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Private wells in Ireland: A quantitative assessment of groundwater quality, consumer awareness, contamination susceptibility and human health risk

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A thesis submitted for the Degree of Doctor of Philosophy

to the University of Dublin, Trinity College

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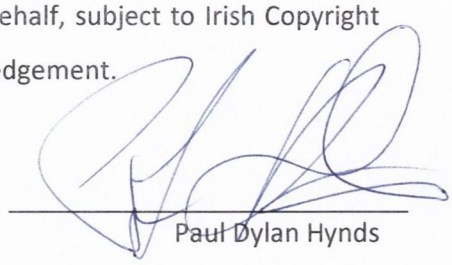
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Paul Dylan Hynds

SUMMARY

A doctoral research study, funded by the Environmental Protection Agency, was undertaken to examine and assess the health risks associated with private groundwater sources in Ireland. Furthermore, the potential contamination pathways and mechanisms, including consumer awareness, were investigated, quantified and modelled, in order to provide risk management and communication tools in the form of a source susceptibility model and guidance for private groundwater consumers in Ireland. Four primary study areas were selected using decision-based criteria for data acquisition and subsequent analyses, with a fifth area used for model verification. Within the four main study areas, 211 private groundwater sources, of differing design and construction characteristics were surveyed using a developed site assessment process. Furthermore, groundwater samples from each assessed source were analysed for a comprehensive suite of physicochemical, chemical and microbial parameters, with a number of these sources tested on a monthly basis for up to 21-months. A groundwater awareness questionnaire was developed and completed with 545 private groundwater users.

The presence of thermo-tolerant *E. coli* was relatively widespread, with 29.4% of "one-off" wells testing positive. Proportionately, hand-dug wells were shown to be significantly associated with *E. coli* presence, and therefore are likely to pose a higher human burden. Schwartz's Bayesian Inference Criterion (BIC) analyses further reflected this, with hand-dug wells (Cluster 3) found to be significantly correlated with high levels of contamination, while boreholes located in limestone/sandstone areas (Cluster 1) and boreholes located in Numurian/granite areas (Cluster 3) were associated with moderate and low levels of *E. coli* presence, respectively. Overall, the level of bacterial contamination encountered in *High/Extreme* vulnerability areas was lower than expected, while conversely, a higher than expected level of bacterial contamination was encountered in the *Low* vulnerability study area, which was also found during regular monthly sampling. This finding suggests that source design and/or construction characteristics are a primary risk factor with regard pathogenic contamination. The majority (93%) of monitored wells (14/15) showed evidence of *E. coli* contamination at least once during the monitoring period. However, only 40% of monitored sources showed evidence of *E. coli* contamination upon initial "one-off" sampling, therefore, it would appear that for microbial analysis, seasonal sampling was revealing, with a high level of variability and seasonality presented. Typically, *E. coli* presence and magnitude (where present), was increased during mid to late summer and early to mid winter at all monitored sources, due to increased precipitation and seasonal anthropogenic activities. Approximately 5% of samples tested positive for *Cryptosporidium* spp., which is generally consistent with previous international studies. No samples tested positive for *Giardia* spp. Four "cumulative precipitation periods" prior to sampling (24-hour, 48-hour, 120-hour, 30-day) and their respective association with bacterial contamination were analysed. It was found that higher rainfall during the 120-hour and 30-day periods prior to sampling had occurred where *E. coli* were present, particularly among hand-dug wells. A high level of association was found in the Low vulnerability area ($r = 0.785$, $p = 0.021$) between *E. coli* presence and the 120-hour precipitation period prior to sampling, suggesting that rapid ingress of direct runoff or shallow groundwater infiltration is a significant contamination mechanism in *Low* vulnerability areas.

Analysis of the developed awareness surveys found that over 70% of respondents acknowledged the involvement of a water diviner at the well design stage, however, only 16% and 1.2% of

respondents acknowledged the involvement of local authorities and a professional hydrogeologist, respectively, which presents a concern with respect to adequate source protection. Furthermore, a significantly low level of awareness was noted with regard to waterproof sealing of private wells, with borehole users in particular, displayed a very low awareness of appropriate sealing of their source (i.e. 81.5% had no awareness of sealing or grouting of their well). Water treatment, regular water testing and wellhead/on-site maintenance are three "consumer controls" that all private well owners may employ to minimise "consumer (secondary) susceptibility", however, it was found that 68%, 40% and 65% of respondents, respectively, acknowledged that these controls were not in place. A scoring methodology was developed and found a pattern of higher source awareness among hand-dug well users, with a similar pattern found with respect to contamination awareness. Overall, a median contamination awareness of 60% was found, which is considered low. No significant relationship was found to exist between overall awareness and *E. coli* presence or magnitude (where present) in the respondents well, and therefore no correlation was found between respondent awareness and "source susceptibility" i.e. the probability of a well becoming contaminated. However, significant associations were found between the level of consumer awareness and the level of "consumer control" in place, with those respondents exhibiting higher levels of awareness found to have higher levels of consumer control in place and therefore a lower "consumer susceptibility".

Developed multivariate susceptibility models predict that point sources of contamination, particularly septic tank systems and point agriculture, are the most significant hazard sources. Likewise, source design and construction characteristics are significant within developed models, with liner clearance and wellhead cover being the design parameters of most importance. Antecedent precipitation was also identified as a significant predictor within a number of models, with 120-hour and 30-day rainfall the most significant rainfall periods, indicating that both rapid ingress/infiltration at or adjacent to the wellhead and gradual recharge to the aquifer are both significant modes of contaminant transport. However, within the total generalized model, only 120-hour precipitation was a significant predictor, therefore direct ingress of surface runoff and rapid shallow groundwater infiltration are adjudged to be the most significant modes of contaminant transport, particularly in hand-dug wells and poorly constructed boreholes. The *Low* vulnerability model included 48-hour precipitation as a predictor variable, indicating rapid ingress at the wellhead (surface runoff) as the primary contamination pathway. The significance of on-site and wellhead radius conditions as risk factors in a number of models (*High/Extreme* vulnerability; Cluster 2; Generalised model), suggests that "maintenance issues" are associated with source susceptibility and therefore, by extension, consumer control and consumer awareness is involved in source susceptibility. All individual study area models had overall prediction accuracies of approximately 90%, while the total generalised model providing an "effective fit" in excess of 80% in relation to the verification set. Sensitivity analyses found that the effect of septic tank setback distance from the wellhead was approximately linear, as was the effect of precipitation. Sources located in limestone areas and/or in areas with <3 m of overlying subsoils were more susceptible to bacterial ingress, as were sources with uncovered wellheads, sources with no liner clearance, and sources with categorically poor wellhead radius ground conditions.

A stochastic Monte Carlo simulation model was developed to evaluate the annual risk of infection from microbial contamination (*E. coli* and *Cryptosporidium* spp.) of private groundwater sources in

Ireland, in addition to three previously used deterministic models being employed. All three deterministic models and the stochastic risk assessment model predict higher crude incidence rates of waterborne illness attributable to *E. coli* among users of hand-dug wells than borehole users, while the same is true with respect to *Cryptosporidium* spp., although to a lesser extent. Sensitivity analysis of the stochastic model found that the *E. coli* loading rate (i.e. amalgamation of E. coli presence and magnitude) is the input with the greatest overall contribution on model outputs and therefore, improved source protection (i.e. location, design, construction and maintenance), particularly in the case of existing wells, is necessary in order to minimize the overall health burden attributable to private groundwater sources in Ireland. In the case of modelled cryptosporidial illness, sensitivity analyses found that the dose-response rate associated with *Cryptosporidium* spp. is the highest overall contributor to model outputs, suggesting that vulnerable sub-populations in particular are likely to contribute significantly to the overall health burden, and therefore "consumer protection" (i.e. boil water notices, regular testing, increased levels of treatment, improved diagnostics) should be a priority. Overall outputs by the stochastic model predict that approximately 0.7% and 0.18% of private groundwater users in Ireland will contract a symptomatic illness attributable to waterborne *E. coli* and *Cryptosporidium* spp. per year, respectively.

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Chapter 1: Introduction

1.1 Background

In recent years, concerns over groundwater quality in Ireland have been highlighted due to the risks posed by anthropogenic activities with contamination arising from both diffuse (typically agricultural) and point sources, particularly farmyards (manure and silage storage), and on-site wastewater treatment systems. As a case in point, recent statistics (EPA, 2008) have revealed that 36% of private group water schemes (most of which are supplied by a groundwater source) showed evidence of faecal contamination, with a higher proportion expected among unregulated individual supplies. Furthermore, Daly & Craig (2009) have suggested that there are areas in Ireland which may experience intermittent faecal contamination in up to 70% of local private sources on an annual basis. On-site treatment and disposal of domestic sewage is the sole method of wastewater renovation for over 400,000 Irish dwellings, equating to over one third of the population (DELG/EPA/GSI, 2000). Additionally, the land area of Ireland is 6.9million hectares, of which 4.4 million hectares is used for agriculture or about 64% of total land area (CSO, 2007). The majority of this agricultural land (80%) is given over to grazing and animal production.

The main groundwater contaminants of concern in terms of human health are microbiological pathogens, due to low dose-response rates, the proximity and profusion of microbial hazard sources, and not least, the severe potential human health effects which may occur upon ingestion. Microbial contaminants derived from human faecal matter in particular represent a particular threat due to their inherently human pathogenicity/virulence. The microbial contaminants of concern include:

- Bacteria, such as verotoxin-producing *Escherichia coli (VTEC)*;
- Protozoa, including *Cryptosporidium* (*parvum* and *hominus*) and *Giardia lamblia*;
- Viruses, notably rotavirus and adenovirus

There is a wide range of potential illnesses that may result from waterborne microbial infection; the most commonly diagnosed being generalised acute gastrointestinal illness (AGI). Symptoms include nausea, vomiting and diarrhoea. Typically, microbial-related illnesses are of short duration, relatively minor and self limiting, however, in some populations, severe illnesses or fatalities may occur, particularly among vulnerable sub-populations such as the elderly, the very young and the immuno-compromised. HPSC (2008b) have recently stated that the rise in VTEC cases in Ireland may be associated with consumption from private wells.

Groundwater resources currently supply water to approximately 26% of the Irish population, with 17% of the population, in excess of 750,000 people, using private groundwater supplies from two main source types: small private supplies (SPS) - private, unregulated groundwater supplies typically serving individual households - and private group water schemes (PrGWS), which are committee or shareholder run schemes serving fewer than 50 people, or supplying <10m^3/day (Brady & Gray, 2010). Individual domestic supplies are currently unregulated, and it is likely they are also largely untreated and unmonitored, therefore, the risk to public health from contaminated private supplies is a very real concern and emphasises the need to protect this resource from contamination. To date

however, the risks posed have not been accurately established or quantified, in order that appropriate risk management and/or communication strategies may be put in place.

Moreover, the health risks associated with private drinking water supplies depend upon a variety of factors, including; the contaminant type and loading rate; the contamination source; the type, location, design and construction of the source; the presence or absence of water treatment; previous maintenance works; and, perhaps, the awareness (or lack thereof) of these aforementioned risk factors amongst the owners and users of the source. To date however, the effect of these risk factors on the overall private source contamination have not been investigated in Ireland, nor has the level of awareness of groundwater issues amongst private groundwater consumers.

Progress has been made in recent years in improving the standards of drinking water in Ireland, under the European Communities (Drinking Water) (No.2) Regulations 2007, the Water Framework Directive Monitoring Programme to meet the EU Water Framework Directive (2000/60/EC), National Regulations implementing the Water Framework Directive (S.I. No. 722 of 2003) and National Regulations implementing the Nitrates Directive (S.I. No. 788 of 2005). Important developments have included the National Rural Monitoring Committee Action Plan for rural drinking water for 2003-2006 (NRWMC, 2003), IGI Guidelines for drilling wells for private water supplies (IGI, 2007) and A Handbook on the Implementation of the Regulations for Water Services Authorities for Private Water Supplies (EPA, 2010a).

1.2 Project Aims and Objectives

The overall aim of the research is to investigate and assess the health risks associated with private rural groundwater sources in Ireland and in so doing provide risk management and communication tools in the form of a source susceptibility model and guidance for private groundwater consumers in Ireland. In order to achieve this overall objective, the project was delineated into four primary objectives, with the successful completion of these objectives expected to aid in the reduction of adverse health impacts associated with groundwater contamination in the future. These four primary objectives have been defined as follows:

1. To carry out a survey to determine the awareness amongst private well owners of the potential contamination risk factors and associated health impacts from contaminated groundwater supplies, and investigate the role of user awareness (or lack thereof) as a source susceptibility risk factor.

2. To investigate the link between the contamination risk, and hence health risk, and the various factors associated with private groundwater supplies, including geological/hydrogeological factors (groundwater vulnerability, bedrock and superficial geology, aquifer importance), well location with respect to potential contaminant sources, well design, construction and maintenance.

3. To undertake a quantified assessment of the magnitude of risk to health from microbial contamination of private well supplies in Ireland.

4. To provide guidance to the owners and users of new and existing private domestic wells on the location, design, construction and operation of these sources.

1.3 Thesis Outline

Chapters 2, 3, 4 and 5 provide an extensive literature review focusing on the key areas of research, as follows: (i) the hydrogeology of Ireland and current terminology; (ii) private groundwater source location, design, construction and maintenance, including both hand-dug and bored wells; (iii) groundwater contamination, including biological and chemical contaminants, their respective sources and previous national and international instances of groundwater contamination, with particular emphasis placed on contamination of private groundwater wells, and (iv) human health risk assessment, including both deterministic and probabilistic methods of assessment.

Chapter 6 describes the overall study methodology including the selection of four primary study areas, site assessment development and awareness survey development and completion, as well as the groundwater sample collection and laboratory analysis protocols.

Chapter 7 presents the fieldwork results from "one-off" sampling and analysis of 211 private groundwater sources, in addition to temporal sampling of 15 such sources over a 21 month period. Bivariate analyses of pollutant loadings and associated source data including source location, hazard location and source design and construction are also presented, providing a basis upon which the "source susceptibility" model is constructed.

Chapter 8 presents statistical analysis of 545 completed groundwater awareness surveys. An awareness scoring methodology has been developed and utilised, with results used to investigate the level of interaction and/or correlation between private groundwater consumer awareness and the instance and rate of microbial contamination in private groundwater sources. Again, these results are used to inform the development of both the "source susceptibility" model and guideline framework.

In Chapter 9, a "source susceptibility" predictive model is developed and tested. This chapter outlines the analytical methods used, the input parameters and associated data and model outputs. In addition, a test dataset is presented and used to refine the model, while the use of sensitivity analysis allows for the quantification of model input effects.

Chapter 10 presents the results of a quantified assessment of the magnitude of risk to public health from bacterial and protozoan contamination of private well supplies in Ireland. Both deterministic and probabilistic methods are used and compared, as well as sensitivity analysis being carried out in order that the primary risk factors to private groundwater consumers are identified and quantified.

As an overall project output, a framework for future guidelines for the owners of new and existing private domestic wells in Ireland on the location, construction and operation of private groundwater sources has been developed and is presented in Chapter 11 (Research Objective 4). Finally, the overall project conclusions are presented in Chapter 12, including recommendations for further research.

Chapter 2: Groundwater in Ireland

2.1 Introduction

For thousands of years, wells and natural springs have supplied clean, abundant groundwater to human communities throughout the world. Groundwater is the source of drinking water for many people around the world, particularly in rural areas. Groundwater comprises approximately 30.1% of global freshwater, accounting for approximately 135,000 km^3 (Shiklomanov, 1993).Therefore, as shown in *Table 2.1*, groundwater constitutes the largest global source of readily available freshwater, accounting for 97% of the world's supply of unfrozen freshwater, although estimates vary and range from a global resource of 7M – 60M km^2 (Konokow & Kendy, 2005). Globally, it supplies more than 50% of drinking water, 40% of irrigation water and 25% of water used for industrial purposes (Chernicoff *et al.*, 2002).

The following chapter provides a review of the (bedrock and superficial) geology and aquifers of Ireland, with particular emphasis placed on groundwater occurrence. The principles of aquifer importance and groundwater vulnerability are outlined.

Table 2.1 Global Water Reserves (Shiklomanov, 1993)

Source	Distribution Area (10^3 km^3)	Total Water (%)	Freshwater (%)
World Ocean	361,300	96.5	-
Groundwater	134,800	0.76	30.1
Glaciers*	16,227	1.74	68.7
Ground Ice**	21,000	0.022	0.86
Lakes (Freshwater)	1,236	0.007	0.026
Lakes (Saline)	822	0.006	-
Swamp Water	2,682	0.0008	0.03
River flows	148,800	0.0002	0.006
Atmospheric Water	510,000	0.001	0.04
Soil Water		0.001	0.05

**Incl permanent snow cover*

***Incl permafrost*

2.2 The bedrock geology of Ireland

The Republic of Ireland has an area of approximately 70,273 km^2, consisting of a central lowland lying between 50-100 m above sea level, surrounded by coastal highlands, with peaks ranging from 500-1000 m (Commission of the European Communities, 1982).Typically, as the majority of groundwater storage and transmission in Ireland occurs in bedrock cracks and fissures (i.e. secondary porosity), the bedrock type is paramount with regard to the overall groundwater-bearing potential of Irish aquifers. For example, in Ireland limestone rocks typically have greater groundwater potential than granite rocks, meaning that the water-bearing fractures, joints and

cracks in limestone rock are usually wider and have higher levels of interconnectivity than those in granites (IGI, 2007).

As shown in *Figure 2.1*, Ireland's bedrock geology is comprised primarily of Precambrian schists and quartzites; Lower Palaeozoic metasediments, volcanic rocks and granites; Devonian sandstone; Carboniferous limestone and shales (European Environment Agency, 1999; Holland, 2001; Holland & Sanders, 2009). A geological time scale is included in the Appendix (A1).The oldest geological formations are the Precambrian (4,600-545 Ma) schists and quartzites of Donegal, Sligo and Mayo in the west and northwest, with some small areas in Co Wexford in the southeast (Commission of the European Communities, 1982). Three subdivisions of the Precambrian exist in Ireland, with the Dalradian Supergroup and pre-Grampian basement units in the west and northwest of the country, and the pre-Monian basement in the southeast (Holland, 2001; Holland & Sanders, 2009).

The Lower Palaeozoic (550-390 Ma) rocks represent a complex geological history and comprise a large range of rock types including greywackes (turbidites), volcaniclastic sediments, lavas, shales, mudstones and cherts. (Graham, 2001; Holland, 2001). The Lower Palaeozoic geological formations of most significance in Ireland belong to the Ordovician (550-460 Ma) and Silurian (460-390 Ma) periods (GSI, 2009). The Ordovician rocks can be subdivided into the Ordovician Metasediments and Ordovician volcanics (GSI, 2009). The rocks within the Ordovician Metasediment group are widely distributed, and have a total areal extent of approximately 4,200 km^2, concentrated in the southeast, the northeast and southwest Co. Mayo. The Ordovician volcanics cover an area of approximately 850 km^2, extending from south Co. Wicklow, throughout Co. Wexford and some of Co. Waterford. Silurian rocks are widely distributed throughout Ireland, with a total extent of 4,216 km^2, or approximately 1.6% of the country (GSI, 2009). There are three main areas of occurrence, namely; the Longford-Down Inlier; the Kilcullen Group in Co. Kildare; and various locations around Co Tipperary (GSI, 2009).

The Devonian (417-354 Ma) strata are of continental non-marine clastics, with the most frequent Devonian formation in Ireland being Old Red Sandstones (Commission of the European Communities, 1982; Graham, 2001), comprising red, purple and green conglomerates and Kiltorcan-type Sandstones (Holland & Sanders, 2009). Upper Devonian-Lower Carboniferous marine clastic formations are widespread in the midlands and south midlands and as shown in *Figure 2.1*, with the aforementioned non-marine facies widely distributed in the south of the country, with significant expanses found in Kerry, Cork, Waterford and Tipperary. There are various other outcrops of Old Red Sandstone including those in Clew Bay/Lough Conn in Co Mayo and The Curlew mountains in Co Roscommon (Simon, 1984; Graham, 2001; Holland & Sanders, 2009; GSI, 2009).

As outlined by Commission of the European Communities (1982) and GSI (2009), the Carboniferous period (354-290 Ma) includes the most important formations in Ireland, with approximately 50% of the country underlain by Carboniferous Limestone (Sevastopulo & Wyse-Jackson, 2001), typically of the Dinantian (Lower Carboniferous) series (360-326 Ma) (*Table 2.2*). The Carboniferous limestones are normally hard and grey to black in colour, and are found in almost every part of Ireland (except Co. Antrim and Co. Wicklow), as shown in *Figure 2.2*, and are particularly widespread in the midlands (GSI, 2009). In addition to limestones, Carboniferous dolomites, shales and sandstone also occur (Commission of the European Communities, 1982). As set out by Sevastopulo & Wyse-Jackson

(2001), Carboniferous formations are of great economic importance in Ireland, particularly with reference to quarrying (aggregates, industrial minerals, and base metal deposits), agriculture and water supply. Chemical solution of the limestones have taken place to varying degrees, particularly clean bedded limestones, resulting in widespread karstification in the west and northwest of the country.

As shown in *Figure 2.1*, three major igneous intrusive areas occur, comprising primarily granites, in the east (Dublin/Wicklow/Wexford/Carlow), west (Galway) and northwest (Donegal).

Table 2.2 Limestone types, extent and locations in Ireland (Sevastopulo & Wyse Jackson, 2001; GSI, 2009)

Limestone Type	Extent (km^2)	% of National Area	Locations
Dinantian (Early) Limestones ("Lower Limestone Shales")	1,300	1.9	Downpatrick formation, Co Mayo, narrow bands across midlands
Dinantian Lower Impure ("Ballysteen/ABL")	3,700	5.4	Widely dispersed , concentrated in midlands
Dinantian Pure Unbedded ("Waulsortian)	n/a	n/a	Midlands, southern valleys, Dublin and Shannon basins
Dinantian Dolomitised	308	0.4	Southwest Dublin, Port Laoise, several areas in Co Tipperary and Co Waterford
Dinantian Upper Impure ("Calp")	n/a	n/a	Much of Cos. Dublin, Meath, Kildare, Westmeath, Offaly, Galway; thin bands also found in Tipperary, Kilkenny and Laois.
Dinantian Pure Bedded ("Burren")	11,600	16.7	Large areas of the west, northwest, midlands and southwest

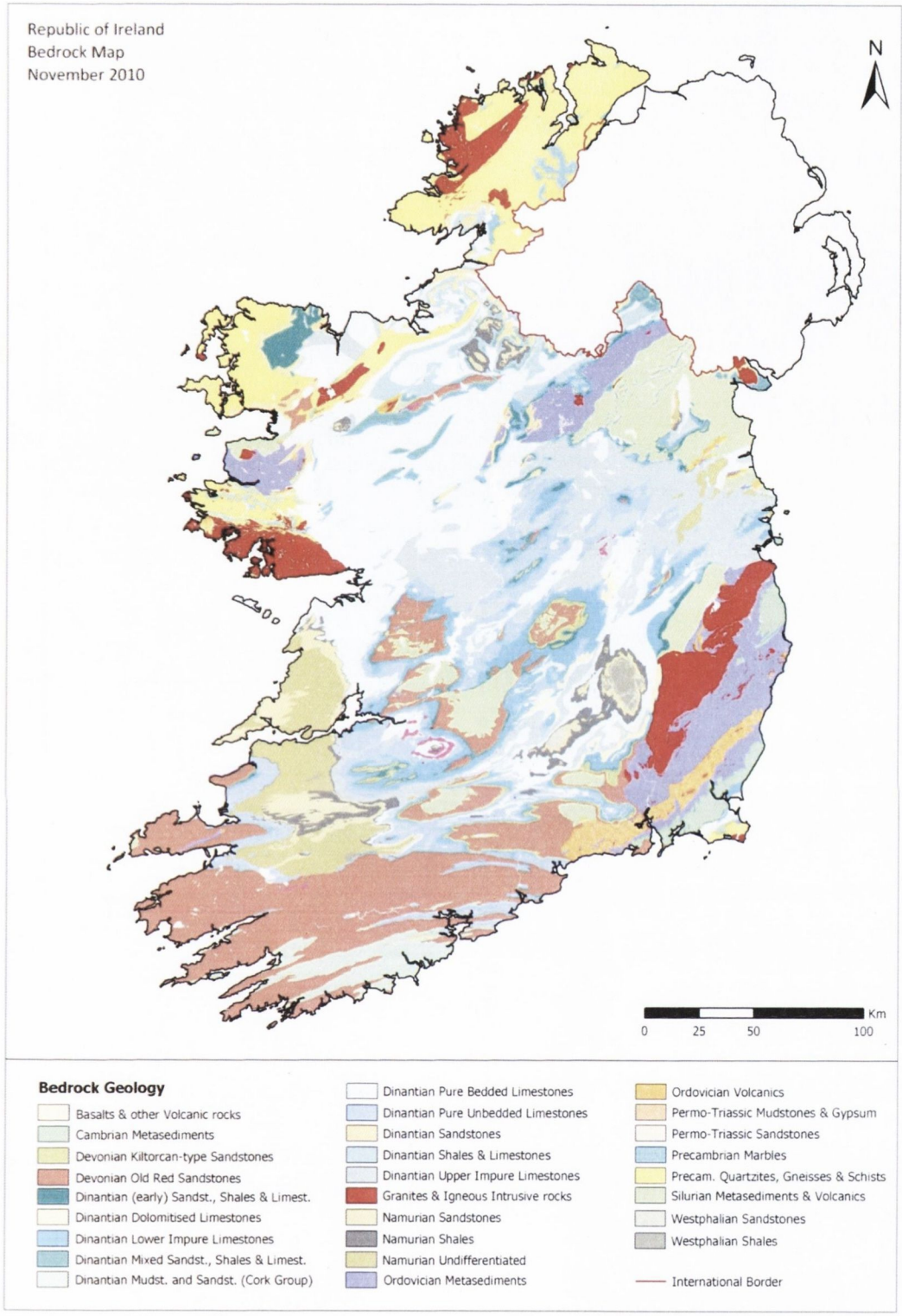


Figure 2.1 Bedrock Geology Map of Republic of Ireland (GSI, 2011)

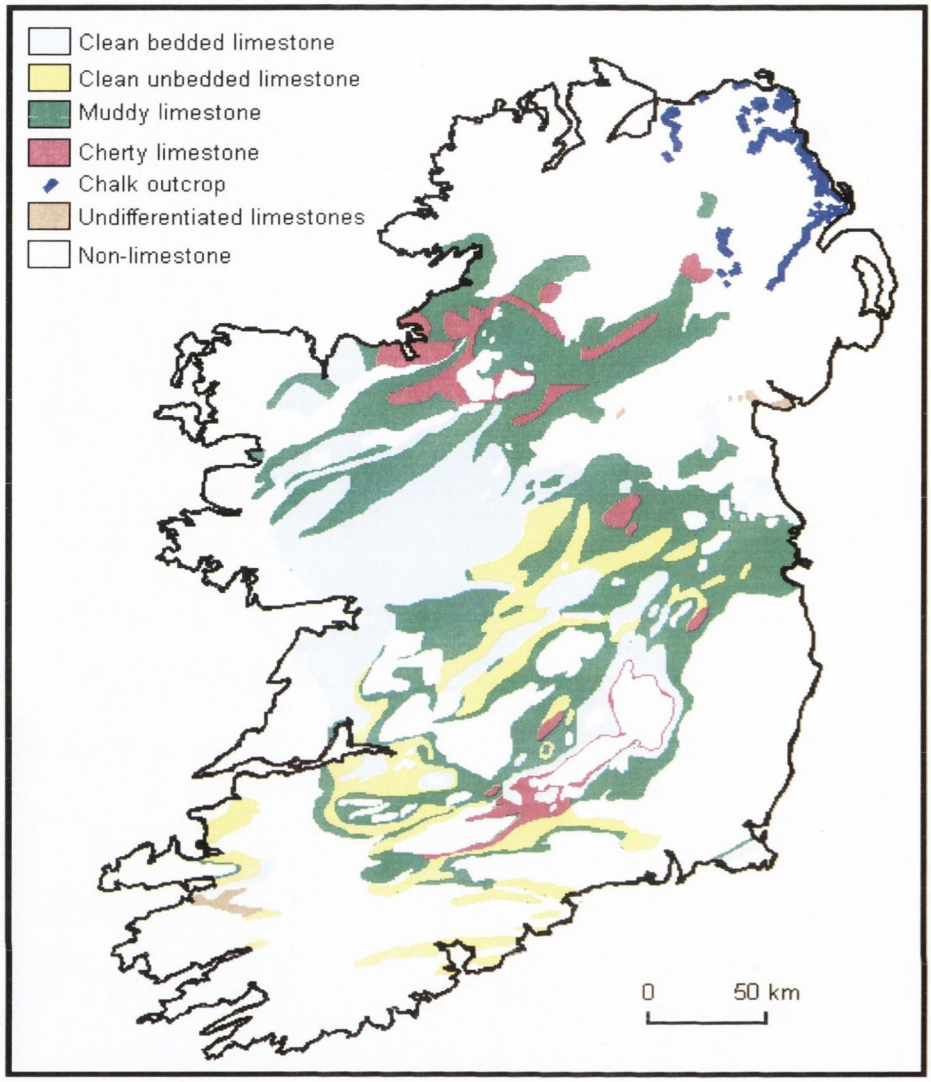


Figure 2.2 Spatial extent and types of limestone in Ireland (GSI, 2011)

2.3 The Quaternary Geology of Ireland

The Quaternary Period is the most recent of the three periods of the Cenozoic era, following the Neogene Period, spanning 2.58 Ma to the present. The Quaternary includes two geologic epochs: the Pleistocene and the Holocene. Ireland's Quaternary (or superficial) geology is dominated by sediments deposited under glacial, periglacial and temperate conditions (Holland & Sanders, 2009), with these glacial deposits being widespread and variable (Coxon, 2001). A sub-soil map of the Dublin area is presented in *Figure 2.3.*

In Ireland, the principal aquifers occur in fractured bedrock and in sand and gravel deposits. Typically, limited attenuation of contaminants occurs in the bedrock as flow is primarily via fissures, therefore, the subsoil acts as a filter and protective layer for the underlying aquifer. Consequently, subsoil type and thickness are the most important natural features influencing groundwater vulnerability (Section 2.8) and groundwater contamination in Ireland (Swartz *et al.*, 2003). Subsoils may be classified as the 'loose' lithified sediments present between topsoil (Appendix A2) and

bedrock. The two most important and common subsoil types in Ireland are glacial deposits (i.e. tills) and sand and gravel deposits.

Most Irish subsoils are derived from glacial drift and are relatively immature (O'Luanaigh, 2008). The Quaternary epoch started at a time of global cooling in northern Europe, marking the onset of what is commonly referred to as the 'Ice Age'. The movement of ice sheets determined the topography of the Irish landscape, providing upwards of 90% of the parent material covering much of the bedrock surface with up to 50 m of tills and gravels. As set out by Daly (1985), there are five main Irish subsoil types, as follows:

- Tills
- Glaciofluvial sands and gravels
- Glaciolacustrine deposits
- Alluvium
- Peat

Tills

Also known as 'boulder clay', these are the most common and widespread Quaternary deposits. They are often tightly packed, unsorted, unbedded, with various particle and clast sizes and types, ranging from angular to subangular.

Glaciofluvial sands and gravels

These are the Quaternary deposits of highest permeability (due to intragranular permeability), and therefore offer relatively low contaminant attenuation. Where these deposits are of sufficient thickness, they are considered aquifers in their own right (i.e. The Curragh, Co Kildare). Where they are not sufficiently extensive, or saturated, they are still considered important as they allow a high proportion of recharge water to enter underlying aquifers (Daly, 1985).

Glaciolacustrine deposits

These consist of sorted gravels, sand, silt and clay and are typically found in wide, flat plains, or in small depressions.

Alluvium

These are a product of river flow and flooding and are usually composed of sands and/or silts.

Peat

The onset of the warmer postglacial period (approximately 14,000 years ago) resulted in widespread disintegration of ice sheets and consequently, the initiation of peat development, particularly in lake basins. In Ireland, peats typically overlie poorly drained glaciolacustrine silts and clays.

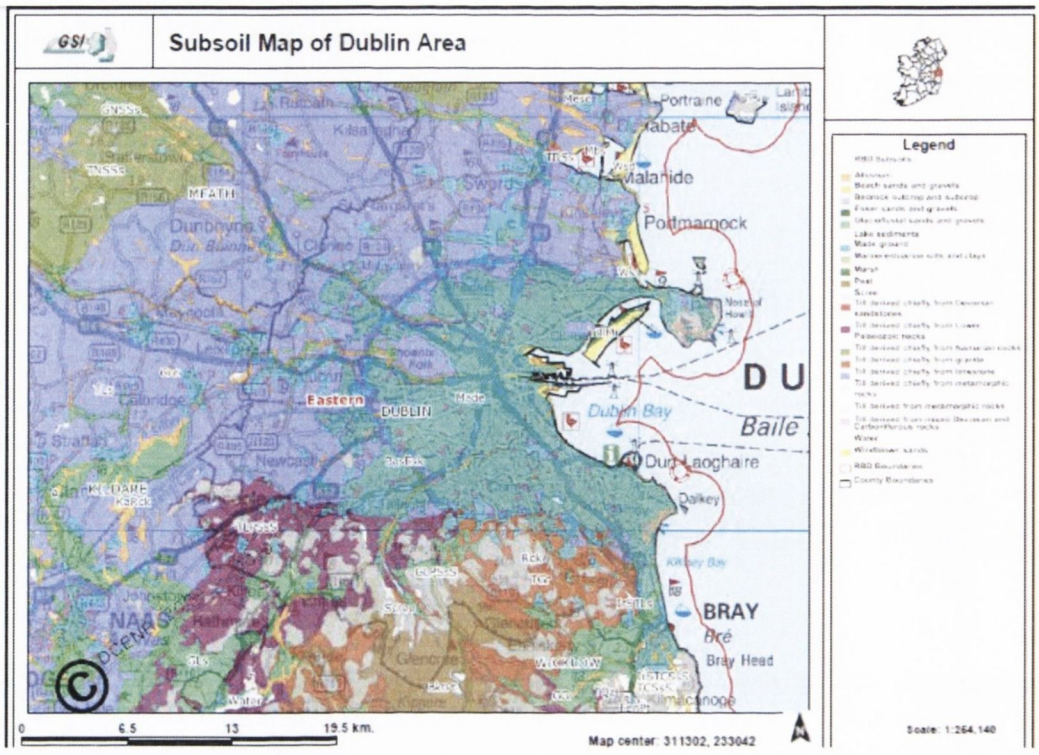


Figure 2.3 Example of a subsoil map of the Dublin area with the urban area chiefly underlain by made ground, surrounded by limestone tills to the north and west (Meath, Kildare) and granite tills and blanket peats to the south (Wicklow) (GSI, 2011)

Perhaps the most characteristic Quaternary features found in Ireland, apart from raised bogs (peats), are eskers (the international name for these features comes from the Irish name, eiscir) formed throughout the country, from rivers flowing beneath the ice, leaving a long narrow ridge of sand and gravel. Around 10% of the country is covered by eskers, and even more by associated outwash deposits of sand and gravel where the rivers came out from under the ice (Smith & Wyse-Jackson, 2009).

A detailed overview of the Tertiary and Quaternary periods (Cenozoic) may be found in Coxon, 2001.

2.4 Subsurface hydrology and aquifers

Subsurface or groundwater hydrology is the study of the occurrence, distribution and movement of water below the surface of the earth, with groundwater commonly understood to mean water occupying all the voids within a geological layer or stratum, including rock and subsoil layers (Todd & Mays, 2005). In the context of groundwater abstraction for human use, which is the focus of this study, it is the water within the saturated zone which is of immediate interest (*Figures 2.4 & 2.5*).

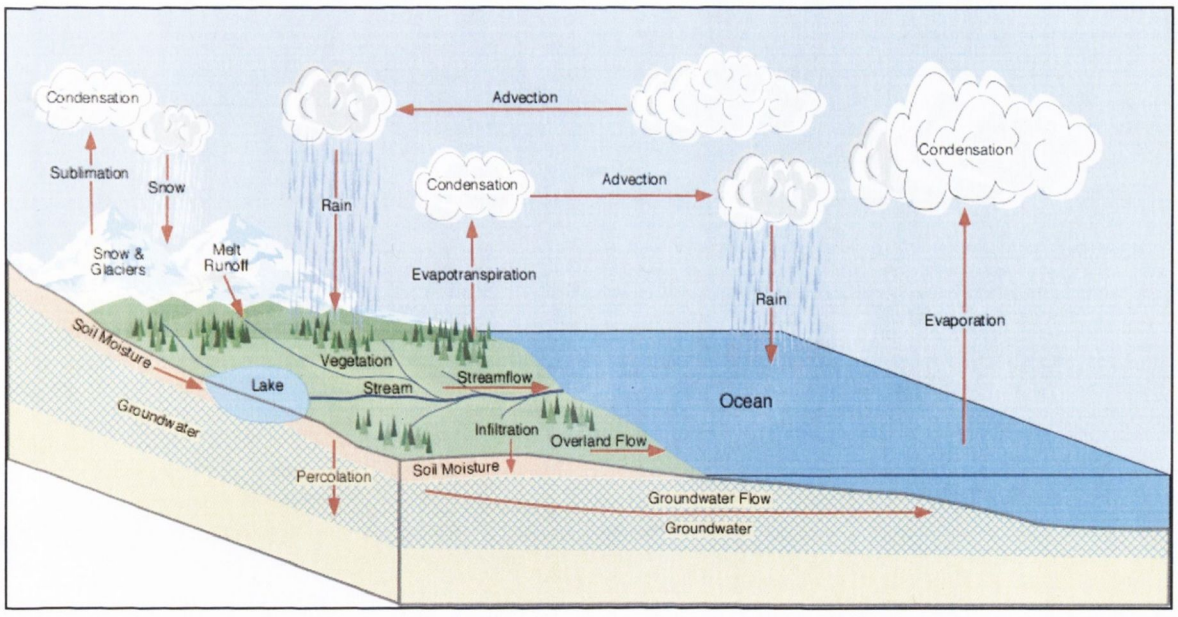


Figure 2.4 The Hydrologic Cycle (USGS, 2007)

2.4.1 Groundwater occurrence

Groundwater typically originates as infiltration from precipitation, streamflow, lakes and reservoirs and is that portion of water in the subsurface that occurs in the saturated zone (*Figure 2.5*). As water percolates through cracks and pores in soil and rock, it passes through the unsaturated (aeration) zone, which is characterised by the presence of both air and water in available pore space, with the interface between the saturated and unsaturated zones known as the phreatic surface or more commonly, the watertable (Bouwer, 1978; Todd & Mays, 2005). Groundwater bearing formations sufficiently permeable to transmit and yield water in useable quantities are called aquifers and are necessary to ensure the resource value of groundwater (Shaw, 1994; Misstear *et al.*, 2006).

