From (m)	To (m)
220	0.0	2.7

Delays, bad access, pits, standing time, etc...
Installation 1 – base 18mbgl 3m slotted with geosock 15.5m plain.
Installation 2 – base 13.5mbgl 3m slotted with geosock 11m plain.
Installation 3 – base 2mbgl 1m slotted with geosock 1.5m plain.
Finished with raised cover and concrete plinth.

DRILLING			
Diameter	From (m)	To (m)	Type
165mm	2.7	18.0	6" hammer

Client.................................

Driller...........Colm O'Neill..................

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JS DRILLING LTD
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CO. KILKENNY

Soft Ground Boring: Daily Record

Location: Offaly	Site: Rhode
Client: DCSEE Trinity	Borehole No: 1
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA (soft firm stiff / loose med.d. dense / colour / clayey silty sandy / fine med. coarse / soil name / with gravel, cobbles, sand bands, etc)
0.0	Start of days boring — Time
1.1	Soils
7.2	Clays and gravels
9.8	Gravels
12.0	Sands
14.0	Sand and gravels
19.0	Gravels and few sands
21.8	Clays and gravels
25.0	Dense clays and gravels
26.3	Gravels
26.3	End of days boring — Time

SAMPLES

No	Type	Depth (m) From	Depth (m) To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	4.0	7.2	25.0		
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

CASING		
Size (mm)	From (m)	To (m)
220	0.0	22.0

Delays, bad access, pits, standing time, etc...
Installation 1 – base 18mbgl 3m slotted with geosock 15m plain.
Installation 2 – base 10mbgl 3m slotted with geosock 7m plain.
Finished with raised cover and concrete plinth.

DRILLING			
Diameter	From (m)	To (m)	Type
165mm	22.0	26.3	6" hammer

Client.................................

Driller...........Colm O'Neill..................

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JS DRILLING LTD
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CHAPEL LANE
THOMASTOWN
CO. KILKENNY

Soft Ground Boring: Daily Record

Soft Ground Boring: Daily Record	
Location: Offally	Site: Rhode
Client: DCSEE Trinity	Borehole No: 1 - a
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA (soft firm stiff / loose med.d. dense / colour / clayey silty sandy / fine med. coarse / soil name / with gravel, cobbles, sand bands, etc)
0.0	Start of days boring Time
1.1	Soils
7.2	Clays and gravels
9.8	Gravels
12.0	Sands
14.0	Sand and gravels
19.0	Gravels and few sands
21.8	Clays and gravels
25.0	Dense clays and gravels
26.3	Gravels
30.0	Weathered bedrock
30.0	End of days boring Time

SAMPLES												
No	Type	Depth (m) From	Depth (m) To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	4.0	7.2	25.0		
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

CASING		
Size (mm)	From (m)	To (m)
167	0.0	30.0

Delays, bad access, pits, standing time, etc...
Installation 1 – base 29.5mbgl 3m slotted with geosock 27m plain.
Finished with raised cover and concrete plinth.

DRILLING			
Diameter	From (m)	To (m)	Type

Client..................................

Driller............Colm O'Neill...................

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JS DRILLING LTD
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Soft Ground Boring: Daily Record

Location: Offally	Site: Rhode
Client: DCSEE Trinity	Borehole No: 2
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA: soft firm stiff	loose med.d. dense	colour	clayey silty sandy	fine med. coarse	soil name	with gravel, cobbles, sand bands, etc
0.0	Start of days boring						Time
0.3	Soils						
2.3	Sands						
3.0	Gravels						
6.5	Sandy clays with few gravels						
8.6	Weathered limestone bedrock						
11.2	Competent limestone bedrock						
13.5	Large fractures / void						
13.5	End of days boring						Time

SAMPLES

No	Type	Depth (m) From	To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	2.0	8.6			24hr after pulling casing
Duration	mins	mins	mins	mins	
Depth to water					
Depth cased					
Depth of hole					

CASING		
Size (mm)	From (m)	To (m)
220	0.0	8.6

Delays, bad access, pits, standing time, etc...
Installation 1 – base 10mbgl 3m slotted with geosock 7.5m plain.
Installation 2 – base 2.5mbgl 1m slotted with geosock 2m plain.
Finished with raised cover and concrete plinth.

DRILLING			
Diameter	From (m)	To (m)	Type
165mm	8.6	13.5	6" hammer

Client.................................

Driller...........Colm O'Neill..................

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JS DRILLING LTD
20 THE BELFRY
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THOMASTOWN
CO. KILKENNY

Soft Ground Boring: Daily Record

Location: Offally	Site: Rhode
Client: DCSEE Trinity	Borehole No: 3
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA						
	soft firm stiff	loose med.d. dense	colour	clayey silty sandy	fine med. coarse	soil name	with gravel, cobbles, sand bands, etc
0.0	Start of days boring						Time
0.3	Soils						
0.6	Firm red clays						
1.1	Light brown clays with gravels						
1.2	Limestone cobbles						
2.4	Soft light brown clays with large gravels						
3.7	Dense cobbles – dry						
4.3	Dense clays with cobbles						
5.2	Soft clays and gravels						
6.5	Black firm clays with cobbles – dry						
7.0	Gravels with clays – damp						
16.5	Black boulder clay						
17.0	Weathered rock						
21.0	Competent bedrock						
21.0	End of days boring						Time

SAMPLES												
No	Type	Depth (m)		U100 Blows	S.P.T / C.P.T (mm)						Remarks
		From	To		0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	6.5	17.0			
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

CASING		
Size (mm)	From (m)	To (m)
220	0.0	17.0

Delays, bad access, pits, standing time, etc…
Installation 1 – base 21mbgl 3m slotted with geosock 18.5m plain.
Installation 2 – base 17mbgl 3m slotted with geosock 14.5m plain.
Installation 3 – base 6.5mbgl 3m slotted with geosock 4m plain.
Finished with raised cover and concrete plinth.

DRILLING			
Diameter	From (m)	To (m)	Type
165mm	17.0	21.0	6" hammer

Client…………………………… Driller………..Colm O'Neill………………

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THOMASTOWN
CO. KILKENNY

Soft Ground Boring: Daily Record	
Location: Kilkenny	Site: Carrigeen
Client: DCSEE Trinity	Borehole No: 1
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA						
	soft firm stiff	loose med.d. dense	colour	clayey silty sandy	fine med. coarse	soil name	with gravel, cobbles, sand bands, etc
0.0	Start of days boring						Time
3.5	Soft clays – damp						
5.0	Weathered rock – very loose						
13.0	Competent bedrock						
20.0	End of days boring						Time

SAMPLES											
No	Type	Depth (m) From	Depth (m) To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	12.0	16.0			
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

CASING		
Size (mm)	From (m)	To (m)
220	0.0	5.7

Delays, bad access, pits, standing time, etc...
Installation 1 – base 20.0mbgl 3m slotted with geosock 17.5m plain.
Installation 2 – base 16.5mbgl 3m slotted with geosock 14m plain.
Installation 3 – base 12.5mbgl 13m slotted with geosock 10m plain.
Finished with raised cover and concrete plinth.

DRILLING			
Diameter	From (m)	To (m)	Type
165mm	5.7	20.0	6" hammer

Client.................................

Driller...........Colm O'Neill..................

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JS DRILLING LTD
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THOMASTOWN
CO. KILKENNY

Soft Ground Boring: Daily Record	
Location: Kilkenny	Site: Carrigeen
Client: DCSEE Trinity	Borehole No: 2
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA (soft firm stiff / loose med.d. dense / colour / clayey silty sandy / fine med. coarse / soil name / with gravel, cobbles, sand bands, etc)
0.0	Start of days boring — Time
0.3	Brown / orange top soil
1.5	Orange clays firm
1.7	Red / orange clays soft damp
9.8	Dense red / orange clays soft damp with large gravels
10.5	Limestone cobbles
11.9	Firm red clays
12.5	Orange clays with sands soft
20.0	Dense brown clays with limestone chippings
20.0	End of days boring — Time

SAMPLES

No	Type	Depth (m) From	To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	3.5	12.5			
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

Delays, bad access, pits, standing time, etc...
Installation 1 – base 14.6mbgl 3m slotted with geosock 11.5m plain.
Installation 2 – base 4mbgl 3m slotted with geosock 1m plain.
Finished with raised cover and concrete plinth.

CASING		
Size (mm)	From (m)	To (m)
220	0.0	20.0

DRILLING			
Diameter	From (m)	To (m)	Type

Client.................................

Driller...........Colm O'Neill..................

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JS DRILLING LTD
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CO. KILKENNY

Soft Ground Boring: Daily Record	
Location: Limerick	Site: Ferrybridge
Client: DCSEE Trinity	Borehole No: 1
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA						
	soft firm stiff	loose med.d. dense	colour	clayey silty sandy	fine med. coarse	soil name	with gravel, cobbles, sand bands, etc
0.0	Start of days boring						Time
3.5	Soft clays – damp						
5.0	Weathered rock – very loose						
13.0	Competent bedrock						
13.0	End of days boring						Time

SAMPLES											
No	Type	Depth (m) From	Depth (m) To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	3.5	5.0			
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

CASING		
Size (mm)	From (m)	To (m)
220	0.0	5.7

Delays, bad access, pits, standing time, etc...
Installation 1 – base 13mbgl 1m slotted with geosock 12.5m plain.
Installation 2 – base 9mbgl 2m slotted with geosock 7.5m plain.
Installation 3 – base 5mbgl 1.5m slotted with geosock 4m plain.
Finished with raised cover and concrete plinth.

DRILLING			
Diameter	From (m)	To (m)	Type
165mm	5.7	13.0	6" hammer

Client..................................

Driller............Colm O'Neill..................

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JS DRILLING LTD
20 THE BELFRY
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CO. KILKENNY

Soft Ground Boring: Daily Record

Location: Limerick	Site: Ferrybridge
Client: DCSEE Trinity	Borehole No: 2
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

DESCRIPTION OF STRATA

Depth to base of strata	soft firm stiff	loose med.d. dense	colour	clayey silty sandy	fine med. coarse	soil name	with gravel, cobbles, sand bands, etc
0.0	Start of days boring						Time
1.5	Clays						
2.5	Weathered rock with clays						
4.3	Weathered rock						
10.3	Competent limestone bedrock						
10.3	End of days boring						Time

SAMPLES

No	Type	Depth (m) From	Depth (m) To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	2.5	3.8			
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

Delays, bad access, pits, standing time, etc...
Installation 1 – base 10.3mbgl 1m slotted with geosock 9.8m plain.
Installation 2 – base 5mbgl 1.2m slotted with geosock 4.3m plain.
Installation 3 – base 2.6mbgl 1m slotted with geosock 2.1m plain.
Finished with raised cover and concrete plinth.

CASING

Size (mm)	From (m)	To (m)
220	0.0	4.3

DRILLING

Diameter	From (m)	To (m)	Type
165mm	4.3	10.3	6" hammer

Client..................................

Driller............Colm O'Neill...................

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20 THE BELFRY
CHAPEL LANE
THOMASTOWN
CO. KILKENNY

Soft Ground Boring: Daily Record

Location: Limerick	Site: Ferrybridge
Client: DCSEE Trinity	Borehole No: 3
Date: May 2010	Sheet: 1 of 1
Ground Level: m (Ordnance datum)	Engineer: Patrick Morrissey
Weather: Fine / Drizzling / Raining / Stormy / Snow	
Rig Type: Beretta T51 top drive rotary rig	

Depth to base of strata	DESCRIPTION OF STRATA (soft firm stiff / loose med.d. dense / colour / clayey silty sandy / fine med. coarse / soil name / with gravel, cobbles, sand bands, etc)
0.0	Start of days boring — Time
0.5	Top soils
3.0	Soft fine clays and gravels
5.2	Weathered blue / white limestone bedrock
5.5	Weathered bedrock with sands
7.5	Mudstone
8.1	Large angular limestone chippings
11.0	Mudstone
12.6	Large angular limestone chippings
23.0	Mudstone
23.0	End of days boring — Time

SAMPLES

No	Type	Depth (m) From	Depth (m) To	U100 Blows	S.P.T / C.P.T (mm) 0 to 75	75 to 150	150 to 225	225 to 300	300 to 375	375 to 450	Remarks

U – Undisturbed D – Disturbed B – Bulk W – Water S – S.P.T. C – C.P. T.

Water Strike (m)	10.5				
Duration	mins	mins	mins	mins	24hr after pulling casing
Depth to water					
Depth cased					
Depth of hole					

Delays, bad access, pits, standing time, etc...
Installation 1 – base 23mbgl 6m slotted with geosock 17.5m plain.
Installation 2 – base 12mbgl 3m slotted with geosock 9.5m plain.
Finished with raised cover and concrete plinth.

CASING

Size (mm)	From (m)	To (m)
220	0.0	4.3

DRILLING

Diameter	From (m)	To (m)	Type
165mm	4.3	23.0	6" hammer

Client.................................

Driller...........Colm O'Neill..................

APPENDIX B

APPENDIX B: SITE CHARACTERISATION FORM

File Reference: Naul Site Characterisation

1.0 GENERAL DETAILS (From planning application)

Prefix:

First Name: n/a

Surname: n/a

Address: Moonlone Lane, Naul, Co. Dublin

Site Location and Townland: Moonlone Lane, Naul, Co. Dublin - Hazardstown

Telephone No: n/a

Fax No: n/a

E-Mail: n/a

Maximum no. of Residents:

No. of Double Bedrooms:

No. of Single Bedrooms:

Proposed Water Supply: Mains ✓ Private Well/Borehole ☐ Group Well/Borehole ☐

2.0 GENERAL DETAILS (From planning application)

Soil Type, (Specify Type): Soil: Gleys Sandstone & Shale Till (Upper Palaeozoic)

Aquifer Category: Regionally Important ☐ Locally Important LI Poor ☐

Vulnerability: Extreme ☐ High ☐ Moderate ☐ Low ✓ High to Low ☐ Unknown ☐

Bedrock Type: Dinantian Upper impure Limestone

Name of Public/Group Scheme Water Supply within 1 km: Bog of the Ring Public water supply

Groundwater Protection Scheme (Y/N): Yes

Source Protection Area: SI ✓ SO ☐

Groundwater Protection Response: R2[4]

Presence of Significant Sites (Archaeological, Natural & Historical): none evident

Past experience in the area: Good conditions have usually favoured the installation of secondary treatment systems

Comments:

(Integrate the information above in order to comment on: the potential suitability of the site, potential targets at risk, and/or any potential site restrictions).

Potential suitability of the site: Fair
Potential targets: Groundwater, locals wells
Potential site restrictions: Low permeability clays present

Note: Only information available at the desk study stage should be used in this section.

3.0 ON-SITE ASSESSMENT

3.1 Visual Assessment

Landscape Position:	Agricultural land - orchards to the east - sloping from south-west to north-east

Slope: Steep (>1:5) ☐ Shallow (1:5-1:20) ☑ Relatively Flat (<1:20) ☐

Surface Features within a minimum of 250m (Distance To Features Should Be Noted In Metres)

Houses:	A cluster of houses exist within 250m - closest is 90m away
Existing Land Use:	Agricultural land - orchards to the east
Vegetation Indicators:	none
Groundwater Flow Direction:	North-easterly
Ground Condition:	Firm underfoot
Site Boundaries:	Ditches and drains on all boundaries except the local roadway to the north
Roads:	Moonlone lane is main access
Outcrops (Bedrock And/Or Subsoil):	none

Surface Water Ponding:	none	Lakes:	An old resovoir to the north (600m)
Beaches/Shellfish:	none	Areas/Wetlands:	none

Karst Features:	None
Watercourse/Stream*:	Local stream passes to the south, wet ditch to the north
Drainage Ditches*:	Drainage ditches on all boundaries - most appear shallow or dry at time of testing
Springs / Wells*:	None within 100m

Comments:
(Integrate the information above in order to comment on: the potential suitability of the site, potential targets at risk, the suitability of the site to treat the wastewater and the location of the proposed system within the site).

The sit seems suitable for disposal of on-site wastewater to ground. The greatest target at risk in the area is groundwater given that the site is within the inner zone of protection for the Bog of the Ring public water supply. The site seems well drained with no obvious karst features and no wells or springs within 500m. The area seems well suited to treat wastewater subject to the appropriate treatment system being installed subject to passing the percolation test criteria.

*Note and record water level

3.2 Trial Hole (should be a minimum of 2.1m deep (3m for regionally important aquifers))

To avoid any accidental damage, a trial hole assessment or percolation tests should not be undertaken in areas, which are at or adjacent to significant sites (e.g. NHAs, SACs, SPAs, and/or Archaeological etc.), without prior advice from National Parks and Wildlife Service or the Heritage Service.

Depth of trial hole (m): 2.10

Depth from ground surface to bedrock (m) (if present):

Depth from ground surface to water table (m) (if present):

Depth of water ingress:

Rock type (if present):

Date and time of excavation: 11/10/2011 10:00

Date and time of examination: 12/10/2011 09:00

	Depth of P/T Test*	Soil/Subsoil Texture & Classification**	Plasticity and dilatancy***	Soil Structure	Density/ Compactness	Colour****	Preferential flowpaths
0.1 m		Topsoil		Granular	Loose		
0.2 m						Brown	Root zone
0.3 m							
0.4 m		SILT/CLAY	Threads 3, 4, 4 Ribbons 110mm	Granular	Well compacted	Brown	None
0.5 m							
0.6 m							
0.7 m							
0.8 m							
0.9 m							
1.0 m							
1.1 m							
1.2 m	T 1,2,3						
1.3 m							
1.4 m							
1.5 m							
1.6 m							
1.7 m							
1.8 m							
1.9 m							
2.0 m							
2.1 m							
2.2 m							
2.3 m							
2.4 m							
2.5 m							
2.6 m							
2.7 m							
2.8 m							
2.9 m							
3.0 m							

Likely T value: 35.00

Note: *Depth of percolation test holes should be indicated on log above. (Enter P or T at depts as appropriate).
** See Appendix E for BS 5930 classification.
*** 3 samples to be tested for each horizon and results should be entered above for each horizon.
**** All signs of mottling should be recorded.

3.2 Trial Hole (contd.) Evaluation:

Soil was fairly uniform CLAY/SILT with few cobbles and was well drained and dry. Bedrock and water table were not met.

3.3(a) Percolation ("T") Test for Deep Subsoils and/or Water Table

Step 1: Test Hole Preparation

Percolation Test Hole	1	2	3
Depth from ground surface to top of hole (mm) (A)	800	800	
Depth from ground surface to base of hole (mm) (B)	1,200	1,200	
Depth of hole (mm) [B - A]	400	400	0
Dimensions of hole [length x breadth (mm)]	300 x 300	300 x 300	x

Step 2: Pre-Soaking Test Holes

	1		2		3	
Date and Time pre-soaking started	11/10/2011	11:00	11/10/2011	11:00		

Each hole should be pre-soaked twice before the test is carried out. Each hole should be empty before refilling.