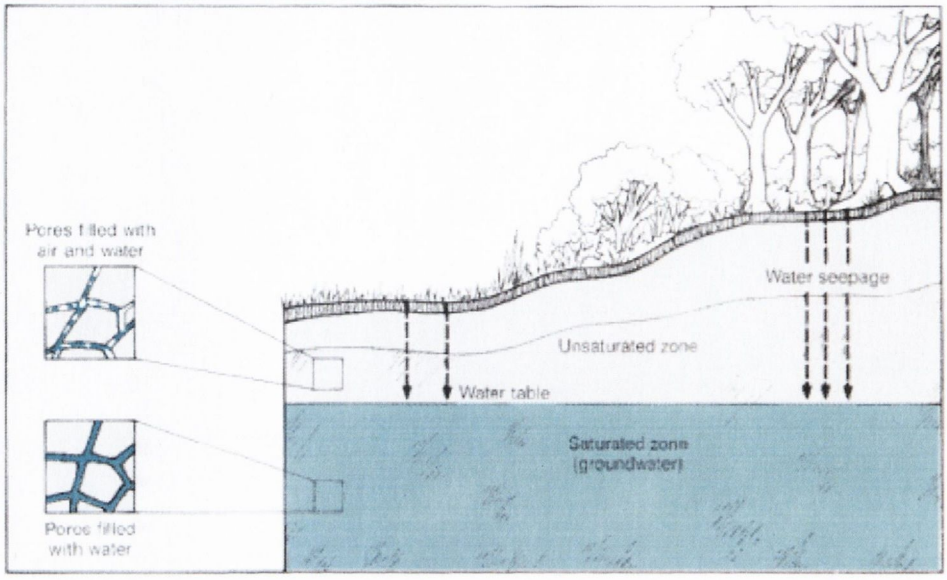


Figure 2.5 The unsaturated and saturated zones (Oregon Dept. of Environmental Quality, 2011)

2.4.2 Aquifers

Typically, whereas the majority of geological formations will store some volume of water, those capable of storing and transmitting significant (useful) volumes of water are referred to as aquifers. Hunter-Williams & Lee (2007) state that groundwater flow in the majority of Irish bedrock aquifers occurs through fissures, fractures and faults (secondary porosity), with the flow volume influenced by the number, size and connectivity of these fractures. Additionally, although to a lesser extent in Ireland, groundwater flows through and is stored in the pore spaces of sand and gravel (unconsolidated) deposits i.e. intergranular permeability.

Irish aquifers are relatively shallow and often small in their lateral extent. In the western parts of the country there are karstic aquifers, which are outlined in greater detail in *Section 2.7*. In Ireland the total aquifer area is estimated to be of the order of 18,870 km^2 (EEA, 1999). Recent figures suggest that total annual groundwater abstraction for public, group and industrial water supplies in Ireland is approximately 200 million m^3 (Moe *et al.*, 2007). Nationally, approximately 26% of water supplies are obtained from groundwater resources. For public water supplies, this value rises to more than 50% in some counties (e.g. Roscommon, Offaly). Groundwater abstractions usually take place by pumping from wells or boreholes, although springwater is also exploited, particularly in limestone areas (Hunter Williams & Lee, 2007).

2.5 Aquifer materials

The geological materials that comprise an aquifer can be categorised into three groups (Clark, 1988), namely:

1. Crystalline Aquifers
2. Consolidated Aquifers

3. Unconsolidated Aquifers

2.5.1 Crystalline Aquifers

Crystalline (also known as hard rock) aquifers are generally those water-bearing formations composed of igneous or metamorphic rocks with little or no primary porosity (1-3%) and no well defined base (Davis & DeWiest, 1966; Whitten & Brooks, 1987; Clark, 1988; Slattery, 1997; Misstear *et al.*, 2006). Aquifers in crystalline rocks, including granites, basalts, marbles, mica-schists, gneisses and quartzites, are heterogeneous and have irregular configurations with variable hydraulic characteristics over short distances. Locally, they constitute isolated groundwater basins with little or no inter-basin exchange. Due to the absence of primary (original) porosity in the majority of crystalline aquifers i.e. intergranular pore spaces are almost negligible, groundwater transmissivity is reliant mainly on secondary porosity, characterised by both weathering and fracturing (Bannerman & Ayibotele, 1984; Todd & Mays, 2005; Misstear *et al.*, 2006), with almost all useful groundwater flow occurring in fractures. Fractured crystalline aquifers constitute groundwater reserves which are important groundwater sources in many parts of the world including South America, Africa and South Asia (Maréchal *et al.*, 2004). As outlined by Banks & Robins (2002), the majority of Norway is underlain by crystalline bedrock and consequently these formations constitute an important drinking water source throughout much of rural Norway.

Crystalline aquifers in Ireland are composed mainly of Cambrian and Precambrian gneisses, schists and quartzites, and granites and other igneous intrusive rocks (Johnston & McConvey, 2003). The Cambrian and Precambrian aquifers are located mainly in the North-West of the country, with a small area in the southeast, just south of Rosslare, Co.Wexford. They include the oldest rocks in Ireland and are typically highly complex metamorphic rocks, particularly metamorphosed sedimentary rocks such as sandstones and mudstones. Approximately 6,500km^2 of Ireland is underlain by Cambrian and Precambrian bedrock (Holland, 2001; Working Group on Groundwater, 2005). There are 4 main areas of igneous intrusive rocks in Ireland, all of which are granite, namely the Leinster granites (1,500 km^2), the Donegal granites (950 km^2), the Connemara granites (700 km^2) and the Newry/Carlingford granites (215 km^2), which collectively underlie approximately 4.8% of Ireland's total land area (Working Group on Groundwater, 2005; GSI, 2009). In all, crystalline aquifers underlie approximately 14.1% of Irelands land area.

As mentioned, the water-bearing voids in crystalline aquifers are typically in the form of fissures, produced as a result of folding and faulting followed by weathering. Due to the "hard rock" nature of crystalline formations, boreholes drilled in these aquifers often do not require physical support i.e. open borehole completion may be permissible (Slattery, 1997), although Misstear *et al.* (2006) state that the upper section of the well should be cased and grouted in order that surface pollution is excluded.

Well depths in the bedrock aquifers, including crystalline aquifers, typically range between 30 – 100 m below the surface, with drill depths >100 m rarely exceeded in crystalline aquifers due to decreasing yields with increasing depth due to a decrease in both weathering and fracturing with depth (Misstear *et al.*, 2006 from Clark, 1985). Although crystalline aquifer yields are typically low in comparison to other bedrock aquifers and unconsolidated aquifers, with median values of 10-20 m^3

day^{-1} (Misstear *et al.*, 2006 from Gustafson, 2002), these yield rates are adequate for individual domestic supplies.

2.5.2 Consolidated Aquifers

Consolidated aquifers are those aquifers composed of sediments which have been consolidated into rock by a number of processes, including compaction, cementation or physical and/or chemical changes in the sediment constituents. In Ireland, consolidated aquifers are typically composed of sandstones, limestones and some smaller shale formations. Examples of consolidated aquifers in Ireland include those located in the Ballysteen Limestones (Midlands), Calp Limestones (Midlands and East) and Devonian Old Red Sandstones (South).

Groundwater in consolidated bedrock aquifers is typically present within interconnected fractures and fissures, in addition to limited pore spaces and rock layers, with permeability in these aquifers therefore characterised as being mainly secondary permeability (Fetter, 2001). The volume of water stored and consequently, the effective yield of these aquifers is highly dependent on the size and interconnectivity of these fractures and fissures (*Figure 2.6*). Generally, consolidated limestone aquifers are categorised as high yield aquifers with sandstone aquifers yielding lower volumes, although still greater than those from crystalline aquifers (Oregon Department of Environmental Quality, 2011).

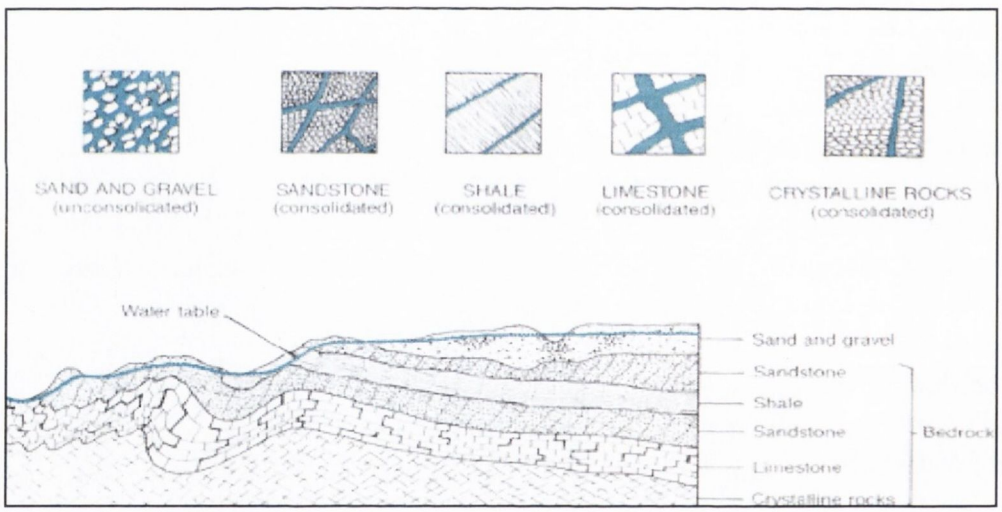


Figure 2.6 Consolidated bedrock layers overlain by unconsolidated sediments (Oregon Dept. of Environmental Quality, 2011)

Similar to wells in crystalline aquifers, wells in "hard rock" consolidated aquifers may generally be constructed without support and therefore open borehole completion may be appropriate (Price, 1996). However, again, in order that surface contamination is not allowed to enter the well, it is recommended that the upper section be cased and sealed (Bloetscher *et al*, 2007).

2.5.3 Unconsolidated Aquifers

Where groundwater is found in useable amounts in unconsolidated deposits of sand, gravel, silt or clay, these deposits may form an abstraction resource in their own right and be classified as aquifers

(Hunter-Williams & Lee, 2007). Additionally, smaller deposits may act as a high-porosity "reservoir" of groundwater overlying a fractured bedrock aquifer. The sand and gravel aquifers that underlie approximately 2% of Ireland are the only aquifers with intergranular permeability (*Figure 2.7*), with these aquifers typically being unconfined and relatively thin in comparison to bedrock aquifers, being rarely being more than 15 m thickness (Hunter-Williams & Lee, 2007; EPA, 2010b). Porosities in unconsolidated aquifers internationally range from 20-90% and are typically 25-65% (Davis & DeWiest, 1966), with specific yield values typically up to 50% (Heidari & Wench, 1997). However, recently drafted porosity and specific yield values relating to unconsolidated aquifers in Ireland are significantly less, at approximately 15% and 14%, respectively (*Table 2.3*) (Hunter-Williams & Lee, 2007; GSI, 2009).

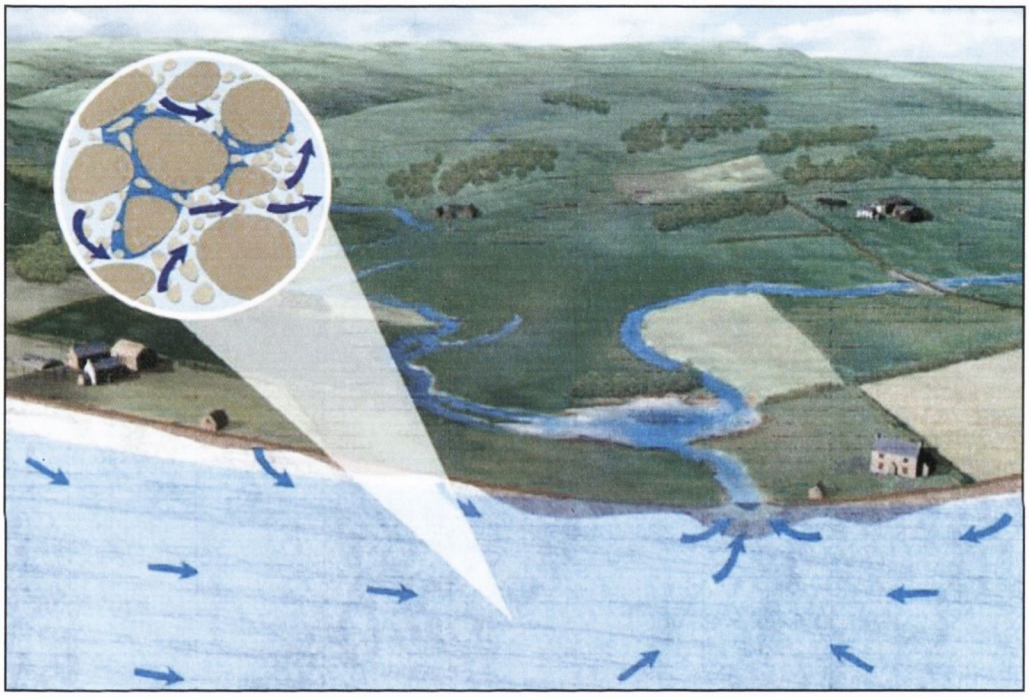


Figure 2.7 Unconsolidated Aquifer with inter-granular flow (WFD Visual, 2010)

Table 2.3 Estimated values of porosity, specific yield and specific retention in Ireland (GSI, 2009)

Aquifer Material	Porosity (%)	Specific Yield (%)	Specific Retention (%)
Sand & Gravel	15	14	1
Clay	50	2	48
Peat	90	15	75
Pure Limestone	2.5	2.4	0.1
Impure Limestone	1	0.9	0.1
Unweathered Granite	0.1	0.09	0.01

In Ireland, these sand and gravel deposits are of Quaternary age and primarily of glaciofluvial origin. They occur in different forms, e.g. as an outwash plain (e.g. The Curragh, Co. Kildare), delta (e.g. Blessington, Co. Wicklow) or eskers (e.g. Rosemount & Ballymore, Co. Westmeath).In Ireland, because of the part-glacial origin of the bulk of these deposits, they tend to be coarse and poorly

sorted (well-graded). Some of Ireland's most productive aquifers are unconsolidated sands and gravels, the largest of which is the Curragh aquifer of Co Kildare (200 km^2), with gravel thicknesses between 5-30 m (Hayes *et al.*, 2001; Hunter-Williams & Lee, 2007).

As outlined by Misstear *et al.* (2006) and Bloetscher *et al.* (2007), due to the nature of unconsolidated formations, wells in unconsolidated aquifers require screening and may require the installation of an artificial gravel pack, to prevent collapse and/or sediment ingress. This is discussed in greater detail in Chapter 3 (*Section 3.4.2*).

2.6 Aquifer Conditions

Aquifers are typically divided into two "stratigraphy" types, which as defined by Whitten & Brooks (1987), are based upon the presence and type of rock layering or bedding (particularly their sequence in time), the rock character and the spatial correlation of these bedding layers. These two stratigraphy types are (Blyth & de Freitas, 1984):

1. Aquifers comprised of a particular strata i.e. strataform aquifers;
2. Aquifers composed entirely of a fracture zone

Slattery (1997) states that strataform aquifers may be further subdivided into confined and unconfined aquifers, with reference to the pressure head conditions within the aquifer.

2.6.1 Unconfined Aquifers

As mentioned, unconfined aquifers are strataform formations, whereby the top of the aquifer coincides with the ground surface, as shown in *Figure 2.8*. The static water level in unconfined aquifers is known as the water table and is coincident with the potentiometric surface (Clark, 1988; Slattery, 1997). The majority of aquifers in Ireland are unconfined hard rock aquifers (Wright, 2000).

2.6.2 Confined Aquifers

Confined aquifers are water-bearing strata (i.e. strataform aquifers) found between impermeable rock layers called aquicludes or very low permeability rock layers known as aquitards (Chernicoff *et al.*, 2002), as shown in *Figure 2.8*. There is no free water surface in such aquifers, with the potentiometric surface consequently at a higher elevation than the top of the aquifer (Slattery, 1997) i.e. the water is generally stored under pressure; or as outlined by Misstear *et al.* (2006), an aquifer is said to be confined when fully saturated and the potentiometric surface lies in a low-permeability confining layer, located above the aquifer.

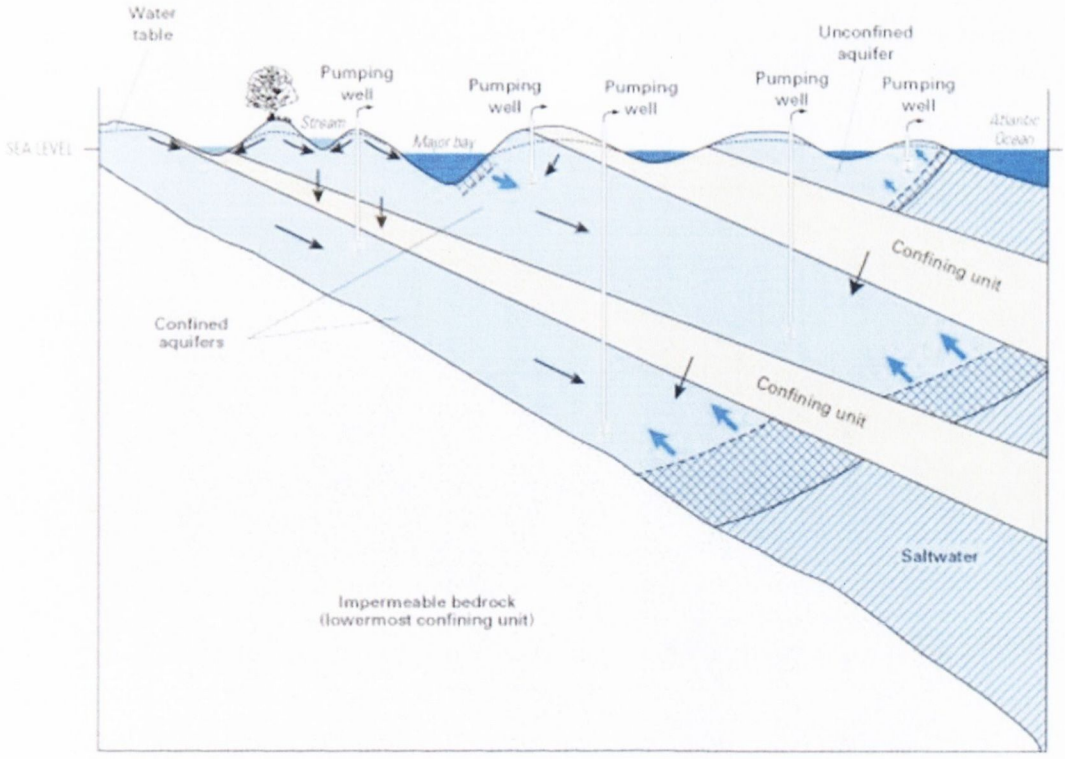


Figure 2.8 Confined and Unconfined Aquifers (Leahy & Martin, 1993)

2.6.3 Multi-Aquifer Systems

A multi-aquifer system may be described as consisting of a number of water-bearing strataform formations inter-bedded by low permeability (aquitard) or impermeable (aquiclude) strata, with the uppermost aquifer being confined or unconfined, as shown in *Figure 2.9*.

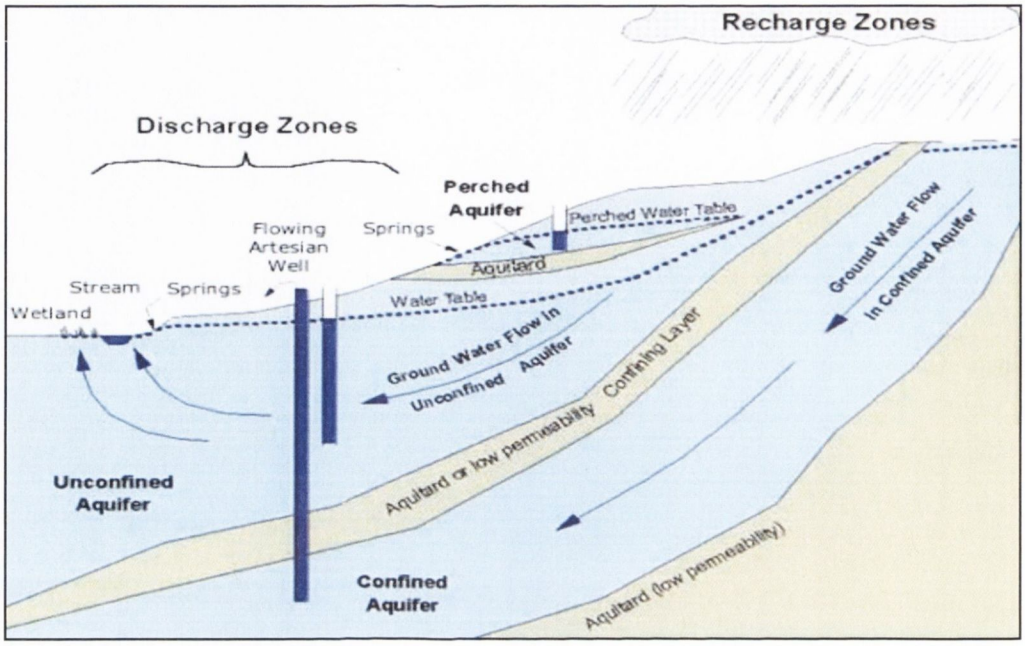


Figure 2.9 A multi-aquifer system with unconfined aquifer in discharge zone (Leahy & Martin, 1993)

2.7 Aquifer Classification

Aquifers and can be classified using a variety of methods to characterise geologic and hydraulic properties, the degree of connection with surface water, level of development, geochemical conditions and perhaps most importantly, the overall importance of the aquifer (Payne & Woessner, 2010; Olaniyan *et al.*, 2010). In the case of aquifer classification in Ireland, the current system is based primarily upon the overall aquifer productivity (potential) yield and areal extent of the aquifer, with these factors amalgamated to formulate an overall aquifer value as a groundwater resource. There are three main aquifer categories defined in Groundwater Protection Schemes (DoELG/EPA/GSI, 1999), further subdivided into nine categories (*Figure 2.10*), as follows:

2.7.1 Regionally Important Aquifers

These are aquifers which are sufficiently productive to yield sufficient water to supply major regional water schemes (>400 m^3 d^{-1}), with a spatial extent typically greater than 25 km^2. Groundwater flow predominantly occurs through fractures, fissures and joints. This category is further divided into: Regionally Important sand/gravel aquifers (Rg); Regionally Important Karstified Bedrock Aquifer (Rk); and Regionally Important Fissured Bedrock Aquifers (Rf). The following descriptions are based mainly on DoELG/EPA/GSI (1999), Misstear & Daly (2000) and Hunter-Williams & Lee (2007).

Rg Regionally Important Sand/Gravel Aquifer

These aquifers are highly permeable, generally more than 10m thick (or a saturated thickness of at least 5m), and extend >10 km^2.Groundwater flows through the pore spaces between sand/gravel grains, with permeability determined by the grain size (larger grains give larger pore spaces) and the

'sorting' of the material (more uniform grain sizes give higher permeability).Groundwater in the aquifer tends to be well distributed, with associated high storage and long flow paths, limited only by the aquifers spatial extent. Due to relatively uniform grain size in the aquifer, 'flat' water tables (low groundwater gradients) are characteristic, resulting in low groundwater velocities. Baseflow is usually high in concurrence with high water tables, and conversely, when surface water levels are high, large volumes of surface water may transmitted to the aquifer, although this is not common in Ireland. Large springs (>2000 m^3 d^{-1}) are frequently associated with sand/gravel aquifers, particularly in lowland systems.

Rk Regionally Important Karstified Bedrock Aquifer

Karstification is the process whereby limestone is slowly dissoluted by percolating waters, resulting in dissolution conduits which permit rapid groundwater movement (Gill, 2009), most often occurring in the upper bedrock layers and along particular strata. This results in uneven distribution of permeability, and the development of distinctive karst landforms at the surface (e.g. swallow holes, caves, dry valleys), some of which provide direct access for recharge/surface water to enter the aquifer. The landscape is characterised by large underground drainage systems, which may be several kilometres long. Groundwater velocities through conduits are characteristically high and aquifer storage is typically low, with the groundwater frequently discharging to large springs (>2000 m^3 d^{-1}), ranging from regular and dependable to highly variable ('flashy'). There is strong interconnection between surface water and groundwater (Gill, 2009). The degree of karstification ranges from low to high, with two types of karst aquifer generally identified; namely those dominated by diffuse flow (Rkd) i.e. the southeast of Ireland, and those dominated by conduit flow (Rkc) i.e. the west and northwest of Ireland (Hunter-Williams & Lee, 2007).

Rf Regionally Important Fissured Bedrock Aquifer

Aquifers in which the network of fractures, fissures and joints, through which groundwater flows, has a high level of connectivity, resulting in highly permeable zones. There is relatively high aquifer storage, and groundwater flow paths can be up to several kilometres in length. Characteristically, high baseflow and large (>2000 m^3 d^{-1}) springs are associated with these aquifers.

2.7.2 Locally Important Aquifers

Moderately productive aquifers are capable of sufficient abstraction to supply villages, small towns or factories (100-400 m^3 d^{-1}), with groundwater flow occurring predominantly through fractures, fissures and joints. They are divided into: Sand/gravel aquifers (Lg); Locally Important Karstified Bedrock aquifer (Lk); Bedrock aquifers which are generally moderately productive (Lm); and Bedrock aquifers which are moderately productive only in local zones (Ll).

Lg Locally Important Sand/Gravel Aquifer

Although similar to a Regionally Important Sand/Gravel Aquifer (Rg) i.e. high permeability, high storage, high flow length, high baseflow, these aquifers are capable of decreased abstraction rates, due to a typically smaller spatial area (1-10 km^2) and/or less consistent permeability (smaller and/or less uniform grain size). The aquifer may supply very high yields, however, due to the decreased areal extent; decreased available recharge limits the potential abstractions.

Lk Locally Important Karstified Bedrock Aquifer
As for Lg aquifers, these are very similar to the Regionally Important Karstified Bedrock Aquifer (Rk), however, decreased areal extent (<25 km^2) limits the volume of recharge available, therefore limiting abstraction.

Lm Locally Important Bedrock Aquifer, Generally Moderately Productive
Aquifers in which the network of fractures, fissures and joints, has a relatively high level of connectivity throughout, resulting in moderate permeability and groundwater flow. Storage is moderate and groundwater flow paths can be up to several kilometres in length. There is likely to be a substantial baseflow and large (>2000 m^3 d^{-1}), dependable springs may be associated with these aquifers. This classification also includes aquifers similar to the Regionally Important Fractured Bedrock Aquifer (Rf), but with a decreased areal extent (25 km^2). The aquifer may supply excellent yields; however the decreased overall size limits the amount of recharge available to meet abstractions.

Ll Locally Important Bedrock Aquifer, Moderately Productive only in Local Zones
These are aquifers with limited water-bearing strata and low associated levels of connectivity, resulting in low permeability which typically decreases with depth. A shallow zone of higher permeability may exist within the top few metres of more fractured/weathered rock, and higher permeability may also occur along fault zones. These zones may be able to provide larger 'locally important' supplies of water. Characteristically, decreased connectivity between limited fractures results in low aquifer storage and flow paths that may only extend a few hundred metres. Due to the low permeability and storage capacity, these aquifers tend to have low 'recharge acceptance'. Baseflow may decrease significantly during drier periods.

2.7.3 Poor Aquifers

Poor aquifers are typically capable of supplying single households, farms or small group water schemes (<100 m^3 d^{-1}), with groundwater flow generally occurring through fractures and joints with low levels of connectivity. As reported by Pilatova & Ofterdinger (2010), poorly productive aquifers are characterised by low porosity and consequently low specific well yields, and underlay approximately 70% of Ireland. They are subdivided into: Bedrock aquifers which are generally unproductive (Pu) and Bedrock aquifers which are generally unproductive except for local zones (Pl).

Pl Poor Bedrock Aquifer, Moderately Productive only in Local Zones
As with Pu aquifers, these aquifers are typically capable of low abstraction rates and secondary permeability predominates. However, larger numbers of fractures and faults with typically greater levels of connectivity lead to higher potential abstraction rates and storage rates than generally observed in Pu aquifers. Also, recharge acceptance, typical flow path length and baseflow (discharge to surface water bodies) are likely to be greater, although still relatively low.

Pu Poor Bedrock Aquifer, Generally Unproductive
These are aquifers with low numbers of fractures and faults and very low levels of connectivity of these water-bearing strata. Low permeability increases with depth as fracture numbers and connectivity decrease. Typically, this results in low aquifer storage, short flow paths and low recharge acceptance, with little or no baseflow.

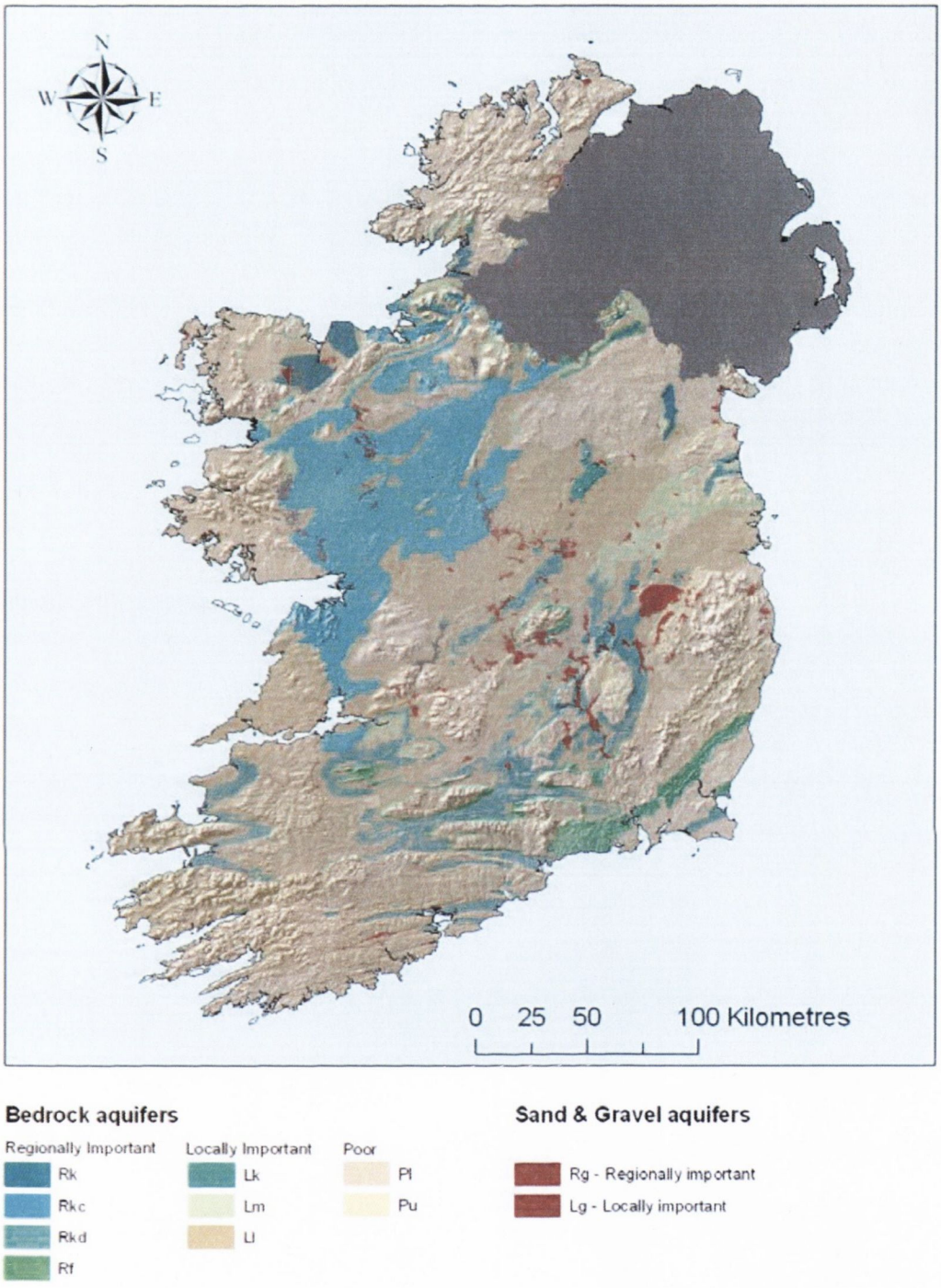


Figure 2.10 Aquifer Map of Republic of Ireland (GSI, 2011)

2.8 Groundwater vulnerability

One of the key environmental objectives of the proposed EU Water Framework Directive is that Member States must prevent the deterioration of groundwater quality (Misstear & Daly, 2000), with groundwater vulnerability categorisation and mapping considered to play a central role in the implementation of this directive.

Vulnerability is 'a term used to represent the intrinsic geological and hydrogeological characteristics that determine the ease with which groundwater may be contaminated by human activities' (Daly & Warren, 1998). Misstear *et al.*, (2009b) state that there are a number of key factors contributing to the vulnerability of an aquifer to contamination, including the volume and intensity of recharge, leaching characteristics of the topsoil, subsoil characteristics such as permeability (hydraulic conductivity) and thickness, unsaturated zone characteristics and the aquifer type and extent. In Ireland, groundwater vulnerability is evaluated qualitatively and subsequently categorised as *Extreme* (E), *High* (H), *Moderate* (M) or *Low* (L) (*Table 2.4*). The vulnerability assessment is ascertained primarily according to the thickness and permeability of the subsoils overlying the aquifer, based on the principle that a thick layer of low permeability subsoil will provide greater contaminant attenuation; conversely, thin (or absent) layers of high permeability subsoil will provide little or no attenuation (DoELG/EPA/GSI, 1999; Misstear & Daly, 2000; Swartz *et al.*, 2003). Subsoil permeability is the defining feature of groundwater vulnerability categorisation, as the time of travel of infiltrating water (and contained contaminants), the quantity of contaminants capable of reaching the groundwater and the overall attenuation capacity of the soil are all dependent on subsoil type (GSI, 2007; Misstear *et al.*, 2009b). Additionally, it is assumed that little or no attenuation of contaminants occurs in Irish bedrocks due to secondary porosity which, as discussed earlier in this chapter, is a characteristic feature of a majority of Irish aquifer systems. Consequently, subsoils are the single most important natural feature influencing groundwater vulnerability and groundwater contamination prevention (Swartz *et al.*, 2003; GSI, 2007).

Table 2.4 Vulnerability Mapping Guidelines (DoELG/EPA/GSI, 1999)

Depth to rock	Hydrogeological Requirements for Vulnerability Categories				
	Diffuse recharge			Point Recharge	Unsaturated Zone
	high permeability (*sand/gravel*)	Moderate permeability (*sandy subsoil*)	low permeability (*clayey subsoil, clay, peat*)	(*swallow holes, losing streams*)	(*sand & gravel aquifers only*)
0–3 m	**Extreme**	**Extreme**	**Extreme**	**Extreme (30 m radius)**	**Extreme**
3–5 m	**High**	**High**	**High**	**N/A**	**High**
5–10 m	**High**	**High**	**Moderate**	**N/A**	**High**
>10 m	**High**	**Moderate**	**Low**	**N/A**	**High**

i *N/A = not applicable.*
ii *Release point of contaminants is assumed to be 1–2 m below ground surface.*
iii *Permeability classifications relate to the engineering behaviour as described by BS5930.*
iv *Outcrop and shallow subsoil (i.e. generally <1.0 m) areas are shown as a sub-category of extreme vulnerability.*

(amended from Deakin and Daly (1999) and DELG/EPA/GSI (1999))

As shown in *Table 2.4*, subsoil permeability in Ireland is classified as being high, moderate or low. A number of studies have focused on the subsoil permeability boundaries, with resulting estimates indicating the numerical boundary between moderate and low permeability is approximately $1x10^{-8}$ - $1x10^{-9}$ m s^{-1} (Swartz *et al.*, 2003) and the boundary between moderate and high permeability in the region of $1x10^{-4}$ m s^{-1} (O'Suilleabhain, 2000). Indirect indicators of subsoil permeability are also used in vulnerability mapping, including groundwater recharge, natural drainage density, artificial

drainage density and vegetation characteristics (Lee, 2000, Misstear *et al.*, 2009a; Misstear *et al.*, 2009b).

Groundwater vulnerability maps (*Figure 2.11*) have now been produced for the majority of local authority areas, and comprise a central role in groundwater protection schemes (Daly & Warren, 1998; Fitzsimons *et al.*, 2003; Misstear *et al.*, 2009b).

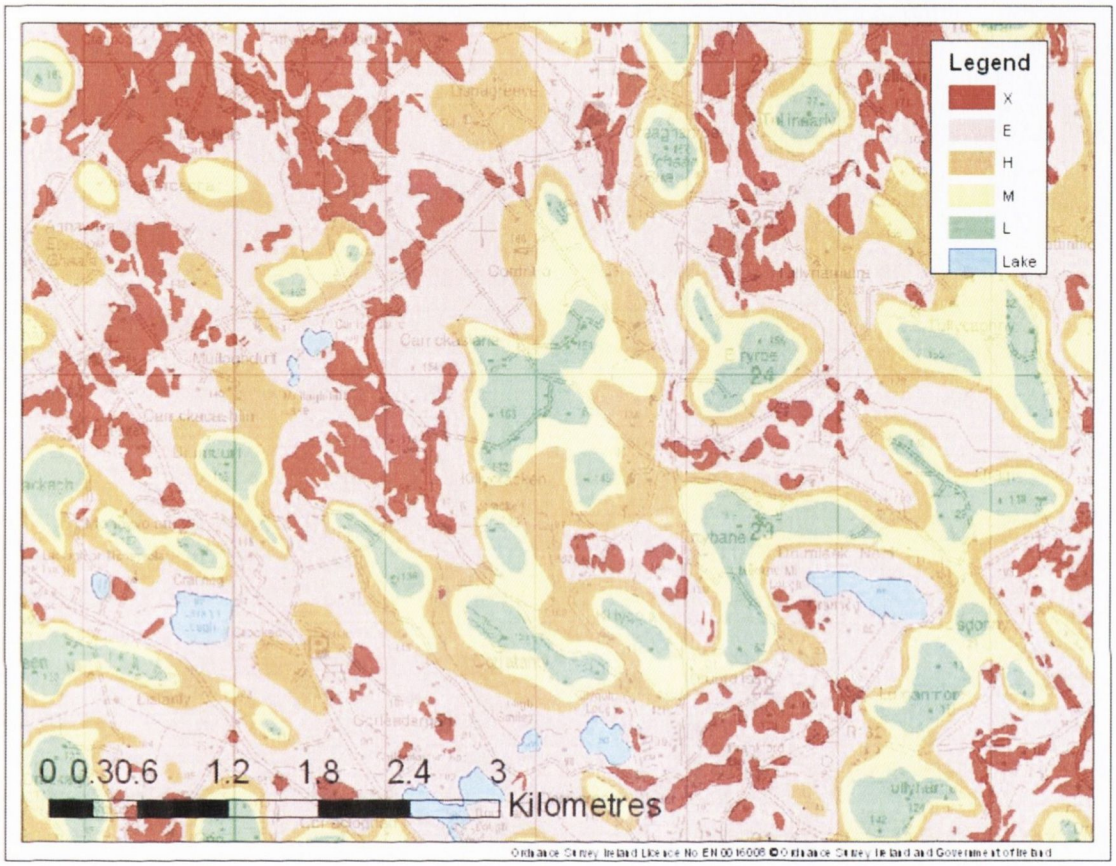


Figure 2.11 Sample Extract from Groundwater Vulnerability map
(GSI, 2011)

Chapter 3: Water Supply Wells

3.1 Introduction

In Ireland, a borehole (or dug well) is almost the only significant feature of a domestic residence which is not regulated by any legislative tools or standards (Wright, 1995; Ball, 2000). The following chapter outlines the principles and main objectives underlying appropriate well location, design and construction in Ireland, focusing on those methods most relevant to Ireland. Furthermore, methods pertaining to appropriate well maintenance/rehabilitation and source treatment are summarised.

3.2 Well Classification

A well, as it will be defined throughout this study, is a vertical hole, dug or drilled into an aquifer below the watertable for the purpose of groundwater abstraction. Wells used for the abstraction of groundwater vary from shallow, hand or machine dug, large diameter shafts to deep, mechanically drilled boreholes. As such, they are generally classified according to the method of construction; typically dictated by the aquifer type and therefore the geological area in which the well is to be located. The two main categories in Ireland are dug wells and bored wells, which are now outlined.

3.2.1 Hand-Dug Wells

Digging is the simplest and oldest method of well construction (Davis & Lambert, 2002; Bloetscher *et al.*, 2007). Shallow (<5 m) hand-dug wells are common in many parts of the world, particularly in alluvial deposits and the weathered sections of carbonate, crystalline, and basaltic bedrock (Mace, 1999). There are a number of types of hand dug well, including traditional hand-dug wells (*Figure 3.1*), radial collector wells and qanat (falaj); however, in Ireland they are typically of the traditional hand-dug type. It is not known how many hand-dug wells are currently in use in Ireland, however, it is estimated that no more than 10% of private domestic wells are of the traditional hand-dug type (Misstear, pers. comm. 2009; Ball, pers. comm. 2010). Dug wells are generally less costly to sink and in areas where high levels of iron, manganese and sulphates are naturally occurring in the bedrock, a well which drains water from glacial overburden may be the preferred option (McCole, 1992).

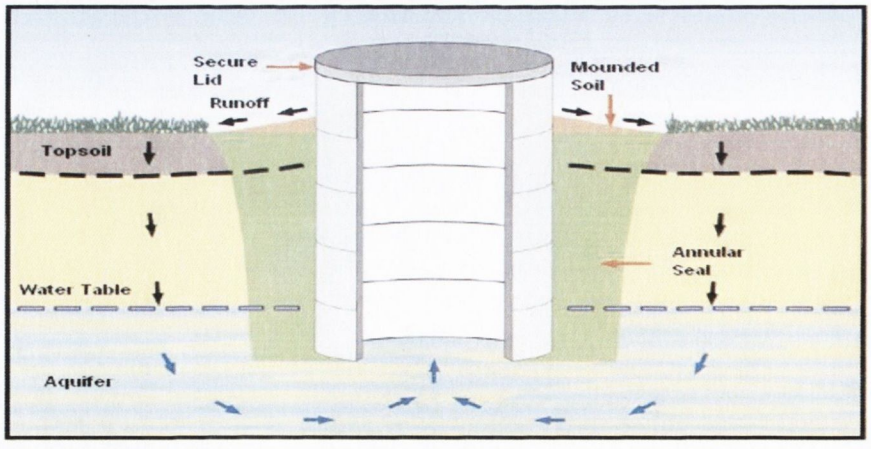


Figure 3.1 Cross section of typical hand-dug well construction (OMAFRA, 2006)

3.2.2 Bored Wells

Bored wells are encountered frequently in Ireland, with upwards of 90% of the estimated 100,000-200,000 wells being of the bored type (Wright, 1999). Bored wells are variously called tube wells, production wells and production boreholes and, as with hand-dug wells, there are a number of bored well types including vertical drilled wells, inclined drilled well and infiltration galleries (Misstear *et al.,* 2006). As the vertical drilled well is undoubtedly the most frequently encountered, this well type is focused on throughout this chapter and referred to as a borehole (*Figure 3.2*).

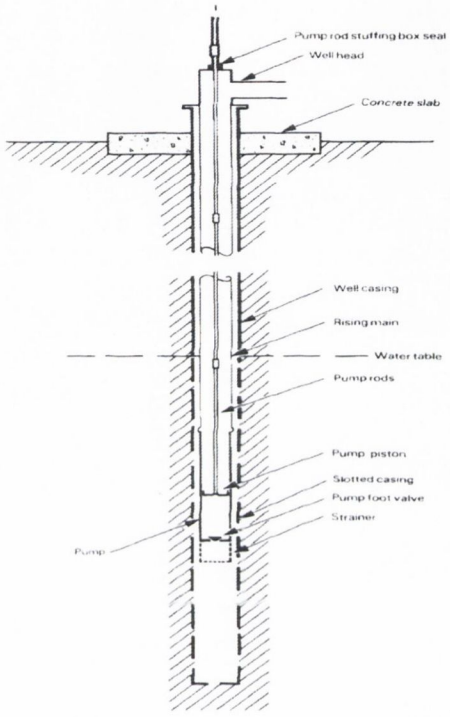


Figure 3.2 Cross section of typical borehole (WHO/FAO, 1986)

3.3 Well Location

As set out by Bloetscher *et al.* (2007), there are a number of factors to consider when locating a new well, the primary issues being:

- Site availability
- Water supply (yield)
- Water quality and,
- Wellhead protection

Additionally, in domestic situations, cost will often be a factor and therefore a balance must be sought between depth, local hydrogeology, local landuse and potential hazards. With regard to water quality, the most important considerations are the distance and hydraulic gradient from potential contaminant sources, including septic tanks, grazing animals, farmyards and landfill sites. No formal standards currently exist in Ireland pertaining to the appropriate siting of a domestic well, however, IGI (2007) recommend that in general "*a well should be located upslope from, and as far*

away as possible from, potential sources of pollution". Specific setback distances are dependent on site characteristics including subsoil type, water table depth, hydraulic gradient, pumping rate and the extent of the potential contaminant source. Suggested distances are set out in *Table 3.1*.

Table 3.1 Recommended distance of a private well from likely pollution sources (IGI, 2007)

Subsoil Type	Subsoil Depth (m above bedrock)	Minimum distance (m); groundwater flow direction known		Minimum distance (m); groundwater flow direction unknown	
		Up Grad	Down Grad		
		Farmyard/Septic tank (m)	Farmyard/Septic tank (m)	Farmyard (m)	Septic tank (m)
Silt or Silt/Clay (low permeability, poor drainage)	1-3	100/40	50/15	10	40
	>3	75/30		75	30
	>8	60/		60	
Sandy clay, silt (moderate permeability)	1-2	120/45	50/15	120	45
	2-8	90/30		90	30
	>8	90/30		90	30
Sand and gravel (high permeability, free draining)	1-2	150/60	50/15	150	60
	2-8	150/40		150	40
	>8	120/30		120	30

3.4 Well Construction

Just as there are a variety of well purposes (water production, geothermal, injection/disposal, aquifer storage and recovery, environmental remediation and monitoring) and well types, there are a variety of options for their construction, including;

- Hand dug
- Large diameter auger
- Hydraulic rotary
- Dual-tube rotary
- Reverse circulation (mud)
- Reverse-air circulation
- Sonic
- Cable tool
- Screened stem auger
- Reverse-air circulation
- Direct push
- Bucket type drilling
- Percussion hammer
- Jetting and driving
- Hollow stem auger
- Solid flight auger
- Casing hammer air rotary
- Mud rotary
- Direct-air circulation
- Horizontal/angle
- Rotary hammer

(Bloetscher *et al.*, 2007)

This section will focus on construction methods of the types of well most commonly encountered in Ireland, including hand-digging, cable tool (percussion) drilling, rotary drilling (direct and reverse circulation) and down the hole hammer drilling.

3.4.1 Manual Construction

Dug wells are typically lined or cased with concrete liners, ranging in diameter from 0.6-1.5 m, and approximately 1 m in length (Watt & Wood, 1997; Davis & Lambert, 2002) and serve to support the well, with the bottom 1-2 liners being perforated to facilitate the passage of water (McCole, 1992). As set out by Davis & Lambert (2002), excavation usually takes place from underneath the lowest casing, and is allowed to sink as digging progresses, with new concrete rings being added at the surface as necessary. On encountering the water table, the excavation is ordinarily dewatered using a surface mounted pump (Misstear *et al.*, 2006). Hand-dug wells should be excavated as far as possible into the water-bearing formation, with the perforated bottom caisson being firmly packed onto a crushed rock base. A sanitary seal should be provided by overlapping and grouting all casings onto lower liner sections, with sealant being provided from inside (GSI, 1979; McCole, 1992; Watt & Wood, 1997; Davis & Lambert, 2002). (See *Figure 3.1*)

3.4.2 Borehole Drilling Procedures

The general procedure for borehole drilling in Ireland will depend on the aquifer type, with two main aquifer types being commonly encountered - bedrock aquifers and unconsolidated (sand & gravel) aquifers. Although, as outlined above there are a plethora of different drilling techniques and equipment, the same order of works are typically applied. Current IGI guidelines (2007) give a recommended 8-step procedure for drilling wells into bedrock aquifers in Ireland (Appendix B1):

In the case of unconsolidated sediments, the borehole walls will typically need to be supported by casing for the entire length of the borehole in order to prevent collapse within the hole, with casing being advanced at the same rate as drilling. Conductor casing is used in much the same way as in consolidated materials. Where water-bearing layers are encountered, slotted well screens are installed in the permanent casing; otherwise blank 150-180 mm casing is installed. As the temporary casing is withdrawn, unconsolidated sediments will collapse to fill the annulus. Sand bridges are installed to prevent grout penetrating the entire borehole length and blocking the productive sections, with grouting being undertaken as outlined above.

It should be emphasised that these are *recommended* procedures for borehole installation, and that existing practice may differ. For example, many boreholes lack proper grout seals around the upper well casing.

3.4.3 Cable Tool Drilling

Cable-tool or percussion drilling was the first method used to drill a borehole and is used to describe the method by which a heavy impact tool called a chisel (due to the chisel like shape) or bit on the end of a wire cable is repeatedly raised and dropped to pulverise the rock (Price, 1996). This cable passes through a pulley mounted on a mast, with power typically supplied to a winch by a diesel engine (Misstear *et al.*, 2006). The drill stem is a solid steel rod which adds weight to the bit, while also maintaining verticality of the borehole (Misstear *et al.*, 2006; Bloetscher *et al.*, 2007). The percussive action is provided by the driller alternately engaging and disengaging power to a cable drum, which lifts the drill string 0.5-1.0 m before dropping again. As outlined by Misstear *et al.*, (2006 from Clark, 1988), the actual cable tool drilling procedure used will depend on the formation

being drilled i.e. tool strings used to drill in softer formations are typically simpler than those used in hard rock formations.

When drilling begins, a short guide tube or conductor pipe is typically drilled or hand sunk into the ground (Clark, 1988). This prevents the formation beneath the rig from collapsing and maintains verticality of the borehole (Slattery, 1997). In consolidated formations, the remainder of the drilling is completed in an open bore, with Cruse (1979) recommending that percussive methods not be used in unconsolidated formations.

3.4.4 Rotary Drilling

As the name suggests, it is the process of constructing a borehole using a rotating drill system, using a drill bit turned by means of a Kelly or driving chuck of the rotary drill system, through which the rotation is transmitted to the drill stem/string (Allaby & Allaby, 1999). The drill stem is a hollow steel pipe, composed of lengths of drill pipe, to which the drill bit is attached, normally consisting of the drill bit, drill collars, stabilisers and drill pipe (Price, 1996; Jellison *et al.*, 2000). In the case of air-hammer rotary, which is typically used in Ireland (Section 3.4.5), compressed air is used to drive the rotating system and also to carry drill cuttings to the surface and improve drilling efficiency (IGI, 2007) i.e. drilling fluid. In some cases, water, drilling mud or foam is used as drilling fluid (Price, 1996; Bloetscher *et al.*, 2007). Drilling fluid is continuously pumped down the drill stem and through the drill bit, both cooling and lubricating the bit and removing cuttings up the annular space between the drill stem and borehole wall, known as direct circulation rotary drilling (Price, 1996; Misstear *et al.*, 2006). The pressure of the drill fluid also helps maintain integrity of the borehole wall in unconsolidated materials (Keely & Boateng, 1987). Where the drilling fluid is drawn up the middle of the drill string and discharged to a sump or settling pit, this is known as reverse circulation rotary drilling (*Figure 3.3*) (Slattery, 1997; Misstear *et al.*, 2006) and is typically used in larger diameter boreholes than direct circulation rotary drilling (Clark, 1988).

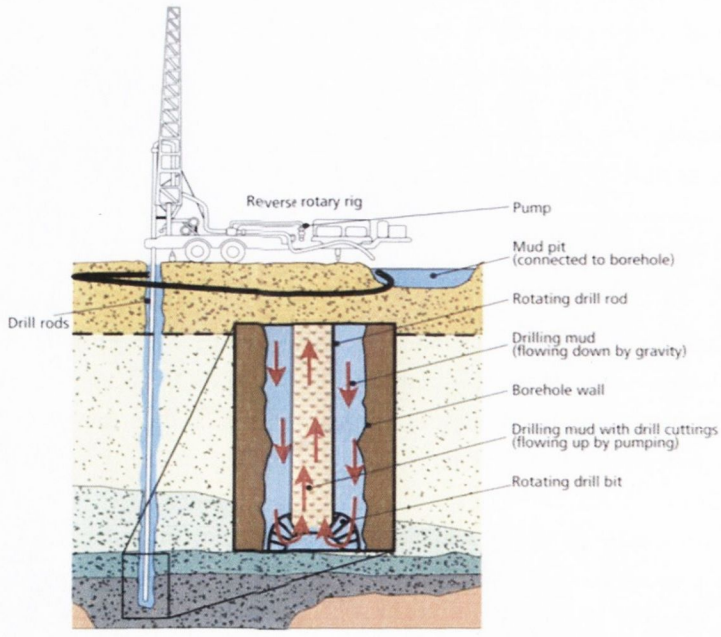


Figure 3.3 Reverse circulation rotary drilling (Midnight Sun Drilling, 2011)

3.4.5 Down-the-Hole (DTH) Hammer drilling

As described by Slattery (1997) and Misstear *et al.* (2006; from Rosen *et al.*, 2001), the hammer methods comprise both a rapid percussive action (1000-2000 strokes/minute) and a slow rotation (20-40 rpm), which ensure an even drilling face. The down-the-hole hammer method was developed, according to Cruse (1979), in the mining industries for drilling small diameter holes where water or mud were unsuitable as circulation fluids. Instead, compressed air is used as the drilling fluid, which is directed down the drill stem through openings in the drill bit (*Figure 3.4*). This method employs very similar rig and drill strings as those used in direct circulation rotary drilling. However, drill bit assemblies differ to allow the use of air as the drilling fluid (Clark, 1988). Water (or foam) is occasionally used to cool the bit and flocculate the drill cuttings, which facilitates extraction of the cuttings (Misstear *et al.*, 2006). Down-the-hole hammer drilling is particularly effective in crystalline and unconsolidated formations, with drilling rates up to 300 mm min^{-1} reported by Misstear *et al.* (2006).

A thorough overview of drilling techniques is provided by Misstear *et al.* (2006) and Bloetscher *et al.* (2007).

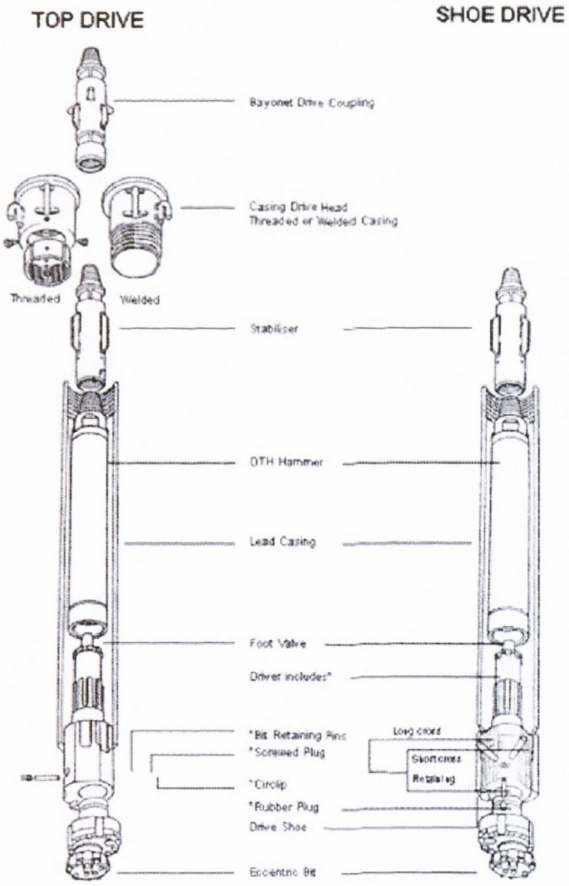


Figure 3.4 Down-the-hole hammer (DTHH) drill bits

(Techno Drill USA Ltd., 2011)

3.5 Well Design

Well design is the determination of the materials and dimensions to be used in the well, which will be restricted by the purpose and necessary yield of the well in addition to financial constraints. However, local hydrogeological settings will primarily govern these determinations. The following section outlines the well design considerations of utmost importance including well casing, well screens and gravel packs, pump type and well depth. It is important to note that the following sections (*3.5-3.11*) provide a summary of ideal well design practice, construction techniques and well completion, and do not accurately reflect current practice in Ireland. This is due primarily to the current lack of regulation in Ireland with regard to well/borehole design and construction. The International Association of Hydrogeologists (Ireland) have, as early as 1992, reported that this lack of regulation is considered the main cause of high levels of contamination in private groundwater sources in Ireland. Current design and construction practices with regard to private groundwater sources in Ireland have been investigated as part of this study, and are therefore considered in detail in Chapters 7, 8, 9 and 11.

3.5.1 Well Depth

The depth of a well is a function of both the intended yield of the well and the hydrogeological setting in which the well is to be located. For example, as previously mentioned (*Section 2.5.1*) the water bearing properties of crystalline aquifers are typically dependent on fracture/fissure flow i.e. secondary porosity, with these fractures/fissures often less abundant at greater depths (Davis & Turk, 1964; Todd & Mays, 2005). Well depth is also an economic consideration as typically, the cost of overall well construction is highly correlated with well depth (Maréchal, 2009). Gustafson (2002) report that, in drilling crystalline aquifers in Sweden, it is standard practice to abandon drilling at depths of 100 m, where a sustainable yield has not been established. Well design for depth in some thick, consolidated aquifers may be similar to that in crystalline aquifers due to a lack of data pertaining to the distribution and size of these water bearing fissures (Misstear *et al.*, 2006; Bloetcher *et al.*, 2007). Therefore, well design for depth in these areas is often based upon past drilling experience and existing boreholes. In unconsolidated aquifers (*Section 2.5.3*), shallower wells are more common due to higher watertables, with shallow hand-dug wells more common in unconsolidated aquifers.

Misstear *et al.* (2006) state that in aquifers of limited thickness, boreholes are typically drilled to provide full penetration of the aquifer in order to maximise yield, with well depths in thick aquifers being primarily based upon necessary yield and cost. In many areas, however, wells are drilled to greater depths than necessary for a sustainable supply in order to provide additional water storage in the well. Price (1996) notes that, although a borehole need not fully penetrate an aquifer, the total transmissivity of the aquifer will not be fully exploited if only a fraction of the formation is penetrated. Another notable point in regard to well depth is water quality. A number of studies, including Exner & Spalding (1979), Exner & Spalding (1990), Goss *et al.* (1998), Hudak (2000), Zimmerman *et al.* (2001), Tabbot & Robson (2006) and Gonzales (2008) have found significant relationships between groundwater quality and domestic well depth, with all the aforementioned studies typically reporting elevated levels of contamination (particularly faecal coliforms and nitrate) in shallower wells.

3.5.2 Well Casing

The casing houses the pumping equipment i.e. pump chamber, and serves as a conduit for water. It is a lining for the drilled (or dug) hole that maintains the open hole from the land surface to the water bearing formation and typically fulfils two purposes. The first is to provide structural reinforcement and stabilisation for the well. The second is the exclusion of contaminated water from the surface and/or formations above the aquifer into the well (Bloetscher *et al.*, 2007; IGI, 2007). Pump protection is another well casing function although this may be seen as being equivalent to structural reinforcement (Slattery, 1997). In the case of hand-dug wells, casing (concrete caissons or masonry) also provides protection to the well diggers against cave-in during construction (Davis & Lambert, 2002).

In order to provide adequate well reinforcement and stabilisation, the casing must possess sufficient strength to withstand all possible earth and hydrostatic pressures that may be encountered during completion, development and the working lifespan of the well (Roscoe Moss Co., 2006). This is achieved by suitable casing selection insofar as the casing material and dimensions must provide adequate strength (collapse, bending and bending) in the hydrogeological setting in which it is to be utilised. In deeper wells, it is preferable to have multiple casings i.e. a casing string (as opposed to single casings which should be restricted to shallow wells) which makes drilling easier in formations at risk of collapse. Multiple casings are also desirable in corrosive and/or contaminated aquifers (Misstear *et al.*, 2006; Bloetscher *et al.*, 2007).

3.5.2.1 Casing Material

Steel

Steel is the most commonly used material for borehole casing as it possesses the high yield and tensile strengths necessary (*Figure 3.5 a.*) (Roscoe Moss Co., 2006). It is available in a variety of grades and weights with the type to be used depending on local conditions: heavy high-grade carbon steel is most suitable for deep wells whereas lower-grade mild steel is often used for shallow wells and consequently, most commonly used in domestic well installations (Driscoll, 1986, Clark, 1988, Slattery, 1997). Steel casing must always be used where the casing is driven, pulled back or if the casing is installed in holes liable to collapse (Misstear *et al.*, 2006), with grade 1 mild steel casing preferable where casing is to be driven (Briody, 1995). The main advantages of steel are its high yield and tensile strength, high burst strength and suitability for welding (allowing for deeper casing strings and subsequent installation to greater depths) (Bloetscher *et al.*, 2007). Additionally, steel's suitability for welding expedites appropriate field installation (Roscoe Moss Co., 2006).

Disadvantages of steel are its susceptibility to corrosion from galvanic and microbial contamination, making it potentially unsuitable in aquifers with high chloride concentrations or microbial populations. These conditions may be counteracted by using stainless steel which has the same advantages of carbon steel but limits corrosion problems (Driscoll, 1986; Bloetscher *et al.*, 2007). Stainless steel is however significantly more expensive than mild/carbon steel and can be more prone to microbial attack (Misstear *et al.*, 2006; Bloetscher *et al.*, 2007).

Thermoplastic

There are a number of polymers used for water well casing including Acrylonitrile butadiene styrene (ABS), Polyvinyl chloride (PVC) and rubber-modified polystyrene (SR), Polypropylene and

Polytetrafluoroethylene (PTFE) (Kurt & Johnson, 1982; IGI, 2007). Thermoplastics are now widely used in shallow aquifers as they are cheap and highly resistant to corrosion (*Figure 3.5a.*) (IGI, 2007). Thermoplastic casings have been used since the late 1940s in extremely corrosive waters with their use growing significantly since the late 1960s with the development of stronger materials and advances in extrusion technology (Driscoll, 1986). Other advantages of thermoplastic casings are their ease of installation and they are lightweight. Plastic casing is more flexible than steel, however their strengths are significantly less than that of steel (see Appendix B1). A sudden formation collapse can cause plastic casing to break. Additionally, plastic casing can be cracked/crushed by surface shocks, combined pressure and setting temperature of cement grouting, external hydrostatic pressures during well development or maintenance, when excessive drawdown may occur (Clark, 1988; Misstear *et al.*, 2006; IGI, 2007), particularly at elevated temperatures (Johnson *et al.*, 1980). Therefore, physical strength considerations limit the use of the majority of polymeric well casing materials to relatively shallow wells. If it protrudes above ground level, the casing must be carefully protected, as it may be easily damaged by vehicles or moving equipment. Long term exposure to sunlight can also reduce the impact strength of the material. With regard to contaminated aquifers, plastics can be permeable in the presence of certain chemicals and therefore thermoplastic casing may not be suitable for use in areas prone to groundwater contamination. PVC is degraded by several neat organic solvents (Plastics Design Library, 1994), and high concentrations of these chemicals in aqueous solution can also degrade PVC (Vonk, 1985; Ranney & Parker, 1997). Plastics are used widely in the Middle East (Misstear *et al.*, 2006) and are recommended for use in Ireland due to their non-corrosive properties (IGI, 2007).

Fibreglass

Glass-reinforced-plastic (GRP) has, according to Clark (1988), the advantages of being both relatively strong and corrosion resistant, being significantly stronger than plastic casings. GRP has however significantly weaker collapse and tensile strengths than steel and for this reason is not generally suitable for use in deep wells being both weaker and more fragile than steel. Additionally, fibreglass products 'have not been received as favourably for potable water well construction due to perceived and actual problems with fibreglass sloughing' (NGWA, 1998). As with plastics, heat can significantly reduce the collapse resistance of fibreglass casing, which is a concern in the case of heat caused by grout curing (Johnson *et al.*, 1980), however, for most water well applications, this reduction is minimal. Fibreglass is slightly permeable allowing gradual ingress of water, thus contamination can occur in formations where ubiquitously contaminated water is cased off above potable water (Slattery, 1997). Fibreglass is commonly found in Pakistan and Bangladesh, particularly in irrigation and drainage wells (Misstear *et al.*, 2006), and in areas where groundwater is known to be corrosive (Roscoe Moss Co., 2006), but has not been widely used in Ireland.

Concrete

The lining of hand-dug wells is most commonly of pre-cast concrete rings (caissons) or masonry (Davis & Lambert, 2002) due to the larger diameters which are necessary in order to accommodate well diggers or well digging equipment (*Figure 3.5 b.*). The use of concrete or masonry casing in bored wells however is extremely rare (Misstear *et al.*, 2006). Two types of concrete rings are used for lining hand-dug wells, permeable and impermeable. In soft formations where the water table is relatively shallow, the aquifer is lined with permeable caissons, while above the water table impermeable caissons are used (Davis & Lambert, 2002; Misstear *et al.*, 2006). Caissons are usually

0.9-1.5 m in diameter, 0.5-1 m high and 50-100 mm thick (Davis & Lambert, 2002). As they are usually transported on-site, steel or other reinforcement materials such as ferro-cement or polypropylene strands are often incorporated in the concrete to ensure structural integrity during transport. Disadvantages of concrete casings are their weight and consequent handling difficulties (Roscoe Moss Co., 2006). Additionally, moulded blocks, bricks and stone masonry (*Figure 3.5 c.*) are all suitable for in-situ lining of stable excavations, however as their use is atypical in Ireland, they will not be discussed here. Tables outlining typical collapse and tensile strengths of casing materials are presented in Appendix B1.



Three well types; (a) steel lined borehole with thermoplastic inner liner, Co Wicklow, (b) concrete lined hand-dug well Co Westmeath and (c) stone lined hand dug well Co Offaly

Figure 3.5 Well types utilising various well casing materials

3.5.2.2 Casing Diameter, Depth and Jointing

Typically, for domestic boreholes, the casing diameter selection is determined by pump size, which is based upon the required discharge rate. There should be adequate clearance between the pump and well casing to accommodate both installation and maintenance where necessary (Driscoll, 1986). Misstear *et al.,* (2006) suggest that while 25 mm clearance may prove adequate, 50 mm clearance is desirable (i.e. a casing with an ID 100 mm greater than the maximum pump OD), which allows for ample flow around the pump motor, resulting in higher pump efficiency. Bloetscher *et al.,* (2007) advise that when selecting a casing diameter, the minimum casing and screen diameter should be no less than one casing-size greater than the maximum pump OD, for maximum hydraulic efficiency of the well. Driscoll (1986) in turn recommends that the casing diameter should be two pump sizes greater than the nominal diameter of the pump. However, as set out by Misstear *et al.,* (2006), greater casing diameters will increase the overall cost of the well and therefore a balance must be sought between adequate yield, well size and well cost. In cases where greater discharge rates are sought, increasing the well diameter, and consequently increasing intake area, is in most cases not financially advisable. As Price (1996) points out, in many cases doubling the well diameter may only decrease drawdown by 10% for the same yield.

Another important consideration with regard to well design is casing depth. Where a well has been constructed to maximise specific capacity and consequent well yield, it is important to determine the appropriate casing depth at which the well is protected from contamination and ensure highly productive waterbearing zones are not cased off. Slattery (1997) maintains that the casing depth (or length) is a function of the screen length, which will largely depend on aquifer type. In deep consolidated aquifers and crystalline aquifers, it is recommended that the grouted surface casing should extend several metres into the solid rock to ensure an adequate seal against surface contamination, with screen sections being incorporated into the string if highly fissured shallow zones are to be exploited (Misstear *et al.*, 2006) i.e. the well should be watertight to such depth as necessary to exclude contaminants. AWWA (2003) recommend that in unconsolidated aquifers the well casing should extend to 8-9 m, with the well screen being set below this depth, in order to provide adequate protection. However, where a well is developed at less than the recommended depth in unconsolidated materials, a greater level of wellhead protection is necessary.

A protective well casing string shall have watertight joints throughout its length to ensure the well is impervious to potentially contaminating layers. Misstear *et al.* (2006) outline the three most commonly used jointing systems, as follows:

- Threaded couplings (steel and plastic casing)
- Welded joints (steel casing; plastic casing may be 'solvent-welded')
- Spigot and socket with key locks (plastic and fibreglass casing)

3.5.3 Well Screens and Gravel Packs

The well screen is a type of casing which allows for the controlled flow of water from the aquifer into the well through a series of slots, which may run vertically or horizontally (IGI, 2007). A gravel or filter pack, as described by Ives & Coad (1987) and Price (1996) is an envelope of coarse sand or fine gravel (2-6 mm diameter), preferably silaceous rather than calcareous (IGI, 2007), placed between the borehole wall and screen used to settle-out fine grained particles that may otherwise enter the well and to increase the effective hydraulic diameter of the well. The purpose of the screen is to allow water to flow into and up the well efficiently, without incurring large head losses, while preventing sand and other fine materials from clogging the well or damaging the pump (Misstear *et al.*, 2006). The main factor driving the necessity of well screen use is typically the aquifer type in which the well is sited. As set out by Misstear *et al.* (2006) and IGI (2007), screens and gravel packs are generally not required in crystalline aquifers and their use in consolidated aquifers is frequently unnecessary. They are however, always required in unconsolidated aquifers in order to exclude unconsolidated sediments from the well.

There are three main methods of screened well intake completion, outlined by Slattery (1997) as follows:

1. Artificial filter packs
2. Natural filter packs
3. Formation stabilisers

Driscoll (1986) defines artificial filter packing as the removal of a proportion of formation material surrounding the well screen and subsequent replacement with pre-graded, permeable materials. No concise definition for natural filter packing is provided, however, formation stabilisers are defined as materials with no specific filtration requirements. As the majority of filter packs used in Irish boreholes are of the artificial variety, these are discussed here.

Well Screen and Gravel Pack Selection

Screen selection is an important well design criterion, as they must fulfil all of the following (Driscoll, 1986; Slattery, 1997; Misstear *et al.,* 2006; Roscoe Moss Co., 2006):

- Permit the entry of water;
- Prevent the entry of sediment by retaining the filter pack (where utilised) and stabilise both the aquifer and filter zone;
- Lend structural support to the borehole

There are a variety of well screen types, including continuous slotted screens, torch slotted screens, machine slotted screens, bridge slot screens and louvred screens (Slattery, 1997). The major design differences between the various screen types are their respective slot widths and consequently the open area provided by the screen (*Figures 3.6 & 3.7*). These criteria are of importance because they directly affect the overall well yield and are typically designed to allow the maximum amount of water in with minimal entry of formation sediments, with screens manufactured with various slot sizes and shapes, to match the characteristics of the aquifer (AMAFRD, 2001). Plastic screens are the most widely used because they are strong and relatively able to withstand corrosive water (Misstear, *pers. Comm.,* 2011). The total open area of the screen is additionally dependent on the length and diameter of the screen. Typically, screen diameter is determined by the casing diameter and therefore the major design decision after screen type has been established will be the screen length utilised. A number of borehole liner and screen assemblies are illustrated in *Figure 3.6.*

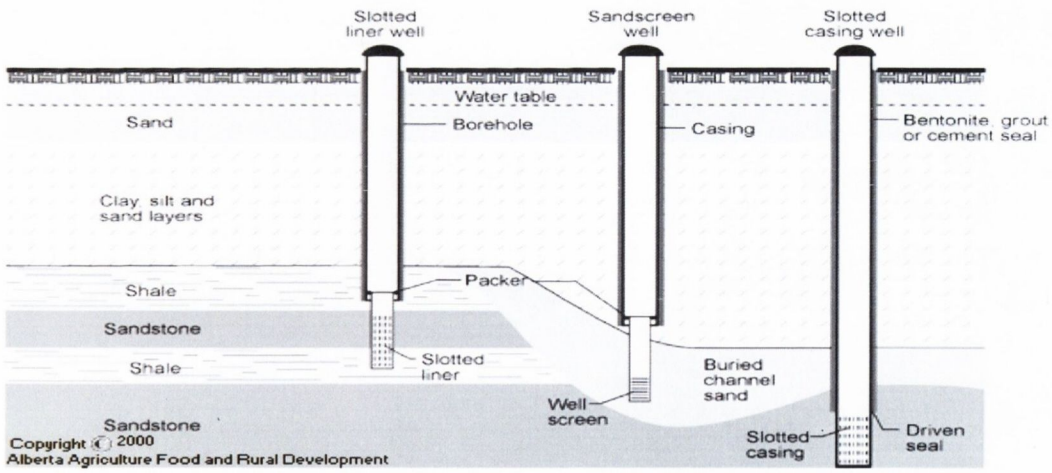


Figure 3.6 Example of three well liner and screen assemblies (AMAFRD, 2001)

The screen slot size must also account for the gravel pack, where used, as the screen size must also prevent the gravel pack itself from passing through and into the well. Briody (2007) and Bloetscher

et al. (2007) state that screen openings should be selected to ensure that >90% of gravel pack remains after well development. Both Misstear *et al.* (2006) and Bloetscher *et al.* (2007) advocate the use of grain size curves for gravel pack size selection; with Price (1996) recommending that the gravel pack grain size should be 5-10 times that of the natural aquifer sediments. Bloetscher *et al.* (2007) recommend a figure of 4-6 times that of the natural aquifer sediments.

In a borehole that is to be cased and screened with a single casing string, once it has been drilled to the final depth and the aquifer boundaries and productive zones have been located, the casing and screen string are assembled on the surface in manageable lengths. These are then lowered down the borehole and positioned. The next string length is then connected (solvent welded, welded, etc) and subsequently lowered. This continues until the relevant well screens are in place i.e. productive sections (Misstear *et al.,* 2006; Bloetscher *et al.,* 2007).

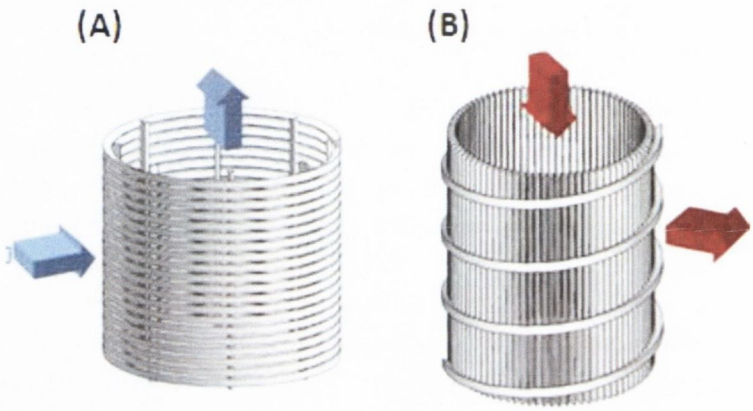


Figure 3.7 Example of well screen types; (A) Standard Screen Construction (External Circumference Wire) and (B) Rerolled Screen Construction (Internal Axial Wire) (Completion Products Ltd, 2011)

The gravel pack, once selected, in the case of hand-dug wells or shallow boreholes, is typically poured into the annular space between the well screen and aquifer/borehole side, with the pack retaining aquifer particles in its outside boundary. Although Price (1996) maintains that in theory the gravel pack need only be a few grains thick to function effectively, generally a gravel pack thickness of 60-80 mm is utilised, with IGI (2007) recommending a minimum thickness of 50 mm. In the case of deeper boreholes, the gravel pack is transported via tremie pipe to the annular space, evenly around the screen in 0.6-1.2 m layers, with the casing and tremie being raised as required (Bloetscher *et al.,* 2007). The entire screen should be completely surrounded by the gravel pack, extending 0.5-1.0 m over the highest screen height (Bloetscher *et al.,* 2007).

3.5.4 Grout Seal

Sealing the well protects the well's producing zone from contamination, particularly microbial contaminants from surface water entry. The diameter of the borehole is larger than the casing being installed and therefore, boreholes that remain unsealed below the final design depth may create a vertical conduit for preferential flow. Hence, grouting of water wells is commonly undertaken to increase the well life and to protect the water supply (Johnson *et al.,* 1980). The annulus (space

between borehole wall and casing exterior) is typically filled with a thin expandable concrete (Price, 1996) above the screen to the surface around the well (Cohen *et al.,* 1986). Grouting also provides structural integrity once the permanent casing has been placed, while sealing off the formation and providing protection against deterioration of the casing as a result of microbial activity (Bloetscher *et al.,* 2007). It is imperative that the annulus has adequate clearance for grouting, with Misstear *et al.* (2006) recommending a minimum clearance of 50 mm (after NGWA, 1998). In shallow, hand-dug wells (<10 m), grout may simply be poured directly from the surface or by using a bailer (Bloetscher *et al.,* 2007). In the case of deeper bored wells however, this is not an adequate grouting method as uniform grout placement is unlikely to result. In deeper wells, it is recommended that the well be pressure grouted from the ground surface to the rock through a tremie pipe (Bloetscher *et al.,* 2007), with the tremie pipe initially set at the bottom of the annulus and raised as grouting advances until grout becomes evident at ground surface.

Bloetscher *et al.*(2007) state that typically, three materials are used for grouting; concrete - sand cement or neat cement grout - with admixtures such as bentonite (an absorbent aluminium phyllosilicate clay which can expand 10-15 times its original volume) often added to reduce shrinkage, increase viscosity and reduce permeability (Lehr *et al.,* 1988). As outlined by Kurt & Johnson (1982), the permeability of grout and cement mixes is an important design parameter, particularly in well design, as the permeability of the mix directly effects the efficacy of the sanitary seal and therefore the potential human health hazard posed by the well. Additionally, Christman *et al.* (2002) conclude that the structural stability of the sealant used is important in quantifying the overall effectiveness of the well seal.

The necessary length of grout seal will vary according to aquifer type and well application (Misstear *et al.,* 2006), with the current IGI guidelines (2007) recommending the casing and grout depths (*Table 3.2).*

Table 3.2 Recommended Casing and Grout Depths (IGI, 2007)

Depth to bedrock (m)	Minimum Depth of Casing and Grout
<10	20 m
>10	10 m into bedrock
Bedrock not met (Unconsolidated Aquifer)	> 6 m, and as deep as possible up to 20 m

3.5.5 Pump Type

There are typically two main pump types used for groundwater abstraction; variable displacement pumps and positive displacement pumps (Driscoll, 1986; Misstear *et al.,* 2006). Variable Displacement Pumps (also known as rotodynamic pumps) are characterised where the pumping head has an inverse relationship to the discharge rate (Chadwick & Morfett, 1986) and include centrifugal pumps (vertical turbine pumps, submersible pumps and suction lift pumps), jet pumps and air-lift pumps. The most important category of variable displacement pump is the centrifugal pump (Misstear *et al.,* 2006) due to an abundance of electrical power and spare parts (Davis & Lambert, 2002). As outlined by Price (1996), centrifugal pumps may be used both at the surface or

submersed below the water table. Where installed at the surface, even where a pump is capable of creating a "perfect vacuum" i.e. reduce the pressure to zero, atmospheric pressure would raise the water to a maximum height of approximately 10 m above the water level. Due to mechanical limitations and the process of cavitation, most centrifugal pumps cannot lift where the water level is greater than 7 m below the pump. Therefore, surface mounted centrifugal pumps are typically used to draw water from shallow hand-dug wells, whereas, lifts greater than 7 m are typically undertaken by submersible centrifugal pumps, containing an impeller arrangement and waterproof motor (Chadwick & Morfett, 1986; Bloetscher *et al.,* 2007). A pump may include a number of impellers arranged in series (or parallel); with each impeller in effect operating as a centrifugal pump, thereby increasing pumping head (Price, 1996). Turbine well pumps are also available, which have a vertical shaft motor connected to a drive shaft, which operates the suspended centrifugal pump (Chadwick & Morfett, 1986; Bloetscher *et al.,* 2007). These are not commonly used in Ireland. Surface mounted jet pumps are used for low-yielding boreholes for single domestic supplies (IGI, 2007). These pumps use a water jet propulsion system in order to lift water, however again atmospheric pressure limits pumping head to 7-8 m (Chadwick & Morfett, 1986).

Positive Displacement Pumps are characterised by the discharge (Q) remaining relatively constant at a given pump speed, not depending strongly on the pumping head (Misstear *et al.,* 2006). Positive displacement pumps are typically used for groundwater sampling as opposed to domestic (or public) groundwater abstraction and include rotary pumps, peristaltic pumps and piston pumps. They are therefore not described in detail here.