Step 3: Measuring T_{100}

Percolation Test Hole No.	1	2	3
Date of test	12/10/2011	12/10/2011	
Time filled to 400 mm	09:30	09:30	
Time water level at 300 mm	10:15	10:21	
Time to drop 100 mm (T_{100})	45.00	51.00	0.00
Average T_{100}			32.00

If $T_{100} > 300$ minutes then T-value >90 – site unsuitable for discharge to ground
If $T_{100} \leq 210$ minutes then go to Step 4;
If $T_{100} > 210$ minutes then go to Step 5;

Step 4: Standard Method (where $T_{100} \leq 210$ minutes)

Percolation Test Hole	1			2			3		
Fill no.	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)
1	11:36	12:46	70.00	11:36	12:46	70.00	11:36	12:46	70.00
2	12:51	14:20	89.00	12:51	14:20	89.00	12:51	14:20	89.00
3	14:25	16:22	117.00	14:25	16:22	117.00	14:25	16:22	117.00
Average Δt Value			92.00			92.00			92.00
	Average Δt/4 = [Hole No.1]		23.00 (t_1)	Average Δt/4 = [Hole No.2]		23.00 (t_2)	Average Δt/4 = [Hole No.3]		23.00 (t_3)

Result of Test: T = 23.00 (min/25 mm)

Comments:

Step 5: Modified Method (where $T_{100} > 210$ minutes)

Percolation Test Hole No.	1				2				3			
Fall of water in hole (mm)	Time Factor = T_f	Time of fall (mins) = T_m	$K_{fs} = T_f / T_m$	T – Value = 4.45 / K_{fs}	Time Factor = T_f	Time of fall (mins) = T_m	$K_{fs} = T_f / T_m$	T – Value = 4.45 / K_{fs}	Time Factor = T_f	Time of fall (mins) = T_m	$K_{fs} = T_f / T_m$	T – Value = 4.45 / K_{fs}
300 - 250	8.1	31	0.26	17.03	8.1	31	0.26	17.03	8.1	31	0.26	17.03
250 - 200	9.7	47	0.21	21.56	9.7	47	0.21	21.56	9.7	47	0.21	21.56
200 - 150	11.9	53	0.22	19.82	11.9	55	0.22	20.57	11.9	55	0.22	20.57
150 - 100	14.1	83	0.17	26.20	14.1	83	0.17	26.20	14.1	83	0.17	26.20
Average T- Value	T- Value Hole 1= (t_1)			21.15	T- Value Hole 1= (t_2)			21.34	T- Value Hole 1= (t_3)			21.34

Result of Test: T = 21.28 (min/25 mm)

Comments:

3.0 ON-SITE ASSESSMENT

3.1 Visual Assessment

Landscape Position: Pasture lands with a gentle slope from north to south

Slope: Steep (>1:5) ☐ Shallow (1:5-1:20) ☑ Relatively Flat (<1:20) ☐

Surface Features within a minimum of 250m (Distance To Features Should Be Noted In Metres)

Houses: A cluster of houses exist within 250m - the closest being approx 80m away

Existing Land Use: Agriculture

Vegetation Indicators: Vegetation on the site indicates good drainage

Groundwater Flow Direction: Groundwater would seem to be flowing with the ground contours from north to south

Ground Condition: Undisturbed agricultural land - firm underfoot

Site Boundaries: Site is bounded to the north by a local road and on all other sides by existing ditches

Roads: Local road to the north which connects the R441 with the R400

Outcrops (Bedrock And/Or Subsoil): None

Surface Water Ponding: None Lakes: None

Beaches/Shellfish: None Areas/Wetlands: Wetland area 400m to the south

Karst Features: None

Watercourse/Stream*: Local ditch/stream 435m to the south with a water depth of approx. 350mm

Drainage Ditches*: a series of shallow ditches drain the land to a local stream/ditch to the south some 435m away

Springs / Wells*: None

Comments:
(Integrate the information above in order to comment on: the potential suitability of the site, potential targets at risk, the suitability of the site to treat the wastewater and the location of the proposed system within the site).

The sit seems suitable for disposal of on-site wastewater to ground. The greatest target at risk in the area is groundwater. The site seems well drained with no obvious karst features and no wells or springs within 500m. The area seems well suited to treat wastewater subject to the appropriate treatment system being installed.

*Note and record water level

3.2 Trial Hole (should be a minimum of 2.1m deep (3m for regionally important aquifers))

To avoid any accidental damage, a trial hole assessment or percolation tests should not be undertaken in areas, which are at or adjacent to significant sites (e.g. NHAs, SACs, SPAs, and/or Archaeological etc.), without prior advice from National Parks and Wildlife Service or the Heritage Service.

Depth of trial hole (m): 2.10

Depth from ground surface to bedrock (m) (if present):

Depth from ground surface to water table (m) (if present):

Depth of water ingress:

Rock type (if present):

Date and time of excavation: 10/07/2011 11:00

Date and time of examination: 11/07/2011 10:00

	Depth of P/T Test*	Soil/Subsoil Texture & Classification**	Plasticity and dilatancy***	Soil Structure	Density/ Compactness	Colour****	Preferential flowpaths
0.1 m		Topsoil		Granular	Soft	Brown	Root zone
0.2 m							
0.3 m		SILT with pebbles	Threads 2,2,3 Ribbons 80-90mm	Blocky	Firm	Brown/Black	None
0.4 m							
0.5 m							Possible downward barrier (ironpan)
0.6 m		Presence of Ironpan				Orange gold	
0.7 m							
0.8 m		SILT/CLAY with frequent pebbles and some cobbles	Threads 3,3,4 Ribbons 100 -110	Blocky - subangular	Well compacted	Brown	None
0.9 m							
1.0 m							
1.1 m							
1.2 m	T1,2,3						
1.3 m							
1.4 m							
1.5 m							
1.6 m							
1.7 m							
1.8 m							
1.9 m							
2.0 m							
2.1 m		END					End
2.2 m							
2.3 m							
2.4 m							
2.5 m							
2.6 m							
2.7 m							
2.8 m							
2.9 m							
3.0 m							

Likely T value: 40.00

Note: *Depth of percolation test holes should be indicated on log above. (Enter P or T at depts as appropriate).
** See Appendix E for BS 5930 classification.
*** 3 samples to be tested for each horizon and results should be entered above for each horizon.
**** All signs of mottling should be recorded.

3.2 Trial Hole (contd.) Evaluation:

Trial hole showed the presence of an orange/gold ironpan at 0.6m depth. this may be limiting downward flow of water from above. Soil seemed well drained and neither the water table or the bedrock were met.

3.3(a) Percolation ("T") Test for Deep Subsoils and/or Water Table

Step 1: Test Hole Preparation

Percolation Test Hole	1	2	3
Depth from ground surface to top of hole (mm) (A)	800	800	
Depth from ground surface to base of hole (mm) (B)	1,200	1,200	
Depth of hole (mm) [B - A]	400	400	0
Dimensions of hole [length x breadth (mm)]	300 x 300	300 x 300	x

Step 2: Pre-Soaking Test Holes

	1		2		3	
Date and Time pre-soaking started	10/07/2011	14:00	10/07/2011	14:00		

Each hole should be pre-soaked twice before the test is carried out. Each hole should be empty before refilling.

Step 3: Measuring T_{100}

Percolation Test Hole No.	1	2	3
Date of test	11/07/2011	11/07/2011	
Time filled to 400 mm	10:45	10:46	
Time water level at 300 mm	11:14	11:26	
Time to drop 100 mm (T_{100})	29.00	40.00	0.00
Average T_{100}			23.00

If T_{100} > 300 minutes then T-value >90 – site unsuitable for discharge to ground
If $T_{100} \leq$ 210 minutes then go to Step 4;
If T_{100} > 210 minutes then go to Step 5;

Step 4: Standard Method (where $T_{100} \leq 210$ minutes)

Percolation Test Hole	1			2			3		
Fill no.	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)
1	11:14	12:51	97.00	11:14	12:51	97.00	11:14	12:51	97.00
2	12:15	14:48	153.00	12:15	14:48	153.00	12:15	14:48	153.00
3	14:48	17:32	164.00	14:48	17:32	164.00	14:48	17:32	164.00
Average Δt Value			138.00			138.00			138.00
	Average Δt/4 = [Hole No.1]		34.50 (t_1)	Average Δt/4 = [Hole No.2]		34.50 (t_2)	Average Δt/4 = [Hole No.3]		34.50 (t_3)

Result of Test: T = 34.50 (min/25 mm)

Comments:

Adequate subsoil percolation

Step 5: Modified Method (where $T_{100} > 210$ minutes)

Percolation Test Hole No.	1				2				3			
Fall of water in hole (mm)	Time Factor = T_f	Time of fall (mins) = T_m	$K_{fs} = T_f / T_m$	T – Value = 4.45 / K_{fs}	Time Factor = T_f	Time of fall (mins) = T_m	$K_{fs} = T_f / T_m$	T – Value = 4.45 / K_{fs}	Time Factor = T_f	Time of fall (mins) = T_m	$K_{fs} = T_f / T_m$	T – Value = 4.45 / K_{fs}
300 - 250	8.1	29	0.28	15.93	8.1	29	0.28	15.93	8.1	29	0.28	15.93
250 - 200	9.7	51	0.19	23.40	9.7	51	0.19	23.40	9.7	51	0.19	23.40
200 - 150	11.9	72	0.17	26.92	11.9	72	0.17	26.92	11.9	72	0.17	26.92
150 - 100	14.1	114	0.12	35.98	14.1	114	0.12	35.98	14.1	114	0.12	35.98
Average T- Value	T- Value Hole 1= (t_1)			25.56	T- Value Hole 1= (t_2)			25.56	T- Value Hole 1= (t_3)			25.56

Result of Test: T = 25.56 (min/25 mm)

Comments:

Adequate subsoil percolation

APPENDIX B: SITE CHARACTERISATION FORM

File Reference: Carrigeen Site Characterisation

1.0 GENERAL DETAILS (From planning application)

Prefix:

First Name: n/a

Surname: n/a

Address: Carrigeen, Co. Kilkenny

Site Location and Townland: Cooleigh, Carrigeen, Co. Kilkenny

Telephone No:

Fax No:

E-Mail:

Maximum no. of Residents:

No. of Double Bedrooms:

No. of Single Bedrooms:

Proposed Water Supply: Mains ☐ Private Well/Borehole ☐ Group Well/Borehole ☐

2.0 GENERAL DETAILS (From planning application)

Soil Type, (Specify Type): Tills derived from Devonian Sandstones

Aquifer Category: Regionally Important: Rf Locally Important: Poor:

Vulnerability: Extreme ☐ High ☑ Moderate ☐ Low ☐ High to Low ☐ Unknown ☐

Bedrock Type: Upper Palaeozic Devonian rocks chiefly Old Red Sandstone (ORS)

Name of Public/Group Scheme Water Supply within 1 km: none

Groundwater Protection Scheme (Y/N): No

Source Protection Area: SI ☐ SO ☐

Groundwater Protection Response: R2²

Presence of Significant Sites (Archaeological, Natural & Historical): none

Past experience in the area: Previous T-tests in Cooleigh from Local Authority records resulted in a T-value of 11.5mins/25mm

Comments:

(Integrate the information above in order to comment on: the potential suitability of the site, potential targets at risk, and/or any potential site restrictions).

The site would seem to be suitable in principal for the construction of an on-site wastewater treatment system with the disposal of treated water to groundwater subject to percolation test. The most likely potential target is groundwater with a High vulnerability rating.

Note: Only information available at the desk study stage should be used in this section.

3.0 ON-SITE ASSESSMENT

3.1 Visual Assessment

Landscape Position: Part way down a gentle to steep slope on Corluddy hill

Slope: Steep (>1:5) ☑ Shallow (1:5-1:20) ☐ Relatively Flat (<1:20) ☐

Surface Features within a minimum of 250m (Distance To Features Should Be Noted In Metres)

Houses: Existing cluster of houses in the centre of the village

Existing Land Use: To the rear and east is the village of Carrigeen. To the south and west is agricultural land

Vegetation Indicators: None - grass fields

Groundwater Flow Direction: North

Ground Condition: Firm underfoot

Site Boundaries: Public road to the east. Hedgerows on all other boundaries

Roads: Local road through the village to the east

Outcrops (Bedrock And/Or Subsoil): None

Surface Water Ponding: None

Lakes: Small lake 2km to the north

Beaches/Shellfish: None

Areas/Wetlands: None

Karst Features: None

Watercourse/Stream*: Local stream some 800m to the north

Drainage Ditches*: Local ditches to the west of the site draining the area

Springs / Wells*: Two private wells further up the hill to the south within 800m

Comments:
(Integrate the information above in order to comment on: the potential suitability of the site, potential targets at risk, the suitability of the site to treat the wastewater and the location of the proposed system within the site).

The sit seems suitable for disposal of on-site wastewater to ground. The greatest target at risk in the area is groundwater given the High vulnerability rating. The site seems well drained with no obvious karst features and no wells or springs within 500m. The area seems well suited to treat wastewater subject to the appropriate treatment system being installed.

*Note and record water level

3.2 Trial Hole (should be a minimum of 2.1m deep (3m for regionally important aquifers))

To avoid any accidental damage, a trial hole assessment or percolation tests should not be undertaken in areas, which are at or adjacent to significant sites (e.g. NHAs, SACs, SPAs, and/or Archaeological etc.), without prior advice from National Parks and Wildlife Service or the Heritage Service.

Depth of trial hole (m): 2.10

Depth from ground surface to bedrock (m) (if present): 0.00

Depth from ground surface to water table (m) (if present): 0.00

Depth of water ingress: 0.00

Rock type (if present):

Date and time of excavation: 21/08/2012 09:00

Date and time of examination: 21/08/2012 10:00

	Depth of P/T Test*	Soil/Subsoil Texture & Classification**	Plasticity and dilatancy***	Soil Structure	Density/ Compactness	Colour****	Preferential flowpaths
0.1 m		Topsoil		Crumb	Loose		Root zone
0.2 m						Brown/Orange	
0.3 m							
0.4 m		gravely CLAY	Threads 1, 1, 1 Ribbons 60 - 80mm	Structureless	Firm	Orange brown	None
0.5 m							
0.6 m							
0.7 m							
0.8 m							
0.9 m							
1.0 m		CLAY	Threads 1, 1, 2	Massive	Firm	Orange brown	None
1.1 m							
1.2 m							
1.3 m							
1.4 m							
1.5 m							
1.6 m							
1.7 m							
1.8 m							
1.9 m							
2.0 m							
2.1 m							
2.2 m							
2.3 m							
2.4 m							
2.5 m							
2.6 m							
2.7 m							
2.8 m							
2.9 m							
3.0 m							

Likely T value: 25.00

Note: *Depth of percolation test holes should be indicated on log above. (Enter P or T at depts as appropriate).
** See Appendix E for BS 5930 classification.
*** 3 samples to be tested for each horizon and results should be entered above for each horizon.
**** All signs of mottling should be recorded.

3.2 Trial Hole (contd.) Evaluation:

Soil seemed suitable for the installation of an on-site wastewater treatment system

3.3(a) Percolation ("T") Test for Deep Subsoils and/or Water Table

Step 1: Test Hole Preparation

Percolation Test Hole	1	2	3
Depth from ground surface to top of hole (mm) (A)	800		
Depth from ground surface to base of hole (mm) (B)	1,200		
Depth of hole (mm) [B - A]	400	0	0
Dimensions of hole [length x breadth (mm)]	300 x 300	x	x

Step 2: Pre-Soaking Test Holes

	1		2		3	
Date and Time pre-soaking started	21/08/2012	12:00				

Each hole should be pre-soaked twice before the test is carried out. Each hole should be empty before refilling.

Step 3: Measuring T_{100}

Percolation Test Hole No.	1	2	3
Date of test	22/08/2012	22/08/2012	22/08/2012
Time filled to 400 mm	08:55	08:55	08:55
Time water level at 300 mm	09:22	09:22	09:22
Time to drop 100 mm (T_{100})	27.00	27.00	27.00
Average T_{100}			27.00

If T_{100} > 300 minutes then T-value >90 – site unsuitable for discharge to ground
If $T_{100} \leq$ 210 minutes then go to Step 4;
If T_{100} > 210 minutes then go to Step 5;

Step 4: Standard Method (where $T_{100} \leq 210$ minutes)

Percolation Test Hole	1			2			3		
Fill no.	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)
1	09:22	10:31	69.00	09:22	10:31	69.00	09:22	10:31	69.00
2	10:35	11:56	81.00	10:35	11:56	81.00	10:35	11:56	81.00
3	11:29	13:02	93.00	11:29	13:02	93.00	11:29	13:02	93.00
Average Δt Value			81.00			81.00			81.00
	Average Δt/4 = [Hole No.1]		20.25 (t_1)	Average Δt/4 = [Hole No.2]		20.25 (t_2)	Average Δt/4 = [Hole No.3]		20.25 (t_3)

Result of Test: T = 20.25 (min/25 mm)

Comments:

Step 5: Modified Method (where $T_{100} > 210$ minutes)

Percolation Test Hole No.	1				2				3			
Fall of water in hole (mm)	Time Factor $= T_f$	Time of fall (mins) $= T_m$	$K_{fs} = T_f / T_m$	T – Value $= 4.45 / K_{fs}$	Time Factor $= T_f$	Time of fall (mins) $= T_m$	$K_{fs} = T_f / T_m$	T – Value $= 4.45 / K_{fs}$	Time Factor $= T_f$	Time of fall (mins) $= T_m$	$K_{fs} = T_f / T_m$	T – Value $= 4.45 / K_{fs}$
300 - 250	8.1	23	0.35	12.64	8.1	23	0.35	12.64	8.1	23	0.35	12.64
250 - 200	9.7	38	0.26	17.43	9.7	38	0.26	17.43	9.7	38	0.26	17.43
200 - 150	11.9	57	0.21	21.32	11.9	57	0.21	21.32	11.9	57	0.21	21.32
150 - 100	14.1	69	0.20	21.78	14.1	69	0.20	21.78	14.1	69	0.20	21.78
Average T- Value	T- Value Hole 1= (t_1)			18.29	T- Value Hole 1= (t_2)			18.29	T- Value Hole 1= (t_3)			18.29

Result of Test: T = 18.29 (min/25 mm)

Comments:

APPENDIX B: SITE CHARACTERISATION FORM

File Reference: Site characterisation at Faha, Co. Limerick

1.0 GENERAL DETAILS (From planning application)

Prefix: 	First Name: n/a	Surname: n/a

Address: Faha, Co. Limerick

Site Location and Townland: Faha, Kildimo, Co. Limerick

Telephone No: 	Fax No:

E-Mail:

Maximum no. of Residents: 	No. of Double Bedrooms: 	No. of Single Bedrooms:

Proposed Water Supply: Mains ✓ Private Well/Borehole ☐ Group Well/Borehole ☐

2.0 GENERAL DETAILS (From planning application)

Soil Type, (Specify Type): Till derived chiefly from (Carboniferous) limestone

Aquifer Category: Regionally Important: Rk Locally Important: Poor:

Vulnerability: Extreme ✓ High ☐ Moderate ☐ Low ☐ High to Low ☐ Unknown ☐

Bedrock Type: Waulsortian Mudbank, Pale-grey Massive Limestones

Name of Public/Group Scheme Water Supply within 1 km: none

Groundwater Protection Scheme (Y/N): Yes	Source Protection Area: SI ☐ SO ☐

Groundwater Protection Response: $R2^3$

Presence of Significant Sites (Archaeological, Natural & Historical): None

Past experience in the area: To the north 100m, trial hole with water table at 1.6m and T-value of 7, P-value of 10

Comments:

(Integrate the information above in order to comment on: the potential suitability of the site, potential targets at risk, and/or any potential site restrictions).

Construction of an on-site wastewater treatment system will only be acceptable if extra unsaturated subsoil is provided beneath either a percolation area or a polishing filter. Vulnerability is extreme and a high water table was noted in the area. Groundwater is a high risk in this area.

Note: Only information available at the desk study stage should be used in this section.

3.0 ON-SITE ASSESSMENT

3.1 Visual Assessment

Landscape Position: Generally flat floodplain lands with a hill outcrop to the west

Slope: Steep (>1:5) ☐ Shallow (1:5-1:20) ☑ Relatively Flat (<1:20) ☐

Surface Features within a minimum of 250m (Distance To Features Should Be Noted In Metres)

Houses: Cluster of houes to the south

Existing Land Use: Agriculture

Vegetation Indicators: none

Groundwater Flow Direction: to the north-west - very flat gradients

Ground Condition: Firm to soft

Site Boundaries: Cluster of houes to the south. Hedgerows on all other boundaries

Roads: Local road to the south - local laneway provides access to the site

Outcrops (Bedrock And/Or Subsoil): A number of bedrock outcrops were noted in all directions. A hill 100m to the south of rock

Surface Water Ponding: none

Lakes: 1km to the west

Beaches/Shellfish: Foreshore of River Maigue to the south

Areas/Wetlands: none

Karst Features: None - bedrock outcrops noted

Watercourse/Stream*: River Maigue 500m to the south

Drainage Ditches*: Series of local dirches draining the lands to the River Maigue

Springs / Wells*: A number o local wells within 250m

Comments:

(Integrate the information above in order to comment on: the potential suitability of the site, potential targets at risk, the suitability of the site to treat the wastewater and the location of the proposed system within the site).

The area is relatively flat and it is difficult to estimate the groundwater flow direction. The hill 100m to the south appears to be a pure rock outcrop. Give the bedrock outcrops and the vulnerability rating it is likely that bedrock in the area is shallow. The risk in the area is groundwater due to the likely shallow depth and thin cover of unsaturated subsoil. The presence of wells within 250m is also a high risk.

*Note and record water level

3.2 Trial Hole (should be a minimum of 2.1m deep (3m for regionally important aquifers))

To avoid any accidental damage, a trial hole assessment or percolation tests should not be undertaken in areas, which are at or adjacent to significant sites (e.g. NHAs, SACs, SPAs, and/or Archaeological etc.), without prior advice from National Parks and Wildlife Service or the Heritage Service.

Depth of trial hole (m): 1.10

Depth from ground surface to bedrock (m) (if present): 1.10

Depth from ground surface to water table (m) (if present): 0.80

Depth of water ingress: 0.70

Rock type (if present): Limestone

Date and time of excavation: 06/08/2012 10:00

Date and time of examination: 06/08/2012 10:30

Depth	Depth of P/T Test*	Soil/Subsoil Texture & Classification**	Plasticity and dilatancy***	Soil Structure	Density/ Compactness	Colour****	Preferential flowpaths
0.1 m – 0.2 m		Silt loam topsoil		Loose	Compact	Brown	Root zone
0.3 m – 0.5 m		SILT	Dilant Threads 1, 2, 2 Ribbons 80mm	Blocky	Loose	Dark brown	None
0.6 m – 1.1 m		silty CLAY Water ingress at 0.7m	Threads 4, 4, 5 Ribbons 130mm	Soft Saturated from 0.7m	Soft	Dark brown	None
1.2 m – 3.0 m		BEDROCK					

Likely T value: 150.00

Note: *Depth of percolation test holes should be indicated on log above. (Enter P or T at depts as appropriate).
** See Appendix E for BS 5930 classification.
*** 3 samples to be tested for each horizon and results should be entered above for each horizon.
**** All signs of mottling should be recorded.

3.2 Trial Hole (contd.) Evaluation:

Water ingress at 0.7m - bedrock at 1.1m - therefore T-test not undertaken as conditions not suitable for standard percolation area.

3.3(a) Percolation ("T") Test for Deep Subsoils and/or Water Table

Step 1: Test Hole Preparation

Percolation Test Hole	1	2	3
Depth from ground surface to top of hole (mm) (A)			
Depth from ground surface to base of hole (mm) (B)			
Depth of hole (mm) [B - A]	0	0	0
Dimensions of hole [length x breadth (mm)]	x	x	x

Step 2: Pre-Soaking Test Holes

	1		2		3	
Date and Time pre-soaking started						

Each hole should be pre-soaked twice before the test is carried out. Each hole should be empty before refilling.

Step 3: Measuring T_{100}

Percolation Test Hole No.	1	2	3
Date of test			
Time filled to 400 mm			
Time water level at 300 mm			
Time to drop 100 mm (T_{100})	0.00	0.00	0.00
Average T_{100}			0.00

If T_{100} > 300 minutes then T-value >90 – site unsuitable for discharge to ground
If $T_{100} \leq$ 210 minutes then go to Step 4;
If T_{100} > 210 minutes then go to Step 5;

Step 4: Standard Method (where T_{100} < 210 minutes)

Percolation Test Hole	1			2			3		
Fill no.	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)	Start Time (at 300 mm)	Finish Time (at 200 mm)	Δt (min)
1			0.00			0.00			0.00
2			0.00			0.00			0.00
3			0.00			0.00			0.00
Average Δt Value			0.00			0.00			0.00
	Average Δt/4 = [Hole No.1]		0.00 (t_1)	Average Δt/4 = [Hole No.2]		0.00 (t_2)	Average Δt/4 = [Hole No.3]		0.00 (t_3)

Result of Test: T = 0.00 (min/25 mm)

Comments:

Step 5: Modified Method (where T_{100} > 210 minutes)

Percolation Test Hole No.	1				2				3			
Fall of water in hole (mm)	Time Factor = T_f	Time of fall (mins) = T_m	K_{fs} = T_f / T_m	T – Value = 4.45 / K_{fs}	Time Factor = T_f	Time of fall (mins) = T_m	K_{fs} = T_f / T_m	T – Value = 4.45 / K_{fs}	Time Factor = T_f	Time of fall (mins) = T_m	K_{fs} = T_f / T_m	T – Value = 4.45 / K_{fs}
300 - 250	8.1				8.1				8.1			
250 - 200	9.7				9.7				9.7			
200 - 150	11.9				11.9				11.9			
150 - 100	14.1				14.1				14.1			
Average T- Value	T- Value Hole 1= (t_1)			0.00	T- Value Hole 1= (t_2)			0.00	T- Value Hole 1= (t_3)			0.00

Result of Test: T = 0.00 (min/25 mm)

Comments:

APPENDIX C

Year	Month	BH-N1b		BH-N1c		BH-N2a		BH-N2b		BH-N3b		BH-N3c	
Year	Month	Temp (°C)	pH	Temp (°C)	pH	Temp (°C)	pH	Temp (°C)	pH	Temp (°C)	pH	Temp (°C)	pH
2011	May	11.4	7.12	11.2	7.14	10.9	7.22	10.2	7.34	n/a	n/a	10.6	6.78
	Jun	11.7	7.1	11.6	7.09	11.1	7.18	10.4	7.31	n/a	n/a	10.2	7.32
	Jul	11.5	7.04	11.1	7.06	11.5	7.15	10.9	7.29	n/a	n/a	11.1	7.22
	Aug	11.6	7.07	11.4	7.04	11.7	7.16	11	7.33	n/a	n/a	10.8	7.41
	Oct	11.9	7.1	11.6	7.16	11.1	7.24	10.5	7.36	n/a	n/a	10.6	6.85
	Nov	9.9	6.94	10.4	7.06	10.4	7.09	10.1	7.22	10.2	6.86	10	6.97
	Dec	10.6	6.97	10.1	7.12	9.8	7.19	9.4	7.24	n/a	n/a	9.7	7.13
2012	Jan	9.4	7.05	9.8	7.17	8.9	7.25	9.6	7.25	n/a	n/a	9.5	7.24
	Feb	10.2	7.08	9.9	7.14	9.6	7.18	9.9	7.19	n/a	n/a	10.1	7.3
	Mar	10.9	7.03	10.8	7.09	10.2	7.2	10.2	7.28	n/a	n/a	10	7.44
	Apr	11.1	7.07	10.5	7.05	10.9	7.28	10.5	7.33	10.6	6.89	10.4	7.14
	May	10.8	6.95	10.1	7.11	10.4	7.16	10.3	7.14	10.4	6.92	10.6	7.09
	Jun	11.5	6.98	11	7.13	10.8	7.08	10.5	7.11	11.2	6.94	10.1	7.04
	Jul	11.8	6.91	11.4	7.04	11.1	7.05	10.7	7.24	12.4	6.81	11.2	6.93
	Aug	12.1	6.96	11.6	7.06	11.5	7.11	10.9	7.41	12.9	6.74	11	7.13
Min		9.4	6.91	9.8	7.04	8.9	7.05	9.4	7.11	10.2	6.74	9.5	6.78
Max		12.1	7.12	11.6	7.17	11.7	7.28	11	7.41	12.9	6.94	11.2	7.44
Avg		11.1	7.02	10.8	7.10	10.7	7.17	10.3	7.27	11.3	6.86	10.4	7.13

Nitrate concnetrations (mg-N/l)

Year	Month	BH-N1b	BH-N1c	BH-N2a	BH-N2b	BH-N3b	BH-N3c
2011	Apr	0	0	2.3	0	0	0
	May	0	0	0	0	1.2	0
	Jun	0.5	0	0	0	0	0
	Jul	0	0	0	0	0	0
	Aug	0	0	1.2	0	0	0.7
	Oct	0	0	0	0	0	0
	Nov	0	0	0	0	0	0
	Dec	0.4	0	0	0	0	0
2012	Jan	0	0	0	0	0	0
	Feb	0.4	0	0.9	0.4	0	0
	Mar	0	0	0	0	0	0
	Apr	0	0	0	0	0	0
	May	0.6	0	0	0.6	0	0
	Jun	1.1	1.3	0.6	0	0.6	1.1
	Jul	0.6	0	0	0	0.4	0

Nitrite concnetrations (mg-N/l)

Year	Month	BH-N1b	BH-N1c	BH-N2a	BH-N2b	BH-N3b	BH-N3c
2011	Apr	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	May	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Jun	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Jul	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Aug	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Oct	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Nov	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Dec	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
2012	Jan	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Feb	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Mar	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Apr	Stopped sampling for nitrite					
	May						
	Jun						
	Jul						

Ammonium concnetrations (mg-N/l)

Year	Month	BH-N1b	BH-N1c	BH-N2a	BH-N2b	BH-N3b	BH-N3c
2011	Apr	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	May	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Jun	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Jul	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Aug	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Oct	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Nov	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Dec	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
2012	Jan	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Feb	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Mar	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Apr	Stopped sampling for ammonium					
	May						
	Jun						
	Jul						

Total phosphorus concentrations (mg-P/l)

Year	Month	BH-N1b	BH-N1c	BH-N2a	BH-N2b	BH-N3b	BH-N3c
2011	Oct	0.062	0.047	0.031	0.027	0.041	0.009
	Nov	0.096	0.071	0.045	0.032	0.048	0.006
	Dec	0.152	0.063	0.112	0.069	0	0.057
2012	Jan	0.084	0.101	0.022	0.044	0	0.017
	Feb	0.125	0.067	0.133	0.124	0	0.036
	Mar	0.096	0.064	0.078	0.042	0	0.028
	Apr	0.031	0.025	0.03	0.024	0.048	0.015
	May	0.079	0.031	0.099	0.008	0.074	0.026
	Jun	0.191	0.079	0.206	0.233	0.214	0.108
	Jul	0.034	0.026	0.088	0.04	0.051	0.038

Water levels (mOD)

Year	Month	BH-N1b	BH-N1c	BH-N2a	BH-N2b	BH-N3b	BH-N3c
2011	Apr	75.56	75	49.97	55.3	-	48.91
	May	75.5	74.95	50.34	56.58	-	49.13
	Jun	75.67	75.1	50.18	56.16	-	49.27
	Jul	75.72	75.15	50.54	55.7	-	49.23
	Aug	75.57	75	50.5	55.95	-	49.09
	Oct	75.97	75.6	50.72	56.4	53.83	50.04
	Nov	77.22	76.75	51.82	57.1	43.35	49.81
	Dec	75.87	75.3	51.12	55.7	-	49.66
2012	Jan	75.92	75.35	50.68	55.3	-	49.85
	Feb	75.8	75.4	50.62	55.66	-	49.35
	Mar	75.57	75.13	50.12	55.35	-	49.15
	Apr	75.92	75.35	49.96	55.66	41.13	49.5
	May	75.52	75	49.87	56.04	38.8	49.05
	Jun	76.47	76	50.09	56.39	50.51	50.66
	Jul	77.17	76.5	50.38	57.25	46.6	49.61

E-coli numbers (CFU/100ml)

Year	Month	BH-N1b	BH-N1c	BH-N2a	BH-N2b	BH-N3b	BH-N3c
2011	Apr	0	0	0	0	0	0
	May	0	0	0	0	0	0
	Jun	1	0	2	0	0	0
	Jul	0	0	0	0	0	0
	Aug	0	0	0	0	0	0
	Oct	0	0	0	0	1	0
	Nov	0	0	0	0	3	0
	Dec	0	0	0	0	0	0
2012	Jan	0	0	0	0	0	0
	Feb	5	0	3	0	0	0
	Mar	0	0	0	0	0	0
	Apr	0	0	2	0	1	0
	May	1	0	0	0	0	0
	Jun	2	1	4	0	2	0
	Jul	0	0	0	0	0	0

Enterococci numbers (CFU/100ml)

Year	Month	BH-N1b	BH-N1c	BH-N2a	BH-N2b	BH-N3b	BH-N3c
2011	Apr	0	0	0	0	0	0
	May	0	0	1	0	0	0
	Jun	2	0	1	0	0	0
	Jul	0	0	0	0	0	0
	Aug	1	0	0	0	0	0
	Oct	0	0	0	0	5	0
	Nov	0	0	1	0	0	0
	Dec	1	0	0	0	0	0
2012	Jan	2	0	1	0	0	0
	Feb	0	0	0	0	0	0
	Mar	0	0	1	0	0	0
	Apr	0	0	0	0	0	0
	May	1	0	3	2	1	0
	Jun	0	0	1	0	0	0
	Jul	0	0	0	0	0	0

Chloride concentrations (mg/l)

Year	Month	BH-N1b	BH-N1c	BH-N2a	BH-N2b	BH-N3b	BH-N3c
2011	Apr	21.7	15.7	39.1	42.8	n/a	23.5
	May	12.3	8.7	37.2	40.3	n/a	22.3
	Jun	36.5	34.8	42.7	43.6	n/a	39.9
	Jul	27.5	22.3	24.2	28	n/a	31.6
	Aug	29.5	28.3	21.9	22.3	n/a	33.9
	Oct	25.6	21.5	22	25.2	n/a	20.8
	Nov	35.7	31.6	21.8	25.6	n/a	27
	Dec	28.7	23.6	25.3	27	n/a	20.3
2012	Jan	23.4	14.1	39.3	40.5	n/a	27.6
	Feb	28	27	34.4	36.6	16.5	18.7
	Mar	12.7	3.8	18.2	21.2	24.1	28.5
	Apr	35.1	27	25	26.3	10.4	35.7
	May	31.5	25.2	33.9	37.7	27.3	33.4
	Jun	33	28.1	32.9	35.1	42.1	25.2
	Jul	30.4	29.5	42.2	45.7	9.4	18

		BH-O1a			BH-O1b			BH-O1c			BH-O2a			BH-O2b			BH-O2c			BH-O3b		
Year	**Month**	**Temp (°C)**	**EC (µS)**	**pH**	**Temp (°C)**	**EC (µS)**	**pH**	**Temp (°C)**	**EC (µS)**	**pH**	**Temp (°C)**	**EC (µS)**	**pH**	**Temp (°C)**	**EC (µS)**	**pH**	**Temp (°C)**	**EC (µS)**	**pH**	**Temp (°C)**	**EC (µS)**	**pH**
2010	Nov	9.6	471	7.14	9.6	886	7.01	9.6	871	6.86	10.4	775	6.89	10.3	756	6.91	9.1	771	7.07	10.6	366	7.13
	Dec	7.9	712	7.21	8.2	661	7.18	9.9	672	6.98	11.6	1551	6.79	11.3	1259	7.04	11.1	961	7.19	10.8	1468	7.05
2011	Jan	8.2	576	7.26	8.2	521	7.18	8.6	527	6.97	9.6	876	7.04	9.5	784	7.11	9.2	654	7.15	9.3	724	7.11
	Feb	9.7	498	7.3	9.6	412	7.28	10.1	406	7.06	10.2	745	6.97	10.1	708	7.06	10.4	621	7.04	10.6	717	7.22
	Mar	9.2	1042	7.41	10	933	6.94	10.1	848	6.86	11.1	848	7.27	10.8	906	7.12	11.3	806	6.84	11.7	763	7.26
	Apr	10.1	934	7.11	10.6	724	7.2	10.6	711	7.22	11.1	703	7.09	11.5	739	7.08	11.4	758	7.18	10.6	1056	7.08
	May	10.8	826	7.08	11.4	711	7.22	11.3	693	7.16	11.5	682	7.12	11.5	751	7.11	11.6	687	7.15	10.9	973	7.09
	Jun	11.6	618	7.26	11.4	550	7.18	10.8	568	7.09	12	654	7.19	12	522	7.16	11.3	501	7.09	11.3	476	7.32
	Jul	11.4	845	7.35	11.6	776	7.31	10.7	754	7.22	11.7	712	7.15	11.4	725	7.17	10.5	612	6.95	10.9	648	7.28
	Aug	11..1	789	7.29	11	648	7.21	10.4	641	7.19	11.4	498	7.04	11.1	467	7.06	10.4	422	7.14	10.8	345	7.23
	Oct	10.7	984	6.94	10.7	792	7.02	9.6	775	7.13	11.1	841	6.81	11	764	6.97	10.1	722	7.12	10.1	651	7.07
	Nov	11	924	7.08	11.1	818	7.05	9.9	841	6.96	10.8	889	7.1	10.7	641	7.14	10.2	609	7.11	9.4	846	7.22
	Dec	9.5	975	7.19	9.4	998	7.17	9.4	984	7.08	9.6	748	7.27	10.1	584	7.31	9.7	555	7.14	9.9	444	7.16
2012	Jan	9.1	840	6.96	9	824	7.03	9.5	811	7.09	9.1	511	6.88	9.7	452	6.98	9.8	384	7.06	9.7	276	7.11
	Feb	10.1	870	7.16	10.2	783	7.17	10.2	743	7.15	10.4	741	7.28	10.3	344	7.21	10.2	339	7.18	10.3	421	7.23
	Mar	10.5	951	7.14	10.5	811	7.36	10.4	764	7.26	10.7	819	7.14	10.5	841	7.44	10.4	665	7.27	10.4	582	7.29
	Apr	9.2	913	7.37	9.3	865	7.64	9.7	888	7.68	9.7	820	7.45	9.6	796	7.68	9.6	786	7.69	10	806	7.74
	May	11.1	718	7.41	11	654	7.35	10.4	649	7.13	10.9	728	7.34	10.8	911	7..39	10.4	712	7.22	10.1	643	7.64
	Jun	11.4	548	6.82	11.3	612	6.94	10.6	628	7.01	11.3	n/a	n/a	11.2	n/a	n/a	10.5	346	6.98	10.2	n/a	n/a
	Jul	12.1	798	6.86	11.9	715	6.92	10.9	687	6.98	11.8	784	6.87	11.7	769	7.02	11.1	645	7.22	11.1	789	7.15
	Aug	11.8	723	6.92	11.8	668	6.95	10.7	661	6.94	11.7	889	6.93	11.6	912	6.97	10.9	497	7.07	10.9	748	6.92
	Sep	11.5	810	6.97	11.3	703	7.01	10.8	674	6.96	11.1	684	7.18	11.1	516	7.23	10.3	434	7.27	10.5	387	7.51
	Oct	11.7	931	7.69	10.7	869	7.88	10.5	851	7.89	10.1	820	7.62	10.5	761	7.67	10.7	792	7.45	10.6	732	7.93
Min		7.9	471	6.82	8.2	412	6.92	8.6	406	6.86	9.1	498	6.79	9.5	344	6.91	9.1	339	6.84	9.3	276	6.92
Max		12.1	1042	7.69	11.9	998	7.88	11.3	984	7.89	12	1551	7.62	12	1259	7.68	11.6	961	7.69	11.7	1468	7.93
Avg		10.4	795.5	7.17	10.4	736.3	7.18	10.2	723.8	7.12	10.8	787.2	7.11	10.8	723.1	7.16	10.4	620.8	7.16	10.5	675.5	7.26

Year	Month	BH-O1a	BH-O1b	BH-O1c	BH-O2a	BH-O2b	BH-O2c	BH-O3b
2010	Nov	22.1	32.4	39.1	27.4	23	26	16.2
	Dec	29.4	19	14.5	112	96.2	69.3	136
2011	Jan	32	34.5	28.6	52.4	41	34.6	44.8
	Feb	17	15.1	12.6	27.4	19.6	11.4	22.6
	Mar	132	53	59	56	88	53	71
	Apr	43.4	23	19.4	18.6	23.4	19	109.4
	May	42	44.2	36.7	24.5	22.3	14.6	87
	Jun	36	32.1	27.4	27.4	16.4	16	39.2
	Jul	59	57.2	36.4	36.1	24.5	22.1	42.9
	Aug	46.1	42.4	45.7	41	43.4	27.1	28
	Oct	23.1	17.4	12.5	14.3	12	10.4	19.2
	Nov	72	41	26.4	41	24.5	17.1	52.4
	Dec	82	89.4	34.2	33.1	35.2	22.1	63.2
2012	Jan	30	26	14.5	22	14.6	11	24.6
	Feb	47	79	54	59	56	61	99
	Mar	68.8	86	58.3	66.2	63	46.7	36.2
	Apr	39.9	28.7	22	22.4	22.1	20.3	27.4
	May	27.1	44	30.3	57.4	49.6	36	38.7
	Jun	19.4	12.1	14.3	15	15	14.2	15
	Jul	16.1	17.2	15.6	22.1	12.3	13.6	9.4
	Aug	12.3	10.3	13.4	18.6	11.3	12.9	11.2
	Sep	50	61	42	75	62	49	51
	Oct	72	63	56	50	51	62	57

BH-O1c	BH-O2c	BH-O3b
39.1	26	16.2
14.5	69.3	136
28.6	34.6	44.8
12.6	11.4	22.6
59	53	71
19.4	19	109.4
36.7	14.6	87
27.4	16	39.2
36.4	22.1	42.9
45.7	27.1	28
12.5	10.4	19.2
26.4	17.1	52.4
34.2	22.1	63.2
14.5	11	24.6
54	61	99
58.3	46.7	36.2
22	20.3	27.4
30.3	36	38.7
14.3	14.2	15
15.6	13.6	9.4
13.4	12.9	11.2
42	49	51
56	62	57