3.6 Well Development

When well construction has been completed and final casing/screen assembly installed, it is necessary to clean and develop the well (IGI, 2007). The main objectives of well development are the removal of wall smear or mud cake derived from bentonite based muds/drill fluids and the removal of the fine fraction from both the borehole and surrounding aquifer (Slattery, 1997), thereby bringing the well to its maximum efficiency. Misstear *et al.* (2006) state that, although biodegradable polymer-based muds may be employed during construction, their use does not eliminate the need for appropriate well development. The common methods for developing wells generally depend on either physical or chemical techniques to "wash" the well face and mobilize material removal from the well and surrounding aquifer. These are variously described by Driscoll (1986), Aller *et al.* (1991), Bloetscher *et al.* (2007) and Misstear *et al.* (2006), all of which if carried out correctly may improve abstraction efficiency and overall yield. These include:

- Pumping and overpumping,
- Backwashing,
- Surging (moving a tightly fitting plunger up and down the well),
- Bailing (moving a bailer up and down the well),
- Jetting (circulating a high pressure water jet in the well),
- Airlift pumping (discharge of silty water under air pressure through an annular inductor pipe),
- Air Surging,
- Mud dispersants,

- Acidisation,
- Hydrofracturing,
- Polyphosphate dispersion,
- Wall scratching.

The most commonly used methods employ hydraulic pressure to draw loose sediment from the well wall, well screen, gravel-pack and aquifer matrix into the well, allowing for subsequent removal by pumping, airlifting (*Figure 3.8*) or bailing. These techniques include backwashing, surging, bailing, jetting and airlift pumping (Clark, 1988). Other techniques, such as acidisation, are used in carbonate (commonly limestone or chalk) aquifers, whereby hydrochloric or sulfamic acid is used to dissolve calcium carbonate ($CaCO_3$) which may be blocking fissures, thereby decreasing drawdown and increasing well yield (Misstear *et al.,* 2006). Hydrofracturing, which involves the injection of chlorinated water under pressure (5,000 – 20, 000 kPa) into discrete well sections in order that waterbearing fractures may be opened and maintained, may be combined with acidisation, particularly in the case of low permeability carbonate aquifers (Banks *et al.,* 1996; Misstear *et al.,* 2006).

3.7 Wellhead Completion

As set out by IGI (2007), borehole wellhead completion (headworks) is of primary importance for the exclusion of contaminants and should be completed above ground where possible, although, conversely, GSNI (2006) recommends below ground wellhead completion for reasons such as land shortage and visual intrusion. Gaut (2005) found that wellhead completion was one of the contributing factors to microbial contamination of water supply wells in Norwegian crystalline aquifers and echoed the IGI recommendations (2007) by advising wellhead completion not less than 400 mm above ground level (*Figure 3.8*).

A surface concrete pad (or apron) is recommended (IGI, 2007; Bloetscher *et al.*, 2007), as a measure used to minimise surface contamination of the well (*Figure 3.9*). IGI (2007) recommend this protective pad be installed around the well casing to a depth of 500 mm, a radius of 500 mm and at a gradient of 1:10 to allow surface water to drain away from the wellhead. It is also recommended that it is extended to a radius of at least 2 m around the wellhead at a minimum thickness of 150mm. Alternatively, where a wellhead chamber (housing) is utilised, 25 mm drainage pipes should be installed to prevent ponding directly adjacent to the wellhead (AMAFRD, 2001). A protective steel casing should protrude above the concrete pad and be suitably covered. Additionally, a manhole cover should be utilised where possible, particularly where the wellhead is completed below ground level, to protect from vehicular traffic (*Figures 3.8 & 3.9*).

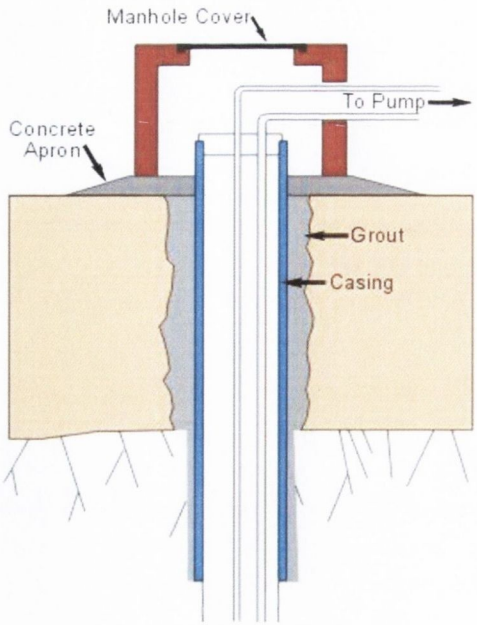


Figure 3.8 Recommended above ground wellhead completion (Gaut, 2005)

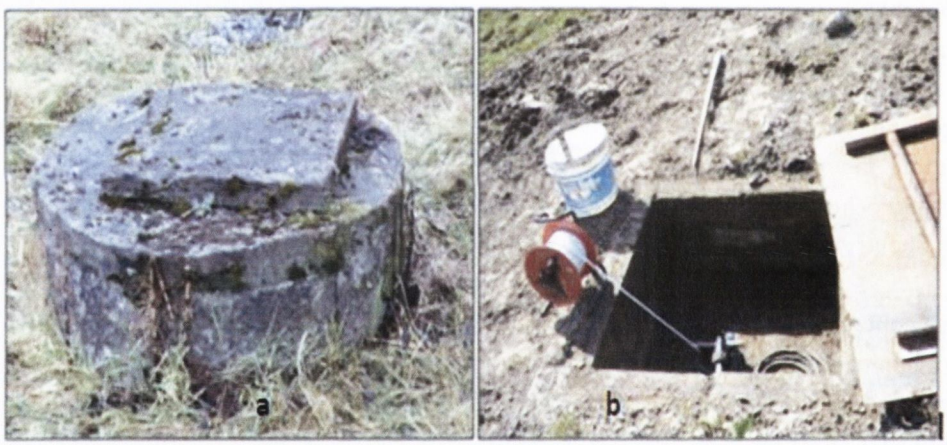


Figure 3.9 Typical wellhead completion in Ireland; (a) hand-dug well finished above ground Co Meath and (b) borehole finished below ground Co Westmeath

3.8 Disinfection

Disinfection of the well is the last step in well construction and is recommended to ensure a safe groundwater supply (Macler & Merkle, 2000), with Lehr *et al.* (1988) suggesting that this procedure is often neglected by drilling contractors and/or well diggers. Craun *et al.* (1997) reported that of all waterborne disease outbreaks in the US during the period 1971-1994 attributable to private groundwater sources, inadequate disinfection and lack of disinfection were cited in 28% and 53% of outbreaks, respectively.

In addition to groundwater contamination as a result of anthropogenic activities such as wastewater disposal and agriculture, drilling/digging operations may introduce bacteria into the well by tools,

drilling mud, the casing and column pipes, and the screen and therefore disinfection after construction and sanitary protection is imperative (AWWA, 2003; Bloetscher *et al.*, 2007). A number of guidelines are available for recommended well disinfection procedures including HSE South East (2000), AWWA (2003), Bloetscher *et al.* (2007) and CNA Environmental Inc (2005), with the majority of disinfection methods involving chlorine in some form, typically hypochlorite (Westerhoff *et al.*, 2005; IGI, 2007).

3.9 Yield testing

A well pump test or yield test should be carried out after cleaning, development and disinfection of the well and as set out by IGI (2007), the well pumping test should:

- Confirm that the well can produce water of sufficient quality and quantity to meet the consumer needs, allowing for a factor of safety for dry periods,
- Determine the maximum sustainable yield of the well, and
- Provide baseline data on well performance.

The pump test procedure for new wells should consist of:

1. A step-drawdown test, to assess individual well performance with regard to yield and adjacent aquifer properties, followed by;
2. A constant-rate test to further assess hydraulic properties of the aquifer.

Tests are undertaken by measuring the static water level in the well, after which the well is pumped at a maximum rate until the water level in the well stabilizes. Depth to the watertable is noted, with the difference in depths being the drawdown, and the discharge-drawdown ratio used to estimate the specific capacity of the new source (Todd & Mays, 2005). A number of published procedures exist regarding the appropriate employment of both the above tests, including Driscoll (1986), Helweg (1994), Boonstra & Kselik (2001), IGI (2007), and Piscopo & Summa (2007).

3.10 Well Maintenance and Rehabilitation

As outlined by Misstear *et al.* (2006), preventative well maintenance is the process of periodic upkeep of the well and its components, including cleaning, which is undertaken in order to maintain the well at or near its original performance. Conversely, well rehabilitation (or repair) is the restoration of a supply to its original performance after a period of disuse or neglect. Misstear *et al.* (2006) also point out that techniques used for maintenance and rehabilitation are often similar, with the primary differentiation between the two being the frequency with which they are carried out. The results of two separate surveys, presented by Howsam (1990) detail the most commonly encountered well problems requiring well maintenance (*Table 3.3*).

Table 3.3 Commonly encountered well problems requiring maintenance (Howsam, 1990)

Problem	Positive Responses	Total (%)
Sand pumping	63/169	32.7
Hydrogen sulphide (H_2S)	54/169	31.9
Corrosion	47/169	27.8
Iron bacteria	39/154	25.3
H_2S & Sand pumping	22/154	14.2
H_2S & Corrosion	20/154	12.9
Fe/Mn clogging	14/169	8.2
Mineral clogging	9/169	5.3
Fe/Mn & H_2S	4/154	2.5

Some of the problems outlined in *Table 3.3* may result in reduced hydraulic performance of the well system including well casing, screens, pump and distribution lines, reduced water quality, loss of structural integrity of well components and increased water costs due to one or more of the aforementioned factors. An exhaustive list of well rehabilitation and maintenance aims and associated methods is presented by Howsam *et al.* (1995) and Misstear *et al.* (2006).

3.11 Abandoned Wells

Due to recent trends of urbanisation, reduced numbers of small and medium-scale agricultural enterprises, and the extension of public water systems to rural areas, a large number of abandoned wells are likely to be found throughout Ireland. As outlined by Gass *et al.* (1977) and Scherer & Johnson (2007), abandoned wells comprise a number of potential hazards, including:

- Safety – poorly covered large diameter wells pose a safety hazard to both humans and animals,
- Groundwater contamination – abandoned wells comprise a preferential flow conduit from the ground surface directly to the associated aquifer, without filtration or amelioration by soils or subsoils (*Figure 3.10*)
- Mixing of groundwater – in the case of a well connecting two or more aquifers i.e. deep and shallow aquifers, a conduit such as an abandoned well may allow contamination to cross from one aquifer to another. In the case of coastal aquifers, saline intrusion from a deep or high pressure aquifer may be transported to a shallow aquifer, rendering it undrinkable (Nordbotten *et al.*, 2004),
- Loss of pressure head – in the case of artesian or high pressure aquifers, pressure head may be diminished by water discharging to unsealed abandoned wells, leading to watertable decline.

Therefore, all abandoned wells should be appropriately decommissioned, which entails restoring the geological conditions that existed before the well was drilled. Backfilling the well casing with cement grout, concrete, bentonite or clay is advised, however, a number of procedures pertaining to specific aquifer formations are available (Water Systems Council, 2003; Nordbotten *et al.*, 2004; Texas

Groundwater Protection Committee, 2010). Currently, no standard procedures exist for decommissioning abandoned wells in Ireland.

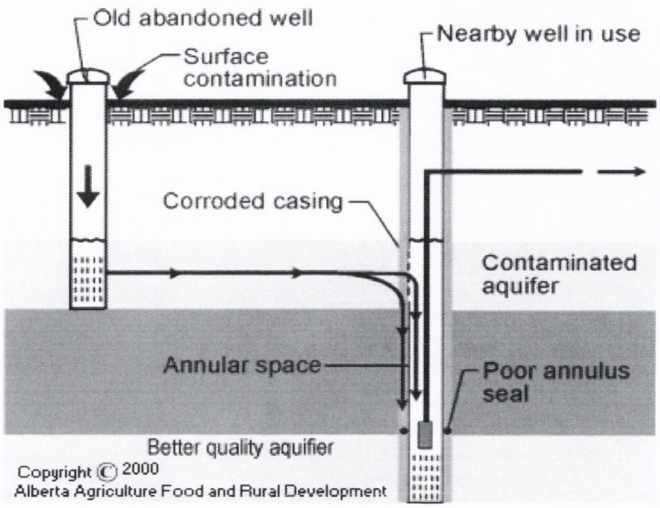


Figure 3.10 Example of abandoned well as a potential contamination source (AMAFRD, 2001)

3.12 Water Treatment

The following section outlines the most commonly used water treatment processes, their modes of action and their reasons for use.

3.12.1 Ion Exchange

Water hardness is primarily caused by a high mineral content, typically multivalent cations, with the most common causes of hard water being calcium (Ca^{2+}), magnesium (Mg^{2+}), bicarbonates (HCO^{3-}), and to a lesser extent sulphates (SO_4) and metallic cations. The main causes of hardness in groundwater are calcium carbonate ($CaCO_3$) from limestones (particularly in Ireland) or chalk; or magnesium predominantly from dolomite ($CaMg(CO_3)_2$). Calcium sulphate ($CaSO_4$) from mineral deposits is another potential cause of hardness.

Hardness is generally not considered a health issue, although previous research has provided both positive and negative correlations between hardness and a number of adverse human health effects including rectal cancer (Yang *et al.*, 1999), atopic eczema (McNally *et al.*, 1998), cardiovascular disease (Marque *et al.*, 2003) and kidney stones (Bellizzi *et al.*, 2000). In Ireland however, hardness is generally perceived as being an aesthetic issue, with hardness causing scaling of pipes and fittings (Masters & Ela, 2008), and is widespread, particularly in limestone areas, which as previously mentioned, comprise approximately 50% of bedrock units.

During ion exchange, both calcium and magnesium cations, the primary causes of water hardness, are targeted during the ion-exchange process (Holl & Flemming, 1991, as cited in Vaaramma & Lehto, 2003). During the process, raw water is passed through a column containing solid synthetic resin beads. The beads are comprised of ion exchange resins, recharged by sodium or potassium

(typically sodium in the form NaCl or potassium in the form KCl), and as the raw water passes through, the hardness mineral ions are preferentially absorbed, replacing them with sodium or potassium ions, thereby forming soluble salts. It is important to note that ion-exchange processes generally do not affect pathogen or microbial indicator populations in raw water.

3.12.2 Chlorination

The most commonly used method of disinfection of public water supplies and group water schemes, is free chlorine disinfection (Virto *et al.*, 2005). This is primarily due to it being cheap, relatively reliable and easy to use, with the main sources of free chloride being chlorine gas ($Cl_{2(g)}$), sodium hypochlorite (NaOCl) and calcium hypochlorite ($Ca(OCl)_2$) (Masters & Ela, 2008). These react in solution to form dissolved chlorine gas ($Cl_{2(aq)}$), hypochlorous acid (HOCl) and hypochlorite (OCl^-), forming so-called free chlorine.

Hypochlorous acid is a powerful nonselective antimicrobial substance due to its potential oxidizing capacity. At the concentrations used, it is difficult for microorganisms to survive. Its primary mode of action in cellular inactivation is interruption of pathogen metabolic processes through reaction with subcellular compounds (Dukan & Touati, 1996). It is particularly effective at disrupting Gram-negative bacteria i.e. *E. coli, Shigella, Salmonella,* by selectively and rapidly inhibiting cell division, changing membrane permeability, fragmenting proteins and reacting with nucleotides (Dukan & Touati, 1996; Venkobachar *et al.,* 1997; Virto *et al.,* 2005)

A principal advantage of chlorination of drinking water supplies is the formation of a chlorine residual after primary disinfection, which provides a form of secondary bacterial disinfection , resulting in inhibition of bacterial growth in distribution system after treatment (Neden *et al.,* 1992), preventing potential contamination in the water distribution system (Masters & Ela, 2008).

It is important to note however, that although chlorination is particularly effective in limiting the growth of heterotrophic bacteria, including coliforms, effectiveness is significantly less with regard to protozoan cysts and viruses (Loret *et al.,* 2005). Additionally, previous studies have confirmed that elevated levels of chlorine in solution, may lead to the formation of halogenated disinfection by-products (DBP), including brominated and chlorinated acetonitriles (Bull *et al.*,1985), trichloroethylene (Herren-Freund *et al.,* 1987), trihalomethanes (Dunnick & Melnick, 1993) and haloacetic acids (Masters & Ela, 2008); all of which are causes of public health concern. Masters & Ela (2008) maintain that concerns regarding the formation of free chlorine DBPs have been the main driver behind much work on alternative primary disinfectants including chlorine dioxide (ClO2) and ozone (O3) (Batterman *et al.*, 2000). Chlorination is rarely employed in private domestic settings i.e. private groundwater sources in Ireland.

3.12.3 Membrane Filtration/Separation

Membrane filtration of raw water, both surface and groundwaters, is a generalised term for a number of pressure-driven membrane processes, which are increasingly used for drinking water treatment (Jacangelo *et al.*, 1997). Traditionally, these processes are categorised into five separation processes on the basis of the membrane size range/pore space and the type of contaminant removed; namely conventional *media filtration, microfiltration (MF), ultrafiltration (UF),*

nanofiltration (NF) and *reverse osmosis (RO)* (Van der Bruggen *et al.*, 2003a; Masters & Ela, 2008). Masters & Ela (2008) offer a more intuitive classification, with *"particulate separation processes"*, including MF and UF, designed to exclude particles, including pathogens, and *"solute separation processes"*, including NF and RO, designed to reject dissolved substances such as salts and hardness. *Table 3.4* outlines these processes and their respective characteristics.

Table 3.4 Membrane Filtration Process Pore Size Ranges and Contaminant Separation (From Taylor & Wiesner, 2000, as cited by Masters & Ela, 2008)

Separation Process	Media Filtration	Microfiltration (MF)	Ultrafiltration (UF)	Nanofiltration (NF)	Reverse Osmosis (RO)
Pore Size Range (µm)	$3x10^{0} - 3x10^{2}$	$5x10^{-2} - 5x10^{0}$	$9x10^{-4} - 8x10^{-2}$	$1x10^{-4} - 3x10^{-3}$	$2x10^{-5} - 8x10^{-4}$
Contaminants Excluded	Bacteria, Algae, Cysts, Silt, Sand, Clays	Bacteria, Algae, Cysts, Clays, Viruses	Viruses, Humic Acids, Aqueous Salts, Metal Ions	Humic Acids, Aqueous Salts, Metal Ions	Aqueous Salts, Metal Ions

All the membrane filtration processes mentioned above employ semi-permeable barrier technologies, which allow the passage of water and some of its constituents, while stopping or severely restricting the passage of other constituents (contaminants) (Van der Bruggen *et al.*, 2003b).

Reverse osmosis is somewhat different to MF, UF and NF (Kiely, 1997), although Drewes *et al.* (2003), characterise RO and NF as being operationally similar with analogous applications. A semi-permeable membrane is used which, as shown in Table 3.5, effectively separates dissolved salts and inorganic molecules from solution, making it suitable for desalination in some cases (Fritzmann *et al.*, 2007). RO membranes do not have identifiable pores as in MF, UF or NF (Kiely, 1997). By applying a pressure difference across a semi-permeable polymer membrane, which although highly permeable for water, has a very low permeability for dissolved substances, the water contained in the feed is forced through the membrane (Meerganz von Medeazza, 2004). In order to overcome the "feed side" osmotic pressure, relatively high feed pressures are required (Fritzmann *et al.*, 2007), making the process relatively expensive on a domestic scale, which have limited its use in Ireland.

3.12.4 UV Disinfection

In recent years, UV disinfection for the inactivation of pathogens from drinking water supplies has been increasingly used. It was utilised for the first time in water treatment in Ft. Benton, MT, USA in the early 1970s (Wolfe, 1990). UV light has been shown to effectively inactivate a number of waterborne pathogens, including *E. coli* 0157:H7 (Tosa & Hirata, 1999), *Giardia* spp. (Craik *et al.*, 2000), *Cryptosporidium* spp. (Craik *et al.*, 2001), and enteric viruses (Shin *et al.*, 2005), with the limited formation of disinfection by-products considered a major advantage of the process (Peldszus *et al.*, 2004). Chang *et al.*, (1985) report a significantly narrower necessary UV dose range for the

inactivation of pathogens when compared with chlorine disinfection. Additionally, *Cryptosporidium* spp. are considered highly resistant to conventional chlorination (Shin *et al.,* 2001; Betancourt & Rose, 2004) and therefore UV disinfection is now the primary disinfection process with regard to waters likely to be contaminated with this protozoan pathogen.

The primary mode of action of UV disinfection of waterborne pathogens is the absorbance of UV photons by DNA molecules in the range 200-390nm, with peak absorption occurring in the shorter range of 250-280 nm and at approximately 260nm (Kiely, 1997; Zimmer & Slawson, 2002). UV absorption damages the pathogen DNA by altering nucleotide base pairing, creating new linkages on the DNA strand, which if unrepaired results in cessation of DNA replication and ultimately cell death (Zimmer & Slawson, 2002). For the purposes of pathogen inactivation of drinking water supplies, low pressure mercury UV lamps are most often used, causing output at approximately 254 nm, corresponding with the associated peak absorption (Bolton, 1999).

Although UV disinfection of pathogens in drinking water is effective and increasingly used, however, there are disadvantages resulting in their not being widely used in domestic/small scale settings including (i) expensive to install and operate, (ii) interference by turbidity, (iii) no lasting residual disinfection effect i.e. low secondary disinfection (Betancourt & Rose, 2004).

In addition to those water treatment methods outlined above, a number of other treatment methods for the inactivation of waterborne pathogens are also currently in use, including oxidation using ozone (O_3), peroxide (H_2O_2) or chloride dioxide (ClO_2), as well as ultrasound techniques.

Chapter 4: Groundwater and Public Health

4.1 Introduction

A wide variety of materials have been identified as groundwater contaminants including synthetic organic chemicals, hydrocarbons, inorganic cations, inorganic anions, pathogens and radionuclides. An extensive listing of these is presented by Misstear *et al.* (2006). Groundwater contamination is the introduction of a harmful substance or product into an environment, with the receiving environment in this case being an aquifer on a large scale or a groundwater production well on a small scale. The following chapter provides a review of potential groundwater contaminants, particularly those most often encountered in rural settings, their sources and pathways and subsequent potential human health effects. A number of previous groundwater borne illness outbreaks are also discussed, with the focus on groundwater pathogens.

4.2 Pathogenic Organisms and Diseases

Pathogens are those organisms that either produce or are involved in the production of a disease, which include bacteria, viruses and protozoa. Gleick (2002) and Gray (2004) outline the four main categories of water related diseases based on the direct or indirect action of pathogens, as first characterised by Bradley (1974):

Water-borne diseases are enteric diseases caused by the ingestion of water contaminated by human or animal faeces or urine containing pathogenic bacteria, protozoa or viruses. Disease transmission typically occurs when organisms previously excreted by an infected person or animal, make their way into a drinking water source and are subsequently consumed by another person or animal, initiating further infection, with infections generally being diarrhoeal in nature. The most widely recognised water-borne diseases tend to be low-infective dose infections, namely cholera, typhoid and leptospirosis. Other water-borne diseases are mainly high-infective dose infections and include infectious hepatitis and amoebic and bacillary dysentery. Gray (2004) states that all diseases typically associated with water-borne transmission routes may also follow alternative transmission routes that allow faecal material to be ingested, including person to person and food-borne transmission routes. In the context of this research therefore, which focuses on water-borne infection, it is important to note that, unless an infection is positively confirmed as being water-borne, alternative transmission routes, particularly food-borne, are always possible. This, when coupled with likely low diagnosis rates associated with gastroenteric infections, may result in the true incidence of water-borne illness not being reflected in currently reported outbreak statistics.

Water-washed diseases are most commonly caused by poor personal hygiene and/or skin or eye contact with contaminated water and have been shown to occur in areas where clean water is scarce. Consequently, they are most commonly associated with tropical areas and include infections of the intestinal tract, the skin and the eyes. All water-washed intestinal infections are faecal in origin and may include all the water-borne infections, if associated with poor personal hygiene. For example, previous studies (Feacham, 1977; Maurer & Sturchler, 2000) have provided evidence that shigellosis is more closely linked to personal hygiene than water quality. The majority of water-washed intestinal infections are diarrhoeal diseases responsible for high mortality rates among

infants in the developing world (Gray, 2004; Glass, 2006). The infections of the skin and mucous membranes are non-faecal in origin and include bacterial skin sepsis, scabies and cutaneous fungal infections (such as ringworm). Diseases spread by fleas, ticks and lice are also included in this category, such as typhus, rickettsial typhus and louse-borne fever (Matzger & Moench, 1994; Gleick, 2002; Webber, 2010).

Water-based diseases are caused by organisms that reside primarily in water, require water or require an intermediate aquatic host for part or all of their life-cycle. The main causes of water-based diseases are parasitic worms, with increased numbers of worms directly affecting the severity of infection. Two common water-based diseases are Schistosomiasis carried by the trematode *Schistosoma* spp. and Guinea worm which is the nematode *Dracunculus medimensis (Gray, 2004).* Stenimann *et al.* (2006) regard Schistosomiasis as *"the most important water-based disease from a global public-health perspective", with* an estimated 779 million people currently at risk of schistosomiasis, the majority of those in sub-Saharan Africa (Contis & David, 1996). *Schistosoma* worms use aquatic snails as intermediate hosts, and the Guinea worm uses the small crustacean *Cyclops* spp. as its intermediate host (Gray, 2004).

*Water-related diseases (water-related insect vectors)*are diseases caused by insects which breed and/or feed adjacent to contaminated waters (Gleick, 2002) and are therefore not usually associated with groundwater resources. These diseases are often severe and prevention is extremely difficult as it necessitates control of the insect vectors. Globally, important water-related diseases include viral infections, namely, yellow fever transmitted by the mosquito *Aedes* spp. and dengue, which is carried by the mosquito *Aedes aegypti.* Other *water-related diseases include* Gambian sleeping sickness, trypanosomiasis, which is a protozoan infection transmitted by the riverine tsetse fly (*Glossina* spp.), and malaria, also a protozoan infection (*Plasmodium* spp.) and transmitted by the mosquito *Anopheles* spp. (Gray, 2004). These infections are rarely if ever seen in temperate climates such as Ireland, unless travel-related, as the disease vectors are indigenous to tropical regions.

Pathogens transmitted via drinking water are predominantly of faecal origin and therefore known as enteric pathogens (Ashbolt *et al.,* 2001). Due to the potential consequences of water-borne infections, microbial contamination is still considered to be the most critical risk factor in drinking water quality (Fawell & Miller, 1992; Gleeson & Gray, 1997; WHO, 2006). Current estimates of global water related diarrhoeal morbidity and mortality suggest in excess of 1 billion infections and 2.2-5 million deaths per year (Gleick, 2002). These pathogens are prevalent in rural areas in both developed and developing countries, including Ireland (Kisteman *et al.,* 2002; Ramirez *et al.,* 2009), due to an abundance of hazard sources including septic tanks (Spruill *et al.,* 2002; Gardner & Vogel, 2005; Gill *et al.,* 2009b), grazing animals (Schets *et al.,* 2005) and other agricultural activities (Mawdsley *et al.,* 1995; Gagliardi & Karns, 2000; Gerba & Smith, 2005).

Contamination derived from human faecal matter is generally understood to comprise a greater overall risk to human health, as faeces may contain human-specific enteric pathogens, including *Salmonella enterica* serovar Typhi, *Shigella* spp., hepatitis A virus, and Norwalk-group viruses. Animals also serve as reservoirs for a variety of enteric pathogens including various serotypes of *Salmonella*, *Escherichia coli*, and *Cryptosporidium* spp. (Scott *et al.,* 2002).

There are three groups of microorganisms that can be transmitted via drinking water: bacteria, protozoa and viruses. In the presence of adequate hygiene, particularly in developed countries, transmission occurs via the faecal-oral route and subsequent infection is therefore considered water-borne. Typically, contamination occurs as a direct result of human or animal waste ingress to groundwater resources i.e. aquifers or wells. Craun *et al.* (1997) report that during the period 1971-1994, 58% of water-borne disease outbreaks in the USA were caused by contaminated groundwater systems, with 70% of these considered to be attributable to direct contamination of a groundwater source, as opposed to contamination within a distribution system. More recent figures (Reynolds *et al.*, 2008) suggest that consumption of contaminated groundwater systems, in all, account for approximately 6.5 million illnesses per year in the USA, or 34% of total water-borne infections, including non-gastroenteric illnesses. Reynolds *et al.* (2008) go on to suggest that approximately 17% of illnesses resulting from contaminated groundwater supplies are derived from "non-community systems" i.e. one-off, private sources. Additionally, other pathogenic organisms may be transmitted via drinking water, including nematodes (roundworm, hookworm, etc) and cestodes (tapeworm). However, as these are unlikely to be transmitted via groundwater, these pathogens will not be reviewed here.

As shown (*Table 4.1*), a number of pathogenic water-borne outbreaks associated with drinking water have been reported both in Europe and North America.

Table 4.1 Water-borne outbreaks and illnesses in the USA (1991–2002) and Europe (1986–1996)

	USA 1991-2002		Europe 1986-1996*	
Pathogen	**Outbreaks**	**Illnesses**	**Outbreaks**	**Illnesses**
Cryptosporidium	15	408,371	11	4,171
Giardia	25	2,283	5	1,896
E. coli 0157:H7	11	288	16	2,583
Campylobacter	7	360	10	1,614
Salmonella	3	833	4	646
Norovirus	12	3,361	4	578
Rotavirus	-	-	2	289
Total	73	415,496	52	11,777

**data from 19 European Countries (Sources: Kramer et al., 2001; Craun et al., 2006)*

4.3 Bacteria

The kingdom *Monera* consists of unicellular prokaryotic organisms called bacteria (Stanier *et al.*, 1963). In terms of frequency of isolation in drinking water and the number of reported water-borne disease outbreaks, bacteria are the microbial group of most importance (Gray, 1997). *Table 4.2* below outlines the most commonly reported bacterial pathogens associated with groundwater sources in temperate climates, including associated infections and major vectors (Ashbolt, 2004; Soller *et al.*, 2010).

Table 4.2 Water-borne Bacterial Pathogens

Bacterial Pathogen	Health Effect	Major Reservoirs
Shigella spp.	Shigellosis	Human faeces
Salmonella spp.		
S. typhimurium	Salmonellosis	Human and animal faeces
S. typhi	Typhoid fever	Human faeces
S. paratytyphi	Paratyphoid	Human faeces
Enterotoxigenic		
*Escherichia coli (*ETEC*)*	Gastroenteritis	Human faeces
Campylobacter spp.	Gastroenteritis	Human and animal faeces
Vibrio cholera	Gastroenteritis/Cholera	Human faeces
Yersinia enterocolitica	Gastroenteritis	Human and animal faeces
Legionella pneumophila	Acute respiratory illness	Thermally enriched water
Leptospira spp.	Leptospirosis	Human and animal urine
Streptococcus spp.	AGI/Meningitis/Endocarditis	Human and animal faeces
Mycobacteria	Pulmonary illness/TB	Soil and water

Adapted from Ashbolt, 2004; Soller et al., 2010

Although all of the above bacteria may and have been found to contaminate aquifers, some are more prevalent than others. The most widespread water-borne disease throughout history has been cholera, with the first major epidemic in Europe killing over a million people during 1830–1832 (Wyn-Jones, 2000). However, many of these traditional water-borne diseases i.e. cholera and typhoid, have largely disappeared from Europe and North America, due to advances in microbiology, epidemiology and medicine, as evidenced by Craun *et al.* (2006), who report that during the period 1920-1940, 70% of water-borne disease outbreaks in the United States presented as typhoid fever, compared to 22% and 11% of water-borne outbreaks during the periods 1941-1960 and 1961-1970, respectively. Although typhoid fever has not figured as a major causative agent of water-borne infection since then, many of the "classical" water-borne infections are still of major importance in poorer developing countries. The main bacterial pathogens found in groundwater in temperate regions, and particularly Ireland, are now outlined.

4.3.1 Escherichia coli

The coliform group of bacteria comprises of several genera belonging to the family Enterobacteriaceae. Among the common genera of the group are *Escherichia, Citrobacter, Enterobacter* and *Klebsiella* (Bonde, 1977). *Escherichia coli (E. coli)* was discovered by German paediatrician and bacteriologist Theodor Escherich in 1885 and is now classified as part of the Enterobacteriaceae family of gamma-proteobacteria (Gleeson & Gray, 1997; Feng *et al.,* 2002).

E. coli are Gram-negative rod-shaped (bacillus) bacteria, commonly found in the lower intestine of warm blooded animals including humans (Haas *et al.,* 1999). *E. coli* cells are typically 2 µm long and 0.5 µm in diameter, with cell volume ranging from 0.6-0.7 μm^3 (Madigan & Martinko,

2006). They are facultative anaerobic and non-sporulating (Gray, 2004).*E. coli* uses mixed-acid fermentation in anaerobic conditions, producing lactate, succinate, ethanol, acetate and carbon dioxide (Madigan & Martinko, 2006). *E. coli* are generally non-pathogenic, however, a number of strains are capable of causing infection (virulent), with these pathogenic strains being characterised into 5 main groups or virotypes (Todar, 2007). These are listed below with examples of common serotypes causing gastrointestinal illness given in parentheses (Nataro *et al.,* 1998; Gray, 2004; Todar, 2007):

1. Enteropathogenic *E. coli* (EPEC) (18, 26, 44, 86, 111, 114, 119, 125, 126)
2. Enteroinvasive *E. coli* (EIEC) (28ac, 112ac, 136, 143, 144, 152, 164)
3. Enterotoxigenic *E. coli* (ETEC) (6, 8, 15, 25, 27, 63, 78, 115, 148, 153, 154)
4. Enterohaemorrhagic *E. coli* (EHEC) (157)
5. Enteroaggregative *E. coli* (EAggEC) (42)

As shown (*Table 4.3*), *E. coli* are ubiquitous in the faecal material of all humans and animals, in varying concentrations.

Table 4.3 Level of E. coli found in various species, compared with sewage and sewage effluent

Species	Faecal production g day^{-1}	Ave number of *E. coli* per g faeces	Daily Load *E. coli*
Man	150	13×10^6	1.9×10^9
Cow	23,600	0.23×10^6	5.4×10^9
Pig	2,700	3.3×10^6	8.9×10^9
Sheep	1,130	16×10^6	18.1×10^9
Duck	336	33×10^6	11.1×10^9
Turkey	448	0.3×10^6	0.13×10^9
Chicken	182	1.3×10^6	0.24×10^9
Gull	15.3	131.2×10^6	2.0×10^9

Jones & White, 1984, as cited in Gray, 2004

Virulent strains of *E. coli* can cause gastroenteritis, urinary tract infections, and neo-natal meningitis, with gastroenteritis being the most prevalent form of illness associated with water-borne EPEC. Survival of *E. coli* is similar throughout the serotypes and in warm nutrient-rich conditions (e.g. septic tank effluents), they are able to multiply in water (Gray, 2004). Approximately 100 organisms are required to cause infection, with gastroenteric symptoms ranging from profuse watery diarrhoea, with little mucus, nausea and dehydration. The disease does not cause fever and is rarely serious in healthy adults (Galbraith *et al.,* 1987). *Table 4.4* below outlines the main features of *E. coli* infections.

Table 4.4 Features of E. coli Infections

	EPEC	ETEC	EIEC	EHEC
Prevalence	10-40% of hospitalised diarrhoea	40-60% of travellers' diarrhoea	Rare	Developed countries
Outbreaks	Nurseries	Travellers, infants	Food-borne	Food-borne, Water-borne
Age	<2 years	Adults, infants	Unknown	Adults, children
Illness	Diarrhoea, Respiratory symptoms	Mild, cholera-like	*Shigella*-like	Diarrhoea
Duration	7 days	5 days	7 days	8 days
Mortality Rate	High in young children (5-6%) and neonates (16%)	≤0.1%	≤0.1%	0.2% average; 10% if HUS untreated

(adapted from Haas et al., 1999)

A number of *E. coli* infection outbreaks of varying scales associated with private groundwater supplies are documented in the literature (Jackson *et al.,* 1998; Licence *et al.,* 2001; Olsen *et al.,* 2002; Hrudey *et al.,* 2003; Said *et al.,* 2003; Schets *et al.,* 2005). The majority of these focus on the verotoxin-producing, EHEC strain 0157:H7.

Verotoxigenic (or verotoxin-producing) *E. coli* (VTEC) including serotype O157:H7 is associated with a wide range of severity of illnesses from mild diarrhoea to haemorrhagic colitis and haemolytic uraemic syndrome (HUS), which causes the intravascular lysis of red blood cells (Garvey *et al.,* 2010a). In severe cases, HUS can result in irreversible renal failure, requiring haemodialysis, and in a minority of cases, may prove fatal (Coffey *et al.,* 2007). HUS associated with VTEC is the most common cause of renal failure in children, with approximately 10% of confirmed VTEC cases in Ireland resulting in HUS and associated renal failure (Garvey *et al.,* 2010a). Animals, and in particular cattle and other ruminants, are a primary reservoir in which VTEC occurs naturally in the gut (Szewzyk *et al.,* 2000; Rosen, 2000; Garvey & McKeown, 2008b), resulting in this virulent *E. coli* strain's association with rural areas and consequently water-borne outbreaks associated with private groundwater sources. Drinking troughs and moist mixed cattle rations have been cited as sources of VTEC on farms (Hancock *et al.* 2001).

In 1999 a serious outbreak of *E. coli* affected the state of New York with more than 1,000 people affected and 2 deaths. It was reported that a total of 122 cases of *E. coli* O157:H7 were confirmed,65 people were admitted to hospital and 11 children developed haemolytic uraemic syndrome (HUS). Two people (a 3-year old girl and a 79-year old man) died (Charatan, 1999; Hrudey & Hrudey 2004). The source of the infection was a contaminated well at the Washington County Fair. Runoff from cow manure after torrential rain was thought to have been responsible for the well's contamination (Charatan, 1999). A study by Schets *et al.* (2005) reported *E. coli* 0157:H7 presence in 2.7% of private groundwater supplies in the Netherlands, with pulsed field gel electrophoresis (PFGE) analysis

suggesting that cattle were the most probable source. This would seem to agree with Irish experience, with annual VTEC peaks occurring in late summer, when the majority of livestock are grazing (Garvey *et al.,* 2010a).

More recently, both O'Sullivan *et al.* (2008) and Garvey *et al.* (2009) have suggested that a rise in VTEC infections in Ireland may be associated with water consumption from untreated private wells in rural areas, with at least 12 VTEC outbreaks in 2009 linked to private well supply exposure. As shown in *Figure 4.1*, the instance of VTEC infection has been rising steadily in Ireland over the period 1999-2009, with VTEC 0157 comprising the majority of cases. Between 2003 and 2007, the reported incidence of VTEC infection in Ireland rose from 2.4 to 3.9 per 100,000, with only two countries in the European Union reporting a higher VTEC incidence rate in 2007 (Garvey *et al.,* 2009).

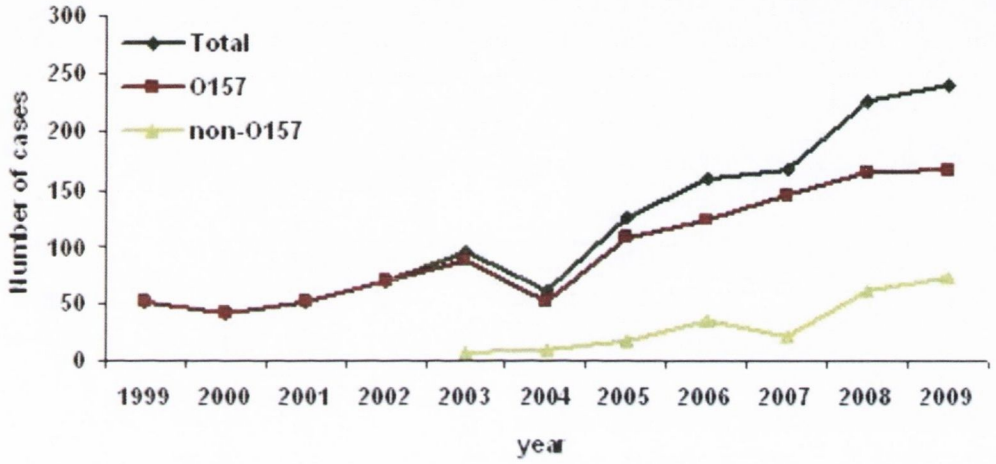


Figure 4.1 Confirmed VTEC cases in Ireland, 1999-2009 (NDSC, 2010)

4.3.2 Campylobacter spp.

*Campylobacter (*meaning 'twisted bacteria') are a genus of Gram-negative spirally shaped bacteria, typically 2000-5000 nm in length and comprising of 2-6 coils (Ryan & Ray, 2004). There are at least 17 known species, with *C. jejuni, C. coli, C. fetus* and *C. rectus* the most important from a human health perspective (Altekruse *et al.,* 1999; Allos, 2001; Zia *et al.,* 2003). They have a characteristic darting motility due to a single polar flagellum (Engberg *et al.,* 2001). They are oxidase-positive and reduce nitrates (Gray, 2004).