		E-coli numbers (CFU/100ml)							Enterococci numbers (CFU/100ml)						
Year	**Month**	**BH-O1a**	**BH-O1b**	**BH-O1c**	**BH-O2a**	**BH-O2b**	**BH-O2c**	**BH-O3b**	**BH-O1a**	**BH-O1b**	**BH-O1c**	**BH-O2a**	**BH-O2b**	**BH-O2c**	**BH-O3b**
2010	Nov	1	0	0	1	0	0	3	0	0	0	0	0	0	0
	Dec	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2012	Jan	0	0	0	1	0	0	0	0	0	0	0	0	0	0
	Feb	2	0	0	0	0	0	3	0	0	0	0	0	0	0
	Mar	3	1	0	7	1	6	26	9	2	0	30	5	8	86
	Apr	0	0	0	0	0	0	4	0	0	0	0	0	0	0
	May	0	0	0	0	0	0	0	0	0	0	1	0	0	0
	Jun	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Jul	2	2	0	1	0	0	1	0	0	0	0	0	0	0
	Aug	2	0	0	1	0	0	1	1	0	0	0	0	0	0
	Oct	4	1	0	2	3	0	5	5	0	0	2	1	0	12
	Nov	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dec	0	0	0	0	0	0	0	0	0	0	0	0	0	0
2012	Jan	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Feb	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Mar	2	0	0	0	0	0	4	2	0	0	1	0	0	7
	Apr	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	May	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Jun	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Jul	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Aug	1	0	0	0	2	0	0	0	0	0	0	0	0	0
	Sep	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Oct	1	0	0	0	3	1	4	0	0	0	0	0	0	2

Total phosphorus concentrations (mg-P/l)

Year	**Month**	**BH-O1a**	**BH-O1b**	**BH-O1c**	**BH-O2a**	**BH-O2b**	**BH-O2c**	**BH-O3b**
2011	Nov	0.048	0.046	0.047	0.051	0.05	0.052	0.056
	Dec	0.045	0.051	0.046	0.053	0.048	0.042	0.05
2012	Jan	0.052	0.05	0.061	0.042	0.053	0.05	0.071
	Feb	0.063	0.063	0.052	0.064	0.049	0.048	0.061
	Mar	0.081	0.078	0.077	0.084	0.081	0.086	0.089
	Apr	0.078	0.076	0.053	0.067	0.087	0.077	0.096
	May	0.071	0.067	0.075	0.069	0.081	0.085	0.07
	Jun	0.053	0.029	0.067	0	0	0.049	0
	Jul	0.026	0.022	0.019	0.046	0.034	0.042	0.031
	Aug	0.035	0.041	0.042	0.037	0.036	0.038	0.041
	Sep	0.038	0.037	0.044	0.033	0.027	0.03	0.044
	Oct	0.047	0.036	0.046	0.04	0.042	0.045	0.044

		Nitrate concentrations (mg-N/l)								Ammonium concentrations (mg-N/l)					
Year	Month	BH-O1a	BH-O1b	BH-O1c	BH-O2a	BH-O2b	BH-O2c	BH-O3b	BH-O1a	BH-O1b	BH-O1c	BH-O2a	BH-O2b	BH-O2c	BH-O3b
2010	Nov	8.6	6.6	6.9	4.9	5.2	3.8	5.5	0.21	0.11	<0.05	0.19	<0.05	<0.05	0.15
	Dec	9.8	2.8	3.6	5.2	2.7	2.2	2.2	0.38	0.23	<0.05	0.21	0.06	0.22	0.32
2011	Jan	7.9	3.2	3.6	2.8	1.9	1.8	3.2	0.12	0.08	<0.05	0.1	<0.05	<0.05	0.07
	Feb	4.8	4.1	3.5	4.9	3.8	1.9	6.4	0.08	<0.05	<0.05	0.06	<0.05	<0.05	0.1
	Mar	12.3	4.6	6.3	1	3.7	4.4	2.8	0.23	<0.05	0.07	0.09	<0.05	<0.05	0.12
	Apr	13.2	4.1	4.1	3.5	3.9	4.3	5.1	0.08	<0.05	<0.05	0.14	0.09	<0.05	0.06
	May	6.7	3.1	3.4	6.8	4.9	4.4	5.2	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Jun	4.1	1.8	1.1	4.2	2.6	2.8	3.6	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Jul	6.1	5	5.1	2.7	2.1	1.8	1.1	0.08	<0.05	<0.05	0.05	<0.05	<0.05	0.06
	Aug	5.7	3.5	4.2	3.4	4.1	2.6	3.1	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Oct	3.7	4.4	3.7	2.9	1.8	0.5	0.9	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Nov	8.7	5.2	4.6	3.1	3.2	4.1	6.3	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Dec	10.6	7.4	7.6	4.1	4.2	3.9	2.6	0.16	0.09	<0.05	0.05	<0.05	<0.05	0.07
2012	Jan	8.8	2.9	3.4	2.4	1.5	1.2	4	0.1	0.11	<0.05	0.08	<0.05	<0.05	0.09
	Feb	5.6	4.4	4.5	2.5	4.1	4.9	3.7	<0.05	0.1	<0.05	0.21	0.11	0.12	<0.05
	Mar	9.2	6.7	4.6	4.1	4.8	3.7	2.9	0.09	0.1	<0.05	<0.05	<0.05	<0.05	<0.05
	Apr	8.3	5.2	3.9	3.5	3.4	3.6	2.1	0.07	<0.05	<0.05	0.21	<0.05	<0.05	0.25
	May	5.9	4.1	2.9	5.9	6.4	4.6	5.1	0.1	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Jun	3.7	2.1	1.1	n/a	n/a	0.9	n/a	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Jul	2.9	2.3	1.4	3.1	2.8	5.2	1.1	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Aug	1.6	1.2	0.7	2.2	2.6	0.8	3.1	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05
	Sep	5.1	4.8	3.4	0.9	1.1	2.6	4.1	0.2	0.15	<0.05	0.22	<0.05	<0.05	0.12
	Oct	3.7	4.4	3.7	2.9	1.8	0.5	0.9	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05

		Nitrite concentrations (mg-N/l)						
Year	Month	BH-O1a	BH-O1b	BH-O1c	BH-O2a	BH-O2b	BH-O2c	BH-O3b
2010	Nov	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Dec	0.11	<0.02	0.02	<0.02	<0.02	0.05	0.04
2011	Jan	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Feb	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Mar	0.07	<0.02	<0.02	0.04	<0.02	<0.02	<0.02
	Apr	0.04	<0.02	<0.02	0.05	<0.02	<0.02	<0.02
	May	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Jun	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Jul	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Aug	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Oct	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Nov	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Dec	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
2012	Jan	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Feb	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Mar	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Apr	<0.02	<0.02	<0.02	0.04	<0.02	<0.02	<0.02
	May	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Jun	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Jul	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Aug	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Sep	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Oct	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02

Year	Month	BH-K1a			BH-K1b			BH-K2a			BH-K2b			BH-K2c		
		Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH
2010	Nov	10.4	341	7.08	10	365	7.05	11.1	462	7.12	11.4	451	7.16	11.2	432	7.23
	Dec	10.3	309	6.67	10.1	312	6.97	11.5	486	7.04	11.4	441	6.96	11.7	498	6.84
2011	Jan	8.9	316	6.78	9.3	324	6.89	10.7	456	6.56	10.9	466	6.59	11.2	511	6.61
	Feb	9.6	368	7.26	9.2	379	7.33	11	440	7.51	11.1	472	7.34	10.9	459	7.55
	Mar	8.4	352	6.89	11	364	6.73	11.4	572	6.78	11.5	542	6.91	11.2	432	6.89
	Apr	10.9	256	7.22	11.1	284	7.14	11.7	294	6.85	11.5	312	6.74	11.4	292	6.66
	May	12.1	268	6.74	12.3	295	6.78	11.4	331	6.86	11.1	347	6.97	11	351	7.08
	Jun	12.8	289	6.69	12.6	276	6.78	11.8	351	6.55	11.4	319	6.69	11.6	341	6.89
	Jul	13.2	264	7.04	13.4	291	7.11	12	284	7.08	12.1	341	7.09	12.4	326	7.21
	Aug	14.1	274	6.89	14.3	279	6.94	12.8	327	6.92	12.6	336	7.06	12.7	331	7.19
	Oct	12.8	401	7.22	12.9	386	7.18	11.2	352	7.76	11.5	348	7.29	11.6	362	7.38
	Nov	11.4	346	6.59	11.5	322	6.65	11.6	375	6.57	11.5	371	6.67	11.3	366	6.62
	Dec	10.4	321	7.14	10.6	334	7.22	11.2	380	7.31	11	364	7.26	11.5	419	7.28
2012	Jan	8.6	298	6.88	8.3	301	6.94	10.9	412	7.09	11.2	375	7.03	11.1	421	6.96
	Feb	8.5	271	6.71	8.4	282	6.65	11.1	312	6.54	10.9	322	6.61	10.8	328	6.83
	Mar	8.9	317	6.79	8.8	300	6.77	11.5	352	6.68	11.4	366	6.72	11.1	355	6.77
	Apr	10.1	313	7.41	10.2	338	7.36	11.2	412	6.84	11.1	393	7.18	10.2	389	7.21
	May	12.4	298	6.86	12.5	305	6.82	11.5	374	6.76	11.4	380	6.82	11.6	369	6.89
	Jun	13.9	312	7.11	13.6	444	7.31	12.6	401	7.28	12.6	423	7.25	12.7	413	7.21
	Jul	13.2	326	7.15	13.1	362	7.18	11.9	425	7.25	11.7	444	7.18	11.8	435	7.26
	Aug	13.7	376	6.97	13.5	372	9.98	12.4	411	6.42	12.2	398	6.51	12.1	409	6.6
	Sep	14.3	318	7.08	14.5	343	7.07	12.6	484	6.84	12.4	406	6.94	12.2	414	7.04
	Oct	12.6	312	8.02	12.8	316	7.91	11.4	477	7.89	11.3	396	7.78	11.3	391	7.67
Minimum		8.4	256	6.59	8.3	276	6.65	10.7	284	6.42	10.9	312	6.51	10.2	292	6.6
Maximum		14.3	401	8.02	14.5	444	9.98	12.8	572	7.89	12.6	542	7.78	12.7	511	7.67
Average		11.4	315.0	7.01	11.5	329.3	7.16	11.6	398.7	6.98	11.5	391.9	6.99	11.5	393.2	7.04

		Nitrate concentration (mg-N/l)					Water level (mOD)		
Year	Month	BH-K1a	BH-K1b	BH-K2a	BH-K2b	BH-K2c	BH2a	BH2b	BH2c
2010	Nov	6.6	6.4	5.4	6.3	4.3	30.848	31.418	31.454
	Dec	2.7	1.9	2.6	2.8	1.8	31.988	32.088	32.024
2011	Jan	10.3	6.4	7.1	5.3	4.9	30.087	30.037	29.984
	Feb	13.2	12.7	8.6	8.1	7.8	31.598	31.708	31.774
	Mar	9.9	9.5	6.4	6	5.9	31.014	31.117	30.996
	Apr	6.5	6.3	6.1	7.4	8.8	28.288	28.528	28.484
	May	4.2	3.9	3.9	4.2	4.1	28.458	28.578	28.634
	Jun	8.8	7.9	7.1	6.8	8.1	31.196	31.218	31.023
	Jul	9.1	9	8.5	6.4	7.9	31.398	31.568	31.574
	Aug	5.7	6.3	6.9	7.2	7.7	30.598	30.628	30.774
	Oct	4.8	4.6	5.9	6.2	7.1	31.648	31.528	31.474
	Nov	6.9	5.8	4.7	5.1	4	31.358	31.503	31.529
	Dec	4.9	3.8	3.2	2.6	2.5	31.308	31.328	31.334
2012	Jan	10.9	8.7	7.8	6.3	6.1	30.298	30.668	30.604
	Feb	8.1	7.2	7.5	6.2	6.6	30.758	30.868	30.834
	Mar	0.9	0.2	0.8	0.6	0.7	28.538	28.678	28.714
	Apr	5	4.1	3.5	2.9	4.9	28.358	28.698	28.714
	May	3.6	3.1	2.2	2.1	2.2	30.738	30.728	30.734
	Jun	9	8.5	6	4.8	5.2	31.778	31.888	31.864
	Jul	8	7.7	5.4	4.5	3.9	30.718	30.908	30.934
	Aug	4.9	5.1	3.2	3.1	3.7	31.688	31.858	31.894
	Sep	6.7	8.3	4	4.1	4.3	27.838	28.308	28.174
	Oct	8.5	7.5	4.9	3.9	4.8	28.968	28.868	29.174

		Nitite concentration (mg-N/l)					Ammonium concentration (mg-N/l)				
Year	Month	BH1a	BH1b	BH2a	BH2b	BH2c	BH1a	BH1b	BH2a	BH2b	BH2c
2010	Nov	<0.02	<0.02	0.24	0.21	0.13	0.11	<0.05	<0.05	<0.05	<0.05
	Dec	<0.02	<0.02	<0.02	<0.02	0.02	<0.05	<0.05	<0.05	0.08	0.07
2011	Jan	<0.02	<0.02	<0.02	<0.02	0.04	<0.05	0.11	0.09	<0.05	0.24
	Feb	<0.02	<0.02	<0.02	0.03	<0.02	<0.05	<0.05	<0.05	<0.05	<0.05
	Mar	<0.02	<0.02	<0.02	<0.02	<0.02	<0.05	<0.05	0.07	0.07	<0.05
	Apr	0.16	0.17	0.21	0.09	0.13	0.35	0.36	0.21	0.16	0.22
	May	<0.02	<0.02	<0.02	<0.02	<0.02	<0.05	<0.05	<0.05	<0.05	<0.05
	Jun	<0.02	<0.02	<0.02	<0.02	<0.02	<0.05	0.12	<0.05	0.08	<0.05
	Jul	<0.02	<0.02	<0.02	<0.02	<0.02	<0.05	<0.05	<0.05	<0.05	<0.05
	Aug	<0.02	<0.02	<0.02	<0.02	<0.02	<0.05	<0.05	0.07	<0.05	0.05
	Oct	<0.02	<0.02	<0.02	<0.02	<0.02					
	Nov	<0.02	<0.02	<0.02	<0.02	0.04	<0.05	<0.05	<0.05	<0.05	<0.05
	Dec	<0.02	<0.02	<0.03	0.02	0.05	<0.05	<0.05	<0.05	<0.05	<0.05
2012	Jan	<0.02	<0.02	<0.02	<0.02	<0.02	<0.05	<0.05	0.05	<0.05	0.08
	Feb	<0.02	<0.02	0.02	0.03	0.04	0.21	<0.05	<0.05	<0.05	<0.05
	Mar	<0.02	<0.02	<0.02	<0.02	<0.02	<0.05	<0.05	0.05	0.06	0.06
	Apr	<0.02	<0.02	0.02	0.04	0.02	<0.05	<0.05	<0.05	<0.05	<0.05
	May	<0.02	<0.02	0.02	0.03	0.02	<0.05	<0.05	<0.05	<0.05	<0.05
	Jun	<0.02	0.02	0.03	0.02	<0.02	0.32	0.11	<0.05	<0.05	<0.05
	Jul	<0.02	<0.02	<0.02	<0.02	<0.02	0.26	0.08	<0.05	<0.05	<0.05
	Aug	<0.02	<0.02	<0.02	<0.02	<0.02	<0.05	<0.05	<0.05	<0.05	<0.05
	Sep	<0.02	<0.02	<0.02	<0.02	<0.02	<0.05	<0.05	<0.05	<0.05	<0.05
	Oct	<0.02	<0.02	0.02	<0.02	<0.02	<0.05	<0.05	<0.05	<0.05	<0.05

		COD concentration (mg-N/l)				
Year	Month	BH1a	BH1b	BH2a	BH2b	BH2c
2011	Nov	<4	<4	<4	<4	<4
	Dec	<4	<4	<4	<4	<4
2012	Jan	<4	<4	<4	<4	<4
	Feb	<4	<4	<4	<4	<4
	Mar	<4	<4	<4	<4	<4
	Apr	<4	<4	<4	<4	4.6
	May	<4	<4	<4	<4	<4
	Jun	<4	<4	<4	<4	<4

		Total Phosphorus concentration (mg-P/l)					Chloride concentration (mg/l)				
Year	Month	BH-K1a	BH-K1b	BH-K2a	BH-K2b	BH-K2c	BH-K1a	BH-K1b	BH-K2a	BH-K2b	BH-K2c
2010	Nov	-	-	-	-	-	12.1	11.6	10.2	11	10.8
	Dec	-	-	-	-	-	22.4	22.1	19.8	20.4	18.7
	Jan	-	-	-	-	-	24.6	22.9	18.6	18.2	16.4
	Feb	-	-	-	-	-	8	8.5	7.9	10.5	10.7
	Mar	-	-	-	-	-	19.8	19.4	19.6	18.9	19.4
	Apr	-	-	-	-	-	79	74	86	62	67
	May	-	-	-	-	-	63	67	46.5	45.4	42
	Jun	-	-	-	-	-	30	29.3	27.4	26.5	26.6
	Jul	-	-	-	-	-	24	23.1	20.5	21.2	21.1
	Aug	-	-	-	-	-	32	32	26.2	25.5	25
	Oct	-	-	-	-	-	17.4	18.2	15	16.2	15.6
2011	Nov	0.0695	0.181	0.071	0.089	0.097	19.4	23.7	19.3	19.1	19.6
	Dec	0.052	0.104	0.1	0.087	0.092	28	25	19	18	19
2012	Jan	0.107	0.089	0.048	0.036	0.042	16.2	16.9	19.2	20.7	20
	Feb	0.061	0.102	0.052	0.087	0.082	27	26	27.1	26.4	26.9
	Mar	0.076	0.086	0.024	0.069	0.1	12.5	13.6	11.1	10.8	12.8
	Apr	0.126	0.041	0.054	0.091	0.064	59	57	50	63	60
	May	0.027	0.036	0.061	0.059	0.07	39.4	39.1	30.5	29.6	29
	Jun	0.048	0.035	0.063	0.039	0.058	9.2	9.8	8.6	8.4	8.6
	Jul	0.012	0.011	0.015	0.023	0.048	13	12	12.4	12.2	11.7
	Aug	0.017	0.019	0.064	0.051	0.032	14.4	14	12.7	12.1	12.1
	Sep	0.044	0.033	0.045	0.071	0.06	53	55	51	45	50
	Oct	0.041	0.039	0.074	0.055	0.047	51	53.2	49	51.6	51.4

		E-coli numbers (CFU/100ml)					Enterococci numbers (CFU/100ml)				
Year	Month	BH-K1a	BH-K1b	BH-K2a	BH-K2b	BH-K2c	BH-K1a	BH-K1b	BH-K2a	BH-K2b	BH-K2c
2010	Nov	7	6	3	1	1	2	0	1	0	1
	Dec	2	3	4	2	1	11	9	6	7	4
2011	Jan	1	0	0	1	0	2	1	2	1	0
	Feb	17	14	6	12	5	28	18	9	14	6
	Mar	10	2	1	0	2	0	0	1	0	0
	Apr	6	1	1	13	1	0	0	0	1	1
	May	0	0	0	1	0	2	1	2	0	1
	Jun	2	1	0	0	1	0	0	1	1	3
	Jul	5	3	4	8	3	12	5	4	2	7
	Aug	0	0	0	0	0	0	0	1	0	0
	Oct	3	2	1	4	1	1	0	0	0	6
	Nov	5	8	0	0	0	54	1	0	0	0
	Dec	2	0	0	1	0	3	1	2	2	1
2012	Jan	0	0	0	1	5	0	0	0	1	1
	Feb	8	5	5	0	3	15	9	22	13	26
	Mar	0	0	0	0	0	0	0	0	0	0
	Apr	1	0	0	0	0	4	0	0	0	0
	May	0	0	0	1	0	2	1	0	0	1
	Jun	201	1	1	0	0	26	0	32	12	0
	Jul	24	1	2	0	0	11	3	131	3	0
	Aug	3	0	0	1	1	5	2	4	1	0
	Sep	38	4	0	0	0	0	0	0	0	0