The Campylobacters (*Campylobacter jejuni* and *Campylobacter coli*) are one of the most common bacterial causes of gastroenteritis worldwide (Ashbolt, 2004). In Ireland, campylobacteriosis is the most commonly reported bacterial cause of infectious intestinal disease (HPSC, 2007) with illness being characterised by severe diarrhoea and abdominal pain (Humphrey *et al.,* 2007). Symptoms may subside after a number of days or may persist for weeks. Rarely, more severe symptoms may develop such as reactive arthritis, Reiter's syndrome, or HUS and approximately one in every 1,000 cases leads to a severe neurological disorder called Guillain-Barré Syndrome (HPSC, 2007).

Campylobacter species are found in the faeces of humans, livestock and a number of wildlife species, and are therefore common in the environment (Szewzyk *et al.,* 2000). The primary routes of transmission are ingestion of contaminated food and untreated water, with *C. jejuni* now recognised as being a major cause of foodborne infection in developed countries (Moore *et al.,* 2005). A number of campylobacter outbreaks have been positively associated with water-borne transmission routes, particularly in Scandinavian countries including Denmark (Engberg *et al.* 1998), Finland (Hänninen *et al.,* 2003) and Sweden (Andersson *et al.,* 1997). Survival in water can be from several weeks up to 3 months provided the temperature is low (Altekruse *et al.,* 1999; Nicholson *et al.* 2005). Chlorination is a highly effective barrier against *Campylobacter* species in water and therefore, where water-borne illness occurs, it is largely associated with untreated private water supplies (Coffey *et al.,* 2007), with higher instances in untreated surface water than untreated groundwater (Haas *et al.,* 1999). One water-borne *C. jejuni outbreak in* Denmark in 1995 caused an estimated 2,400 symptomatic infections. The contamination of the water supply was positively traced to contamination of groundwater due to a break in a sewage pipe (Engberg *et al.,* 1998).

To date, no water-borne Campylobacter outbreaks have been reported in Ireland. Human campylobacter infection became a statutorily notifiable disease in January 2004 under the Infectious Diseases (Amendment) Regulations 2003.

4.3.3 Salmonella spp.

Salmonella spp. are a genus of rod-shaped Gram-negative bacteria that inhabit the intestine human intestine during infection. They are aerobic or facultatively anaerobic, and most are motile (Oxford Dictionary of Science, 2010). As shown in *Table 4.2*, they are present in both human and animal faeces and cause a range of enteric infections depending on the serotype involved, including gastroenteritis, salmonellosis, typhoid fever and paratyphoid fever (Baumler *et al.,* 2000; Gill *et al.,* 2003). The most common symptoms of *Salmonella* gastroenteritis include diarrhoea, fever, and abdominal cramps (Bemrah *et al.,* 2003).

Salmonella serovars other than Typhi and Paratyphi may be found in drinking water from sources that have been contaminated with the faeces of infected humans or animals (Ashbolt, 2004). Various pathways may be responsible for the contamination of potable water with *Salmonella*, including sewage overflows, polluted storm water runoff, and agricultural runoff (Coffey *et al.,* 2007). Once infection has taken place, large numbers of organisms are excreted in the faeces ($>10^8$ g^{-1}) resulting in up to 2,000 organisms per 100ml in contaminated waters (Feachem *et al.,* 1983). Gray (2004) reports that the number of typhoid fever cases in the UK has fallen to less than 200 per annum with the majority (85%) of cases contracted abroad (of which, relatively few are the result of contaminated drinking water).

One documented water-borne outbreak of *Salmonella* associated with a groundwater supply occurred in Gideon, Missouri in 1993. Angulo *et al.* (1997) reported that this outbreak affected approximately 650 persons, 15 of whom were hospitalised, with 7 associated deaths. In this case the outbreak was associated with community wells, all of which tested positive for *S.* typhimurium.

To date, no water-borne *Salmonella* spp. outbreaks have been reported in Ireland.

4.3.4 Shigella spp.

Shigella is a genus of Gram-negative, non-spore forming rod-shaped bacteria closely related to *Escherichia coli* and *Salmonella* (Ryan & Ray, 2004), although it has a shorter environmental survival rate than both of these organisms (Bitton, 1999). During infection, it typically causes dysentery. The genus is named after Kiyoshi Shiga, who first discovered it in 1898 (Mims *et al.,* 1993; Potter, 2006).

Shigella species are classified by four serogroups:

- Serogroup *A*: *S. dysenteriae* (12 serotypes)
- Serogroup *B*: *S. flexneri* (6 serotypes)
- Serogroup *C*: *S. boydii* (23 serotypes)
- Serogroup *D*: *S. sonnei* (1 serotype)

Serogroups *A–C* are physiologically similar; *S. sonnei* (group *D*) can be differentiated on the basis of biochemical metabolism assays (Hale & Keusch, 1996).Three *Shigella* groups are the major disease-causing species: *S. flexneri* is the most frequently isolated species worldwide, accounting for up to 60% of cases in the developing world; *S. sonnei* causes 77% of cases in the developed world, compared to only 15% of cases in the developing world; and *S. dysenteriae* is usually the cause of dysentery epidemics (Hale & Keusch, 1996; DuPont, 2010). The disease is characterised by a short period of watery diarrhoea with intestinal cramps and general malaise, followed by permanent emission of bloody stools, known as haematochizia (DuPont, 2010).

Alamanos *et al.* (2000) report that a water-borne outbreak of shigellosis AGI which occurred in a community of 2,213 persons located near the city of Ioannina, in northwestern Greece during October 1999. Two hundred and eighty-eight inhabitants of the village of Eleoussa, suffered from gastro-enteritis, with the peak of the epidemic occurring during the first 3 days. The highest risk of developing gastro-enteritis was observed in the age group 0–14 years (41·4%) and decreased significantly with age ($p< 0·01$). Patients over 65 years were more frequently hospitalised than those in other age groups ($p< 0·05$). *Shigella sonnei* was isolated from both water samples and faeces of patients (Alamanos *et al.,* 2000). Environmental conditions suggested that contamination was caused via contaminated groundwater sources. Additionally, Koutsotoli *et al.* (2006) present an epidemiological overview of four recent water-borne *Shigella* outbreaks, occurring in separate Greek regions.

During the period 1991-2002, 9 water-borne disease outbreaks in the USA were attributable to *Shigella* spp., leading to 663 human shigellosis cases, however, it is not clear what proportion of these (if any), were attributable to groundwater systems (Craun *et al.,* 2006). Garvey *et al.* (2010b) report that there were, on average, 53 notifications of shigellosis per annum in Ireland during the period 2004-2008, with 11 outbreaks reported during the same period. None of these, however, was directly related to water-borne transmission routes with Garvey *et al.* (2010b) subsequently stating that with a crude incidence rate (CIR) of 1.25 (95% CI 0.91-1.58) per 100,000 population, compared to an EU average of 2.1 per 100,000 (range 0.0-9.7 per 100,000), the incidence of shigellosis in Ireland is low with person-to-person spread the most commonly reported transmission route and many sporadic cases identified as being travel related.

4.4 Protozoa

Protozoa (from the Greek words *proton*, meaning "first", and *zoa*, meaning "animals") are a sub - kingdom of microorganisms that are classified generally as *unicellular non-fungal eukaryotes* (Alcamo & Warner, 2009). They are unicellular organisms surrounded by a cytoplasmic membrane covered by a protective structure called a pellicle. Most protozoans are free-living, some of which can cause disease, whereas others are obligate parasites (Haas *et al.,* 1999). They form structures called cysts, oocysts or spores (depending on the life cycle of the organism) that are resistant to adverse environmental conditions such as dessication, starvation, high temperatures and disinfection, with some found naturally in soil (*Naegleria)* and water (*Acanthamoeba)* (Fayer *et al.,* 2004; Karanis *et al.*, 2007).

Five groups of protozoan containing members that are pathogenic to humans and having the potential for transmission through water are (Haas *et al.,* 1999):

1. Amoebas (*Rhizopoda)*
2. Flagellates (*Mastlgohora)*
3. Ciliates *(Cilratea)*
4. *Sporozoa*
5. *Microsporidia*

Table 4.5 outlines a number of important protozoa transmitted through water-borne routes.

Table 4.5 Water-borne Protozoa

Protozoa	Resistant Stage (Size, µm)	Illness or Disease
Amoebas		
Entamoeba histolytica	Cyst (10-20)	Diarrhoea, abscesses
Naegleria fowlei	Cyst	PAM*
Flagellate		
Giardia lamblia	Cyst(8-15)	Diarrhoea
Cilate		
Balantidium coli	Cyst (45-65)	Diarrhoea
Sporozoans		
Cryptosporidium parvum	Oocyst (3-8)	Diarrhoea
Toxoplasma gondii	Oocyst (10-12)	Congenital infection
Cyclospora	Oocyst (8-10)	Diarrhoea

*(from Marshall et al., 1997; Haas et al., 1999); *Primary meningoencephalitis*

Pathogenic intestinal protozoa are taxonomically divided into the amoebas, flagellates and coccidian (meaning "globose in shape") and are primarily excreted in the faeces or urine of infected human or

animal hosts. They are transmitted via the infectious cyst and oocyst stages by the faecal-oral route and being obligate parasites, can only reproduce in the host (Marshall *et al.,* 1997; Haas *et al.,* 1999).

There are two protozoa frequently found in drinking water supplies in developed countries which are known to be responsible for outbreaks of disease. These are *Cryptosporidium* spp. and *Giardia lamblia.*

4.4.1 Cryptosporidium spp.

Cryptosporidium spp. are coccidian protozoan parasites which belong to the phylum Apicomplexa (Barer & Wright, 1990). There are currently 21 recognised species of *Cryptosporidium* (see Appendix C1) of which two are of particular importance in the context of human infection: *C. parvum* and *C. hominus*. *C. parvum* is the species primarily responsible for clinical illness in humans and animals (Rose, 1988; Fayer *et al.,* 2000; Hrudey & Hrudey, 2004), which despite its discovery in 1912, was not identified as a cause of human infection until much later, with the first case of human infection recorded in 1976 (Gray, 1994; Marshall *et al.,* 1997).

Upon ingestion of the mature thick-walled oocysts, excystation of the oocyst occurs within the small intestine and the released sporozoites parasitize epithelial cells of the gastrointestinal tract. Sporozoites penetrate enterocytes and develop into trophozoites (*Figure 4.2*). This stage divides by asexual multiplication into forms with six to eight nuclei, each maturing into a merozoite. Zygotes formed after fertilization develop into environmentally resistant thick-walled oocysts that undergo sporogony to form sporulated oocysts. These sporulated oocysts are released in faeces and are immediately infectious upon excretion, as well as being capable of survival outside the intestine for considerable periods of time (Fayer & Ungar, 1986; Current & Garcia, 1991; Sterling & Arrowood, 1993; Marshall *et al.,* 1997; Fayer *et al.,* 2000). For example, a study by Robertson & Gjerde (2006) found that *Cryptosporidium* oocysts with viable morphology were detected after approximately 20 weeks, during winter in an aquatic environment in Norway. However, as stated by Reinoso *et al.* (2008), the interaction between pH, temperature and exposure time may have adverse effects on the survival of *C. parvum* oocysts in water and soils, and therefore an exact environmental survival rate is difficult to ascertain.

Sporulated oocysts are produced by approximately 80% of zygotes (Smith & Rose, 1990). The remaining 20% of oocysts are thin-walled and responsible for auto-infection (Gray, 2004), with this mode of auto-infection permitting *Cryptosporidium* to complete its life cycle within a single host, resulting in a host becoming a life-time carrier (Fayer & Ungar, 1986).

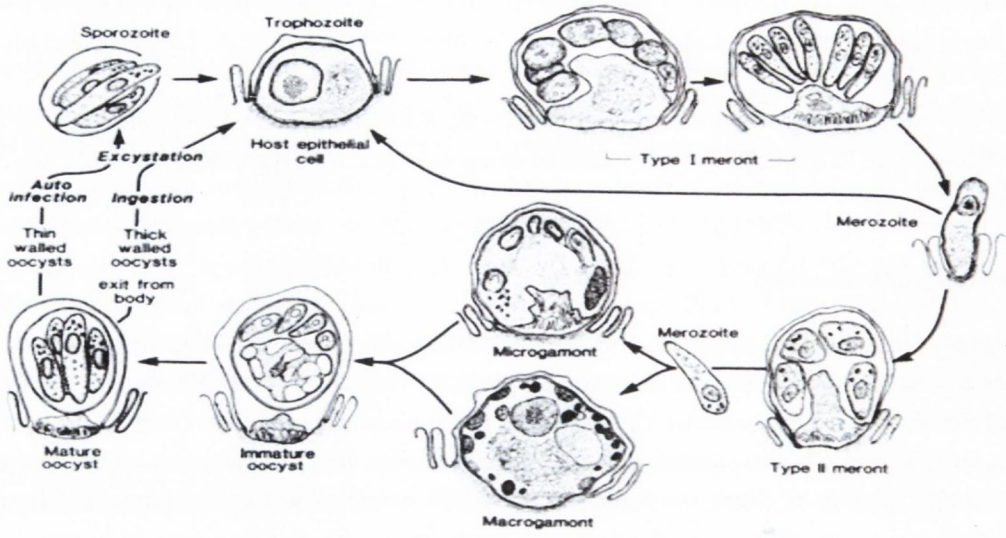


Figure 4.2 Cryptosporidium Life Cycle (CDC, 2009)

Cryptosporidiosis is now recognised as being a significant gastrointestinal disease in infants, young children and immune-compromised individuals (Chen *et al.,* 2002; Cohen *et al.,* 2006), with both food-borne (Quiroz *et al.,* 2000) and water-borne (Fayer, 2004) transmission routes reported. It is generally acquired by ingestion of viable oocysts (Barer & Wright, 1990). The incubation period after ingestion of viable oocysts is 5 to 28 days with a mean of 7.2 days (Marshall *et al.,* 1997; Haas *et al.,* 1999) with symptoms lasting for up to two weeks. The HPSC (2010a) note that cryptosporidiosis is a relatively common gastroenteric illness in Ireland. Clinical symptoms include an influenza-like illness, diarrhoea (characteristically cholera like), malaise, abdominal pain, anorexia, nausea, flatulence, malabsorption, vomiting, mild fever and weight loss (Smith & Rose, 1990; Gray, 2004), with infection typically non-fatal and self-limiting in healthy adult hosts. However, the severity and duration of the disease is significantly higher among immuno-compromised individuals, in whom 2-3 L d^{-1} of diarrhetic stool is common, with significant dehydration, malnutrition and profound weight loss (Gatei *et al.,* 2006). In AIDS patients the disease becomes chronic in nature and can lead to death (Cama *et al.,* 2007), with a study on the filtration of drinking water (Sorvillo *et al.,* 1994) finding a higher instance of cryptosporidiosis among homosexual and bisexual men linked to sexual transmission. Low level exposure to oocysts (1-10) is capable of initiating infection (Casemore, 1990; Gray, 2004), with studies indicating that a single oocyst may be adequate to cause infection (Blewett *et al.,* 1993; Boak & Packman, 2001). Rose (1997), provides ranges for the occurrence of *C. parvum* oocysts in a range of media including raw sewage, filtered treated wastewater, surface waters and groundwaters, with a range of 0.004-0.922 oocysts l^{-1} provided for groundwater samples, suggesting a background level of oocysts existing in groundwater bodies with potential for human infection. This background level may rapidly increase through accidental point or non-point pollution.

This low human dose-response and high environmental resistance to environmental factors including disinfection by chlorination, has lead to a number of large cryptosporidiosis outbreaks associated with contaminated drinking water throughout the world. Major water-borne outbreaks have been confirmed in Northern Ireland (Glaberman *et al.,* 2002), Britain (Bridgman *et al.,* 1995;

Craun *et al.* 1998), France (Dalle *et al.,* 2003), Norway (Robertson *et al.,* 2005), The Netherlands (van Asperen *et al.,* 1996), the USA (MacKenzie *et al.,* 1993), Canada (Craun *et al.,* 1998) and Ireland (McDonald *et al.*, 2011).

The largest and most widely publicised outbreak of Cryptosporidiosis occurred during March and April 1993 in Milwaukee, Wisconsin (MacKenzie *et al.,* 1993). In this case, a water distribution system serving 800,000 people was contaminated by raw water from a river swollen by spring run-off. In all 403,000 people were symptomatically infected, including at least 4,400 hospitalisations and 104 deaths (Hoxie *et al.*, 1997).

Smith & Rose (1998) report that during the 10-year period 1988-1998, *Cryptosporidium* oocysts have been shown to be common contaminants of water, causing at least 19 water-borne outbreaks of cryptosporidiosis in the US, affecting over 425, 000 individuals. Three of these outbreaks were associated with contaminated wells and one with a suspected spring (Smith & Grimason, 2003). Willocks *et al* (1998) report on 345 confirmed cases of cryptosporidiosis in North Thames during spring 1997. A case control study, in addition to oocyst detection rates and descriptive epidemiology associated with the outbreak, indicated that the outbreak was associated with drinking unboiled tap water that originated from a deep chalk borehole. Garvey & McKeown (2005) report that a significant decrease in cryptosporidiosis cases in northwest England during the period 1997-2000, was likely attributable to developments in public water supplies, which would suggest that some burden of illness was associated with private water supplies.

On January 1, 2004, human cryptosporidiosis became a notifiable disease in Ireland (Garvey & McKeown, 2008a). Due to this, the burden of human cryptosporidial disease in Ireland prior to 2004 remains undetermined. Two Irish studies on paediatric populations have shown that of those cases with a laboratory-confirmed diagnosis, approximately 4% were caused by cryptosporidiosis (Corbett-Feeney, 1987; Carson, 1989). Data from the Infoscan Laboratory Surveillance System, covering 3 of the 8 public health authorities (HSE) and a population of approximately 1.2 million, show that there has been an average of 121 laboratory confirmed cases per annum of cryptosporidiosis in Ireland, a rate of 10.1/100,000 population (NDSC, 2004). However, as shown in *Figure 4.3*, incidence has shown an overall, albeit relatively moderate, increase since 2004, probably due to cryptosporidiosis becoming a notifiable disease. The number of laboratory-confirmed cases is likely to be a significant underestimate of the actual number of cases however, as a large proportion of diarrhoeal illness is considered to go undiagnosed and/or unreported. Currently, annual case numbers are typically in the range 350-600 annum^{-1}, with associated crude incidence rates of 8-14/100,000 (Garvey & McKeown, 2009). This is outlined in greater detail in Chapter 10 (*Section 10.3.2*).

The first reported outbreak of cryptosporidiosis in Ireland strongly associated with drinking water occurred in April 2002 in the Midland Health Board region (Jennings & Rhatigan 2002; Hanly, 2003). The water source in this case was a spring-fed lake (Lough Owel) serving a population of approximately 25,000 people. There were 26 confirmed cases of cryptosporidiosis, with a further seven unconfirmed cases reported. In all, four cases were admitted to hospital. Genotyping identified that seven of eight stool samples were *C. parvum* type 2 (NDSC 2004). In 2005 a cryptosporidiosis outbreak in the Carlow town water supply resulted in 31 confirmed cases (18 female/13 male) of the illness in the area between February and May 2005, with private

groundwater suspected of being a primary risk factor. Overall, eight people were hospitalised (Coffey *et al.,* 2007).

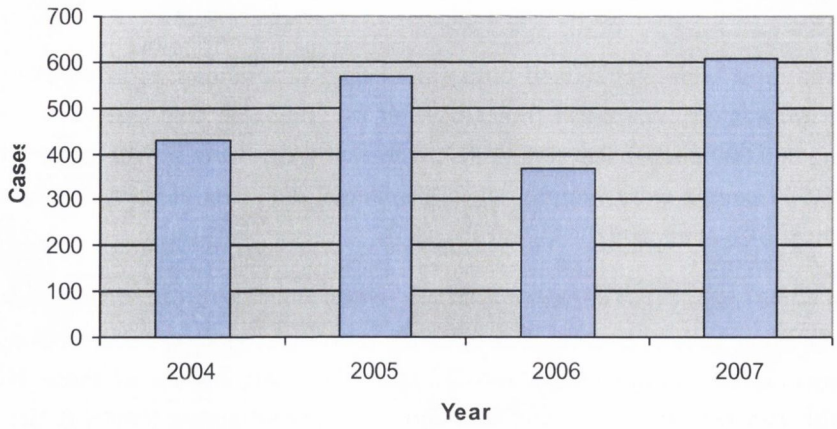


Figure 4.3 Confirmed cryptosporidiosis cases in Ireland, 2004-2007 (Garvey & McKeown, 2009)

A large cryptosporidiosis outbreak occurred during the period February-March 2007 in Galway City, Ireland, which has been linked to a public drinking water supply. In all 182 cases of cryptosporidiosis were laboratory confirmed with both *C. parvum* (28%) and *C. hominis* (72%) found in typed stool samples from infected individuals (Pelly *et al.,* 2007).

A more recent study by Zintl *et al.* (2009) has shown that *C. parvum* (80%) is more commonly found to be the causative agent of cryptosporidiosis in Ireland as a whole, with species other than *C. parvum* and *C. hominis* rarely found in infected human hosts. An annual peak in cryptosporidiosis cases is typically observed in Ireland during the period March-May, which is generally thought to be associated with agricultural cycles (HPSC, 2010a).

The prevalence rate for *Cryptosporidium* is estimate to be 1-3% in Europe and North America and considerably higher in developing countries, ranging from 5% in Asia to 10% in Africa (Marshall *et al.,* 1997). Gray (2004) suggests that previous *Cryptosporidium* outbreaks have frequently been associated with inadequate methods of water treatment which do not effectively eradicate the parasite. Typically, in the case of public water supplies, water treatment processes ultimately terminate with chlorination. It has been established that chlorination at levels used in water treatment is ineffective against oocysts, with concentrations as high as 16,000 μg l^{-1} required to prevent excystation (Korich *et al.,* 1990). Alternative disinfectants for water treatment and inactivation of *Cryptosporidium* oocysts have been investigated including chlorine dioxide, monochloramine, hydrogen peroxide and ozone (Korich *et al.,* 1990; Corona-Vasquez *et al.,* 2002).

4.4.2 Giardia lamblia

Giardia lamblia is a flagellated protozoan belonging to the class Zomastigophorasida, order Diplomonadonda and family Hexamitidae (Hunter & Thompson, 2005). In 1681, Leeuwenhoek ("the father of microbiology") described parasites similar to *G. lamblia* in his own faeces, however, the first report was credited to Lamb in 1859 (Marshall *et al.,* 1997). *G. lamblia* is the most commonly diagnosed flagellate in the intestinal tract and recognised as a significant cause of gastroenteritis

ranging from mild to severe with a worldwide distribution (Kappus *et al.,* 1994), particularly in children in developing countries (Bryan *et al.,* 1994).

Like *Cryptosporidium* spp., *G. lamblia* is found in a wide range of animals, existing in the free-living trophozoite form, with these trophozoites easily recognised, due to their pear or kite-shaped bodies (Gray, 1994; Gray, 2004). Cysts are capable of surviving for long periods in both groundwater and surface water environments, particularly at cold temperatures during the winter months (2-3 months at <10°C; up to 1 month at 21°C) (LeChavallier, 1991; Teunis & Havelaar, 2002; Coffey *et al.,* 2007). Unlike *Cryptosporidium* oocysts, which are round, *Giardia* cysts are oval and larger (8-14 µm long, 7-10 µm wide) (Coffey *et al.,* 2007). Transmission pathways are similar to *Cryptosporidium,* however, *Giardia* is more widespread than *Cryptosporidium* with most surface water containing *Giardia* cysts (LeChavallier, 1991), with these factors resulting in *Giardia* being a pathogen of public health concern in drinking water.

The life cycle of *Giardia,* as depicted in *Figure 4.4*, consists of two stages: an actively multiplying trophozoite and a resistant cyst stage. Cysts are excreted in the faeces of infected persons. Upon (food-borne or water-borne) cyst ingestion, the acid environment of the stomach triggers excystation, typically occurring in the duodenum. Trophozoites subsequently attach themselves to the intestinal lining of the infected host via a ventral disc, replicating by binary fission. Cyst formation takes place as the trophozoites move through the colon or ileum (Marshall *et al.,* 1997; Haas *et al.,* 1999). The cysts are subsequently passed from the body in faeces, with 1-5 x 10^6 cysts per gram of faeces (Lin, 1985).

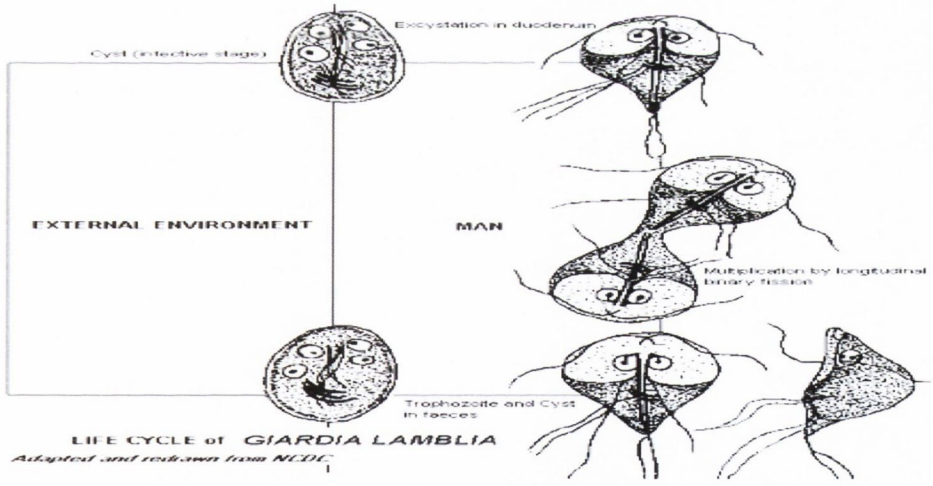


Figure 4.4 Giardia Life Cycle (CDC, 2009)

Ingestion of water-borne *Giardia* cysts causes *Giardia*sis, which is the most common protozoan infection of the human intestine worldwide (Haas *et al.,* 1999). *Giardia*sis occurs throughout temporal and tropical regions with prevalence ranging from 2-5% in the developed world to 20-30% in the developing world (Kappus *et al.,* 1994; Marshall *et al.,* 1997). Symtomatic illness is characterised by an acute onset of diarrhoea, foul-smelling stools, upper or mid-epigastric cramps, bloating and flatulence, with patients usually expressing feelings of malaise, nausea and sulfuric belching. The prolonged duration of diarrhoea, frequently continuing for 2-4 weeks is a typical

feature of *Giardia*sis (Haas *et al.,* 1999; Ashbolt, 2004). *Giardia*sis has an incubation period of 1-2 weeks. Following cyst ingestion, approximately 5-15% of persons become asymptomatic cyst passers, 25-50% will become symptomatic of *Giardia*sis, with the remaining 35-70% exhibiting no sign of infection. Asymptomatic cyst passage has been documented for children in day-care centres to last as long as 6 months (Pickering *et al.,* 1984; Haas *et al.,* 1999).

During the period 1980-1985, there were 502 outbreaks of water-borne disease in the USA; *Giardia* was positively confirmed as the causative organism in 52% of these cases. One large outbreak occurred at a Colorado ski resort, when sewage contamination of a groundwater supply resulted in 123 people contracting symptomatic infection. During the period 1995-2004, *Giardia* has been identified as the primary causative agent in 13% of 108 reported gastroenteric outbreaks associated with drinking water in the US (Yoder & Beach, 2007). It should be noted that the proportion of these outbreaks directly attributable to contaminated groundwater could not be established.

*Giardia*sis became a notifiable disease in Ireland on January 1, 2004, following the Amendment to the Infectious Disease legislation (SI 707). In 2004, 53 cases of *Giardia*sis were reported in Ireland, with a similar number in 2005, when 57 *Giardia*sis cases were notified. These figures resulted in an approximate notification rate of 1.5/100,000. This figure increased slightly in 2006 up to 62 notifiable cases (Coffey *et al.,* 2007). It is not known how many of these cases were water related. Three family *Giardia*sis outbreaks were reported to the HPSC between 2004 and 2006; one family outbreak each year. No general outbreaks were reported in this time period. In total, seven people were affected as a direct result of these three outbreaks. For one outbreak, the reported transmission route was person-to-person spread; the transmission routes for the other two family outbreaks were reported as unknown (Coffey *et al.,* 2007).

Unlike cryptosporidiosis, *Giardia*sis can be treated by a number of drugs (quinacrine, metronidazole, tinidazole, furazolidone, paromomycin), however, currently no means of treating infection (as opposed to symptomatic illness) exists (Marshall *et al.,* 1997; Gray, 2004).

4.4.3 Entamoeba histolytica

Another potential protozoan pathogen of interest in temperate climates is *Entamoeba histolytica,* which is an anaerobic parasitic protozoan, of the genus *Entamoeba,* with the active (trophozoite) stage existing in both the host and in fresh faecal matter (Ryan & Ray, 2004). As with *Cryptosporidium* and *Giardia*, *E. histolytica* cysts survive outside the host in water, soils and on foods, especially under moist conditions on the latter (AWWA, 2006). When cysts are swallowed they cause infections by excysting (releasing the trophozoite stage) in the digestive tract, consequently causing amebiasis/amoebic dysentery (Zlobl, 2001; Ryan & Ray, 2004). Amebiasis is severely debilitating, with infection centred in the large intestine where the amoebae multiply and adhere to enterocytes causing ulceration. The symptoms include mid-abdominal pain, diarrhoea alternating with constipation, or chronic watery diarrhoea with a discharge of mucus and blood (Blessmann & Tannick, 2002; Gray, 2004). Amebiasis is the third leading cause of parasitic death worldwide (Walsh, 1988)

E. histolytica is asymptomatically carried by approximately 10% of the population in Europe and 12% in the USA (Gray, 2004). In the USA, six water-borne disease outbreaks have been attributed to *E. histolytica* between 1946 and 1980 (Lippy & Waltrip, 1984). In countries where the disease is widespread the highest rates of incidence occur in those groups with unprotected water supplies, including groundwater sources, and poor personal hygiene (Barwick *et al.,* 2002; Gray, 2004). Barwick *et al* (2002) report on a relatively large amebiasis outbreak that occurred in Tbilisi, Republic of Georgia in 1998. The outbreak was found to be caused by interruptions in the water supply and decreases in water pressure leading to contamination of municipal water in the distribution system. In all 177 cases of amebiasis were confirmed in the outbreak. To date, no water-borne *E. histolytica* outbreaks have been recorded in Ireland.

4.5 Viruses

Human enteric viruses are causative agents in both developed and developing countries of many non-bacterial gastrointestinal tract infections, respiratory tract infections, conjunctivitis, hepatitis and other more serious infections with high morbidity and mortality in immuno-compromised individuals such as meningitis, encephalitis and paralysis (Okoh *et al.,* 2010).There are over 140 distinct known types of human pathogenic viruses (Gray, 2004), those of most concern being the enteric viruses which cause gastrointestinal illnesses including enteroviruses, rotaviruses, astroviruses, calciviruses, hepatitis A virus and norovirus (Oliver *et al.,* 2003). Enteric viruses are typically gut-associated, with related illnesses being primarily gastrointestinal in nature (Sellwood & Dadswell, 1991). The common water-borne enteric viruses and their associated diseases are presented in *Table 4.6.*

Table 4.6 Common water-borne enteric viruses and associated diseases

Virus Group	Serotypes	Associated Diseases
Polioviruses	3	Paralysis, aseptic meningitis
Coxsackievirus		
A	23	Herpangia, aseptic meningitis, respiratory illness, paralysis
B	6	Pleurodynia, aseptic meningitis, pericarditis, nephritis, fever
Echovirus	34	Respiratory infection, meningitis diarrhoea, pericarditis, myocarditis
Enteroviruses (68-71)	4	Meningitis, respiratory illness
Hepatitis A virus (HAV)		Infectious hepatitis
Reoviruses	3	Respiratory disease
Rotaviruses	4	Gastroenteritis
Adenoviruses	52	Gastroenteritis, respiratory disease, acute conjunctivitis
Norovirus	1	Gastroenteritis
Astroviruses	5	Gastroenteritis

From Gray, 2004

From a public health perspective, the three most commonly diagnosed water-borne enteric viruses, particularly in Ireland, are rotavirus, adenovirus and norovirus.

4.5.1 Rotavirus

Rotavirus is a genus of double-stranded RNA virus in the family Reoviridae (Velázquez *et al.,* 1996). It is the most common cause of severe diarrhoea among infants and young children worldwide (Glass, 2006), and is globally responsible for an estimated 610,000 child deaths per year, accounting for approximately 5% of all deaths in those <5 years (Glass, 2006). Rotavirus may be found in water sources such as private wells that have been contaminated with the faeces from infected humans. Sewage overflows or sewage systems that are not working properly are primarily responsible for any water contamination (CDC, 2003). The virus is very resistant in the natural environment and can be present in large amounts in wastewater (Gerba *et al.,* 1996a; Macler & Merkle, 2000). Previous studies (Sobsey *et al.*, 1995; Azadpour-Keeley *et al.*, 2003; Azadpour-Keeley & Ward, 2005) have found that enteric viruses may survive for many weeks in groundwater environments, with Sobsey *et al* (1995) reporting that there was little, if any, inactivation of any virus in samples from a deep well in North Carolina over an 8-week period.

Symptoms of rotaviral infection in humans include vomiting and watery diarrhoea (Gratacap-Cavallier *et al.,* 2000; CDC, 2006a; CDC, 2006b). Infections can occur in winter epidemics with a high level of secondary person-to-person transmission, which is mainly of faecal–oral origin (CDC, 2006b). As few as 10 viral particles are enough to cause symptomatic illness, with the viral particles replicating in the intestinal epithelial cells of infected individuals. Viral toxins cause subsequent sloughing of the gut epithelium leading to diarrhoea, with Glass (2006) reporting that in the absence of rehydration therapy, an infected child may lose up to 10% body weight over a 24-48 hour period.

Water-borne gastroenteritis outbreaks linked to consumption of water from contaminated groundwater wells with rotavirus as a confirmed causative agent have been reported by Kukkula *et al.* (1997), Frost *et al.* (2002), Borchardt *et al.* (2003), Gallay *et al.* (2006), and Koroglu *et al.* (2011).

There is a well documented seasonal pattern in Ireland associated with rotavirus infection, with peaks occurring annually in early spring (Cryan *et al.,* 1997). Rotavirus associated gastroenteritis/AGI is primarily seen as a paediatric illness, with children generally affected in the first 2-3 years of life e.g. in 2007, 97% of notified rotavirus infections in Ireland occurred in those aged 0-4 years (HPSC, 2008d), resulting in an age-specific incidence rate (ASIR) of 746.1/100,000 in this age group. Over the remainder of the population, an ASIR of approximately 2.1/100,000 was reported (HPSC, 2009a). These data should be treated with caution however, as a majority of those laboratories in Ireland routinely analyzing human stool samples for rotavirus only test specimens from children <2 years of age (Lynch *et al.,* 2001), and therefore the health burden associated with those >2 years is likely to be underestimated.

As shown in *Figure 4.5*, as for *Cryptosporidium* spp., the overall incidence of notified rotaviral infections in Ireland has remained relatively constant during the notification period 2004-2009. Although there was a notable increase in notifications between 2004 and 2005, this is likely due to the organism's new notifiable status and consequent underreporting previously. Over the period

2001-2009, there has been a gradual increase in rotaviral cases among children <2 years, although this may also be attributed to the new notification status. However, as noted by Coffey *et al.* (2007), this is almost certainly a small proportion of the total number cases of infection as most patients probably do not seek medical attention. To date, no water-borne rotavirus outbreaks have been reported in Ireland.

Twelve water-borne outbreaks were noted in the USA from 1991–2002 and four water-borne outbreaks were noted in Europe from 1986–1996 (Kramer *et al.*, 2001; Craun *et al.*, 2006) (*Table 4.1; Section 4.2*).

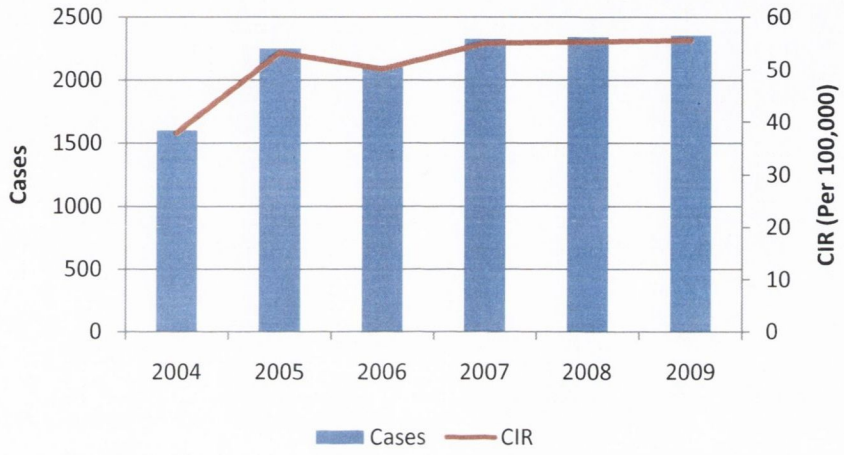


Figure 4.5 Confirmed rotavirus cases and crude incidence rate in Ireland 2004-2009

4.5.2 Norovirus

Norovirus is an RNA virus (taxonomic family Caliciviridae) , identified as the leading cause (≈90%) of epidemic non-bacterial outbreaks of gastroenteritis around the world (Lindesmith *et al.*, 2003). The primary transmission routes are faecally contaminated food or water, person-to-person contact (Goodgame, 2006; Lee, 2011) and aerosolization of the virus and subsequent contamination of surfaces (Said *et al.*, 2008), with contamination of drinking water primarily linked to sewage or human faeces (Nygard *et al.*, 2003). Norovirus outbreaks are primarily associated with closed-community outbreaks i.e. hospitals, retirement homes, cruise ships, etc. (Gallimore *et al.*, 2005).

Norovirus (previously called Norwalk-like viruses and more commonly known as the "winter vomiting bug") causes gastroenteritis in all age groups and has been associated with outbreaks in Irish hospitals during winter (Coffey *et al.*, 2007, Kelly *et al.*, 2008). Symptoms include acute gastroenteritis developing 24-48 hours after exposure, and lasting for 24–60 hours. The disease is usually self-limiting, and characterised by nausea, vomiting, diarrhoea, and abdominal pain; and in some cases, loss of taste. General lethargy, weakness, muscle aches, headache, and low-grade fever may also occur (Zheng *et al.*, 2010).

Noroviruses have been identified as the cause of many outbreaks (Kroneman *et al.*, 2006; Teunis *et al.*, 2008b) and have been identified by Kearney *et al.* (2007) as the leading cause of non-bacterial

gastroenteritis in Ireland. In the US alone, noroviruses have been estimated to cause 23 million infections per year, resulting in 50,000 hospitalizations and 310 fatalities (Mead *et al.,* 1999).

Consumers drinking water from a contaminated well in central Wyoming, which was suspected of having been contaminated by sewage, were found to be 4.5 times more likely to present with gastroenteric symptoms than those who had not consumed from the well (Parshionikar *et al.,* 2003). A recent study (Manuala *et al.,* 2005) has found that most norovirus contamination events occurring in Finland, may be attributed to groundwater systems, with almost 50% of groundwater outbreaks (7/15) associated with private water wells. Reporting on a recent norovirus outbreak associated with a private well in a dolomite aquifer in northeast Wisconsin, Borchardt *et al.* (2010) found that 229 patrons of a new restaurant were affected during the incident, which was positively confirmed as being due to ingress of septic tank effluent into a nearby private well.

No water-borne outbreaks of norovirus have been reported in Ireland in recent years, with a recent study (Kelly *et al.,* 2008) concluding that most reported norovirus outbreaks in Ireland occur in hospitals and residential institutions, with person-to-person spread being the predominant mode of transmission. A water-associated outbreak at a hotel in the west of Ireland in 2001 resulted in 462 illnesses (Coffey *et al.,* 2007).

4.5.3 Adenovirus

Adenoviruses are medium-sized (90-100 nm), non-enveloped icosohedral viruses containing double-stranded DNA, with at least 51 distinct serotypes, approximately one third of which are capable of causing human infection. Adenoviruses are unusually stable to chemical and physical agents and to adverse pH conditions, thus allowing for prolonged survival outside of the body (Mena & Gerba, 2009; CDC, 2010). A recent study has found that adenovirus remained relatively stable in seeded groundwater at 10°C for at least 160 days and up to 301 days (Rigotto *et al.*, 2010). This is a concern, as 10°C is comparable with annual mean groundwater temperatures in Ireland. The same study found an approximate 1-log (90%) adenovirus reduction after 160 days at 19°C.

Although epidemiologic characteristics of the adenoviruses vary by type, all are transmitted by direct contact, faecal-oral transmission, and occasionally water-borne transmission (Wold & Horwitz, 2007). Oral transmission of the virus typically causes no, or only mild, disease. The typical incubation period for gastroenteritis is 3-10 days; for respiratory tract infections it is between 2 and 14 days. Outbreaks of adenovirus-associated respiratory disease have been more common in the late winter, spring, and early summer; however, adenovirus infections can occur throughout the year (CDC, 2010), with children and the immune-compromised generally more severely impacted by adenovirus infections (Mena & Gerba, 2009). Studies have shown that adenoviruses survive longer in water than enteroviruses and hepatitis A virus (Enriquez *et al.,* 1995; Ogorzaly, 2010), which may be due to their double-stranded DNA. Two notable drinking water outbreaks associated with adenovirus are documented in Finland (Kukkula *et al.*, 1997) and Albania (Divizia *et al.*,2004), with Mena & Gerba (2009) reporting that adenovirus is responsible for more recreational water-borne outbreaks than any other enteric virus.

The water-borne adenovirus reported by Kukkula *et al.* (1997) is of particular interest, as ingress of contaminated river water into a wellhead was found to be the source of the outbreak. In all, some 1,500-3,000 people, accounting for 25-50% of the affected Finnish community, were affected. Laboratory findings confirmed adenovirus, a Norwalk-like agent, small round viruses (SRV), and group A and C rotaviruses as causative agents, with Norwalk virus being the main cause of the outbreak. To date no water-borne adenovirus outbreaks have been reported in Ireland.

4.6 Microbial Groundwater Contamination Indicators

Maintenance of the microbial (and chemical) quality and safety of groundwater, and in particular, private groundwater systems used for human consumption, is imperative, as contamination of these systems can cause significant risks to human health. The ability to accurately predict the presence and level of microbial contamination in drinking water supplies is therefore of paramount importance, in order that hazard sources may be located and appropriate remedial systems put in place.

Once the connection had been made between outbreaks of diseases such as cholera and typhoid fever, with contaminated water, attempts were made to develop methods which would isolate the causative organisms. In practice, however, this proved difficult for a number of reasons, not least of which being very low pathogen concentrations and rapid die-off of pathogens after sampling. Additionally, the wide variety of potential pathogens present in contaminated water sources presented limitations as to the practicality of procedures for their enumeration (Pipes, 1982). The isolation of pathogens can never be the sole procedure used in routine water analysis, because, as mentioned, their appearance is intermittent, of short duration and the organisms may be readily attenuated (Bonde, 1977). Furthermore, the isolation and identification of individual pathogenic organisms is typically complex, time consuming and expensive (Gray, 2004). This has lead to the development of the use of indicator organisms to imply the possibility of contamination by faeces and where possible, indicate the likely faecal source.

Gleeson & Gray (1997) and Gray (2004) present the following criteria for selection of indicator organisms:

1. They should be a member of the normal intestinal flora of healthy people
2. They should be exclusively intestinal in habit and therefore exclusively faecal in origin if found outside the intestine
3. Ideally they should only be found in humans
4. They should be present when faecal pathogens are expected to be present
5. They should be present in greater numbers than the faecal pathogen they are intended to indicate
6. They should be unable to grow outside the intestine with a die-off rate slightly less than the pathogenic organisms
7. They should be resistant to natural environmental conditions and to water and wastewater treatment processes in a manner equal to or greater than the pathogens of interest
8. They should be easy to isolate, identify and enumerate
9. They should be non-pathogenic

The non-pathogenic organisms that are always present in the human and animal intestine are excreted along with the pathogens, but in far greater numbers. The most widely used indicator organisms are the non-pathogenic bacteria, in particular coliforms, faecal streptococci and *Clostridium perfringens.*

4.6.1 Escherichia coli and coliforms

The coliform group includes the Gram-negative thermotolerant faecal coliforms (TTC), which are defined as being able to ferment lactose at 44°C, and include not only *E. coli* but also species of the *Klebsiella, Enterobacter* and *Citrobacter* genera. *E. coli* is considered to be the only true faecal coliform as other TTC can be derived from non-faecally contaminated waters (Gray, 2004). McLellan *et al.* (2001) found that β-glucuronidase activity was critical in distinguishing *E. coli* from other faecal coliforms.

E. coli has long been used as an indicator of faecal contamination (Geldreich, 1966; Scott *et al.,* 2002). It has good characteristics of a faecal indicator such as not normally being pathogenic to humans, and is present at concentrations higher than the pathogens it predicts (Scott *et al.,* 2002). Although recent studies have suggested that it may not be a reliable indicator in tropical and subtropical environments (Desmarais *et al.,* 2002; Solo-Gabriele *et al.,* 2000), due to its ability to replicate in contaminated soils, it remains a suitable faecal indicator in temperate regions. A detailed review of coliforms and *E. coli* as contamination indicators is presented by Gleeson & Gray (1997).

It is important to note that, in the EU Drinking Water Directive the term faecal coliform is used specifically to indicate coliforms of faecal origin which it defines as those that are thermotolerant i.e. capable of growth at 44°C. As not all coliforms are faecal in origin, they must be regarded as presumptive faecal coliforms (Gray, 2004). *E. coli* comprises up to 95% of the enterobacteria found in faeces (Waite, 1985) and therefore, throughout this research study, TTC (Chapter 6) are referred to as *E. coli.*

Generally, for the assessment of the microbial quality of untreated waters, it is the faecal coliform (*E. coli*) count which is primarily determined. However, for treated drinking waters enumeration of total coliforms is typically sufficient since it is assumed that waters designated for human consumption should not contain any microorganisms (Cabelli *et al.*, 1983; Gray, 2004).

4.6.2 Faecal streptococci (Enterococci)

The enterococcus group is a subgroup of the faecal streptococci that includes at least five species: *Enterococcus faecalis, Enterococcus faecium, Enterococcus durans, Enterococcus gallinarum* and *Enterococcus avium* (Scott *et al.,* 2002). They are differentiated from other streptococci by their ability to grow in 6.5% NaCl and at high pH (9.6) and at 10°C and 45°C (APHA/AWWA/WEF, 2005), with *E. faecalis* and *E. faecium* being the species most frequently found in humans (Cabelli *et al.,* 1983; Devriese *et al.,* 1993). The faecal streptococci are Gram -positive cocci, occurring in chains of varying length. They are both non-sporulating and non-motile (Pinto, 1999; Nola *et al.*, 2002).Faecal streptococci are considered to have certain advantages over coliforms as pollution indicators (Sinton *et al.,* 1993):

(i) They rarely multiply in water.
(ii) They are more resistant to environmental stress and chlorination than coliforms.
(iii) They generally persist longer in the environment (with the exception of *S. bovis* and *S. equinus* which die off rapidly once outside the animal intestinal tract). (For example, Andrews Jr. *et al.* (2004) have reported that a proportion of *E. faecalis* inoculated into native soils microcosms remained for at least five weeks.)

Additionally, where data pertaining to faecal coliforms (presumptive *E. coli*) and faecal streptococci are available, the faecal coliform:faecal streptococci (FC:FS) ratio may be used to indicate the likely source of faecal contamination (See *Section 4.7.1*).

It should be noted however, that despite its advantage as an indicator organism, there are a number of disadvantages associated with the use of faecal streptococci (Gleeson & Gray, 1997; Gray, 2004):

(i) They are less numerous than coliforms in human faecal matter and therefore a less sensitive indicator of human sources of contamination.
(ii) *E. faecalis* var *liquefaciens* is ubiquitous in nature with a higher persistence in water than similarly exposed faecal coliforms frequently forming a substantial proportion of the low faecal streptococci densities common to good quality waters.
(iii) Lack of standard methodology for their selective enumeration with over 70 distinct media proposed.
(iv) Taxonomical and ecological heterogeneity.
(v) Species in the group found to have differing levels of sanitary significance.
(vi) Selective enumeration is time consuming.

The issue outlined in (ii) above is of particular concern in relation to the use of faecal streptococci in assessment of relatively good quality groundwater and subsequent use of the FC:FS ratio. Pourcher *et al.* (1991) have suggested identification of specific streptococci strains as a viable solution to the problem. The use of antibiotic resistance patterns in faecal streptococci has been used by Wiggins (1996) to differentiate human and animal species.

4.6.3 Clostridium perfringens

There are a number of *Clostridium* species, including *C. perfringens, C. botulinum, C. difficile* and *C. sporogenes. Clostridium perfringens* are enteric, Gram-positive, anaerobic, spore-forming, pathogenic bacteria, found in human and animal faeces and are exclusively faecal in origin (Payment, 1993; Scott *et al.,* 2002; Araujo *et al.*, 2004; Field & Samadpour, 2007). Spores are highly resistant and capable of withstanding temperatures of up to 75°C for 15 minutes (Orsburn *et al.*, 2008).

The use of *C. perfringens* as an indicator organism was first proposed by Klein & Houston in 1899 (Bisson & Cabelli, 1979; Payment, 1993). Clostridial spores can survive in water much longer than both coliforms and streptococci, therefore, clostridial presence may be used to indicate intermittent or occasional contamination (Medema *et al.,* 1997). While the spores are resistant to disinfection which means they are ineffectual in assessing the efficacy of disinfection, they may be used to assess the efficiency of filtration systems (Fujioka & Shizumura, 1985; Gray, 2004). However, although its persistence in the environment has lead to considerable controversy surrounding its use as an indicator of faecal pollution (Scott *et al.,* 2002), several studies have advocated its use, particularly in

situations where the prediction of the presence of viruses or remote faecal contamination is desired (Fujioka & Shizumura, 1985; Sorensen *et al.,* 1989; Payment & Franco, 1993). Payment & Franco (1993) have found that *C. perfringens* counts appear to be the most suitable indicator for the inactivation and removal of viruses in drinking water treatment. They also appear to have a relationship to cysts and oocysts and could be used as indicators of their inactivation and removal. Brooks *et al.* (2005) report high correlations between *C. perfringens* spores and *C. parvum* oocyst concentrations in surface waters. However, it is not know if these correlations exist for groundwaters.

As is the case with the use of any indicator organism, there are a number of disadvantages associated with the use of *C. perfringens* as faecal indicators which has lead to their not being used as an indicator in the USA and limited use for the provision of supplementary information in Europe (Bisson & Cabelli, 1980). A number of these drawbacks are provided by Gray (2004):

(i) The spores are too resistant to chlorination to be of value in assessing treated drinking water quality.

(ii) Unsuitable for use as indicators of recreational water quality as sedimented spores can be re-suspended by bather activity, surf action or areas with significant land runoff.

(iii) Spores can survive for very long periods and in such cases spores may be detected long after a contamination incident has taken place giving rise to false positives and subsequent contamination false alarms.

(iv) A reliable method for isolation and enumeration of spores has not been found, with a number of media available (Araujo *et al.,* 2004). It is generally considered that while a number of these methods, including PCR and gene profiling (Meuller-Spitz *et al.,* 2010), are adequate for research and special investigation, they are time consuming and largely impractical for routine analysis.

4.7 Microbial Source Tracking

Microbial contamination derived from human faecal sources is generally considered as having greater human health consequences than those derived from animals, as human-derived faecal material have an inherently increased likelihood of containing human-specific enteric pathogens. Animals also serve as reservoirs for a variety of enteric pathogens (e.g. various serotypes of *Salmonella, E. coli* and *Cryptosporidium* spp.). As for indicator organisms, understanding the origin of faecal contamination is paramount in assessing associated human health risks in addition to developing appropriate remedial systems while contaminants are present in private groundwater sources.

While the aforementioned (*Section 4.6*) microbial indicators can be useful for predicting the possible presence of faecal contamination in waterbodies, their shortcomings are also increasingly apparent. The advent of microbial source tracking has enhanced the ability of these and traditional indicator organisms to be used as tools for predicting potential sources of faecal contamination in addition to health risks associated with these contamination events. Various microbiological, genotypic, phenotypic, and alternative methods have been proposed to characterise groups of, typically non-pathogenic, organisms (Scott *et al.,* 2002; Field & Samadpour, 2007), that can subsequently be used

to identify the host or environment from which the organisms were derived. For the purposes of this research, the faecal coliform/faecal streptococci ratio was used in an attempt to track the likely source of contamination where found. Therefore, this technique is outlined in some detail below. There are a number of other microbial source tracking organisms and techniques, many of which use molecular and/or biochemical techniques including *Bifidobacterium* spp., *Rhodococcus* spp., Antibiotic resistance (MAR), Coliphages, and biochemical indicators.

4.7.1 Faecal coliform/faecal streptococci ratio

As previously mentioned, faecal streptococci are generally more numerous in animal faecal matter than that of humans, while faecal coliforms are more numerous in human faecal matter than that of animals. This lead to the rationale that the use of a faecal coliform/faecal streptococci (FC/FS) ratio could be successfully used to differentiate between human and animal sources of faecal contamination, with higher FC/FS ratios indicative of human contaminant sources (*Table 4.7*).

Table 4.7 Estimated Faecal Coliform to Faecal Streptococci Ratio from Faeces of Human Beings and Some Animals

Animal	FC:FS Ratio
Human (USA)	4.33
Chicken	0.38
Cow	0.177
Duck	0.6
Pig	0.04
Sheep	0.42
Turkey	0.1
Horse	0.002

Source: Lynch & Poole, 1979; From Gray, 2004

The large FC/FS ratio differences, shown above, between human and animal faeces, are used in the following way (Gleeson & Gray, 1997):

FC/FS Ratio	*Source of Contamination*
>4.0	Strong evidence that contamination is of human origin
>2.0 <4.0	Evidence of the predominance of human wastes
>0.7 <2.0	Evidence of the predominance of animal wastes
<0.7	Strong evidence that contamination is of animal origin

As reported by Geldreich & Kenner (1969), this ratio is valid for the first 24 hours after bacteria have been discharged to a water source, which brings into question its efficacy for use in groundwater quality studies due to the fact that contamination may in some cases take days or weeks to reach an aquifer. Moreover, Pourcher *et al.* (1991) and Maier *et al.* (2009) have shown that even in the first 24 hours after sample collection, the FC/FS ratio is not constant, even for samples of the same origin. A number of previous studies (McFeters *et al.*, 1974; Sinton *et al.*, 1998; Maier *et al.*, 2009) point out that the ratio is dependent on the differential die-off rates of FC and FS, with FS being

more environmentally stable than FC. This was also found by Bitton *et al.* (1983; From Gray, 2004), whereby decay rates for faecal streptococci were lower than for faecal or total coliforms in shallow wells. Therefore, faecal ratios will change after excretion (Feacham *et al.,* 1983; Bitton, 2005).

Although the method provides rapid results and the assay requires minimal expertise to perform (Scott *et al.,* 2002), the approach overall has proven to be unreliable due to bacterial variations already outlined and has been, to some extent, abandoned as a viable approach to faecal source tracking (Pourcher *et al.,* 1991; Clesceri *et al.,* 1998; Sinton *et al.,* 1998; Gray, 2004), although some studies, such as Zmyslowska & Golas (2003) and Sankararamakrishnan & Guo (2005), have continued to use the ratio as a potential microbial indicator.

Although the FC/FS ratio has been used in this research study due to available laboratory resources, the limitations outlined above have been considered and therefore it is employed with caution and only as a potential indicator in concurrence with other, more reliable data.

4.8 Chemical Indicators

There are a number of chemical parameters that, when correctly interpreted, may be used to indicate groundwater quality variations or, in some cases, groundwater contamination. Unlike previously outlined indicator organisms (excluding total coliforms and *E. coli*) and microbial source tracking organisms, these parameters are routinely analysed for and analytical methods are relatively simple. An important point to note with regard to inorganic parameters and their presence in groundwater sources, as anthropogenic activities have had some impact on a high proportion of groundwater bodies in Ireland, and therefore there are relatively few areas where groundwater is in a pristine, natural state.

The following parameters may be used to assess groundwater contamination (*Tables 4.8 & 4.9*):

Table 4.8 Recommended Parameters for Contamination Assessment

Colour	Calcium	Nitrate*	Sediment
Magnesium	Ammonia*	pH	Sodium*
Iron*	Conductivity*	Potassium*	Manganese*
Total hardness	Chloride*	Sulphate*	Alkalinity

Table 4.9 Optional Parameters (depending on local circumstances and sampling rationale)

Fluoride	Fatty acids*	Zinc	Orthophosphate
Trace organics*	Copper	Nitrite*	TOC*
Lead	BOD*	Boron*	Other metals
Dissolved Oxygen*	Cadmium		

**Good indicators of contamination; adapted from Daly, 1994b; Kiely, 1997*

4.8.1 Ammonia

Ammonia is found in its free state as NH_3, or in its ionic state, as NH_4^+ (ammonium) (Cronin & Furey, 1998). Ammonia is normally present in groundwaters in very low levels (<0.1 mg L^{-1} NH_4) (Spain, 2000), due to the process of microbial reduction in unsaturated acidic environments (e.g. subsoils), which oxidises ammonia, firstly to nitrite and then to nitrate (EPA, 2010c). It has a low mobility in soil and subsoil, and its presence in groundwater above background levels indicates a nearby organic waste source, such as farmyard effluent, manure effluent or septic tank effluent, and/or vulnerable conditions (EPA, 2005a; EPA, 2007). Elevated levels of ammonia in groundwater commonly indicate ingress from septic tank effluent, as these contain high NH_4 concentrations, with NH_4 being the main form of N in septic tank effluent (Gill *et al.,* 2009b; O'Luanaigh *et al.,* 2009). Cronin & Furey (1998) suggest that the presence of ammonia >0.1 mg L^{-1} NH_4 may be an indication of organic waste contamination, particularly direct septic tank, sewage, industrial or agricultural contamination. This figure of >0.1mg L^{-1} NH_4 as a potential indicator is again suggested by EPA (2005b), who go on to state that groundwater NH_4 levels were typically low, with almost 85% of groundwater stations exhibiting NH_4 below the guideline level (0.05 mg L^{-1}) during the period 2001-2003. Further data (EPA, 2007) indicate no significant change in groundwater NH_4 levels during the period 2005-2006, with natural attenuation capacities of soils and subsoils stated as being the likely reason for these low levels. Conversely however, it is notable that where groundwater supplies are situated in vulnerable areas (i.e. thin, highly permeable subsoils), particularly shallow, poorly protected supplies, may exhibit higher NH_4 levels and/or more frequent exceedances of guideline levels.

The EU MAC level for ammonium is 0.3mg-NH_4 L^{-1} (0.23 mg-N L^{-1}), with a guide level set at 0.05mg L^{-1}. As set out by EPA (2010c), the EU MAC level is in place as elevated ammonia concentrations indicate the possibility of sewage pollution and consequently the possible presence of pathogenic micro-organisms, and is therefore of human health significance.

4.8.2 Nitrate

One of the major components of groundwater pollution is nitrate (NO_3), due to its diffuse type and the difficulty of controlling and managing it (Behera *et al.,* 2003) and is one of the two forms of nitrogen of concern with regard to groundwater contamination, the other being ammonia. Nitrate in increasing concentrations has been recorded in many developed countries including Ireland (Spain, 2000; EPA, 2005b), with groundwater nitrate concentrations in Europe found to exceed the EU MAC level in 22% of cultivated regions (Sacco *et al.,* 2007). The nitrate ion is not absorbed on clay or organic matter (Kolenbrander, 1975), is highly motile and therefore easily leached out of the root zone through soil and subsoil (Kiely, 1997). This is highlighted by Richards (1999), who identified peaks in groundwater nitrate exceedances in Ireland in 1990 and 1996 during winter drainage periods following drought conditions. As normal concentrations in uncontaminated groundwaters are low (<5.0mg L^{-1}), nitrate can be a good indicator of contamination by fertilisers and waste organic matter (Nolan & Hitt, 2006), due to elevated levels normally being of anthropogenic origin, particularly in shallow groundwaters, which are commonly utilised as private domestic sources in Ireland, with the main sources of nitrate contamination considered to be industrial wastes, municipal refuse, and agriculture, especially the intensive application of nitrogen fertilisers (Wang & Yang, 2008).

High nitrate concentrations in drinking water have been linked with adverse health effects (Hancock *et al.*, 2000; Kimura *et al.*, 2002), although nitrates in themselves are non-toxic, unless excessive quantities are ingested. It is the conversion of nitrates to nitrites and nitrosamines which may result in health risks to humans. Nitrite reduces the oxygen carrying capacity of haemoglobin in the blood, leading to a condition known as methaemoglobinaemia. It has been found to primarily affect infants (also known as blue baby syndrome), normally occurring in small rural water supply areas, of poor microbial quality. A nitrate level of 50mg L^{-1} appears to be the minimum level at which the problem occurs (Spain, 2000). Ten cases were recorded in the UK during the period 1949-1972, however notably, none having been detected since (Croll, 1994). Fewtrell (2004) has stated that *"given the apparently low incidence of possible water-related methaemoglobinaemia, the complex nature of the role of nitrates and that of individual behaviour, it is currently inappropriate to attempt to link illness rates with drinking water nitrate levels".*

Oesophageal and stomach cancers have also been linked to N-nitrous compounds. One study (Ward *et al.,* 1996) has linked high nitrate levels in drinking waters with increased incidence of non-Hodgkins lymphoma, also discussed by Freedman *et al.* (2000) and Wolfe & Patz (2002). Freedman *et al.* (2000) found however that, there was no apparent association between non-Hodgkins lymphoma and nitrate consumption within the range of study exposures (0.1-7.2 mg-NO_3-N L^{-1}).

Although, as mentioned above, nitrate has been associated with adverse human health effects, these tend to be linked with nitrate levels rarely found in private groundwater supplies in Ireland and, as set out by Fewtrell (2004), are part of a complex overall process involving other forms of nitrogen (ammonia and nitrite) and bacteria (often due to poor hygiene), the role of groundwater nitrate is usually that of an indicator of anthropogenic contamination. This will be its primary function throughout this research project. In Ireland, excessive nitrate contamination is usually observed in relatively shallow, poorly productive wells, adjacent to non-point or more typically, point waste sources (EPA, 2005a). Spain (2000; after Daly, 1994b) points out that a groundwater nitrate level >25 mg-NO_3 L^{-1} may be a useful indicator of an organic waste source nearby. EPA (2010c) state that a groundwater nitrate concentration >10 mg NO_3- L^{-1} may be indicative of anthropogenic organic or inorganic inputs, with a recent study reporting a median nitrate concentration of 3.3 mg NO_3-N L^{-1} in Irish groundwater sources (OCM, 2007). Further statistics relating to this study are provided in Appendix C2.

The current EU MAC Level (See Appendix C3) is 50 mg-NO_3 L^{-1} with guide and threshold levels of 25 mg-NO_3 L^{-1} and 37.5 mg-NO_3 L^{-1} respectively.

4.8.3 Chloride

Chloride exists in all natural waters with a range of 16-29 mg-Cl L^{-1} for groundwater sources in Co. Offaly put forward by Spain (2000). Background levels of 12-15 mg-Cl L^{-1} in uncontaminated groundwaters in Co. Offaly were reported by Daly (1994b). Furthermore, a nationwide study of natural background parameter levels among Irish groundwaters reported a median chloride concentration of 18 mg-Cl L^{-1} (OCM, 2007). Potential sources of Cl in groundwater are natural sources (precipitation, saline groundwater), agrichemicals (KCl), animal waste, septic effluent and road salts (Panno *et al.,* 2001). The principal source of chloride in uncontaminated groundwater in

Ireland is rainfall and therefore, depending upon distance from a saline waterbody (seawater contains up to 35,000 mg-Cl L^{-1}) and evapotranspiration, levels of Cl will be reasonably constant (Kiely, 1997; Neal & Kirchner, 2000). Human and animal sewage contain large concentrations of chlorides, as do some industrial effluents and consequently, elevated Cl levels, in addition to elevated Cl:Br and Cl:Fl ratios, which have been previously used to indicate contamination by organic wastes (Vengosh & Pankratov, 1998; Morales *et al.,* 2000; Hudak, 2003a). Spain (2000) found that chloride peaks typically coincided with peaks and spikes in sodium, potassium, sulphate and nitrate, and appeared to be a consistent parameter in relation to contamination events.

The EU MAC level is 250 mg-Cl L^{-1}, with the corresponding guide level set at 25 mg-Cl L^{-1}. The EU MAC level is set at 250 mg-Cl L^{-1}, as above this concentration, water will begin to taste salty, becoming increasingly objectionable with further increases in concentration (EPA, 2001). The EU MAC is therefore an "aesthetic guideline", as groundwaters (or surface waters) at this concentration do not pose any significant health hazard.

The Geological Survey of Ireland (GSI) currently employ a "rule of thumb" gained from experience in investigating contamination of groundwater as follows (Hunter-Williams, *pers comm*, 2009):

- >25 mg L^{-1} Cl indicate potential contamination has occurred
- > 30 mg L^{-1} Cl indicate contamination has occurred

Banks (1998) indicates that Na and Cl typically tend to exhibit similar distributions although, a higher proportion of groundwater Na is derived from geological sources i.e. feldspar weathering, than is the case for Cl and therefore, the Na:Cl ratio may be used as an indicator of water – rock interaction.

4.8.4 Electrical Conductivity

Electrical conductivity is *"a measure of the ability of an aqueous solution to carry an electrical current"* (DeZuane, 1997) and is one of the primary physicochemical parameters in water quality analysis. The electric current is conducted in the solution by the movement of ions and so increased ionic concentrations (i.e. the greater the concentration of dissolved inorganic salts) result in increased ionic mobility and subsequently higher EC (Kiely, 1997; Misstear *et al.,* 2006). Due to the very low concentration of ions in pure water (i.e. deionised water only has H^+ and OH^- ions present), very low values of conductivity are observed (about 0.05 µS/cm), whereas, seawater with high salinity has a conductivity of approximately 40,000 µS/cm. Consequently, a significant empirical relationship exists between EC and total dissolved solids (TDS) as most of the dissolved content in groundwaters tends to be in ionic form (Misstear *et al.,* 2006). Typical conductivity values are outlined in *Table 4.10*. As electrical conductivity is influenced by the aquifer geology, no median/mean figure exists with regard to Irish groundwater, with natural background levels ranging from 67 µS cm^{-1} in granite areas to 721 µS cm^{-1} in confined limestone aquifers (OCM, 2007). Further statistics relating to conductivity values in specific bedrock lithologies are contained in Appendix C2.

A number of studies have used EC as a physicochemical indicator of potential groundwater contamination (Helena *et al.,* 2000; Liu *et al.,* 2003; Mor *et al.,* 2006; Yik-Lin *et al.,* 2010), with changes in groundwater flow regimes or pollutant ingress often highly correlated with sharp increases in measured EC, in addition to precipitation events. Due to the nature of electrical

conductivity as a physico-chemical parameter, strong positive correlation of electrical conductivity with the most abundant groundwater ions are expected, particularly sulphate, chloride, calcium, magnesium, sodium and potassium. For example, Helena *et al.* (2000) report Spearman correlation values in the range r = 0.5-0.87.

Table 4.10 Typical Conductivity Values

Water	Conductivity range (µS cm^{-1})
Chemically pure	0.05
Distilled	0.1-4
Rainwater	20-100
Soft water	40-150
Hard water	200-500
Range of rivers	100-1,000
Groundwater	200-1,500
Estuarine water	200-2,000
Seawater	40,000

(Kiely, 1997)

4.8.5 Iron and Manganese

Iron and manganese are both present under natural conditions in groundwater in some areas, with groundwater from rock types such as dark muddy limestones, shales and sandstones or from peaty areas often presenting high iron concentrations (Daly, 1994b). Both iron and manganese can also be good indicators of contamination by organic wastes, with effluent from these wastes typically causing de-oxygenation, resulting in dissolution of iron and manganese from the soil, subsoil and bedrock into groundwater and subsequent re-oxygenation in the groundwater (Kiely, 1997). High manganese concentrations can also be an indicator of groundwater pollution by silage effluent in addition to other high BOD wastes such as milk, landfill leachate and in some cases soiled water and septic tank effluent (Daly, 1994b; Kiely, 1997), and have been used in a number of previous groundwater contamination studies (Jensen *et al.*, 1998; El Arabi, 1999; McQuillan, 2004). Typical background values in relation to iron and manganese concentrations in Irish groundwaters are 25 µg L^{-1} and 8 µg L^{-1} respectively (OCM, 2007).

EU MAC levels for iron and manganese are 0.2 mg L^{-1} Fe and 0.05 mg L^{-1} Mn, respectively, with guide levels currently set at 0.5 mg L^{-1} Fe and 0.02 mg L^{-1} Mn.

4.8.6 Potassium

As shown by Eiswirth & Hotzl (1997), increased levels of potassium at urban water wells in Germany were indicative of the influence of wastewater (leaking sewers), with increased potassium loading also used by Datta *et al* (1997) to indicate groundwater contamination from fertilisers in the Delhi area. Trauth & Xanthopoulos (1997) suggest that potassium may be a useful indicator of over-fertilised agricultural land in addition to wastewater ingress, with their study showing unpolluted groundwater with typical potassium concentrations of 1-5mg/L, with exceedances noted in agricultural and heavily populated urban areas. Studies by Howard & Beck (1993) and Hellstén *et al* (2005) have also found groundwater potassium exceedances in agricultural and urban areas.

Background potassium levels in most areas of Ireland are typically <3.0 mg-K L^{-1}, although higher concentrations are occasionally found in Ireland where the dominant local bedrock contains potassium i.e. granites and sandstones (Kiely, 1997), with a typical range of 1.1 mg L^{-1} (Devonian bedrock) to 2 mg L^{-1} (Lower Palaeozoic bedrocks) (OCM, 2007). Daly & Daly (1982) have suggested that groundwater potassium concentrations greater than 5.0 mg L^{-1} may indicate local, vegetable-based contamination. A number of studies (Daly, 1994b; Whitehead *et al.,* 1999; Griffioen, 2001; Krapac *et al.,* 2002) have used elevated potassium levels in private groundwater sources as being indicative of organic waste ingress in general, and farmyard and/or animal waste in particular. Elevated potassium levels in groundwater (and consequently elevated K:Na ratios – see Section 4.8.8), should however, be interpreted with caution, and two factors must be taken into consideration. Firstly, as previously mentioned, elevated potassium concentrations may result from leaching of a potassium rich geological material, such as granite or sandstone, or from spreading of potassium fertilisers. Secondly, potassium may be adsorbed by negatively charged colloidal components of soil and rock as it is a positively charged ion, with potassium levels in the subsoil largely controlled through cation-exchange (Griffioen, 2001). These colloidal components may include silicate clay minerals, iron and aluminium oxides and organic colloids, depending on the nature and extent of colloids present e.g. peat will adsorb significantly more potassium than sand, due to a generally higher concentration of highly charged organic colloids (Thorn & Hanna, 1989).

The current EU MAC level for potassium is 12.0 mg-K L^{-1}.

4.8.7 Sodium

Sodium is ubiquitous in all natural waters, with its presence in groundwater largely due to erosion of salt deposits and sodium-bearing rock minerals i.e. it is highly bedrock dependent. Other groundwater sources of sodium are naturally occurring brackish water in some aquifers, salt water intrusion into wells in coastal areas and irrigation and precipitation leaching through soils high in sodium (EPA, 2003). When taken in excess it can cause hypertension or, in conjunction with sulphate, sodium sulphate may form, resulting in laxative effects (US EPA, 2009). Potential sources of elevated sodium in groundwater wells include rock salt treatment of roads during freezing conditions, ingress of detergents, contamination by sewage effluent or infiltration of leachate from landfills or industrial sites (Panno *et al.,* 2001; Urbansky, 2002; The British Colombia Groundwater Association, 2007). Sodium is also a constituent of animal feeds (Mackie *et al.,* 2006).

Normal background levels in uncontaminated groundwater in Ireland are 5-15 mg L^{-1} Na (Spain, 2000), with OCM (2007) reporting a range of 9 mg L^{-1} (Devonian bedrocks) to 19 mg L^{-1} (Lower Palaeozoic bedrocks). Significant departures from these figures are a potential indicator of contamination (Spain, 2000). The current EU MAC Level (see Appendix C3) is 150 mg-Na L^{-1} (80% percentile over 3 years), with a guide level of 20 mg-Na L^{-1}. It is notable however, that due to cation-exchange, whereby sodium replaces calcium during and following the dissolution of calcite, particularly in mature groundwaters, marked increases in sodium may be observed (US EPA, 2009).

Additionally, the potassium:sodium (K/Na) ratio may be utilised in order to characterise groundwater contamination (Daly & Daly, 1982), as discussed in *Section 4.8.8*.

4.8.8 Potassium:Sodium (K:Na) Ratio

Ellis (1980) showed that during decay of vegetable matter, a greater volume of potassium than sodium is released and thus elevated potassium levels indicate vegetable-based contamination. This interpretation has been extended by Daly & Daly (1982) to include the K:Na ratio, whereby potassium levels of greater than 5.0mg L^{-1} coupled with K:Na ratios greater than 0.3 may indicate local, vegetable based contamination i.e. contamination by plant organic matter (e.g. farmyards, landfills). This threshold ratio was later increased to 0.4 (Daly, 1994b; Spain, 2000). Conversely, it is not considered to be an indication of contamination by septic tanks as this effluent normally has a K:Na ratio <0.3 (Spain, 2000). Additionally, Pionke *et al.* (1990) found the K:Na ratio to have a good correlation with NO_3-N in groundwaters, particularly at the higher end of the range, correctly identifying 62.5% of high nitrate groundwater wells.

Similar to the interpretation of K concentrations as potential contamination indicators, caution should be applied in the interpretation of K:Na ratios for the reasons previously discussed i.e. high K levels in granite and sandstone areas and/or areas where significant cation-exchange in the sub-surface is likely. For the purpose of this research, the more conservative K:Na ratio of 0.4, will be adopted as a potential indicator of agriculturally derived contamination.

Spain (2000, After Daly, 1994b) have presented *Table 4.11* for assessing groundwater quality analyses in terms of potential contamination.

Table 4.11 Summary of Indicator Parameters and Interpretation

Indicator Parameter	Assessment
***E. coli* present**	Organic waste source nearby (excl karst areas), usually septic tank or agriculture
***E. coli* absent**	Uncontaminated or die-off due to attenuation or time of travel
NO_3>25mg L^{-1}	Organic fertiliser or organic waste source nearby
NH_4>0.15mg L^{-1}	Organic waste source nearby, fertilisers not an issue
K > 5.0mg L^{-1}	Organic waste source nearby
K:Na Ratio >0.4	Agricultural waste rather than septic tank effluent, if <0.4 no conclusion possible
Cl >30 mg L^{-1}	Organic waste source (excl areas close to coastline <20km)

A number of potential chemical tracers or indicators also have potential for use in contaminant source tracking. These include caffeine, anionic surfactants, fluoride, fluorescence whitening agent (FWA), bile acids, pesticides, polycyclic aromatic hydrocarbons (PAH) and faecal stanols/sterols (Bull *et al.*, 2002; Scott *et al.*, 2002; Sankararamakrishnan & Guo, 2005; Blanch *et al.*, 2006). To date, these have not been extensively used in groundwater research in Ireland, nor have they been utilised in this study; although one recent Irish study (Gill *et al.*, 2009a), investigated the fate of caffeine (as a potential indicator of on-site effluent) in on-site domestic wastewater systems, and found considerable removal in the aerobic subsoil conditions compared to little degradation of caffeine in saturated anaerobic environments.

Chapter 5: Groundwater and Human Health Risk Assessment

5.1 Introduction

Contaminants of public health concern, including enteric pathogens, can gain entry into drinking water supplies in both developing and developed countries. In the case of surface water, which is generally considered more vulnerable to contamination than groundwater, break-through of pathogens due to treatment system failure or ingress through cracked pipes are well documented causes of waterborne outbreaks of cryptosporidiosis (Craun *et al.,* 2002; O'Toole *et al.,* 2004; Cummins *et al.,* 2010) and *E. coli O157*:H7 (Licence *et al.,* 2001; Sayah *et al.,* 2005; Muniesa *et al.,* 2006). In the case of groundwater, contamination typically occurs due to direct ingress of the contaminant into the aquifer in areas of high groundwater vulnerability (Yates *et al.,* 1985; Macler & Merkle, 2000; Ibekwe & Grieve, 2003) or ingress at the wellhead due to poor construction (Exner & Spalding, 1985; Steichen *et al.*, 1988; Borchardt *et al.*, 2003; Borchardt *et al.*, 2004 Simpson, 2004; Gonzales, 2008).

As shown in *Figure 5.1*, during the period 1995-2004, waterborne disease outbreaks in the USA were associated with groundwater sources in >50% of cases, with a maximum of 92% during the period 2001-2002. It is noted that latest figures suggest that over 15 million USA households rely on private, household wells for drinking water (US Census Bureau, 2008). Furtado *et al* (1998) suggest that during the period 1992-1995, private groundwater sources in the UK contributed to approximately 35% of all waterborne disease outbreaks, with these supplies currently serving 5-10% of the UK population (Robins & Misstear, 2000). It is therefore evident that private groundwater sources currently contribute a significant proportion of waterborne pathogenic burden in developed countries

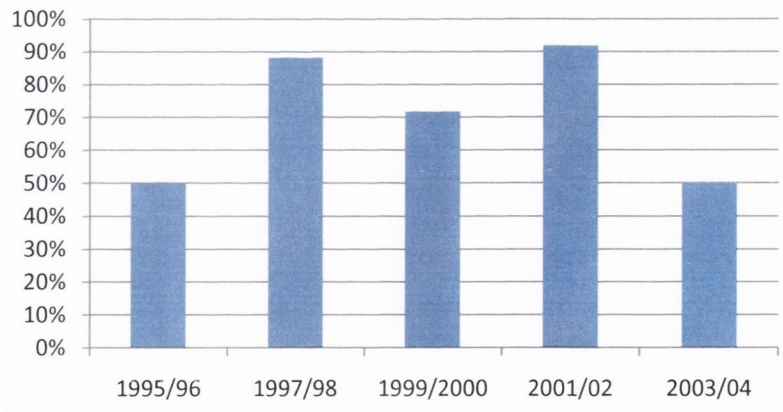


Figure 5.1 Percentage of waterborne disease outbreaks associated with groundwater consumption, USA 1995-2004 (Liang et al., 2006)

Consequently, the accurate prediction or estimation of diseases attributable to particular sources is of importance, be it on a large scale i.e. "how many reported cases of gastroenteritis in Ireland per annum can be associated with groundwater contamination?" or on a smaller scale, i.e. "how many cases of gastroenteritis reported in a particular region are as a direct result of a specific water supply?". Furthermore, it is important to appropriately attribute these infections to the correct

source, so that effective corrective actions may be taken i.e. what proportion of gastroenteric cases are attributable to foodborne exposure routes? Risk assessments are therefore necessary to determine which waterborne contaminants present the greatest risk to humans. Resources and control strategies (i.e. groundwater protection, wellhead protection, guidance) may then be focused on those agents or locations which have the greatest potential for adversely impacting human health.