		BH-L1a			BH-L1b			BH-L1c			BH-L2a			BH-K2b			BH-L3a			BH-L3b			BH-L3c		
Year	Month	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH	Temp (°C)	EC (µS)	pH
2010	Nov	10.4	851	7.21	10.7	832	7.29	10.5	886	7.32	9.4	945	6.56	9.4	998	6.57	9.5	921	6.86	9.3	908	6.91	9.6	875	6.94
	Dec	9.2	587	7.08	9.1	621	7.14	10.1	634	7.22	8.9	843	6.84	8.8	874	6.87	8.7	789	7.86	8.4	776	7.78	8.9	745	7.91
2011	Jan	8.6	612	7.41	9.1	578	7.54	10.2	974	7.56	8.6	745	7.18	5.5	721	7.21	8.2	698	7.05	8	725	7.02	8.3	734	6.94
	Feb	8.9	751	7.34	9.5	708	7.35	9.9	712	7.42	9.1	874	7.58	9.1	912	7.68	9.1	777	6.74	9.1	812	6.82	8.9	841	6.91
	Mar	10.5	632	7.04	10.2	624	7.11	10.8	601	7.19	10.4	812	7.09	10.3	734	7.14	10	822	7.72	9.9	784	7.79	10.2	789	7.89
	Apr	11	894	7.26	10.8	854	7.33	11.1	794	7.51	10.6	1068	6.89	10.2	1122	6.88	10.7	922	6.89	10.6	940	6.78	10.4	931	6.9
	May	11.4	645	7.31	11.1	634	7.58	10.9	697	7.62	11.4	1141	7.41	11.5	997	7.38	11.1	979	6.96	10.9	948	7.03	11.2	941	7.01
	Jun	12.3	676	7.34	11.3	630	7.31	11.2	660	7.41	13.3	1283	7.31	13.1	1311	7.21	13	931	6.91	12.7	922	6.98	13.9	882	7.24
	Jul	11.7	489	7.47	10.9	512	7.76	11.3	777	7.61	11.9	974	7.56	11.8	921	7.65	12.1	871	7.22	12	845	7.31	12.4	824	7.47
	Aug	12.2	677	7.32	11.2	772	7.31	11.1	622	7.44	12.7	844	7.44	12.5	851	7.48	12.9	1011	7.14	12.5	977	7.13	12.8	1054	7.22
	Oct	10.9	812	7.11	10.2	998	7.18	11	723	7.29	11.4	972	7.21	11	1052	7.32	11.2	712	6.82	11.1	684	6.84	11.5	661	6.73
	Nov	10.7	766	7.25	10.4	741	7.26	10.9	723	7.35	11.1	908	7.46	11.1	964	7.52	11	874	6.96	11.3	888	7.01	11.4	912	7.06
	Dec	9.6	512	7.32	10.2	509	7.4	10.8	484	7.55	10.6	666	7.78	10.4	708	7.66	10.6	632	7.45	10.4	678	7.49	10.6	641	7.48
2012	Jan	9.4	576	7.09	9.5	547	7.07	10.5	513	7.18	9.6	712	7.25	9.4	597	7.33	9.1	879	7.76	8.9	942	7.65	9.4	911	7.86
	Feb	9.6	784	7.2	9.8	772	7.25	10.3	745	7.48	9.7	854	7.62	9.8	862	7.68	9	706	7.45	8.7	691	7.41	9.2	725	7.76
	Mar	10.8	724	7.39	10.8	736	7.44	10.9	674	7.76	11.1	906	7.54	11.2	913	7.43	9.7	1151	7.08	9.6	1032	7.06	9.7	1067	7.04
	Apr	10.7	884	7.76	10.9	862	7.69	11.1	812	7.71	11.3	921	7.75	11.4	943	7.77	11.9	1011	7.44	12.3	1201	7.64	12.6	894	7.62
	May	11.4	941	7.55	11.5	986	7.67	11.3	887	7.81	11.6	1021	7.94	11.6	945	7.83	11.7	987	6.84	11.5	941	6.86	11.1	955	6.76
	Jun	12.2	964	7.29	12.2	951	7.35	11.7	840	7.32	12.7	1201	7.64	12.5	1184	7.6	12.4	1241	7.02	12.3	1222	7.06	11.7	1094	7.14
	Jul	12.9	821	7.18	13	910	7.15	11.9	945	7.22	13.4	1084	7.32	13.1	948	7.39	12.8	789	7.49	12.6	741	7.51	11.9	725	7.66
	Aug	13.1	694	7.33	13.1	771	7.39	11.8	698	7.42	13.8	845	7.42	13.2	913	7.37	13.8	812	6.98	13.2	875	6.92	12.1	913	6.84
	Sep	12.8	786	7.49	12.3	1082	7.56	12.6	891	7.72	12.5	722	7.46	12.6	761	7.42	13.5	888	7.24	13.1	789	7.76	12.9	711	8.2
	Oct	10.9	854	7.24	11	941	7.32	11.1	1023	7.56	10.7	687	7.72	10.9	662	7.61	9.9	784	6.97	10.1	721	6.96	10.4	684	7.54
Minimum		8.6	489	7.04	9.1	509	7.07	9.9	484	7.18	8.6	666	6.56	5.5	597	6.57	8.2	632	6.74	8	678	6.78	8.3	641	6.73
Maximum		13.1	964	7.76	13.1	1082	7.76	12.6	1023	7.81	13.8	1283	7.94	13.2	1311	7.83	13.8	1241	7.86	13.2	1222	7.79	13.9	1094	8.2
Average		10.9	736.2	7.3	10.8	764.0	7.4	11.0	752.8	7.5	11.1	914.3	7.4	10.9	908.4	7.4	11.0	877.7	7.2	10.8	871.4	7.2	10.9	848.2	7.3

		Nitrate concentration (mg-N/l)							Nitrite concentration (mg-N/l)						
Year	**Month**	**BH-L1a**	**BH-L1b**	**BH-L1c**	**BH-L2b**	**BH-L3a**	**BH-L3b**	**BH-L3c**	**BH-L1a**	**BH-L1b**	**BH-L1c**	**BH-L2b**	**BH-L3a**	**BH-L3b**	**BH-L3c**
2010	Nov	5.2	2.7	2.2	2.2	9.8	2.8	3.6	<0.02	<0.02	0.05	0.04	0.11	<0.02	0.02
	Dec	0	0	0	0	0	0	0	0.04	<0.02	<0.02	<0.02	0.06	0.04	<0.02
2011	Jan	5.7	1.1	0	0	6.4	4.7	1.6	0.07	0.04	<0.02	0.06	<0.02	<0.02	<0.02
	Feb	0.6	0	0	0	0	0.5	0	0.14	0.07	0.03	0.16	0.12	0.1	<0.02
	Mar	1.1	1.6	4.2	8.2	5.4	6.1	2.3	0.06	0.07	0.03	0.08	0.1	0.04	<0.02
	Apr	1.1	0	0.6	2.5	0	0	0.6	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	May	0	0	0	0	1.1	0.6	0	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Jun	2.9	3	9	5.1	0.4	0.3	3	<0.02	<0.02	<0.02	<0.02	0.04	0.04	0.03
	Jul	0	0	0	0	0	0	0	0.03	<0.02	<0.02	0.05	0.12	<0.02	<0.02
	Aug	0.9	1.1	0.8	6.7	2.5	1.2	0.6	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Oct	0	0.5	0.7	0	0	1.2	2.7	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Nov	4.3	4	3.4	2.9	0	2.5	1.6	0.06	<0.02	0.05	0.05	0.17	0.03	0.02
	Dec	0	0	0	0	1.1	0	0	0.06	0.05	0.06	0.04	0.02	<0.02	<0.02
2012	Jan	6.4	1.9	1.8	4.6	7.5	5.4	4.6	0.11	0.08	0.06	<0.02	0.07	0.05	<0.02
	Feb	0	0	0.3	0.4	5	0	0	0.05	0.07	0.05	0.37	0.03	0.03	0.02
	Mar	1.8	2.5	1.1	0	0	0	0.6	<0.02	<0.02	<0.02	0.11	0.05	0.05	0.03
	Apr	0	0	1.7	0	0	0	1.7	0.03	0.04	<0.02	0.02	0.02	0.03	0.02
	May	0.6	0.6	1.1	1.2	0.5	0.4	0	0.02	<0.02	<0.02	0.04	0.03	<0.02	<0.02
	Jun	1.4	1.6	6.5	3.2	1.6	0	0	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Jul	1.1	0.8	0.4	0.6	0.4	0	0	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Aug	0.2	0.1	0	0.4	0	0	0	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Sep	0.5	0	0.3	0.3	0	1.6	0.3	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02
	Oct	0.1	0	0	0	0.3	0	0.1	0.02	<0.02	<0.02	0.03	0.06	<0.02	<0.02

		Ammonium concentration (mg-N/l)							Chloride concentration (mg/l)						
Year	**Month**	**BH-L1a**	**BH-L1b**	**BH-L1c**	**BH-L2b**	**BH-L3a**	**BH-L3b**	**BH-L3c**	**BH-L1a**	**BH-L1b**	**BH-L1c**	**BH-L2b**	**BH-L3a**	**BH-L3b**	**BH-L3c**
2010	Nov	0.21	0.06	0.06	0.29	0.38	0.28	0.05	32	36	39	31	29	36	42
	Dec	0.07	<0.05	<0.05	0.1	0.15	0.09	<0.05	22	24	32	25	34	27	29
2011	Jan	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	26	28	29	22	27	23	19
	Feb	0.29	0.22	0.08	0.16	0.15	0.09	<0.05	19	18	16	18	21	18	16
	Mar	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	27	24	29	24	26	21	24
	Apr	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	39	34	36	25	31	27	26
	May	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	32	37	27	38	39	42	31
	Jun	0.05	0.11	0.15	0.3	<0.05	<0.05	<0.05	49	52	40	34	52	47	56
	Jul	0.05	<0.05	<0.05	0.05	<0.05	<0.05	<0.05	66	61	54	44	29	28	22
	Aug	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	37	41	29	32	30	34	27
	Oct	0.05	<0.05	<0.05	<0.05	0.05	<0.05	<0.05	27	24	21	29	37	36	30
	Nov	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	38	31	24	32	22	21	17
	Dec	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	30	34	31	27	37	39	27
2012	Jan	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	29	27	24	22	26	27	22
	Feb	0.26	0.22	0.05	<0.05	<0.05	<0.05	<0.05	18.6	19.1	18.9	16.5	17.1	18.2	19.1
	Mar	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	20	19	19	25	26	22	27
	Apr	0.31	0.29	<0.05	<0.05	<0.05	<0.05	<0.05	25.6	26.7	25.9	22.6	24.3	23.5	18.3
	May	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	19	16	15	22	29	28	27
	Jun	0.08	0.06	<0.05	<0.05	0.15	<0.05	<0.05	38	36	30	42	47	44	43
	Jul	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	49	45	40	35	57	55	46
	Aug	0.09	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	38	39.4	44.2	29.4	45	35.2	40.4
	Sep	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	<0.05	42	47	59	48	63	73	69
	Oct	0.15	0.11	0.06	0.32	0.09	<0.05	<0.05	49	55	57	51	72	76	68

Total Phosphorus concentration (mg-P/l)

Year	Month	BH-L1a	BH-L1b	BH-L1c	BH-L2b	BH-L3a	BH-L3b	BH-L3c
2011	Nov	0.09	0.085	0.246	0.072	0.105	0.083	0.082
	Dec	0.071	0.066	0.08	0.051	0.093	0.081	0.077
2012	Jan	0.04	0.045	0.061	0.033	0.064	0.061	0.042
	Feb	0.113	0.122	0.081	0.064	0.107	0.078	0.121
	Mar	0.106	0.094	0.067	0.07	0.091	0.084	0.095
	Apr	0.096	0.137	0.053	0.061	0.099	0.099	0.12
	May	0.107	0.082	0.061	0.049	0.076	0.082	0.06
	Jun	0.057	0.051	0.04	0.031	0.038	0.042	0.033
	Jul	0.069	0.072	0.055	0.078	0.06	0.051	0.054
	Aug	0.044	0.038	0.031	0.022	0.047	0.036	0.029
	Sep	0.086	0.078	0.062	0.039	0.094	0.078	0.062
	Oct	0.074	0.073	0.068	0.087	0.066	0.073	0.059

E-coli numbers (CFU/100ml)

Year	Month	BH-L1a	BH-L1b	BH-L1c	BH-L2b	BH-L3a	BH-L3b	BH-L3c
2010	Nov	1	0	0	3	2	0	0
	Dec	1	2	1	0	0	0	0
2011	Jan	0	0	0	0	0	0	0
	Feb	8	2	5	1	0	0	3
	Mar	13	11	34	6	8	41	52
	Apr	3	0	2	0	1	4	38
	May	0	0	1	0	1	0	2
	Jun	2	2	1	0	0	0	1
	Jul	0	0	6	3	1	0	4
	Aug	4	0	7	0	5	2	11
	Oct	1	3	64	8	11	34	81
	Nov	0	0	0	0	0	0	0
	Dec	0	0	0	0	0	0	0
2012	Jan	1	0	1	0	0	0	2
	Feb	1	0	14	0	0	0	0
	Mar	1	15	29	12	7	19	66
	Apr	1	0	0	0	0	0	240
	May	0	0	4	0	0	0	0
	Jun	1	1	3	0	0	2	0
	Jul	1	0	0	4	0	0	6
	Aug	1	1	7	4	0	0	5
	Sep	2	5	189	16	6	278	291
	Oct	0	1	4	0	0	2	17

Enterococci numbers (CFU/100ml)

Year	Month	BH-L1a	BH-L1b	BH-L1c	BH-L2b	BH-L3a	BH-L3b	BH-L3c
2010	Nov	0	0	1	2	0	0	6
	Dec	0	0	0	0	0	0	0
2011	Jan	1	0	0	1	0	0	0
	Feb	5	0	1	0	0	0	1
	Mar	6	7	19	7	5	17	29
	Apr	0	0	7	0	0	4	15
	May	0	0	0	0	0	0	0
	Jun	1	0	0	0	0	0	0
	Jul	0	0	2	0	0	0	0
	Aug	2	1	0	0	2	1	4
	Oct	2	7	41	4	2	19	32
	Nov	1	0	1	0	0	1	0
	Dec	0	0	0	0	0	0	1
2012	Jan	0	0	1	0	0	0	0
	Feb	0	0	0	0	0	0	0
	Mar	4	3	36	11	5	2	14
	Apr	2	0	0	0	0	0	13
	May	1	0	7	0	0	0	0
	Jun	0	0	0	0	3	0	0
	Jul	0	0	0	1	0	0	2
	Aug	0	0	10	0	0	0	1
	Sep	1	2	19	3	2	36	23
	Oct	0	2	3	1	0	4	6

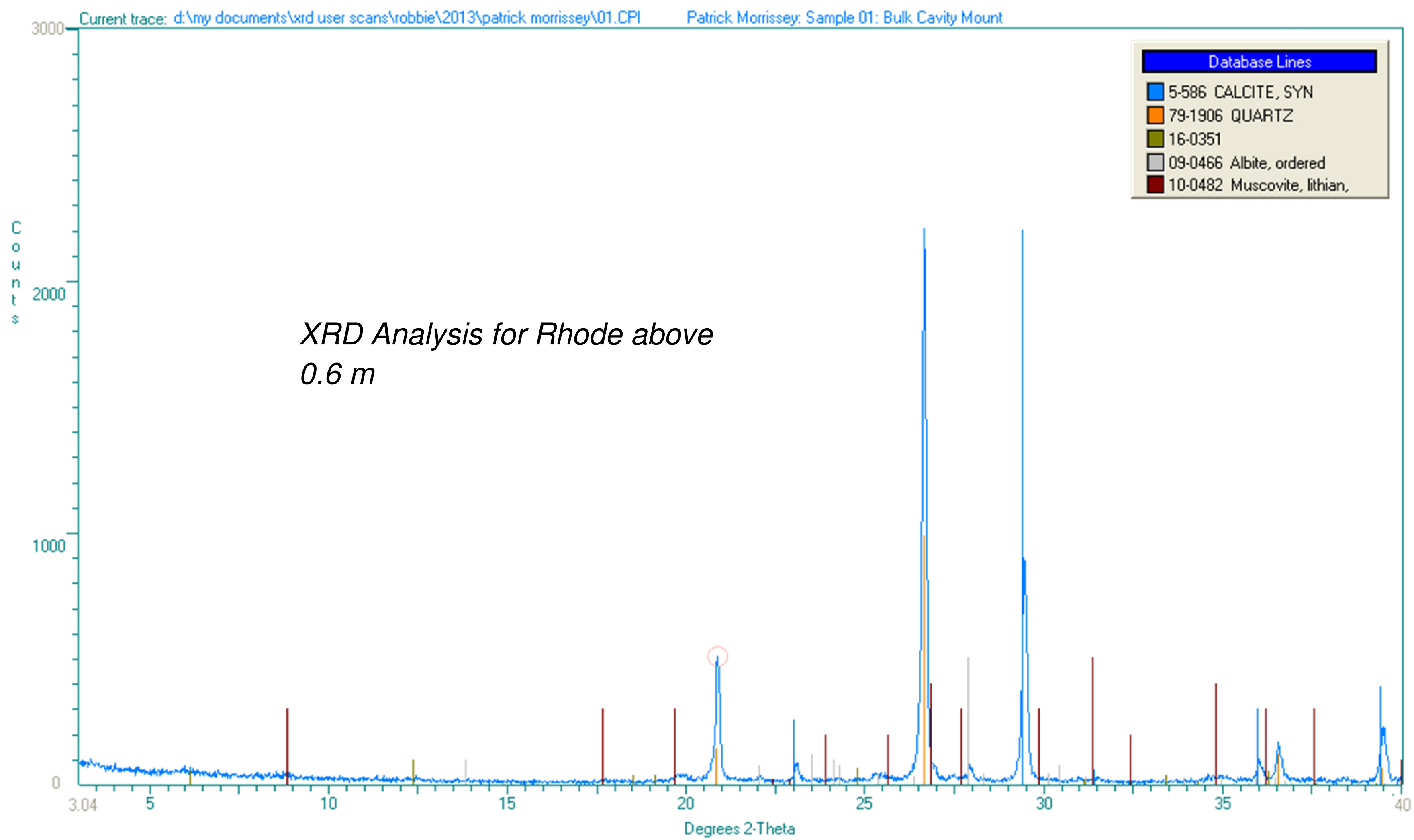
COD concentration (mg/l)

Year	Month	BH1a	BH1b	BH1c	BH2a	BH2b	BH3a	BH3b	BH3c
2012	Jan	<4	<4	<4	<4	<4	<4	<4	<4
2012	Feb	<4	<4	<4	<4	<4	<4	<4	<4
2012	Mar	<4	<4	<4	<4	<4	<4	<4	<4
2012	Apr	<4	<4	<4	<4	<4	<4	<4	<4
2012	May	<4	<4	<4	<4	<4	<4	<4	<4

Water level (mODl)

Year	Month	BH-L1a	BH-L1b	BH-L1c	BH-L2a	BH-L2b	BH-L3a	BH-L3b	BH-L3c
2010	Nov	0.723	0.151	0.386	1.728	1.811	0.655	0.653	0.515
	Dec	2.433	2.671	2.906	-0.732	-0.469	-0.225	-0.217	-0.615
2011	Jan	2.033	2.031	2.266	0.638	0.921	0.795	0.788	0.765
	Feb	0.803	0.661	0.896	1.698	1.891	0.875	0.883	0.815
	Mar	2.373	1.481	1.716	-0.442	-0.289	-0.025	-0.017	-0.155
	Apr	2.232	1.241	1.476	0.018	0.241	-0.325	-0.327	-0.785
	May	1.053	0.721	0.956	1.688	1.911	0.665	0.673	0.615
	Jun	0.733	0.481	0.716	1.598	1.771	0.63	0.628	0.575
	Jul	2.093	1.461	1.696	-0.262	0.001	0.165	0.183	-0.005
	Aug	2.143	1.251	1.486	0.068	0.311	-0.275	-0.277	-0.685
	Oct	1.353	0.611	0.846	1.738	1.931	0.695	0.683	0.645
	Nov	1.433	0.751	0.986	1.188	1.371	0.745	0.753	0.715
	Dec	0.363	0.141	0.376	2.298	2.471	1.025	1.023	0.995
2012	Jan	0.853	0.761	0.996	1.578	1.891	0.605	0.603	0.415
	Feb	2.173	2.331	2.566	1.388	1.461	0.085	-0.047	-0.265
	Mar	1.743	1.451	1.686	-0.512	-0.229	-0.315	-0.317	-0.635
	Apr	2.173	2.331	2.566	1.388	1.461	0.085	-0.047	-0.265
	May	2.463	0.671	0.906	-0.072	0.211	0.055	0.043	-0.025
	Jun	0.533	0.221	0.456	2.258	2.511	0.995	1.003	0.965
	Jul	1.643	1.651	1.886	1.518	1.701	0.645	0.643	0.465
	Aug	1.863	1.751	1.986	0.968	1.211	0.755	0.753	0.735
	Sep	2.078	2.161	2.396	-0.112	0.301	-0.255	-0.387	-0.595
	Oct	1.713	1.451	1.686	0.878	1.211	0.805	0.803	0.775