This chapter outlines the concepts of risk, risk analysis and risk assessment. Particular emphasis is placed on quantitative microbial risk assessment and probabilistic risk modelling as utilised in this research study. Finally, a number of waterborne quantitative risk assessments previously undertaken in Ireland and abroad are detailed.

5.2 Epidemiology

Merriam & Webster (2005) define epidemiology as "*the branch of medicine dealing with the incidence and prevalence of disease in large populations and with detection of the source and cause of epidemics of infectious disease*". Epidemiology examines both endemic and epidemic infections and is based on the observation that most diseases do not occur randomly, but are related to environmental and personal characteristics that vary by place, time, and subgroup of the population.

The first recorded epidemiological investigation was undertaken in London, in 1848 and 1854. Meticulous, logical examination of the figures surrounding cholera epidemics by John Snow (1813–1858) revealed the mode of communication of the disease as being the public water pump on Broad Street in the Soho district (Johnson, 2006). This was carried out through epidemiological mapping and lead to Snow being widely regarded as the founder of modern epidemiology (Beaglehole *et al.*, 1993; Paneth, 2004).

5.3 Risk Analysis and Assessment

Risk

Risk is the likelihood or expected frequency of a specific adverse consequence. It is the combination of the probability of an event occurring (such as contamination of a groundwater resource) and its consequences. The Codex Alimentarius Commission (1999) defines risk as "*a function of the probability of an adverse health effect and the severity of that effect, consequential to a hazard(s) in food (or drinking water)*". An example from the microbial field might be the probability of contracting symptomatic infection from *E coli* O157:H7. In this case the risk is composed of the probability of an individual being exposed to the bacteria (probability of an event) and the impact if the individual is exposed (consequential effect).

Risk Analysis

Risk assessment is part of what is known in a broader context as *Risk Analysis* (RA).RA as a process consists of three separate but intrinsically linked components: risk assessment, risk management, and risk communication, as shown in *Figure 5.2* (Hoppin, 1993; Soller *et al.*, 1999; Vose, 2000; Soller, 2006). The overall objective of risk analysis, as set out by the Codex Alimentarius Commission (1999) is to ensure public health protection.

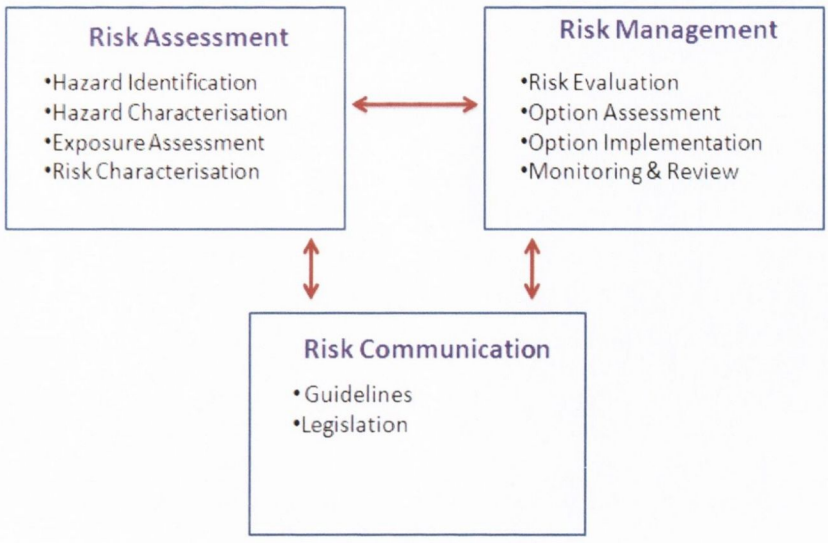


Figure 5.2 Components of Risk Analysis

In essence, risk assessment is the estimation (quantitative where possible) of risk associated with a particular source or process; risk management is the development of strategies or guidelines with the objective of risk minimisation or, where possible, risk eradication; and risk communication is the exchange of information and developed strategies with those people potentially affected.

As the quantitative assessment of risk is a key part of this research, this is now reviewed in some detail. Risk management and risk communication are also outlined, in *Section 5.12* and *Section 5.13*, respectively.

Risk Assessment

Risk assessment may be characterised by the questions it seeks to answer, as follows:

- What can go wrong? *(Identification of hazards and outcomes)*
- How likely is it to go wrong? *(Estimation of outcome probability i.e. vulnerability)*
- What would happen if it did go wrong? *(Consequence analysis)*

Risk assessment is defined by Vose (2000) as "*the quantification, either qualitatively or quantitatively, of the probability and the potential impact of some risk*". In the case of a microbial risk assessment, the process involves identification and subsequent estimation of adverse human health effects following exposure to a microbial pathogen (Parkin, 2007). At its very core, risk assessment is therefore, a systematic method to identify hazards and subsequently estimate the risk posed by these hazards.

Numerical models have been developed for a range of waterborne pathogens in drinking water, including *Giardia* (Regli *et al.,* 1991), enteric viruses (Haas *et al.,* 1993), and *Cryptosporidium parvum* (Teunis *et al.,* 1997; Cummins *et al.*, 2010), with the focus of these models being the prediction of the magnitude of waterborne infection within a defined population. The majority of these models represent non-outbreak conditions and model the endemic levels of infection through drinking

water (Gale, 2001) similar to the approach adopted herein. However, models have also been developed for conditions representative of waterborne outbreaks (Gale & Stanfield, 2000)

5.3.1 Risk Assessment Components

The field of risk assessment in its current form has its beginnings in the "Red Book" first published by the National Research Council in 1983 (CAC, 1999; Miliotis & Buchanan, 2009). The "Red Book" also defined the four recommended steps in the risk assessment process, known as the risk assessment paradigm, referred to by Lammerding & Fazil (2000) as the four cornerstones of microbial risk assessment (*Figure 5.3*), as follows:

Hazard Identification (Section 5.4)

Description of the acute and chronic human health effects associated with any particular hazard, including toxicity, carcinogenity, mutagenicity, developmental toxicity, reproductive toxicity, and neurotoxicity.

Hazard Characterisation (Section 5.5)

Characterisation of the relationship between various doses administered and the incidence of the health effect.

Exposure Assessment (Section 5.6)

Determination of the size and nature of the population exposed and the route, amount, and duration of the exposure.

Risk Characterisation (Section 5.7)

Integration of the information from exposure, dose-response, and health steps in order to estimate the magnitude of the public health problem and to evaluate variability and uncertainty.

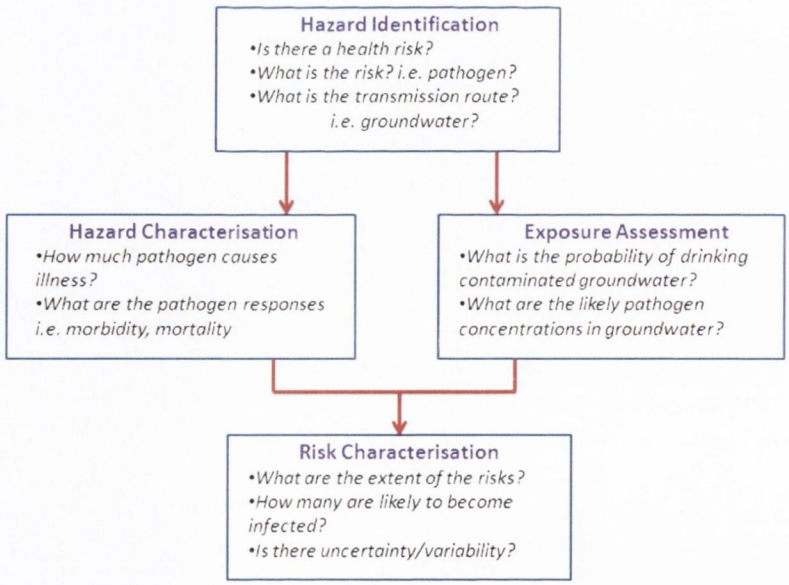


Figure 5.3 4-Step Risk Assessment Approach (CAC, 1999)

5.4 Hazard Identification

The first step in risk analysis and modelling is the recognition and subsequent identification of an existing or potential risk. For the effective management of environmental health risks, as is the case in this study, it is important to identify which environmental sources, pathogens, or situations lead to illness, and to determine the magnitude of the impact these have on human health (Krewski *et al.,* 2004). Therefore, the hazard identification, in the context of this study, is largely a qualitative evaluation of the risk i.e. determination if there is sufficient evidence to consider a waterborne pathogen as the cause of an adverse health effect in Ireland (Lammerding & Fazil, 2000; Pouillot *et al.*, 2004).

This information may be obtained from a variety of sources including scientific literature, industrial databases, government agencies (EPA, HSE, NDSC, CDC), relevant international organisations and through solicitation of expert opinion. Relevant information includes data from areas such as: epidemiological studies and surveillance, laboratory animal studies, investigations of the characteristics of microorganisms, the interaction between microorganisms and their environment through the "consumption" chain from primary development up to and including consumption, and studies on analogous microorganisms and situations (Gibson *et al.,* 1998; Krewski *et al.,* 2004).

The understanding and description of the human health hazards associated with microorganisms has primarily been part of three fields of study. The field of clinical microbiology examines the nature of the infectious microorganism i.e. *E. coli, Cryptosporidium* spp. The field of epidemiology focuses on specific transmission routes i.e. private groundwater sources in Ireland. Finally, the medical field investigates the human response to infection i.e. gastroenteritis, cryptosporidiosis (Gibson *et al.,* 1998; CAC, 1999; Haas *et al.*, 1999; Rowan, 2004; Pouillot *et al.,* 2004).

The cause and effect relationship for microbial hazards can often be measured over relatively short periods of time (hours, days, or weeks) compared to chemical hazards which may have cause and effect relationships measured in decades (Lammerding & Fazil, 2000). This rapid cause and effect relationship allows for a more positive association between a particular pathogen and illness in individuals or a population (Sobsey *et al.,* 1993). Essentially, the hazard identification step is largely an evaluation of the available information, and serves to document the important information known about the pathogen, waterborne frequency and host interaction. For the purposes of this study, the hazard identification component of risk assessment has been largely undertaken and presented in the literature review (*Chapter 4*).

5.5 Hazard Characterisation

Paracelsus (1494-1541) is noted for first proposing the concept that "*the dose makes the poison*". At extremely low doses, a given substance may be non-toxic and even beneficial (a concept known as hormesis) (Mattson, 2008). At intermediate doses it may be toxic, while at high doses it may be lethal. This underscores the importance of understanding the waterborne hazard being studied, and in particular, the human dose-response relationships for the pathogens in question (Rozman & Doull, 2000; Renner, 2004; Mattson, 2008).

Characterisation of the risks associated with waterborne contaminants, particularly waterborne pathogens, is dependent on the availability of information on the population's exposure to the

biological agents. However, exposure data in the absence of hazard characterisation data are insufficient to assess the public health impacts of waterborne pathogens (Haas *et al.,* 1999). This requires the availability of effective dose-response models, which are dependent on a sound understanding of the mechanisms of pathogenicity associated with individual microbial pathogens, based on the interactions comprising the infectious disease triangle (Scholthof, 2006) (also known as the "EPI triangle") (*Figure 5.4*). The biological basis for dose-response models derives from major steps in the disease process as they result from the interactions between the pathogen, the host and the pathogenic pathway or vector (therefore justifying the use of the Hazard-Pathway-Receptor model).

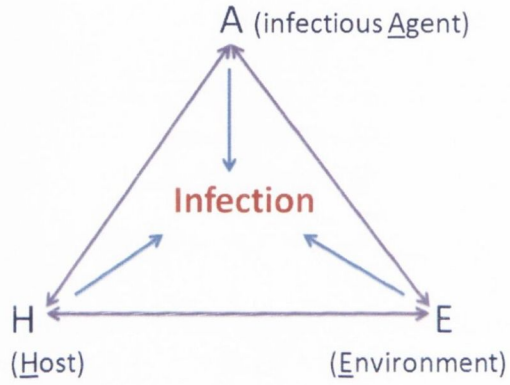


Figure 5.4 The Infectious Disease Triangle (adapted from Scholthof, 2006)

The fundamental goal of a dose-response assessment is to describe the probability of a specified response such as symptomatic infection, to a specified pathogen, as a function of the dose (WHO, 2003) or in essence, quantify the risk of infection once a pathogen or pathogens have been consumed. This relationship is usually expressed in mathematical terms in the form of a dose response model or curve (also known as the dose response envelope).

The development of dose response models is necessary due to a number of issues, the foremost of which are ethical in nature, as to permit the direct assessment of risk through human studies would involve deliberate infection of human subjects. Therefore, data from previous pathogen outbreaks, population based studies, surrogate organisms or animal studies are typically used for development of models (Haas *et al.,* 2000; Teunis *et al.,* 2002; Zmirou-Navier *et al.,* 2006; Teunis *et al.,* 2008a).

An example is given by Haas *et al* (2000) whereby data from an animal exposure study for *E. coliO157*:H7 (Pai *et al.,* 1986) (*Table 5.1*), were used to develop a best fit dose response curve for human infection. In this case, a Beta-Poisson response curve was proposed.

Table 5.1 Dose response data for E. coli O157:H7

Dose	Number of subjects	Positive (infection)
Control	7	0
100,000	3	0
1,000,000	5	2
10,000,000	5	5
100,000,000	13	12
1,000,000,000	5	5
3,000,000,000	2	2
10,000,000,000	6	6

Haas et al., 2000; from Pai et al., 1986

5.5.1 The Dose-Response Curve

Dose-response curves are the most commonly used method for exhibiting mathematical dose-response relationships (Buchanan *et al.,* 2000). Typically, they are simple X-Y graphs relating the magnitude of a stressor (e.g. concentration of a pollutant, pathogen intake) to the response of the receptor (e.g. human or animal). The studied response (endpoint) may be death (mortality), asymptomatic infection or symptomatic illness, with the X-axis plotting the concentration of a particular pathogen and the Y-axis signifying the associated response (Steenland & Deddens, 2004; Ginevan & Watkins, 2010). The response of interest in the context of this study will be the onset of gastrointestinal illness as a direct result of consumption of microbially contaminated groundwater.

Commonly, it is the logarithm of the dose that is plotted on the X axis, and in such cases the curve is typically sigmoidal (Coleman & Marks, 1999; Holcomb *et al.,* 1999). This sigmoidal or S-shaped curve, as shown in *Figure 5.5* below, is also known as a logistic function (von Seggern, 2007).

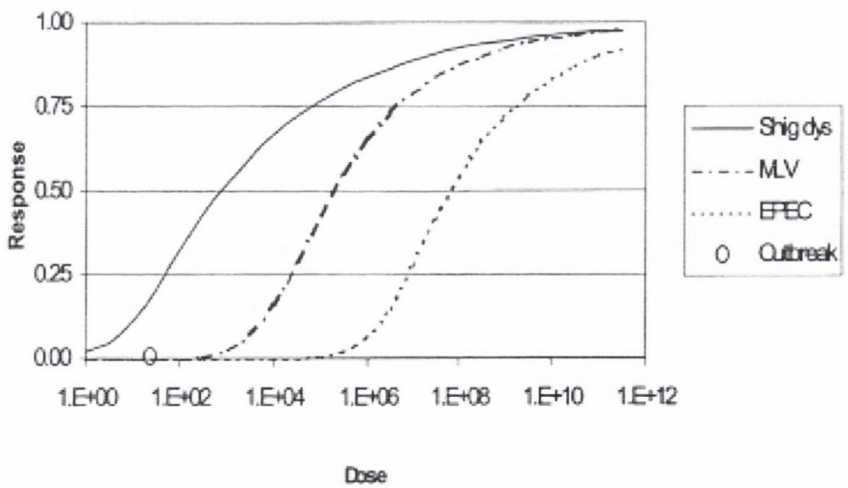


Figure 5.5 Dose-response envelope for E. coli O157:H7 (Powell, 2000)

Figure 5.5 presents dose-response envelopes for both *Shigella dysenteriae* and enteropathogenic *E. coli* (EPEC). As evidenced by Powell (2000), due to the high level of infectivity associated with *E. coli* O157:H7, few human volunteer studies have been carried out with this pathogen. Therefore, S. *dysenteriae* and EPEC have been proposed as upper and lower dose-response limits, with *E. coli* O157:H7 represented as the most likely value (MLV) in *Figure 5.5*. The first point along the graph where a response above zero is reached is usually referred to as a threshold-dose i.e. approximately 1 organism, 100 organisms and 100,000 organisms for *S. dysenteriae*, *E. coli* O157:H7 and EPEC respectively. The more infective a particular pathogen is found to be, the steeper this curve will be. In quantitative situations, the Y-axis usually is designated by percentages, which refer to the percentage of users registering a standard response e.g. infection, illness, death) (DuPont *et al.*, 1995; Powell, 2000).

It is important to note that, a dose-response curve defines the relationship between dose and response based on the following assumptions: 1) response increases as dose increases, and, 2) there is a threshold dose i.e. a dose below which there is no effect (Haas *et al.*, 1999). As shown in *Table 5.1* and *Figure 5.5*, the dose-response data for *E. coli* O157 satisfy both assumptions.

5.5.2 Dose-Response Assessment Methods

The experimental methods commonly used for undertaking dose-response assessments and consequently developing dose-response relationships are:

- Animal Studies
- Analysis of historical data
- Volunteer studies
- Outbreak Investigations
- Epidemiological Studies

(Teunis *et al.*, 1999; Haas *et al.*, 2000; Chen *et al.*, 2006)

Data are fitted to a dose-response relationship which describes observed infectivity as a function of dose (Messner *et al.*, 2001; Teunis *et al.*, 2002). Mathematical models have been used to describe dose-response data empirically for a number of pathogenic microorganisms, in addition to chemical hazards and human diseases. A number of the most widely used numerical dose response models and their respective uses are outlined in *Table 5.2*.

Dose response models range from one parameter models, such as the exponential model to three parameter models, including the Weibull-Gamma and Gompertz models, which are more biologically complex and therefore require more clinical input data, requiring medical evidence at four or more dose levels (Moon *et al.*, 2004). For this reason, two parameter models are often used in microbial risk assessment due to the greater level of complexity compared with the exponential models, but with a lower data requirement (Haas *et al.*, 1999). The most popular of the two parameter models is the Beta Poisson (BP) model, with Moon *et al* (2004) finding that 2-parameter models, particularly the BP function captured at least as much uncertainty as the three-parameter models for the data examined. As shown (*Figure 5.6*), the greatest variation between models is typically observed at extremes i.e. high and low doses.

Table 5.2 Numerical dose-response models and their uses

Dose-response model	Parameter #	Uses
Exponential	1	*Cryptosporidium* (Coupe *et al.*, 2006), *Giardia* (Zmirou-Navier *et al.*, 2006), Adenovirus (Crabtree *et al.*, 1997)
Beta-Poisson	2	*E. coli* O157 (Strachan *et al.*, 2005), Rotavirus (Mena *et al.*, 2007), *Campylobacter jejuni & Shigella spp.* (Latimer *et al.*, 2001)
Weibull	2	Cancers, microbial toxicity, acute ionising radiation (Nyholm *et al.*, 1992; Little & Muirhead, 2000; Anno *et al.*, 2003)
Lognormal	2	Non-typhi *Salmonella* (WHO/FAO, 2002)
Log-Logistic	2	Listeria monocytogenes (Smith *et al.*, 2008), Herbicides (Seefeldt, 1995)
Weibull-Gamma	3	*Listeria monocytogenes(Chen et al., 2003a), Salmonella spp.* and other foodborne pathogens (Holcomb *et al.*, 1999)
Gompertz	3	*Salmonella* spp. and other foodborne pathogens (Coleman & Marks, 1999)

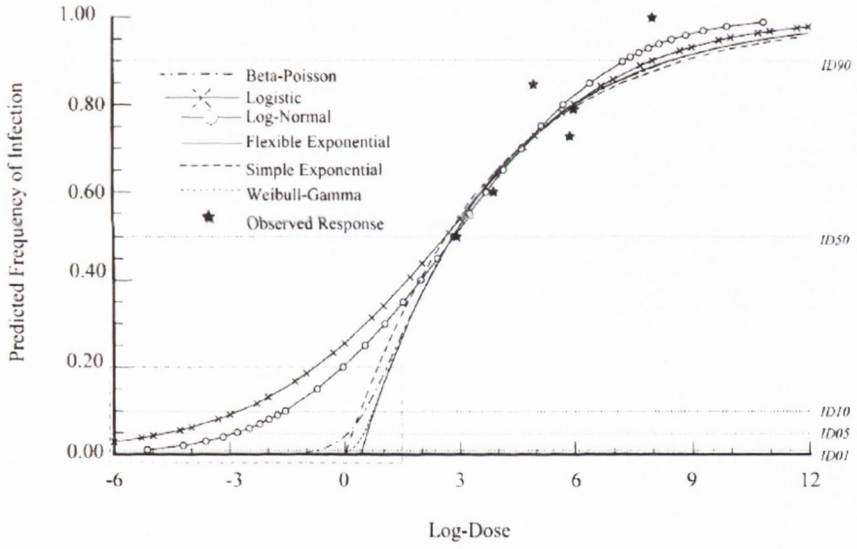


Figure 5.6 Predicted frequency of infection versus log-dose for Campylobacter jejuni (Holcomb et al., 1999)

The two most widely used models for fitting dose-response data, particularly in the case of waterborne pathogens, are the exponential (*Eqns 5.1 & 5.2*) and Beta-Poisson dose-response models (*Eqn 5.3*) (Haas *et al.*, 1999; Moon *et al.*, 2004). These have been used in several studies to describe dose-response relations for a number of different biological agents (Haas, 1983; Rose & Gerba, 1991; Buchanan *et al.*, 2000; Messner *et al.*, 2001; Teunis *et al.*, 2002; Ashbolt, 2004; Howard *et al.*, 2006; Zmirou-Navier *et al.*, 2006; Benke & Hamilton, 2008). Both are based on a number of assumptions, including independence, single hit theory and a Poisson (random) distribution of pathogens (Zmirou-Navier *et al.*, 2006). The assumption of independence refers to the independent action of individual organisms relative to one another (Pinsky, 2000), with single hit theory based on the non-existence

of a "threshold dose" i.e. the survival of at least one organism is enough to initiate human infection (non-zero risk is predicted with any non-zero dose) (WHO, 1998; Fazil, 2005).

The exponential dose-response function

The exponential dose-response function, as described by Moon *et al* (2004) is the simplest dose response model which can be derived from basic biological considerations. It is a one parameter model (r) model and is based on the assumptions outlined above (Haas, 1983; Haas *et al.*, 1999; Teunis *et al.*, 2002).

The assumption is that pathogenic organisms are randomly distributed in a contaminated private well and therefore the probability of consuming a pathogen over a specified period is described by a Poisson (random) distribution. Human hosts have a number of infection barriers which ingested pathogens need to pass in order to initiate infection (independent action). In the case of *i* (number) barriers, the probability that a pathogen successfully traverses all these barriers is p_i. The probability that at least one ingested organism survives and causes infection is the exponential dose response function;

$$P_{inf}[D; p_i] = 1 - e^{-Dp_i} \qquad \textit{Eqn 5.1}$$

Or, as described by Haas *et al* (1999):

$$P_{response} = 1 - exp[-r \times D] \qquad \textit{Eqn 5.2}$$

where *D* is the dose and r is a parameter of the dose-response function and which is interpreted as the probability for one cell to successfully initiate a response (infection/illness, etc). An example of an exponential dose-response function is given below in *Figure 5.7*.

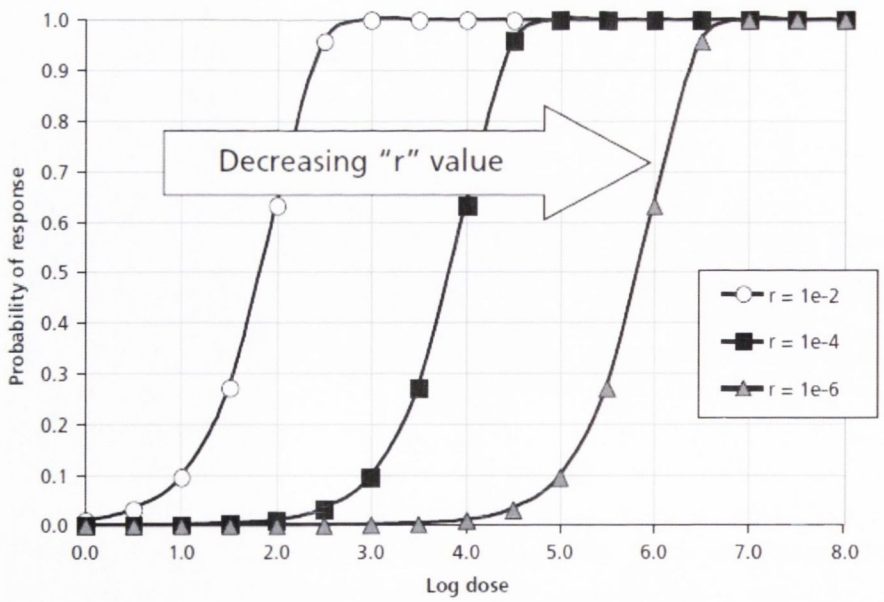


Figure 5.7 Exponential dose-response function, using three different parameter values (Fazil, 2005)

The exponential dose response function has been used to model a number of waterborne pathogens including *Cryptosporidium* (Messner *et al.,* 2001 ; Teunis *et al.,* 2002 ; Coupe *et al.,* 2006), *Giardia* (Rose *et al.,* 1991; Gibson *et al.,* 1998; Zmirou-Navier *et al.,* 2006) and Adenovirus (Crabtree *et al.,* 1997).

The Beta-Poisson dose-response function

The Beta-Poisson (BP) model (also known as the Pareto II model) is a frequently used, single-hit model for predicting dose–response relationships for enteric pathogens, and has been applied to *E. coli O157:H7* (Cassin *et al.,* 1998; Haas *et al.,* 2000; Powell *et al.,* 2000; Strachan *et al.,* 2005) and Rotavirus (Mena, 2007), in addition to *Campylobacter jejuni*, *Shigella* spp. and *Salmonella* spp. (Latimer *et al.,* 2001).

The Beta-Poisson dose-response function expands upon the assumptions made in the exponential dose-response, with the additional assumption that the host-pathogen interaction is beta distributed (Haas, 1983; Haas *et al.,* 1999);

$$P_i[d] = 1 - [1 + d/\beta]^{\alpha} \qquad \textit{Eqn 5.3}$$

where d is the dose, α and β are parameters of the beta distribution, describing the response function shape, therefore, describing the host pathogen interaction. Both α and β are specific to individual pathogens. An example of a Beta-Poisson dose response function is presented in *Figure 5.8*.

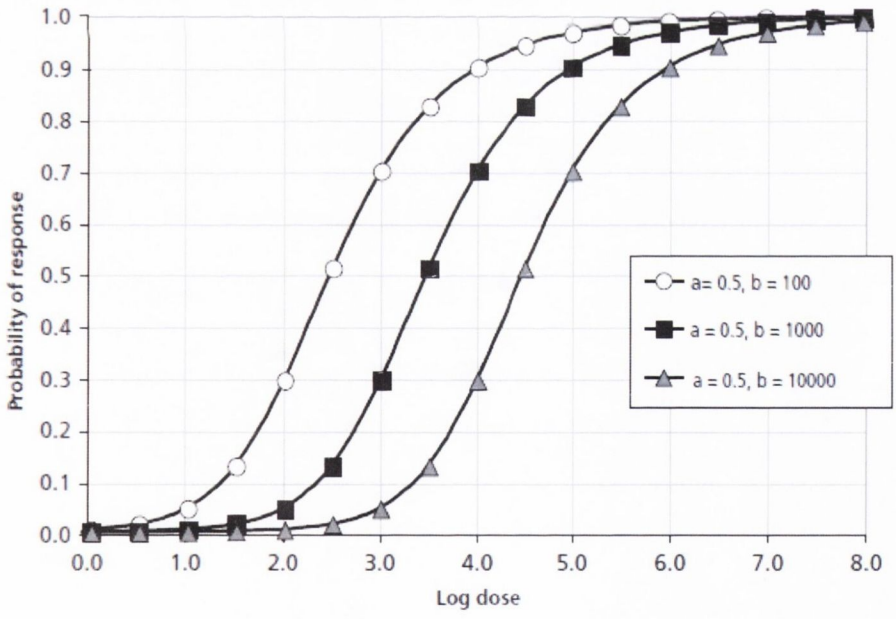


Figure 5.8 Beta-Poisson Dose Response Function; a = α, b = β (Fazil, 2005)

The Beta-Poisson model tends to be more flexible than the exponential model, in its ability to describe data, primarily as a result of the additional parameter i.e. two parameter model. It should

be noted that as α becomes large, the shape of the Beta-Poisson model tends towards the exponential model.

An overview of the various dose response models for use with human data is provided by Moon *et al.* (2004).

5.6 Exposure Assessment

People are exposed to a variety of potentially harmful agents in the air they breathe, the liquids they drink, the food they eat, the surfaces they touch, and the products they use. An important aspect of public health protection is the prevention or reduction of exposures to environmental agents that contribute, either directly or indirectly, to increased rates of infection. In assessing risk to a population, it must be considered that differences exist within a population's exposure level to pathogenic organisms, due to a number of factors. Exposure is defined as *"contact over time and space between a person and one or more biological, chemical or physical agents"* (US NRC, 1991) and therefore exposure assessment is the measurement or estimation of actual human exposure, coupled with appropriate assumptions about associated health effects or safety limits (Ott & Roberts, 1998).

Primarily, exposure assessment is concerned with estimating the likelihood of being exposed to the hazard via the product under consideration (in this case, groundwater for human consumption), and the amount or dose to which that the population or individual is exposed, by combining pathogen occurrence with populations and ingestion rates (Macler & Merkle, 2000; Meliker, 2005). The two primary sources of exposure variability in any waterborne microbial risk assessment are variability in pathogen concentrations and water consumption patterns (Krewski *et al.*, 2004).

In raw and treated drinking water, pathogens are frequently not distributed randomly on either a macro or microscopic scale (El-Shaarawi *et al.*, 1981; Gale *et al.*, 1998; Krewski *et al.*, 2004), accounting for the greatest source of uncertainty in exposure assessment. A number of factors add to the levels of uncertainty associated with pathogen concentration in private groundwater in Ireland, including raw water factors such as variability in rainfall or runoff, seasonality of hazard sources including agricultural cycles and domestic wastewater production, as well as additional variability resulting from treatment processes and distribution. Some of these factors are distributed temporally while others have spatial distributions. To describe the distribution of risks more precisely, such spatial and temporal factors resulting in variability should be characterised (Krewski *et al.*, 2004; Veerman *et al.*, 2005; Coupe *et al.*, 2006; Cummins *et al.*, 2010).

One of the fundamental sources of exposure variability within a population, with regard to a waterborne hazard, is consumption variability, as individuals within any population will consume varying volumes due to various behavioural or physiological factors (Medema *et al.*, 1996; Watanabe *et al.*, 2004; Howard *et al.*, 2007). For example, Craun & Frost (2002) state that with increasing consumption of bottled water, estimation of tap-water consumption has become increasingly problematic.

Additionally, exposure assessment, particularly pathogen concentration and subsequent exposure, relies on the accuracy of microbial isolation and enumeration methods. Krewski *et al* (2004) report

that the main limitations associated with microbial techniques, in the context of exposure assessment, are pathogen specificity (the ability to detect only the pathogen or indicator of interest), sensitivity (the ability to detect very low concentrations of the pathogen or indicator) and viability assessment (the ability to differentiate live and dead cells and subsequently quantify virulence).

When conducting an exposure assessment, it is necessary that the following questions be answered, using a variety of data sources, where available:

- **How many organisms are ingested by the consumer?**

- **How often do they get ingested by the consumer?**

- *sources of contamination: frequency, concentration, and an estimation of the probability and concentration that will be consumed*
- *distribution, growth, inhibition or inactivation from primary contamination,*
- *growth studies, predictive models*
- *animal/zoonotic disease data*
- *composition – pH, nutrient content, presence of antimicrobial substances*
- *population demographics*
- *consumption patterns*

(From Krewski *et al.,* 2004)

Single-hit theory dose-response models applied to pathogens use the product of pathogen concentration and volume of water consumed to calculate the dose (Haas *et al.,* 1999; Howard *et al.,* 2007), which is the overall objective of exposure assessment. Since the two primary sources of exposure variability are variability in concentrations of pathogens or in consumption patterns (Krewski *et al.,* 2004), both pathogen concentration (spatial and temporal) and population consumption should where possible be characterised probabilistically in order to minimise the effects of factor variability and subsequently improve risk estimation.

5.7 Risk Characterisation

Risk characterisationis the concluding task in risk assessment, and typically integrates all of the information generated in the hazard identification, hazard characterisation and exposure assessment steps, to produce a complete picture of the assessed risk (Lammerding & Paoli, 1997; Gibson *et al.,* 1998; Buchanan *et al.,* 2000; Lammerding & Fazil, 2000; Fazil, 2005). The Codex Committee on Food Hygiene (1998) defines the risk characterisation step as a qualitative or quantitative process of estimation, including inherent uncertainties, of the probability of occurrence and subsequent severity of potential adverse health effects in a given population based on the preceding three risk assessment steps. The CAC (2007) and Pouillot *et al* (2009) further refine the risk characterisation as the combination of the results of an exposure assessment with a dose response model (hazard characterisation).

In its most primitive sense, this step could be interpreted to mean simply multiplying the exposure (dose) by the potency to get individual risk, and then multiplying that by the number of people in the population being modelled to give the overall risk estimate. However, this is not the case. Although there are advantages to presenting a simple, single number for a risk i.e. AGI risk to an individual (Haas *et al.,* 1999; Bosgra *et al.,* 2005), appropriate characterisation of risk should be more

comprehensive, detailing the effects of each associated parameter to the overall risk. Additionally, the risk characterisation allows the risk assessor to communicate the relevant level of confidence associated with the results (Williams & Paustenbach, 2002).

It is at this stage of the assessment that the questions or hypothesis, stated in the risk assessment objectives, should be answered and the results presented. For example:

- Expected risk of infection to a "typical" individual (i.e. normal exposure and healthy)
- Expected risk of infection to an immuno-compromised or highly exposed individual
- Upper confidence limit to expected number of illnesses
- Maximum number of illnesses existing in a community at any one time

Masters & Ela (2007) suggest a number of questions that should be addressed in a final characterisation of risk, including:

- What are the statistical uncertainties in estimating the extent of health effects? How are these uncertainties to be computed and presented?
- What are the biological uncertainties? What are their origins? How will they be estimated? What effect do they have on quantitative estimates? How will the uncertainties be described to decision makers?
- Which dose-response assessments and exposure assessments should be used?
- Which population groups should be the primary targets for protection, and which provide the most meaningful expression of the health risks?

These questions have been addressed by the author in carrying out a quantitative risk assessment of waterborne pathogens among private groundwater consumers in Ireland (*Chapter 10*).

In addition to the overall interpretation of the results, the risk characterisation should summarise the effects of assumptions and decisions made in developing the exposure and dose-response assessments. Additionally, insights about the nature of the risk, which are not captured by a simple qualitative or quantitative statement of risk, should be provided. Such insights include, for example, a description of the most important factors contributing to the average risk, the largest contributions to the uncertainty and variability of the risk estimate, and a discussion of gaps in data and knowledge. In depth descriptions of risk assessment components are provided by Haas *et al* (1999) and Miliotis & Buchanan (2009).

5.8 Risk Assessment Models

Risk assessment may be broadly classified as qualitative and quantitative. A qualitative estimate of risk may be conducted by assigning ratings such as negligible, low, medium, or high to the risk factors. However, this doctoral research is concerned with quantitative assessments and therefore mathematical models have been used.

A number of microbial risk assessments have used mathematical modelling to simulate and subsequently predict the transmission of the pathogenic hazard through a chain of processes, from hazard production and ingress until consumption (Gale, 2001; Cassin *et al.*, 1998; Nauta, 2002;

Cummins *et al.,* 2010). The resulting models may then be used to assess the current risk attributable to the process, while also predicting effects of changing input parameters (guidelines, risk mitigation strategies, legislation, treatment, maintenance, etc).

The ability to arrive at a solution to the mathematical model is classified by Fazil (2005) into the following situations:

a) an analytical solution exists and is easily obtained;
b) an analytical solution exists but is highly complex to solve and a solution is not feasible; or
c) an analytical solution is not available.

Where an analytical solution exists and is reasonably easy to obtain, then the analytical solution should be pursued. However, when this is not possible, as in scenarios b) and c) above, then the model needs to be analysed by means of simulation.

Simulation refers to any analytical method which seeks to imitate a real-life system, particularly when other analyses are too mathematically complex or too difficult to reproduce. Without the aid of simulation, a risk assessment model will only reveal a single outcome, generally the most likely or average scenario.

Since each of the steps or factors that contribute to the provision of clean, safe drinking water from groundwater sources i.e. specific contaminant, hydrogeological vulnerability, exposure rate, immune system response of individual, etc., has its own inherent variability and uncertainty, simulation modelling techniques such as Monte Carlo analysis, Bayesian analysis and bootstrapping are increasingly being used to improve the accuracy of quantitative microbial risk assessments (Vose 2000; Vose, 2001). Simulation tools such as these provide a means for considering both the variability of the overall assessment and the impact of individual steps. Although there are a wide range of simulation techniques, Monte Carlo simulation has become the method of choice in developing distributions for derived risk quantities due to its simplicity and the availability of computer software applications (Greenland, 2001; Steenland & Greenland, 2004) and has been applied in the current research (Chapter 10). Simulation models can be classified into three categories, which are now briefly outlined.

5.8.1 Continuous and Discrete Models

Discrete systems can be viewed as systems in which the variables describing the system change instantaneously at a point in time. A continuous system is one in which the changes in the system occur continuously (*Figure 5.9*). However, both are viewed as "dynamic" systems, i.e. systems that change over time (Scheinerman, 2000).

A discrete distribution may take one set of identifiable values, each of which has a calculable probability of occurrence (Vose, 2000; Vose, 2001). Discrete distributions are used to model parameters like the number of bridges a new road scheme may need, or in the case of this work, how many persons may become ill due to a contamination event. Discrete variables such as these can only take specific values i.e. 4.6 people cannot contract an illness. Examples of discrete distributions are: Binomial, Geometric, Hypergeometric and Poisson (Vose, 2000).

The models formulated as part of this study are discrete models due to the discrete nature of data collection and subsequent inputs.

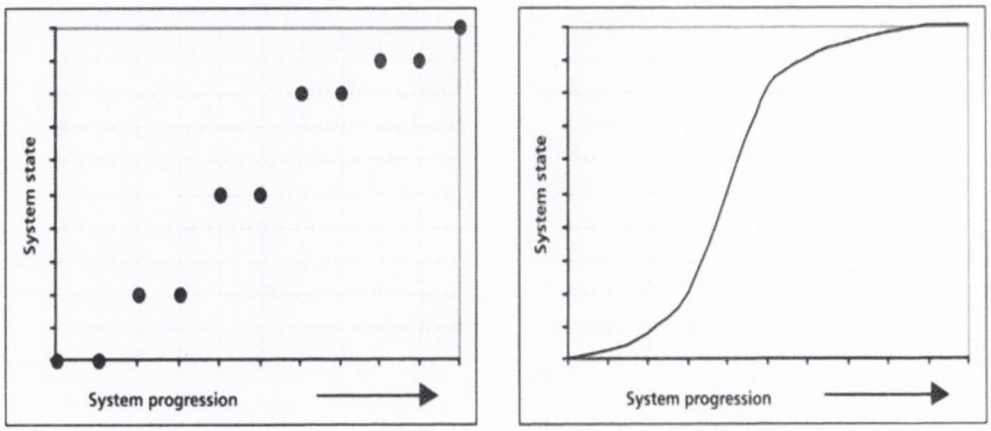


Figure 5.9 Graphical Comparison of Discrete (on left) and Continuous(on right) Models (Fazil, 2005)

5.8.2 Deterministic and Stochastic Models

There are two main approaches to quantitative risk assessment as stated by Lammerding & Fazil (2000), namely 'deterministic' and 'stochastic'. The main difference between these is related to the way in which their inputs are described (Haas *et al.,* 1999).