APPENDIX D



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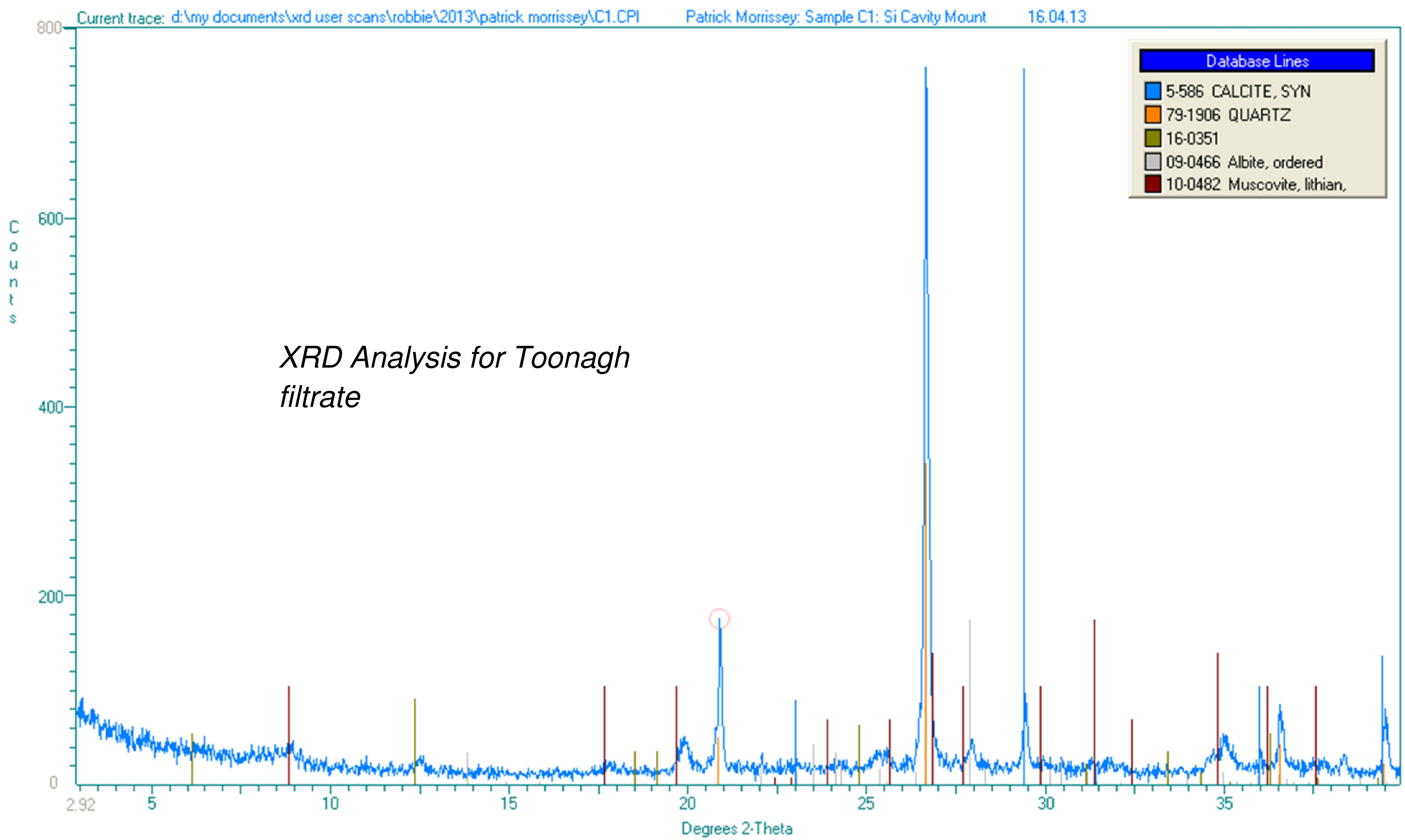
Patrick Morrissey: Sample 01: Bulk Cavity Mount
Database Lines
5-586 CALCITE, SYN
79-1906 QUARTZ
16-0351
09-0466 Albite, ordered
10-0482 Muscovite, lithian,
XRD Analysis for Rhode above
0.6 m
Counts
3000
2000
1000
0
3.04
5
10
15
20
25
30
35
40
Degrees 2-Theta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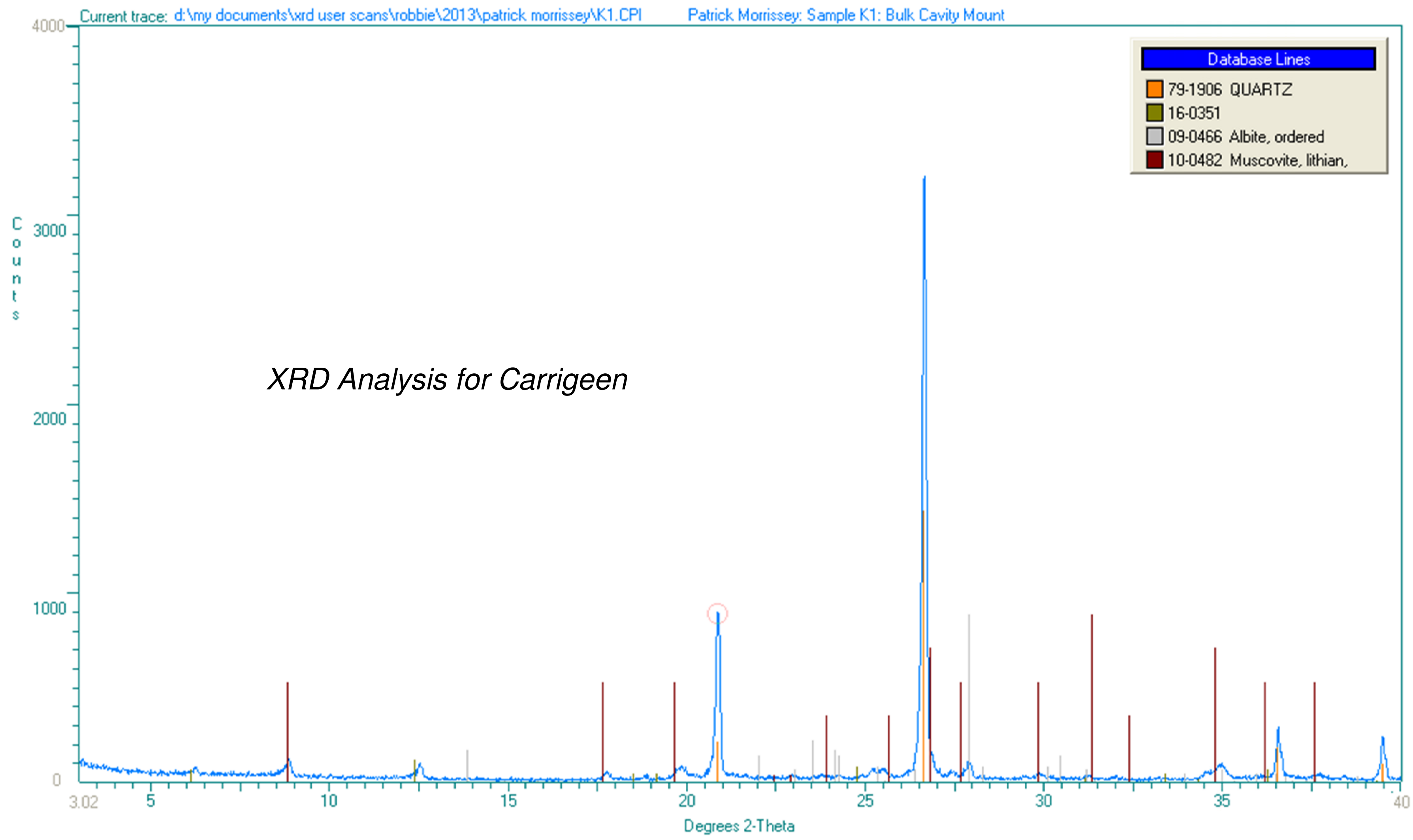
Patrick Morrissey: Sample C1: Si Cavity Mount
16.04.13
Database Lines
5-586 CALCITE, SYN
79-1906 QUARTZ
16-0351
09-0466 Albite, ordered
10-0482 Muscovite, lithian,
XRD Analysis for Toonagh filtrate
Counts
800
600
400
200
0
2.92
5
10
15
20
25
30
35
Degrees 2-Theta



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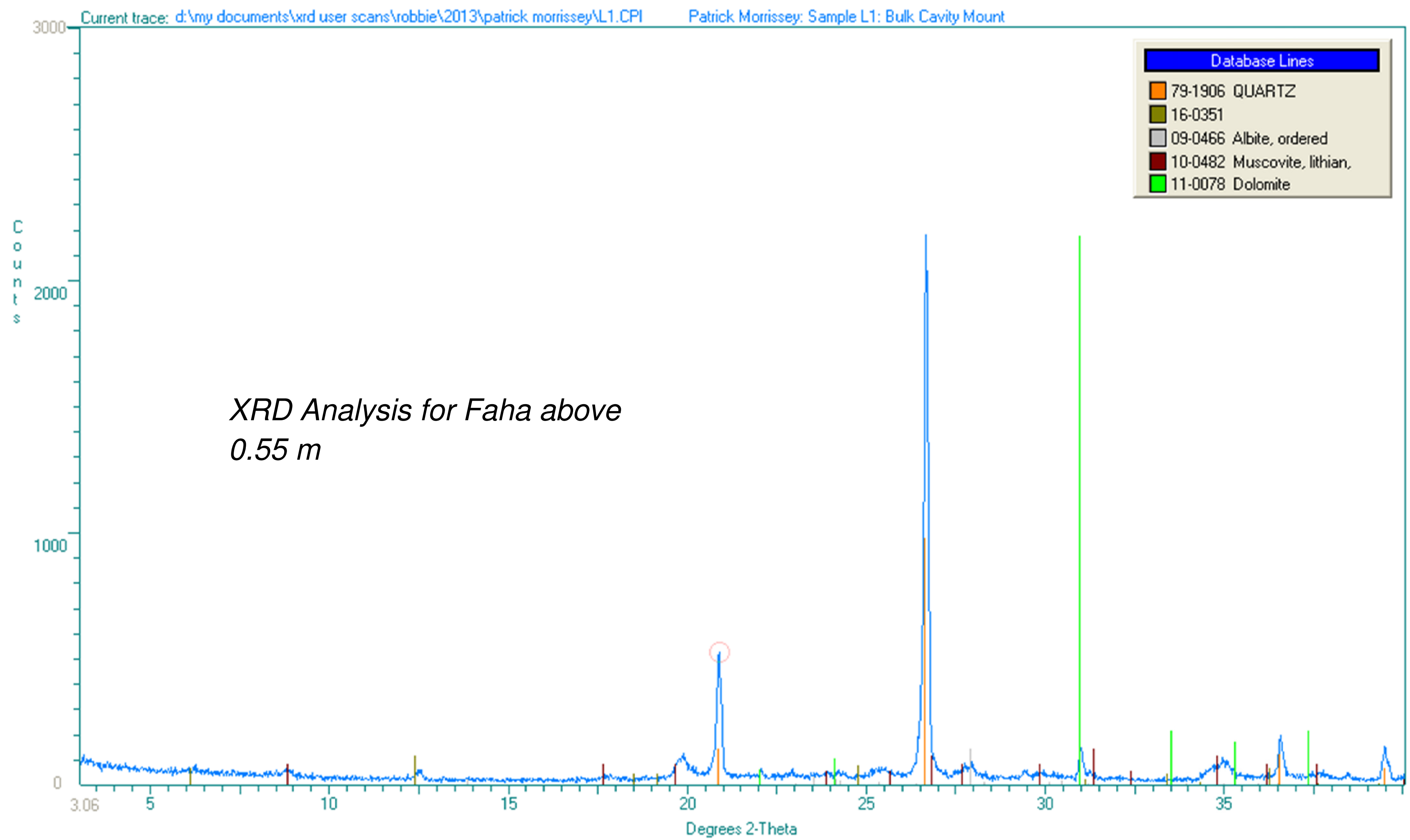
Patrick Morrissey: Sample K1: Bulk Cavity Mount
Database Lines
79-1906 QUARTZ
16-0351
09-0466 Albite, ordered
10-0482 Muscovite, lithian,
XRD Analysis for Carrigeen
Counts
4000
3000
2000
1000
0
3.02
5
10
15
20
25
30
35
40
Degrees 2-Theta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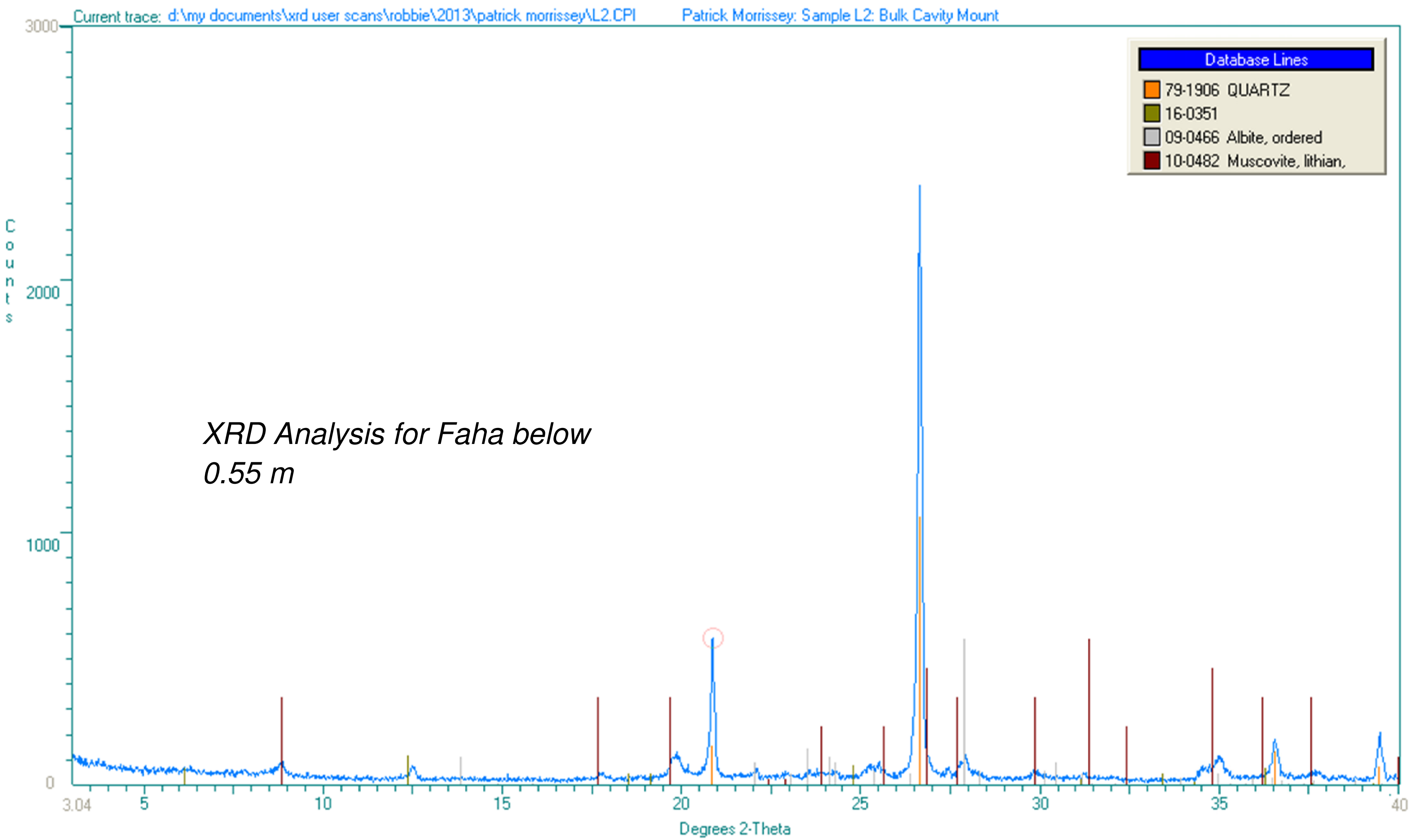
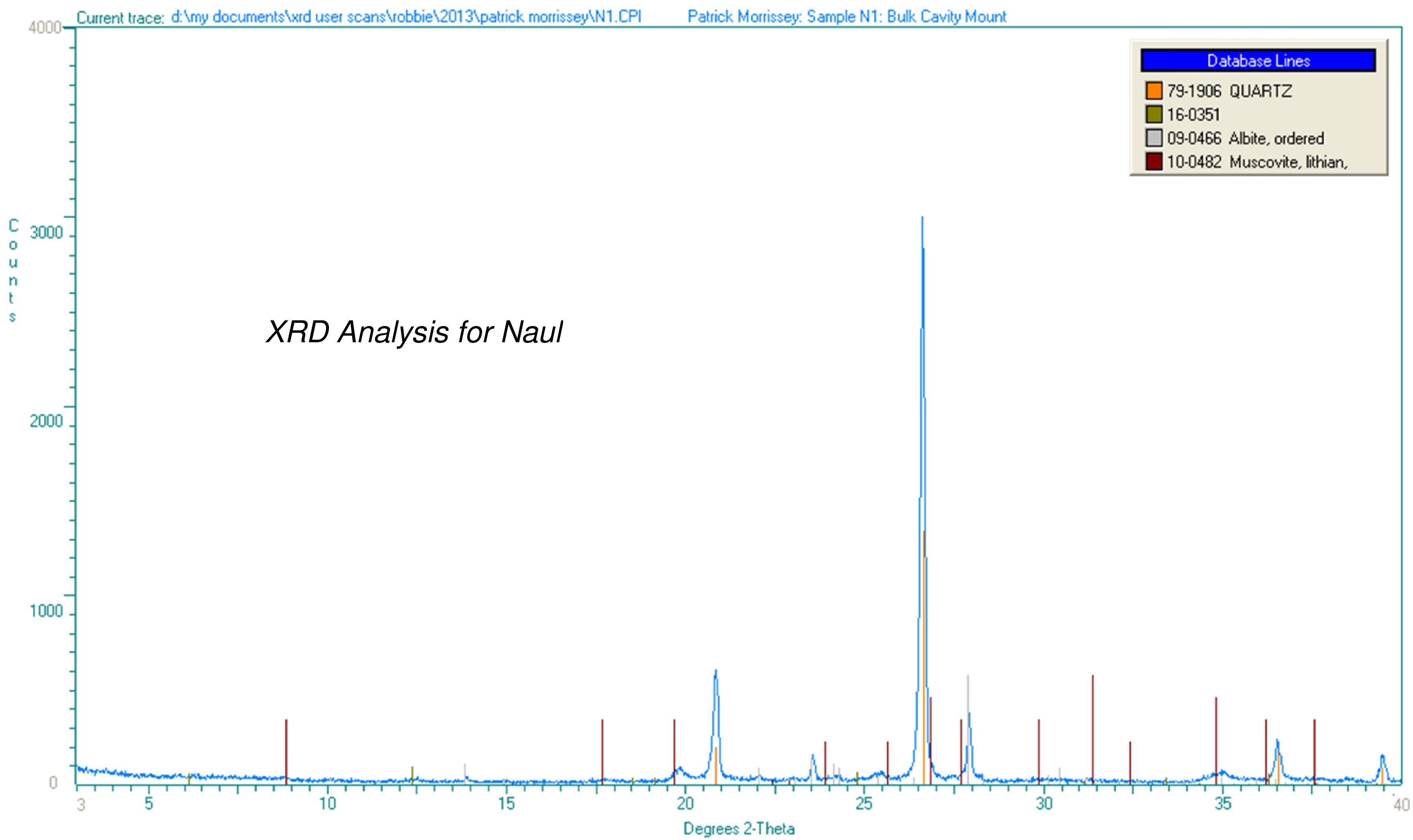
Patrick Morrissey: Sample L1: Bulk Cavity Mount
Database Lines
79-1906 QUARTZ
16-0351
09-0466 Albite, ordered
10-0482 Muscovite, lithian,
11-0078 Dolomite
3000
2000
1000
0
Counts
3.06
5
10
15
20
25
30
35
Degrees 2-Theta
XRD Analysis for Faha above
0.55 m



Current trace: d:\my documents\xrd user scans\robbie\2013\patrick morrissey\L2.CPI
Patrick Morrissey: Sample L2: Bulk Cavity Mount
Database Lines
79-1906 QUARTZ
16-0351
09-0466 Albite, ordered
10-0482 Muscovite, lithian,
XRD Analysis for Faha below
0.55 m
Counts
3000
2000
1000
0
3.04
5
10
15
20
25
30
35
40
Degrees 2-Theta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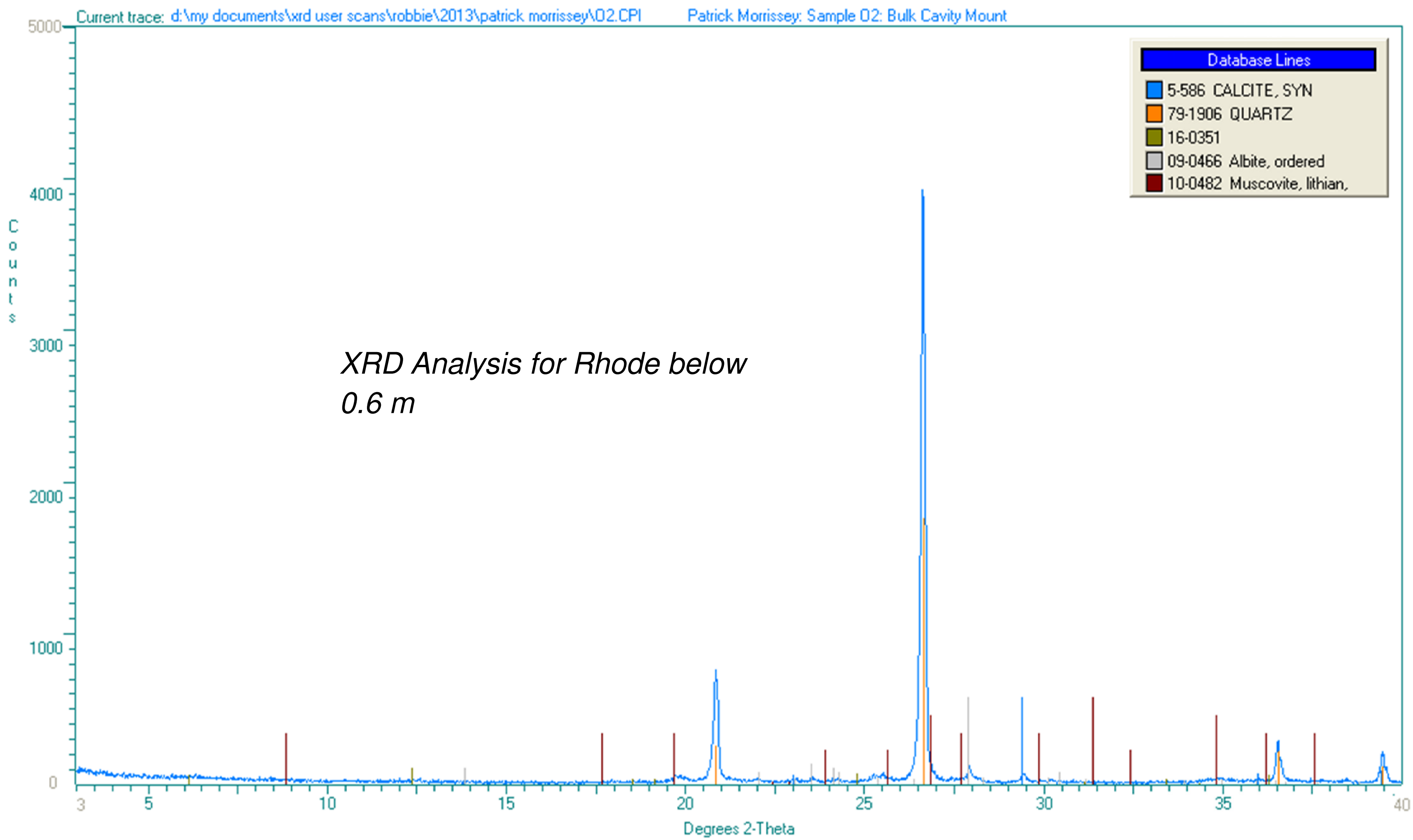
Patrick Morrissey: Sample N1: Bulk Cavity Mount
XRD Analysis for Naul
Database Lines
79-1906 QUARTZ
16-0351
09-0466 Albite, ordered
10-0482 Muscovite, lithian,
Counts
4000
3000
2000
1000
0
3
5
10
15
20
25
30
35
40
Degrees 2-Theta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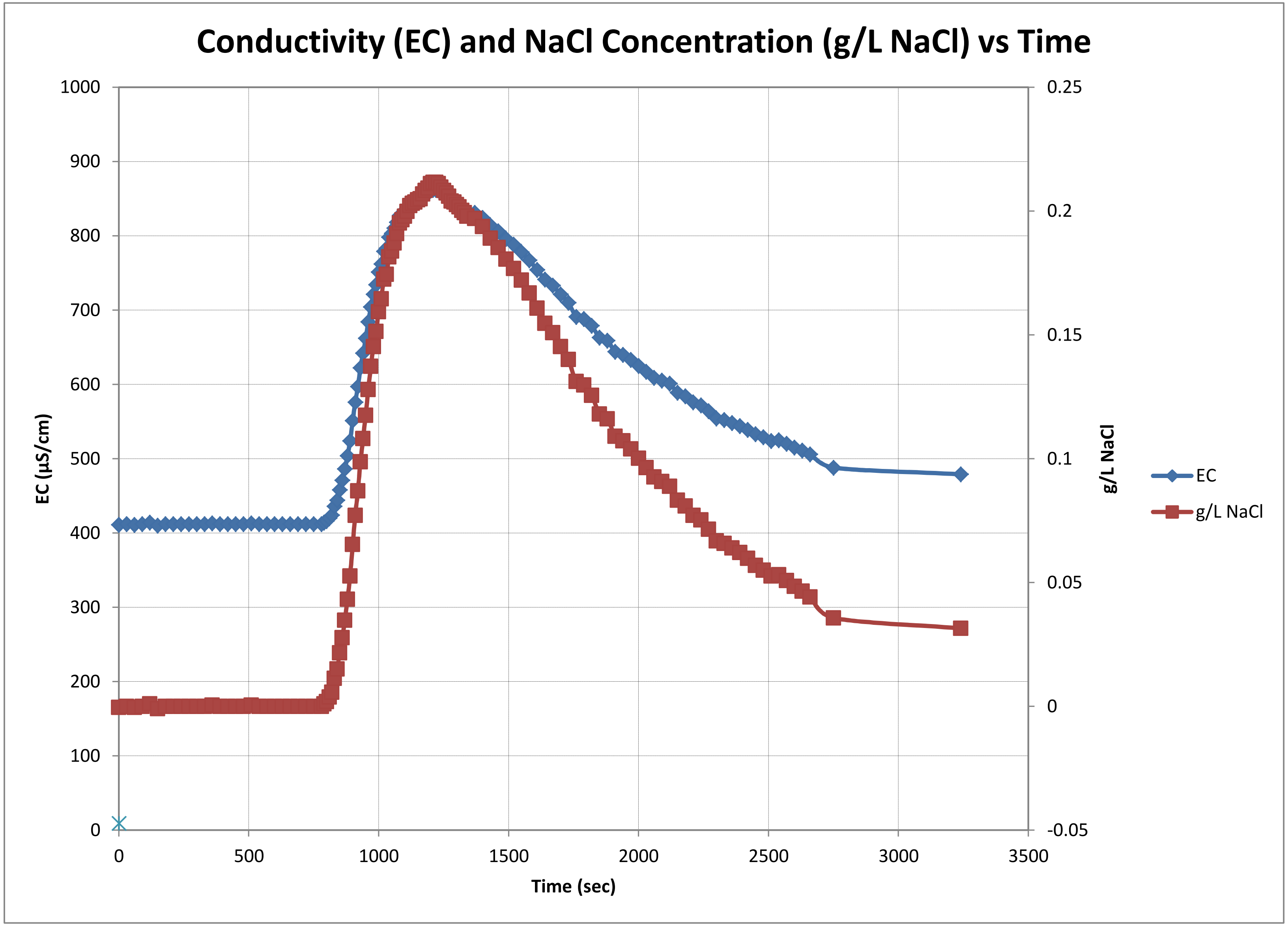
Patrick Morrissey: Sample O2: Bulk Cavity Mount
Database Lines
5-586 CALCITE, SYN
79-1906 QUARTZ
16-0351
09-0466 Albite, ordered
10-0482 Muscovite, lithian,
XRD Analysis for Rhode below
0.6 m
Counts
5000
4000
3000
2000
1000
0
3
5
10
15
20
25
30
35
40
Degrees 2-Theta

APPENDIX E

Sudden Injection Dilution Gauging			
River:	Toreen east	**Location:**	Upstream
Date:	28/05/2013	**Time:**	12:15
Water Temp (C):	12	**Normalising Temp (C):**	25
Weather Conditions:			
Tracer Mass (g NaCl):	2250	**Background EC (µS/cm):**	412
µS/cm EC to g/L NaCl conversion factor:			0.00047
Gauging Structure:			
Grid Reference:		**Stage:**	0.15
Discharge (L/s):		**9.1438**	

Time (sec)	**EC (µS/cm)**	**g/L NaCl**	**Area**
0	411	-0.00047	
30	412	0	-0.00705
60	411	-0.00047	-0.0141
90	412	0	-0.02115
120	414	0.00094	-0.00705
150	410	-0.00094	-0.00705
180	412	0	-0.02115
210	412	0	-0.02115
240	412	0	-0.02115
270	412	0	-0.02115
300	412	0	-0.02115
330	412	0	-0.02115
360	413	0.00047	-0.0141
390	412	0	-0.00705
420	412	0	-0.00705
450	412	0	-0.00705
480	412	0	-0.00705
510	413	0.00047	1.7347E-18
540	412	0	0.00705
570	412	0	0.00705
600	412	0	0.00705
630	412	0	0.00705
660	412	0	0.00705
690	412	0	0.00705
720	412	0	0.00705
750	412	0	0.00705
780	412	0	0.00705
790	414	0.00094	0.01175
800	416	0.00188	0.02585
810	420	0.00376	0.05405
820	424	0.00564	0.10105
830	436	0.01128	0.18565



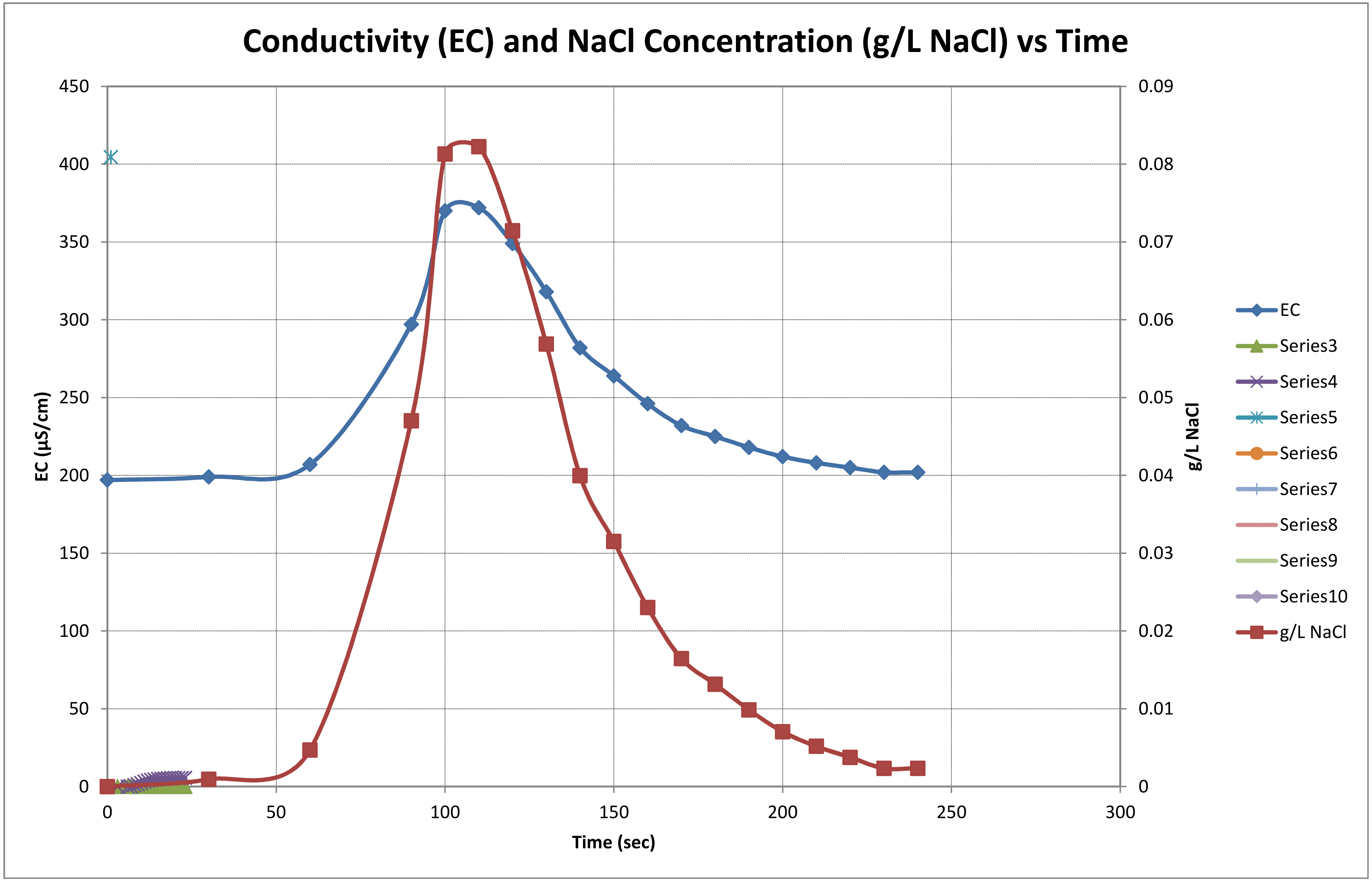
Conductivity (EC) and NaCl Concentration (g/L NaCl) vs Time
EC (µS/cm)
g/L NaCl
Time (sec)
0
100
200
300
400
500
600
700
800
900
1000
0
500
1000
1500
2000
2500
3000
3500
-0.05
0
0.05
0.1
0.15
0.2
0.25
EC
g/L NaCl

Sudden Injection Dilution Gauging			
River:	Toreen east	**Location:**	Downstream
Date:	28/05/2013	**Time:**	
Water Temp (C):	12	**Normalising Temp (C):**	25
Weather Conditions:			
Tracer Mass (g NaCl):	2.25	**Background EC (µS/cm):**	412
µS/cm EC to g/L NaCl conversion factor:			0.00047
Gauging Structure:			
Grid Reference:		**Stage:**	0.25
Discharge (L/s):		**16.6100**	

Time (sec)	EC (µS/cm)	g/L NaCl	Area
0	411	-0.00047	
30	412	0	-0.00705
60	411	-0.00047	-0.0141
90	412	0	-0.02115
120	414	0.00094	-0.00705
150	410	-0.00094	-0.00705
180	412	0	-0.02115
210	412	0	-0.02115
240	412	0	-0.02115
270	412	0	-0.02115
300	412	0	-0.02115
330	412	0	-0.02115
360	413	0.00047	-0.0141
390	412	0	-0.00705
420	412	0	-0.00705
450	412	0	-0.00705
480	412	0	-0.00705
510	413	0.00047	1.7347E-18
540	412	0	0.00705
570	412	0	0.00705
600	412	0	0.00705
630	412	0	0.00705
660	412	0	0.00705
690	412	0	0.00705
720	412	0	0.00705
750	412	0	0.00705
780	412	0	0.00705
790	414	0.00094	0.01175
800	416	0.00188	0.02585
810	420	0.00376	0.05405
820	424	0.00564	0.10105
830	436	0.01128	0.18565
840	444	0.01504	0.31725
850	458	0.02162	0.50055
860	471	0.02773	0.7473
870	486	0.03478	1.05985

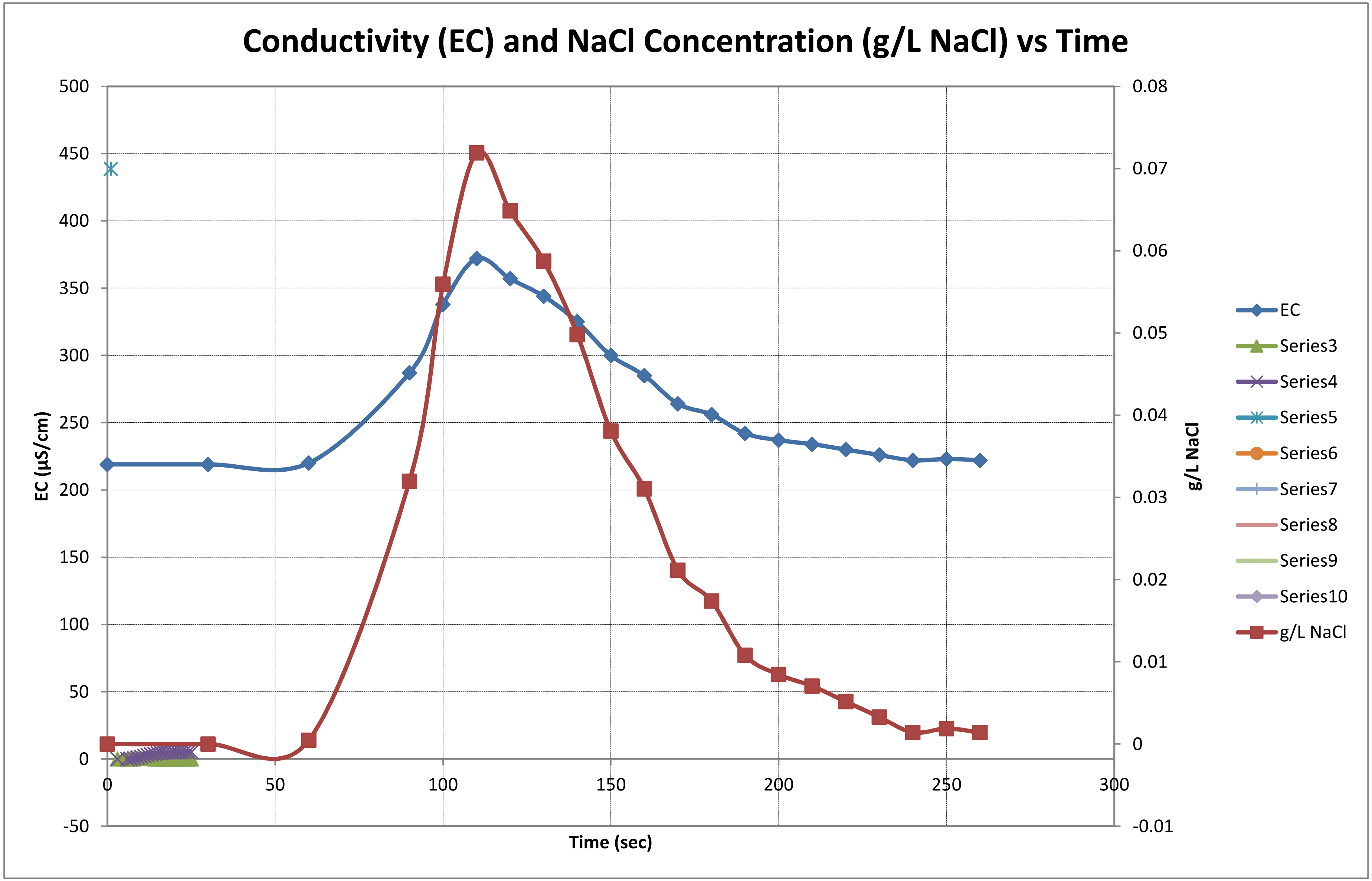
Sudden Injection Dilution Gauging			
River: Toreen East		**Location:** Upstream 1A	
Date: 18-1-13		**Time:** 9.59am	
Water Temp (C):		**Normalising Temp (C):**	
Weather Conditions: Light rain - heavy rain in the previous 12hours			
Tracer Mass (g NaCl):	2250	**Background EC (µS/cm):**	197
µS/cm EC to g/L NaCl conversion factor:			0.00047
Gauging Structure: Stream constriction			
Grid Reference: 529,552,682,501		**Stage:** 0.55m	
Discharge (L/s):		**404.50**	

Time (sec)	EC (µS/cm)	g/L NaCl	Area
0	197	0	
30	199	0.00094	0.0141
60	207	0.0047	0.0987
90	297	0.047	0.8742
100	370	0.08131	1.51575
110	372	0.08225	2.33355
120	349	0.07144	3.102
130	318	0.05687	3.74355
140	282	0.03995	4.22765
150	264	0.03149	4.58485
160	246	0.02303	4.85745
170	232	0.01645	5.05485
180	225	0.01316	5.2029
190	218	0.00987	5.31805
200	212	0.00705	5.40265
210	208	0.00517	5.46375
220	205	0.00376	5.5084
230	202	0.00235	5.53895
240	202	0.00235	5.56245


Conductivity (EC) and NaCl Concentration (g/L NaCl) vs Time
EC (µS/cm)
g/L NaCl
Time (sec)
0
50
100
150
200
250
300
350
400
450
0
0.01
0.02
0.03
0.04
0.05
0.06
0.07
0.08
0.09
0
50
100
150
200
250
300
EC
Series3
Series4
Series5
Series6
Series7
Series8
Series9
Series10
g/L NaCl

Sudden Injection Dilution Gauging			
River: Toreen east		**Location:** Upstream 1B	
Date: 28/05/2013		**Time:**	
Water Temp (C):		**Normalising Temp (C):**	
Weather Conditions:			
Tracer Mass (g NaCl):	2250	**Background EC (µS/cm):**	219
µS/cm EC to g/L NaCl conversion factor:			0.00047
Gauging Structure:			
Grid Reference:		**Stage:**	
Discharge (L/s):		**438.59**	

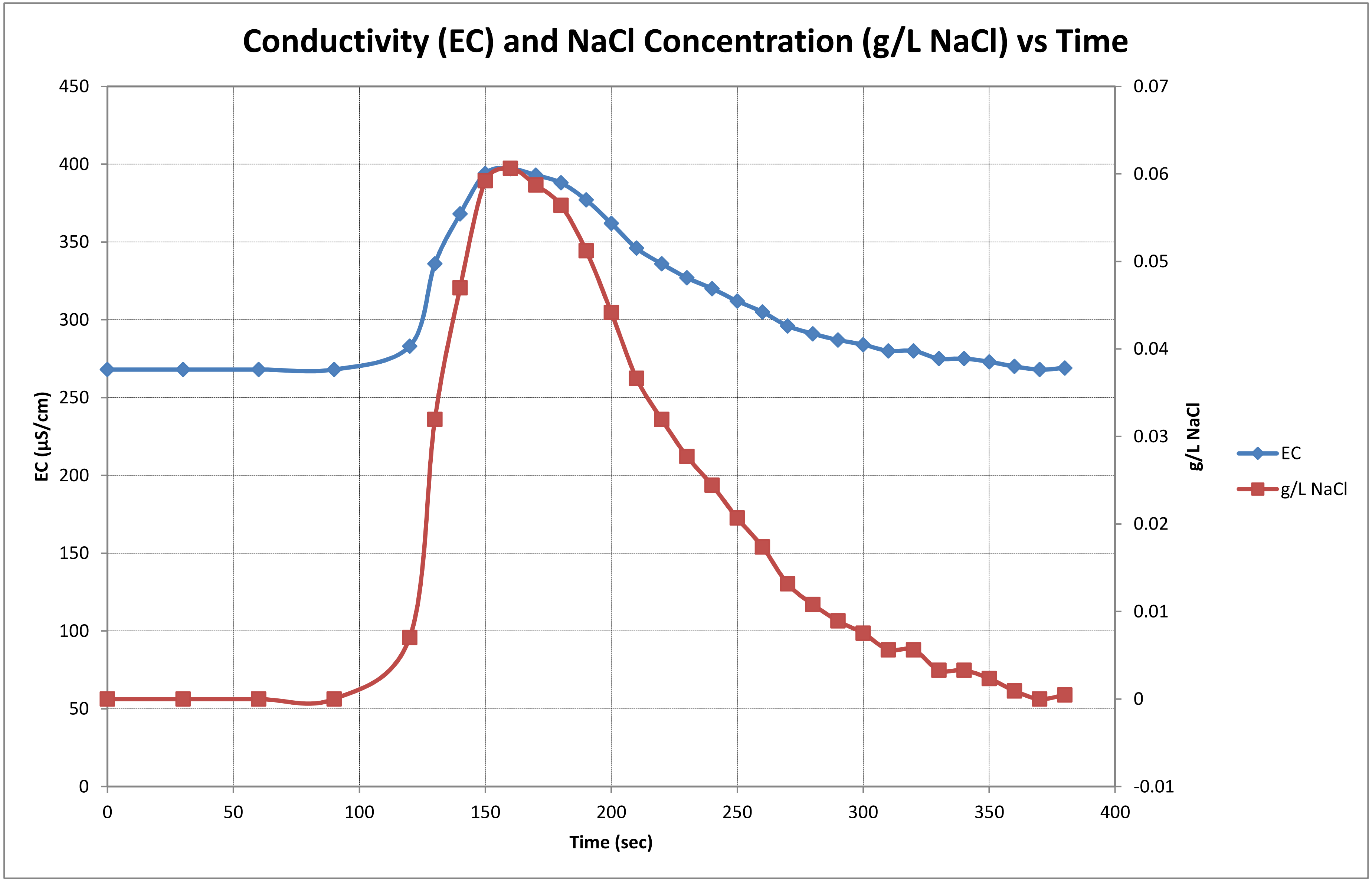
Time (sec)	EC (µS/cm)	g/L NaCl	Area
0	219	0	
30	219	0	0
60	220	0.00047	0.00705
90	287	0.03196	0.4935
100	338	0.05593	0.93295
110	372	0.07191	1.57215
120	357	0.06486	2.256
130	344	0.05875	2.87405
140	325	0.04982	3.4169
150	300	0.03807	3.85635
160	285	0.03102	4.2018
170	264	0.02115	4.46265
180	256	0.01739	4.65535
190	242	0.01081	4.79635
200	237	0.00846	4.8927
210	234	0.00705	4.97025
220	230	0.00517	5.03135
230	226	0.00329	5.07365
240	222	0.00141	5.09715
250	223	0.00188	5.1136
260	222	0.00141	5.13005



Conductivity (EC) and NaCl Concentration (g/L NaCl) vs Time
EC (µS/cm)
g/L NaCl
Time (sec)
500
450
400
350
300
250
200
150
100
50
0
-50
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
-0.01
0
50
100
150
200
250
300
EC
Series3
Series4
Series5
Series6
Series7
Series8
Series9
Series10
g/L NaCl

Sudden Injection Dilution Gauging			
River: Toreen East		**Location:** Upstream 2	
Date: 18-1-13		**Time:**	16:10
Water Temp (C):		**Normalising Temp (C):**	
Weather Conditions: Light rain - heavy rain in the previous 12hours			
Tracer Mass (g NaCl):	2250	**Background EC (µS/cm):**	268
µS/cm EC to g/L NaCl conversion factor:			0.00047
Gauging Structure: Stream constriction			
Grid Reference:	529,552,682,501	**Stage:** 0.35m	
Discharge (L/s):		**349.31**	

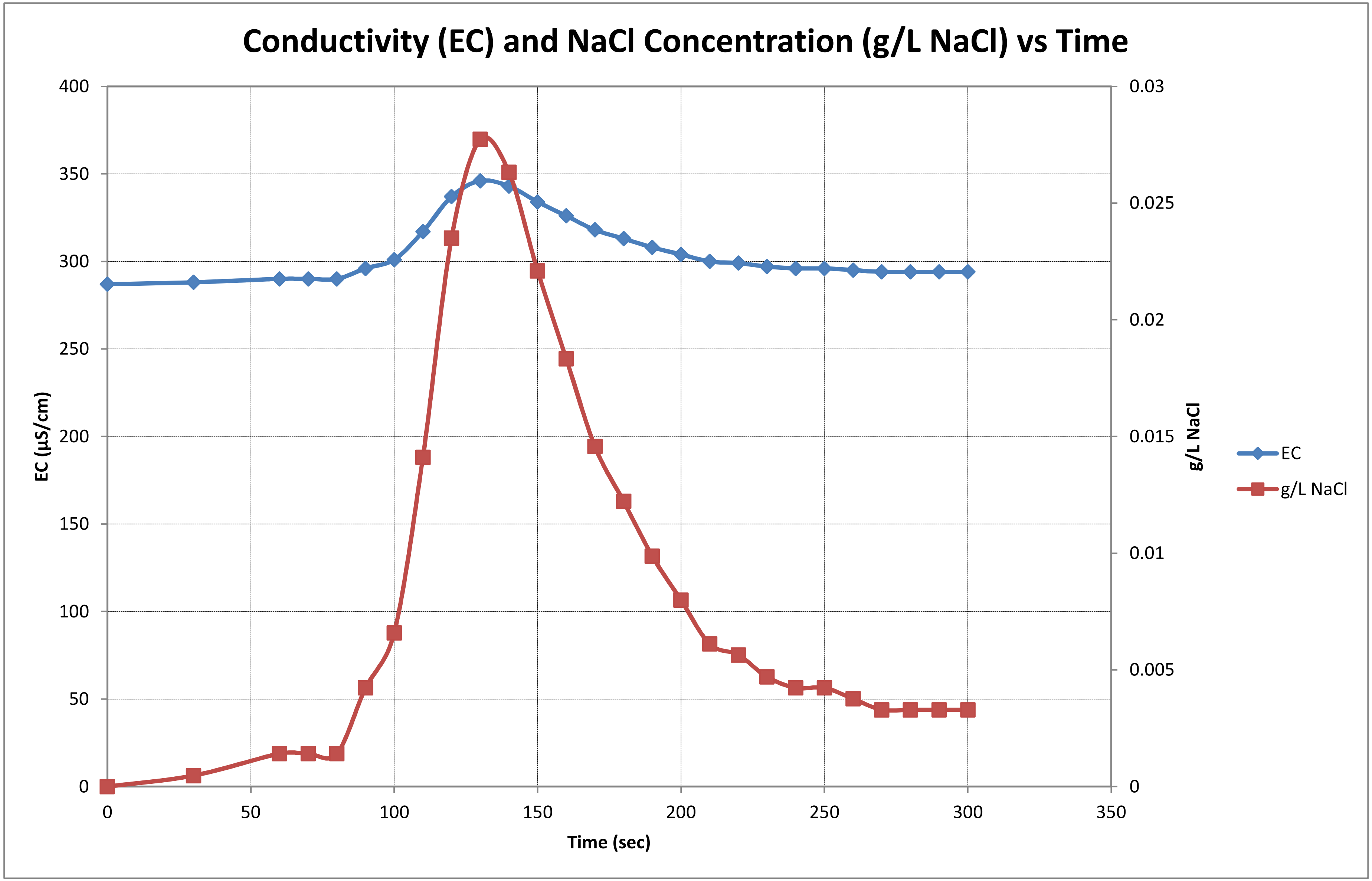
Time (sec)	EC (µS/cm)	g/L NaCl	Area
0	268	0	
30	268	0	0
60	268	0	0
90	268	0	0
120	283	0.00705	0.10575
130	336	0.03196	0.3008
140	368	0.047	0.6956
150	394	0.05922	1.2267
160	397	0.06063	1.82595
170	393	0.05875	2.42285
180	388	0.0564	2.9986
190	377	0.05123	3.53675
200	362	0.04418	4.0138
210	346	0.03666	4.418
220	336	0.03196	4.7611
230	327	0.02773	5.05955
240	320	0.02444	5.3204
250	312	0.02068	5.546
260	305	0.01739	5.73635
270	296	0.01316	5.8891
280	291	0.01081	6.00895
290	287	0.00893	6.10765
300	284	0.00752	6.1899
310	280	0.00564	6.2557
320	280	0.00564	6.3121
330	275	0.00329	6.35675
340	275	0.00329	6.38965
350	273	0.00235	6.41785
360	270	0.00094	6.4343
370	268	0	6.439
380	269	0.00047	6.44135



Conductivity (EC) and NaCl Concentration (g/L NaCl) vs Time
EC (µS/cm)
g/L NaCl
Time (sec)
EC
g/L NaCl
450
400
350
300
250
200
150
100
50
0
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
-0.01
0
50
100
150
200
250
300
350
400

Sudden Injection Dilution Gauging			
River: Toreen East		**Location:** Downstream 1	
Date: 18-1-13		**Time:** 11:30am	
Water Temp (C):		**Normalising Temp (C):**	
Weather Conditions: Light rain - heavy rain in the previous 12hours			
Tracer Mass (g NaCl):	2250	**Background EC (µS/cm):**	287
µS/cm EC to g/L NaCl conversion factor:			0.00047
Gauging Structure: Stream constriction			
Grid Reference:	529,552,682,501	**Stage:** 1.15m	
Discharge (L/s):		**958.41**	

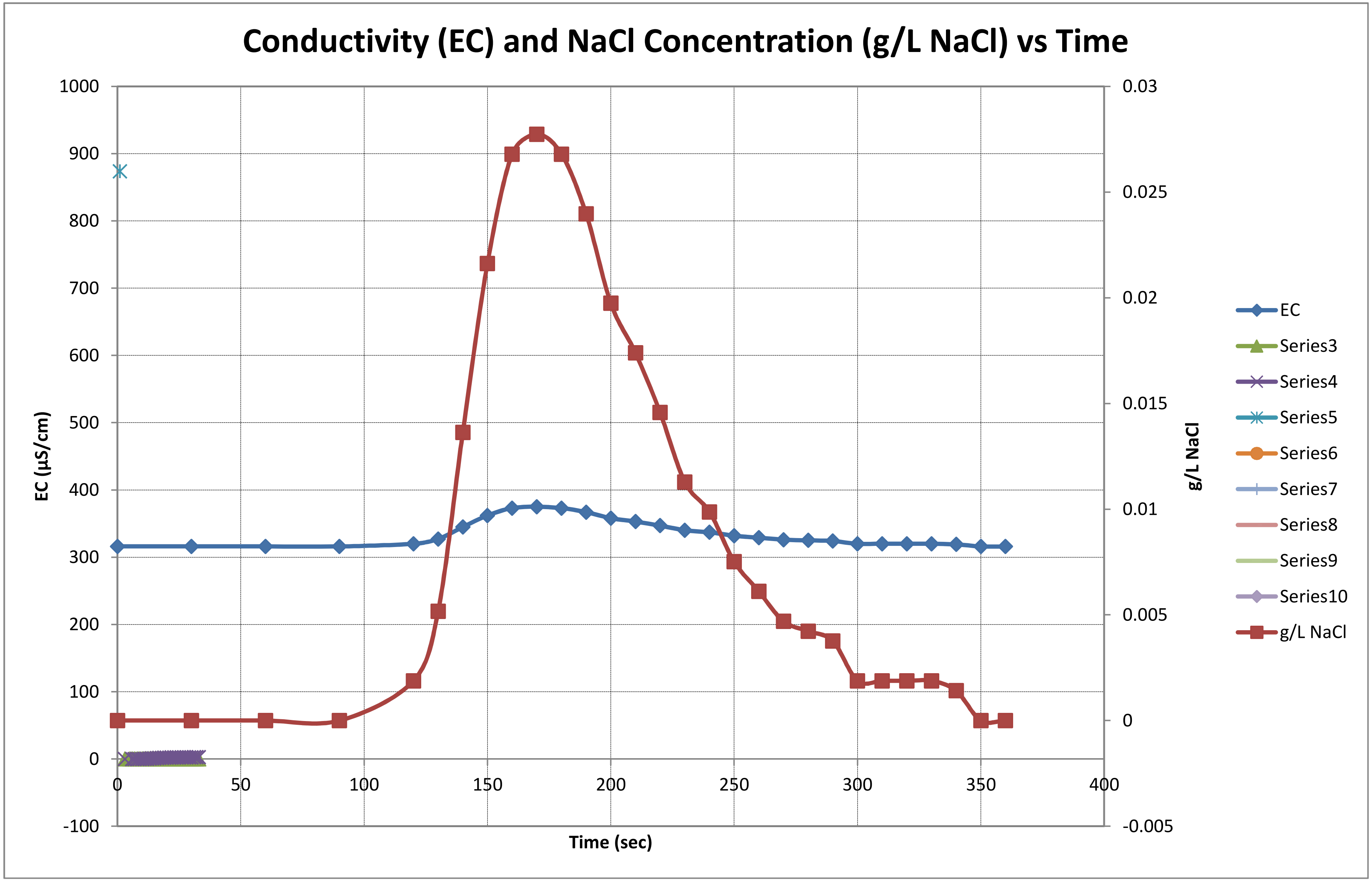
Time (sec)	EC (µS/cm)	g/L NaCl	Area
0	287	0	
30	288	0.00047	0.00705
60	290	0.00141	0.03525
70	290	0.00141	0.04935
80	290	0.00141	0.06345
90	296	0.00423	0.09165
100	301	0.00658	0.1457
110	317	0.0141	0.2491
120	337	0.0235	0.4371
130	346	0.02773	0.69325
140	343	0.02632	0.9635
150	334	0.02209	1.20555
160	326	0.01833	1.40765
170	318	0.01457	1.57215
180	313	0.01222	1.7061
190	308	0.00987	1.81655
200	304	0.00799	1.90585
210	300	0.00611	1.97635
220	299	0.00564	2.0351
230	297	0.0047	2.0868
240	296	0.00423	2.13145
250	296	0.00423	2.17375
260	295	0.00376	2.2137
270	294	0.00329	2.24895
280	294	0.00329	2.28185
290	294	0.00329	2.31475
300	294	0.00329	2.34765



Conductivity (EC) and NaCl Concentration (g/L NaCl) vs Time
400
350
300
250
200
150
100
50
0
0.03
0.025
0.02
0.015
0.01
0.005
0
0
50
100
150
200
250
300
350
EC (µS/cm)
g/L NaCl
Time (sec)
EC
g/L NaCl

Sudden Injection Dilution Gauging			
River: Toreen East		**Location:** Downstream 2	
Date: 18-1-13		**Time:**	15:40
Water Temp (C):		**Normalising Temp (C):**	
Weather Conditions: Light rain - heavy rain in the previous 12hours			
Tracer Mass (g NaCl):	2250	**Background EC (µS/cm):**	316
µS/cm EC to g/L NaCl conversion factor:			0.00047
Gauging Structure: Stream constriction			
Grid Reference:	529,552,682,501	**Stage:**	0.95
Discharge (L/s):		**873.58**	

Time (sec)	EC (µS/cm)	g/L NaCl	Area
0	316	0	
30	316	0	0
60	316	0	0
90	316	0	0
120	320	0.00188	0.0282
130	327	0.00517	0.06345
140	345	0.01363	0.15745
150	362	0.02162	0.3337
160	373	0.02679	0.57575
170	375	0.02773	0.84835
180	373	0.02679	1.12095
190	367	0.02397	1.37475
200	358	0.01974	1.5933
210	353	0.01739	1.77895
220	347	0.01457	1.93875
230	340	0.01128	2.068
240	337	0.00987	2.17375
250	332	0.00752	2.2607
260	329	0.00611	2.32885
270	326	0.0047	2.3829
280	325	0.00423	2.42755
290	324	0.00376	2.4675
300	320	0.00188	2.4957
310	320	0.00188	2.5145
320	320	0.00188	2.5333
330	320	0.00188	2.5521
340	319	0.00141	2.56855
350	316	0	2.5756
360	316	0	2.5756



Conductivity (EC) and NaCl Concentration (g/L NaCl) vs Time
EC (µS/cm)
g/L NaCl
Time (sec)
1000
900
800
700
600
500
400
300
200
100
0
-100
0.03
0.025
0.02
0.015
0.01
0.005
0
-0.005
0
50
100
150
200
250
300
350
400
EC
Series3
Series4
Series5
Series6
Series7
Series8
Series9
Series10
g/L NaCl