'Deterministic' risk assessments (also known as point-estimate assessments) use single value inputs such as the average or worst case, depending on which is judged to be more relevant or indeed which are available; for example, to investigate the average number of a pathogen that an individual may be exposed to, the average level of contamination of a particular water source (or indeed the average contamination level at a national level if the former is not known) is combined with the average consumption rate of an average consumer. This approach produces a single (average, or if selected, worst-case, etc) value for the risk estimate, and in this way, ignores variability or uncertainty in model parameters (Lammerding & Fazil, 2000; Baudrit *et al.*, 2007).

The stochastic approach considers all of the data available and uses probability distributions (as opposed to single values) to describe the parameters that contribute to the risk (FSAI, 2002). This produces a distribution of risk that characterises the range of risk that might be experienced by an individual or population and includes variation and uncertainty associated with each model input parameter (Kodell *& Chen,* 2007; Baudrit *et al.*, 2007). Thompson & Graham (1996) state that, *"the probabilistic approach, despite its increased complexity over point-estimate calculations, is becoming the method of choice for quantitative assessments".* Fazil (2005) affirms that probabilistic models tend to be a better representation of natural systems, given the randomness inherent in nature itself. It would be unlikely that a model to describe a natural system could be deterministic.

In order that both model types and their respective outputs may be compared, both deterministic and probabilistic approaches have been used in this study (*Chapter 10*).

5.8.3 Static and Dynamic Models

Static simulation models attempt to characterise the behaviour of a system at a particular and fixed moment in time. Dynamic models, on the other hand, represent a system as it changes over the course of time (Fazil, 2005). An example of a static model might be an equation that estimates the level of contamination in water given the current surrounding and environmental conditions. A model that predicts the changes in the level of contamination on a day-to-day basis would be an example of a dynamic model. The models produced in this thesis are predominantly static models, due to the static nature of the inputs i.e. "one-off" groundwater samples, static well design and construction parameters. However, the inclusion of dynamic parameters in the form of rainfall volume and temporal sampling introduces a level of dynamic prediction to the overall model (*Chapter 10*).

5.9 Uncertainty and Variability

The concept of risk assessment involves predicting the future, implying that the process must contain inherent weaknesses. Typically this cannot be avoided as a large amount is data is necessary in order to accurately predict an event outcome or consequence. Some of these data can be difficult, impractical or in some cases impossible to obtain, such as accurate exposure levels or dose-response curves. The results of the majority of risk assessments are therefore usually expressed as ranges or probability distribution functions, as outlined in Section 5.8, with associated intervals of likelihood i.e. CI 95%, due to assumptions made during the development of the QRA model (Macler & Merkle, 2000). There are two components of this inability to make precise risk predictions, namely, *uncertainty* and *variability*.

5.9.1 Uncertainty

Uncertainty is a property of the risk assessor (Fazil, 2005) and, as defined by Vose (2000) is the assessor's lack of knowledge about the parameters that characterise the physical system that is being modelled. It may be reducible through further measurement or study, or through consultation with experts, but is by definition subjective as it depends upon the assessor. Due to the multidisciplinary nature of the current research, some level of uncertainty is inherent and therefore unavoidable.

5.9.2 Variability

Variability is essentially a property of nature, a result of natural random processes, and represents the diversity in a well-characterised population or parameter i.e. both the microbial pathogen and human responses are highly variable (Lammerding & Fazil, 2000). The variability of a statistic is described by Moore & McCabe (2006) as the spread of a sampling distribution, meaning that although variability may be characterised, it cannot be eliminated from a system.

The ability to recognise and characterise variability and uncertainty is important since they have different ramifications in the results of a risk assessment and for the risk management decisions pursued as a consequence of this assessment (Nauta, 2000). If variability in a parameter is the driving force that leads to a large risk estimate then better control of the process or factor may be warranted to reduce the risk. If a large risk estimate is the result of uncertainty in one or more parameters, then the management decision may be to focus research activities on collecting more

data to better characterise the important uncertain parameters (Nauta, 2000; Vose, 2000) i.e. water consumption data.

The example of water consumption rates may be used to illustrate both uncertainty and variability in a risk assessment input parameter. An illustration of fundamental uncertainty may be drawn from the description of how much water people consume. With no specific data available for Ireland, a minimum and maximum volume may be estimated. Conducting surveys on water consumption habits provides information on how much water people consume; however, some people will always consume more or less than others regardless of how much information is collected. Uncertainty is thus minimised through further research and may provide accurate estimates of daily consumption volume and frequency, however, variability will remain from individual consumer to consumer, and is not reducible. This variability may be characterised to a degree and thus its effects on final risk estimates quantified. The combination of uncertainty and variability is known as total uncertainty (Vose, 2000).

5.10 Monte Carlo Simulation

In order that uncertainty of risk estimates associated with input parameters may be reflected, Monte Carlo simulation techniques have been extensively used in various fields including atmospheric pollution (Beekman & Derognat, 2005), contaminant transport (Chang *et al.*, 2011), life cycle assessment (McCleese & LaPuma, 2002) and human health risk assessment (Cox & Popken, 2004; Kentel & Aral, 2005; Mara *et al.*, 2007; Buteau & Valcke, 2010)

Monte Carlo simulation uses a computational algorithm which relies on repeated sampling using random number generator seeds (Johnson *et al.*, 1995), to compute its results, with values for uncertain variables being randomly generated over and over (iterations) to simulate a model. A general model may be tested many thousands of times to give likely long run means, medians, upper and lower confidence intervals, etc (Vose, 2000; Vose, 2001; Rubenstein & Kroese, 2007). For each uncertain variable (e.g. water consumption), the possible values are defined with a known or estimated probability distribution or probability density function (Vose, 2000). The type of distribution selected is based on the conditions surrounding that variable e.g. normal, triangular, lognormal. A simulation calculates multiple scenarios of a model by repeatedly sampling values from the probability distributions for the uncertain variables and using those values for the cell (Lammerding & Fazil, 2000), with any required level of precision achieved by increasing the number of modelled scenarios. For example, where a lognormally distributed daily water consumption rate with a mean value of 1 L day^{-1} is a model input, the Monte Carlo algorithm will "randomly" pick a point on the distribution for each recalculation/iteration. However, the algorithm will pick points on the distribution close to 1 L day^{-1} more often than extreme values. In this way, representative figures are used, while still accounting for extremes within the population. Monte Carlo simulation is an example of probabilistic modelling due to the probabilistic nature of the input parameters and subsequent outputs.

Additionally, the behaviour of the developed model may be investigated using Monte Carlo analysis, which is useful when investigating the effects of uncertainty or variability on the model or exploring which input parameters have the greatest effect on final risk estimates i.e. sensitivity analysis (2^{nd} order modelling) (Nauta, 2000; Saltelli *et al.*, 2004).

In the context of this study, Monte Carlo simulation has been used in order that the effects of uncertainty and variability relating to a number of input parameters may be included in the overall risk model. Those input parameters with inherent uncertainty and/or variability include: individual daily water consumption, the use of water treatment processes, the presence of specific waterborne pathogens, and the level of contamination present. The use of simulation modelling in general and Monte Carlo simulation in particular, permits the inclusion of these inherent uncertainties and therefore, their effects on the overall model and consequent outcomes may be studied, resulting in higher levels of assessment accuracy.

5.11 Risk management

As previously outlined (*Section 5.3)*, risk management is considered to be the second step of the integrated risk analysis process. At the broadest level, risk management includes a range of management and policy-making activities: agenda setting, risk reduction decision-making, programme implementation, and outcome evaluation (American Chemical Society, 1998). Similarly, Codex (CAC, 1999) has defined risk management as the process of weighing policy alternatives in the light of the results of risk assessment and, if required, selecting and implementing appropriate control options, including regulatory measures (Guillot & Loret, 2009).

Risk management, as a component of risk analysis, is characterised by the following questions:

- Is the risk acceptable and can it be reduced?
- What decisions arise from risk assessment?
- What control measures are needed to minimise the risk?

Risk management involves amalgamating the risk assessment outputs into some manageable form (Apostolakis, 2004), with the subsequent goal not being risk elimination, rather the dissemination, selection and implementation of scientifically sound, cost-effective, integrated actions that reduce risk to an acceptable level (Notermans & Teunis, 1996; Aven, 2008). The tasks involved in risk management have been described as: the determination of what hazards present more danger than society is willing to accept; consideration of what control options are available, and deciding on appropriate actions to reduce (or eliminate) unacceptable risks (Fazil, 2005).

Risk management methods may include development of consumer guidelines, formal legislative tools, microbial standards or mitigation strategies (Rose & Gerba, 1991).

5.12 Risk Communication

As outlined in *Section 5.3*, the third and final step in an integrated risk analysis is the communication of outcomes to relevant stakeholders. In the contest of this research, the following questions are used to characterise risk communication:

- Is the risk increased or decreased with an increased level of awareness?
- If the risk is decreased, how should this be communicated to those potentially affected?

Risk communication involves the exchange of information between risk assessors, risk managers and stakeholders in the risk issue (FAO/WHO, 1998). Since the primary purpose of risk assessment is to inform the risk management decision-making process, it is important that both assessors and managers understand the specific issue of concern. Haas *et al* (1999) define risk communication as "the communication of risks to managers, stakeholders, public officials and the public; including public perception and the ability to exchange scientific information".

As mentioned in Chapter 1 (Section 1.2), a primary objective of this research has been the development of a guidance framework for the owners and users of new and existing private groundwater sources in Ireland. This framework (Chapter 11) includes recommendations regarding the appropriate location, design, construction and maintenance of groundwater sources, so as to reduce adverse impacts in the future. As such, it is considered that this guidance represents the risk management and communication component of the risk analysis.

The stakeholders in the risk characterisation outcomes include the general public and/or specific vulnerable subpopulations, in addition to other interested parties including public health bodies, medical practitioners, governmental organisations and groundwater professionals.

5.13 Case Studies: Waterborne Risk Assessment

A number of previously used risk assessment techniques are now briefly outlined.

Haas et al., 1993

This methodology was used to estimate the burden of disease attributable to virus in drinking waters in the US. Risk quantification may be based on point-estimate (deterministic) or estimates for entire population based upon frequency distribution of exposure, leading to a risk estimate in the form of a frequency distribution (probabilistic). A probabilistic approach is preferable where risk estimation is for an entire population i.e. the current research. Deterministic (point-estimates) may be converted to frequency distributions upon collection of representative data.

Haas *et al* (1993) utilised the Beta-Poisson dose-response model as follows:

$$P_I = 1 - [1 + N/N_{50}(2^{1/\alpha} - 1)]^{-\alpha} \qquad \text{Eqn. 5.4}$$

where:

P_I= Probability of individual daily infection

N = Number of consumed organisms

N_{50}= Number of organisms eliciting 50% infection

α = Parameter associated with slope of dose-response curve

Monte Carlo simulation (10,000 iterations) was used to estimate the uncertainty distribution of risk, and they found that yearly risks exceed the risk recommendation of 1×10^{-4} at every exposure level. Mena & Gerba (2009) later used a similar technique for the quantification of health burden attributable to waterborne adenovirus. Haas *et al.* (1993) calculated both a point risk estimate of viral infection and an associated uncertainty estimate, accounting for variability and uncertainty,

using Monte Carlo simulation. A point risk estimate of the daily probability of disease was calculated as $7.17x10^{-4}$, equivalent to an annual risk of 0.23. This was favourably compared to a previous study (Payment *et al.*, 1991), which reported similar risk estimates of $8.2x10^{-4}$ and 0.24 respectively. Uncertainty analysis of the daily risk of disease resulted in mean and median daily risk estimates of $4.43x10^{-4}$ and $2.76x10^{-4}$ respectively, with a 95% CI of $3.17x10^{-5} - 1.88x10^{-3}$.

Gerba et al., 1996a

The Gerba *et al.* (1996a) model was used to estimate the public health impacts from exposure to human rotavirus in drinking and recreational waters in the US. The model uses the Beta-Poisson dose-response model, which as previously mentioned, has proven effective in the modelling of waterborne enteric viruses (Haas, 1983; Regli *et al.*, 1991; Haas *et al.*, 1993; Mena *et al.*, 2003). In this study, the Beta-Poisson model was characterised as follows:

$$P_i = 1 - (1+N/\beta)^{-\alpha} \qquad \textit{Eqn. 5.5}$$

where:

P_i= Probability of infection

N = Number of ingested viruses = mV = concentration of viruses x volume consumed

Risks were determined based upon the parameters [β(0.42)] and [α(0.26)], which define the dose-response models for rotavirus (Haas *et al.*, 1993), assuming a consumption rate of 2 L day^{-1} and a rotavirus concentration of 0.004/L in drinking water.

$$P_i = 1 - [1 + (2 \times 0.004)/0.42]^{-0.26} \text{ (risk/affected person/day)}$$

Overall, the analysis indicated that significant annual risks of infection ($5x10^{-1} - 2.45x10^{-3}$) could result from drinking and recreational waters in the US where rotavirus had been detected. Gerba *et al.* (1996a) also state that major limitations were encountered in assessing these risks, most notably in regard to rotavirus occurrence (i.e. lack of monitoring; underreporting of rotaviral illness, etc.) and human exposure.

Teunis et al., 1997

The Teunis *et al.* (1997) model was used to assess the significance of protozoan parasites, namely *Cryptosporidium* and *Giardia*, to public health from drinking water sources. A surface water body (the river Meuse) in the Netherlands was the focus of investigation. An exponential dose-response model was used, calculating for probability of infection per consumer for a specified period, characterised as follows:

$$Dose = C \times 1/R \times I \times 10^{-DR} \times V \qquad \textit{Eqn. 5.6}$$

And

$$P_i = 1 - e^{-r \times Dose} \qquad \textit{Eqn. 5.7}$$

where:

C = Contaminant concentration

R = Recovery rate

I = Pathogen infectivity

DR = Removal or inactivation efficiency of the treatment process, expressed as its decimal reduction factor (DR = 0 when concentrations in drinking water are used)

V= Daily consumption rate

P_i = Probability of infection (a 100% infectivity rate is assumed in this case, i.e. worst case scenario)

r = Dose-response

This study was probabilistic in nature and used the Monte Carlo approach (5,000 iterations) to quantify data uncertainty through random sampling. Simulation results found that uncertainty in the estimated removal efficiency of treatment processes dominated over uncertainties in other factors, with other sources of uncertainty being daily water consumption (*Figure 5.10*), oocyst/cyst viability and recovery, and treatment efficiency. Similar studies using these methods have been carried out by Gibson *et al.* (1998) and Cummins *et al.* (2010).

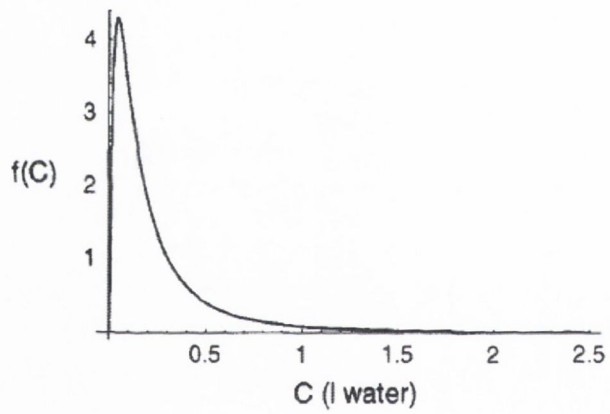


Figure 5.10 Daily water consumption (Lognormal distribution) used by Teunis et al. (1997), based on the Haring et al. (1979) population survey of the Netherlands

Teunis *et al. (1997).* found the median probability of infection from *Cryptosporidium* spp. (*Figure 5.11*) and *Giardia* (*Figure 5.11*) to be just over $1x10^{-4}$ and $5x10^{-4}$ respectively, among people obtaining their drinking water supply from the River Meuse. Assuming mutual independence regarding all input factors, an overall lifetime risk of just over 1% and just under 5% for *Cryptosporidium* and *Giardia* infection respectively were calculated, assuming a life expectancy of 70 years.

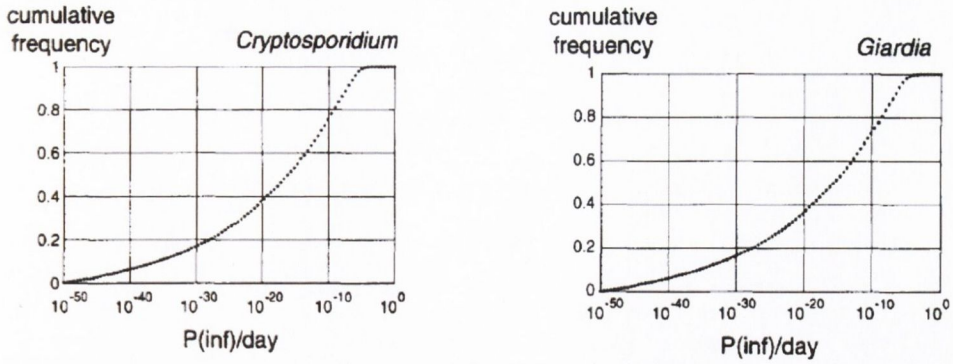


Figure 5.11 Cumulative distribution of the daily risk of infection with Cryptosporidium spp. oocysts and Giardia cysts (Teunis et al., 1997)

Macler & Merkle, 2000

The Macler & Merkle model was originally used to estimate the likely ranges of disease attributable to waterborne rotavirus from groundwater-based community water systems and all public supplies using groundwater in the US. It is a simple deterministic model, as follows:

(Number of persons) x (contamination rate) x (contribution) x (infection rate) x (recovery rate) x (dose-response) x (secondary infection rate) = Number of waterborne infections.

Data were wholly deterministic in nature including contamination and infection rates (i.e. dose-response), from a variety of published sources and resulted in an estimate of 750,000 – 5,000,000 illnesses per year in the USA, attributable to rotavirus infection from consumption of contaminated private groundwater sources. Furthermore, between 1,400 – 9,400 deaths per year, attributable to the same sources, were also predicted as a result of the study. Applying a similar approach as that used by Macler & Merkle (2000), Vaury (2003) estimated that there could be up to 100,000 illnesses per annum in Ireland due to waterborne rotavirus directly attributable to contaminated private groundwater sources in Ireland. This study was however, based upon limited data.

A similar deterministic risk assessment carried out by Howard & Pedley (2004), in which the risk to human health from public water supplies in Uganda for a number of pathogens including *E. coli* O157:H7, *Cryptosporidium* and rotavirus, concluded that had a sufficient volume of data been available, a probabilistic approach would have, and should have, been used.

Food Safety Authority of Ireland (FSAI), 2002

This study sought to assess the risks of fluoride intake of infants <4 months of age in Ireland, as a result of consumption of infant formula reconstituted with fluoridated water. Two probabilistic models were developed to estimate both acute (i.e. range of possible daily fluoride exposures on any one day) and chronic (i.e. average daily exposure to fluoride of babies fed exclusively on formula over the first 4 months of life) exposure, with Monte Carlo simulation being used for both models. All parameter inputs were developed as probabilistic density functions including fluoride concentrations in raw water (*Figure 5.12*), fluoride concentrations in infant formula powder and infant formula consumption per day.

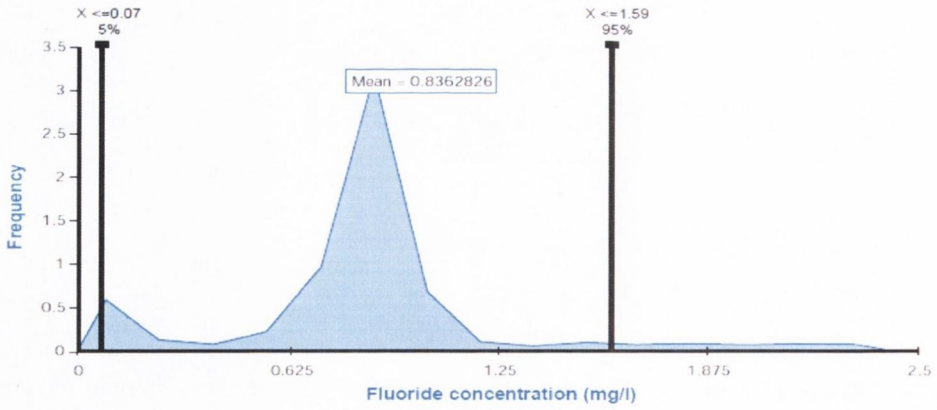


Figure 5.12 Distribution of possible fluoride concentrations in Irish tap water on any single day (FSAI, 2002; Anderson et al, 2004)

As shown (*Figure 5.13*), the mean acute exposure of infants to fluoride on any single day in areas served by 387 fluoridated water supplies was estimated to be between 0.11 and 0.14 mg kg^{-1} body weight depending on age group (95th percentiles 0.2 and 0.26 mg kg^{-1} b.w., respectively). These predicted intakes were well below the intake of fluoride associated with acute toxic effects, which is considered to be 5 mg kg^{-1} body weight. The mean chronic exposure of infants to fluoride was estimated to be between 0.106 and 0.170 mg kg^{-1} b.w./day depending on body weight (95th percentiles 0.108 and 0.172 mg kg^{-1} b.w./day, respectively). Overall, risk characterisation suggested that no definitive link existed between fluoride intake in the first four months of life and moderate fluorosis of permanent dentitition.

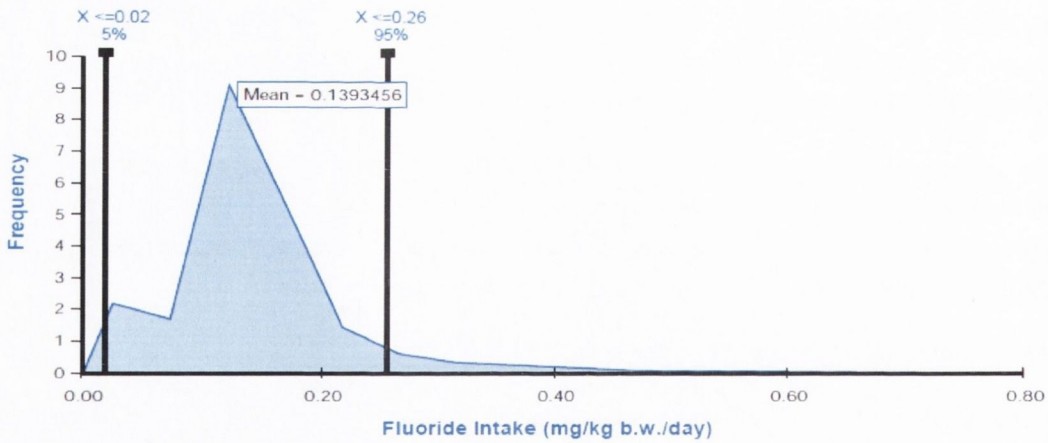


Figure 5.13 Distribution of possible fluoride intake on any single day for babies at 2 weeks of age expressed on a body weight basis resulting from the consumption of infant formula reconstituted with fluoridated tap water (FSAI, 2002; Anderson et al, 2004)

Chapter 6: Data Collection Methodologies

6.1 Introduction

As outlined in Chapter 1 *(Section 1.2)*, in order to achieve the overall research objectives, a number of data collection methodologies have been employed:

- To investigate the link between the contamination risk, and hence health risk, and the hydrogeological factors associated with private groundwater supplies, it was necessary to collect field data on source susceptibility (including source design, source construction and source location) and on water quality associated with private well supplies. The fieldwork programme included selection of four main study areas, surveys of private wells, groundwater sampling and the introduction of a continuous groundwater quality monitoring programme.

- To determine the awareness amongst private well owners of the potential contamination risk factors and associated health impacts from contaminated groundwater supplies, it was necessary to develop and complete an awareness questionnaire with a representative sample of groundwater users in Ireland. Moreover, in order to investigate if a relationship existed between groundwater source and/or contamination and awareness and groundwater quality, it was necessary to complete these surveys in concurrence with an integrated groundwater fieldwork programme.

- To undertake a quantified assessment of the magnitude of risk to health from contamination of private well supplies in Ireland, data compiled from all data collection phases, including site susceptibility assessment, seasonal and "one-off" groundwater sampling and awareness questionnaire completion, have been integrated and used.

The following chapter will outline and discuss all materials and methodologies used for data collection throughout this research study, including:

i. Site Selection and Description
ii. Site Susceptibility Assessment
iii. Sample Collection and Fieldwork Protocols
iv. Laboratory Protocols
v. Acquisition and analysis of meteorological data
vi. Awareness Questionnaire Development and Completion

Quality control (QC) procedures were carried out on two sample batches, with the results discussed. Additionally, in order to quantify meteorological effects (temperature; precipitation) on private groundwater sources and associated contamination rates, meteorological data are presented and discussed.

6.2 Site Selection and Description

Several criteria were identified at the outset of the project that needed to be satisfied with respect to study area suitability and selection. It was determined that three areas of high or extreme vulnerability were to be chosen for source assessment and sampling. Evidence suggests that pathogens must reach the groundwater source within a specified period of time in order for the pathogen to remain viable. Consequently areas of *High* or *Extreme* vulnerability are considered to offer lower resistance to pathogen movement to groundwater, as outlined in Chapter 2 (*Section 2.5*).

In areas of *Moderate* and particularly *Low* vulnerability, decreased subsoil permeability ($<1x10^{-4}ms^{-1}$ for moderate permeability and $<1x10^{-9}ms^{-1}$ for low permeability subsoils) and increasing subsoil thickness will sufficiently impede the rate of pathogen transport, resulting in greatly decreased potential groundwater contamination (O'Suilleabhain, 2000; Swartz *et al.,* 2003). Overall, the subsoil permeability is considered the main groundwater vulnerability driver (Fitzsimmons & Misstear, 2006). Therefore, one area underlain by low permeability subsoil and consequently categorised as being low vulnerability was chosen as a "control" area in order to observe the expected difference in contamination susceptibility between areas of high and low vulnerability.

Study area selection was undertaken primarily as a desk study, by using the aforementioned set of suitability criteria and their ranked importance to decide on the most suitable areas for study.

6.2.1 Site Selection Criteria

The criteria believed to be of most importance in the determination of suitable areas for fieldwork were divided into seven categories in order of perceived importance as follows:

1. *Vulnerability category of aquifer*
2. *Groundwater Protection Scheme Status*
3. *Existence/availability of previous sampling and monitoring data*
4. *Teagasc subsoil data availability*
5. *Access to sampling point and proximity to testing locations*
6. *Well density*
7. *Availability of land use data*

These categories and their associated importance are outlined as follows:

Groundwater Vulnerability Category (VC)

Four study areas were to be chosen for data collection, including three *High/Extreme* vulnerability areas and one *Low* vulnerability area. Therefore, *Moderate* vulnerability areas were excluded, as were those counties which had yet to submit completed Groundwater Protection Schemes (GWPS). Subsequently, a list of 63 potentially suitable study areas drawn from 13 counties were chosen, from which the study areas would be selected. This list (Appendix D1) was developed using a number of data sources including Environmental Protection Agency, Geological Survey Ireland, Local Authorities, and researchers from a number of institutions including National University of Ireland Galway, University College Dublin, Trinity College Dublin and Institute of Technology Sligo.

Groundwater Protection Scheme Status (GPSS)

The status of a county groundwater protection scheme at the time of commencement of fieldwork was of primary importance in study area selection, as completion of the scheme indicates the availability of the following data maps:

- Map 1 – Bedrock Geology Map
- Map 2 – Subsoils Map
- Map 3 – Depth to bedrock Map
- Map 4 – Hydrogeology Map
- Map 5 – Aquifer Map
- Map 6 – Groundwater Vulnerability Map
- Map 7 – Resource Protection Zones

The availability of these maps and the data contained within were considered essential and therefore areas with incomplete Groundwater Protection Schemes were not considered as part of the site selection process. The current status of county groundwater protection schemes may be found in Appendix D2.

The availability of completed GWPS also allowed for classification of aquifers for further analysis.

Previous Sampling/Monitoring Data (PS)

It was considered preferable that some previous research or monitoring had been carried out in the study area previously so that these datasets could be incorporated into the study. Therefore, this factor was included in the site selection procedure.

Relevant data may include:

- Health Service Executive (HSE) Monthly HPSC Surveillance Reports
- EPA Rural Water Quality Monitoring Programme
- EPA *Cryptosporidium* Risk Assessments
- Well improvement grant scheme data
- Local Authority Monitoring
- Water Framework Directive River Basin District Monitoring
- Teagasc Subsoil Classification Project
- Reports and/or theses from universities and other third-level institutions

Existing data, water quality data, would also provide background physico-chemical, chemical and microbial data against which water quality analysis could be compared.

Teagasc Data (TD)

A subsoils classification project undertaken by Teagasc, was used to obtain highly relevant data regarding local soil types, depths and pH. Teagasc outcrop and gravels data-layers could be used to provide information regarding rock outcrops, particularly in high and extreme vulnerability areas.

These data may be used for mapping high permeability/high vulnerability areas and have been integrated into GIS layers.

Well Density (WD)

In carrying out the groundwater sampling regime and completing an adequate number of susceptibility assessments and awareness surveys for meaningful and representative analysis, areas with a high density of production wells were preferable as increased sample numbers could be collected and more assessments completed, making both these components of the research as time and cost effective as possible. For the purpose of site selection, these data were obtained from GSI digital mapping and Water Framework Directive Further Classification studies and are conveyed in wells per square kilometre (wells km^{-2}).

Laboratory Proximity (LP)

Potentially suitable sites were ideally located within 2 hours drive from Trinity College Dublin environmental engineering laboratory to allow for samples to be returned and analysed on the day of collection, as overnight storage is not in keeping with approved analytical standard methods (APHA/AWWA/WEF, 2005).

Land Use (LU)

Several studies, undertaken in the United States, have confirmed correlations between land use and resultant aquifer contamination. For example, Abbaszadegan (2003) established a direct correlation between positive contamination results and distance to pollution hazards, showing that 80.9% of groundwater sites that tested positive for viral contamination were <167m (400 feet) from a sewage source. Scandura & Sobsey (1997) and Ahmed *et al.* (2005) both reported proximity to domestic septic tank effluent percolation areas as being a cause of groundwater contamination, particularly in areas of *High* groundwater vulnerability.

The pathogens of interest in this study, particularly *E. coli* 0157 and *Cryptosporidium parvum*, are both directly related to the agricultural industry, and in particular to animal production cycles. Consequently, the main hazards associated with pathogenic contamination of groundwater sources in Ireland are septic tank systems and agricultural processes (Daly, 1987; Daly, 2003). Therefore, the presence of one or both of these hazards was included as a criterion in site selection. The locality of other potential sources of groundwater contamination such as landfill operations, were also noted and investigated where encountered. Corine Land Cover (CLC) maps and local authority planning files were used to ascertain landuse category.

6.2.2 Study Area Selection Matrix

Using the eight selection criteria set out above, a study area selection matrix was developed. The matrix was used to "score" potential study areas in terms of overall suitability for inclusion in the fieldwork programme. The groundwater vulnerability category and groundwater protection scheme (GWPS) status were considered the two primary selection parameters, with incorrect VC and incomplete groundwater protection schemes precluding selection. VC was therefore scored dichotomously (i.e. in the case of high vulnerability study area selection, a study area dominated by *Low* groundwater vulnerability was given a score of 0, whereas an area dominated by high

vulnerability was given a score of 1; conversely, in the case of *Low* vulnerability study area selection, areas characterised by *Low* vulnerability were given a score of 1, while areas of *High/Extreme* vulnerability were given a score of 0), with potential study areas with a 0 score excluded from further consideration. A similar approach was taken with regard to areas with incomplete groundwater protection schemes. Groundwater protection schemes, where complete, were also scored for currency and availability, as both these issues were considered important. This was done as follows:

GWPS recently completed (<10 years) and digitally available – 3
GWPS recently completed *or* digitally available – 2
GWPS not recently complete and not digitally available – 1
GWPS incomplete – 0

A comprehensive review of existing groundwater quality data was carried out, from a number of potential sources, as previously outlined. The data were reviewed in terms of applicability and currency, with microbial data considered particularly useful. Existing data for each potential study area was then attributed a dichotomous score, based on relevance of existing data, volume of existing data and currency of existing data. Where a significant volume of useful data was available, a score of 1 was given, while a score of 0 was attributed if this was not adjudged to be the case.

In the case of Teagasc outcrop and gravels data, a simple presence/absence score was used, with 1 denoting the presence of Teagasc data and 0 denoting absence of Teagasc data. In terms of the density of private groundwater sources, a target well density of 2 sources km^{-2} was used to assess appropriate well density, using GSI digital maps and Water Framework Directive Further Classification studies. Where an approximate well density of >2 km^{-2} was indicated by the aforementioned data sources, a score of 1 was given; where this figure was <2 km^{-2}, a score of 0 was given.

Areas falling within an approximate 2-hour travel radius of the Environmental Engineering laboratory, Trinity College Dublin were considered preferable and subsequently allocated a selection score of 1, those situated outside this radius were attributed a score of 0. Finally, where adequate agricultural and septic tank data were available, a selection score of 1 was given, and where absent, a score of 0 was used. The final study area selection matrix for potential high vulnerability study areas in Co Offaly is presented in *Table 6.1.*

Table 6.1 Final Site Selection Matrix, Co Offaly

	VC	GPSS	PS	TD	WD	LP	LU	Total
Belmont	1	2	1	1	1	1	1	**8**
Bloomhill	0	N/A	N/A	N/A	N/A	N/A	N/A	
Clonfanlough	0	N/A	N/A	N/A	N/A	N/A	N/A	
Corndarragh	1	2	0	1	0	1	1	**6**
Durrow	1	2	1	1	0	1	1	**7**
Killeigh	1	2	1	1	0	1	1	**7**
Meelaghans	1	2	0	1	1	1	1	**7**
Mountlucas	0	N/A	N/A	N/A	N/A	N/A	N/A	

As shown (*Table 6.1*), the most suitable study area for selection in Co Offaly was adjudged to be Belmont. All total site selection scores were subsequently compared and the four areas with the highest scores were selected as suitable study areas.

6.2.3 Study Areas

After careful consideration using the study area selection matrix set out above, in addition to carrying out site visits and subsequent visual inspections of a shortlist of potentially suitable study areas, four suitable study areas were identified:

- Belmont, Co Offaly
- Crossbridge, Co Wicklow
- Paulstown, Co Kilkenny
- Kilclone, Co Meath

It was considered that due to time, labour and financial constraints, four representative study areas, including one control area, comprising 50-60 private wells/area, were manageable for one researcher. During the study, a suitable additional study area was located for which there was a sizeable volume of relevant data available, namely Birdhill, Co Tipperary. It was decided that a limited amount of fieldwork would also be carried out in this area as part of the overall research.

These study areas (*Figure 6.1*) are now described; with maps outlining site locations, site extent and source location, groundwater vulnerability, aquifer importance, subsoil type and bedrock geology also presented. *Table 6.2* summarises the main study area characteristics.

Table 6.2 Study Area Characteristics

Study Area	Area (km^2)	Vulnerability	Dominant Bedrock	Dominant Subsoil	Aquifer Importance
SA1 (Offaly)	15.5	*High*	Sandstone Limestone	Limestone S&Gs; Limestone Tills	Ll/Lm
SA2 (Wicklow)	20.3	*High/Extreme*	Granite	Granite Tills	Pl/Ll
SA3 (Kilkenny)	30.8	*High/Extreme*	Sandstone Shale Limestone	Sandstone/Shale Tills; Limestone Tills	Pu/Lm/Pl
SA4 (Meath)	15.8	*Low*	Limestone Sandstone	Limestone Tills	Ll/Pl
SA5 (Tipperary)	11	*Extreme(Sur)*	Sandstone Limestone	Sandstone Tills	Ll/Pl

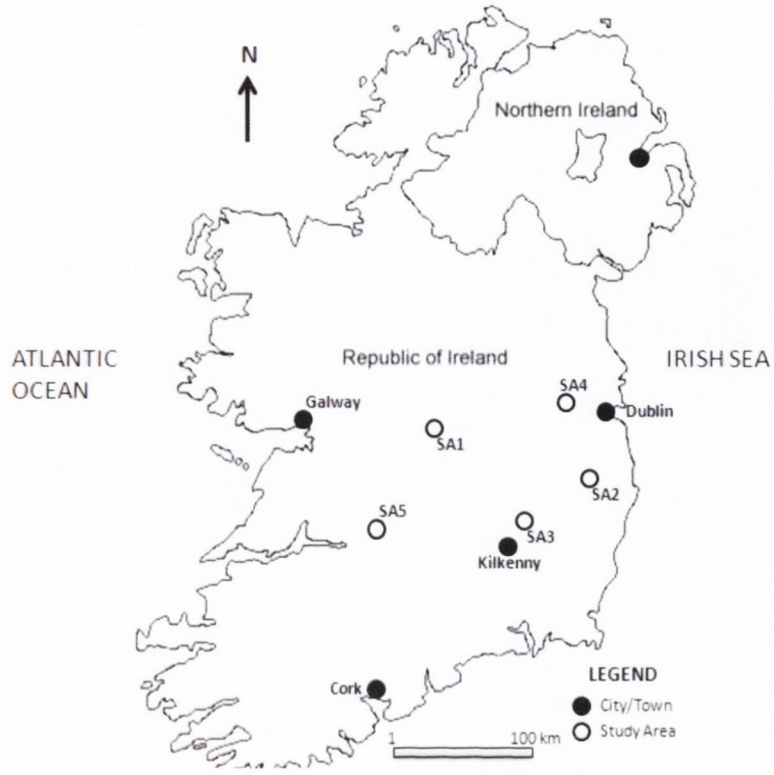


Figure 6.1 Study Area Locations

Study Area 1: Belmont, Co Offaly

Study Area 1 (SA1) (*Figures 6.2 & 6.3*) is located in the northwest of Co Offaly, approximately 136 km from Dublin City and covers an area of approximately 19.8 km^2, with a central point at Belmont village, situated approximately 5.5 km from the Co Galway border. The area is situated in the Shannon River Basin District (RBD).

The superficial geology over SA1 is varied, with Carboniferous Limestone tills covering approximately 30% of the area, cutover peats approximately 10-15% and the remainder being composed of glaciofluvial sands and gravels, presenting as a large esker. There are also some undifferentiated alluviums in the area, following the Brosna River, southwest to northeast across the study area. The bedrock geology is also varied and chiefly composed of Devonian Kiltorcan-type Sandstones (DKS) and Dinantian early Sandstones, Shales and Limestones (DESSL), surrounded by areas of Dinantian Pure Unbedded Limestones (DPUL) and Dinantian Lower Impure Limstones (DLIL). Approximately 80% of the area is composed of DKS and DESSL (*Figure 6.4*).

Over 90% of SA1 is designated as being of *High* vulnerability, with the remainder composed of small pockets of *Extreme* vulnerability (*Figure 6.5*). The aquifer in the area is chiefly characterised as being locally important, with bedrock being moderately productive only in local zones (Ll), however, the area underlain by glaciofluvial sands and gravels is characterised as being locally important, with bedrock being moderately productive (Lm). A proportion of the Lm aquifer is additionally described as a locally important gravel aquifer (Lg). As the study area is entirely un-serviced by public water supply, all residences in the vicinity were reliant upon private groundwater supplies, with both HD SPS and Bored SPS being used. The primary landuse in the area was agricultural with mixed grazing

of beef cattle and sheep being the dominant agricultural activity. The Grand Canal approximately bisects the study area.

An additional small area, namely Ballynahowan, Co Offaly, situated approximately 10km north of Belmont village was studied in order to maximise the number of sample sources (*Figure 6.3*). This area is constrained by the Co Westmeath border to the north and comprises a 2 km stretch of tertiary road with un-serviced domestic households on both sides, comprising a total area of 1.1 km^2. Overall, approximately 90% of this area is overlain by Carboniferous limestone tills, with the remaining 20% being overlain by a mixture of cut peat and glaciofluvial sands and gravels. The area is wholly underlain by Dinantian pure unbedded limestones. Over 80% of the area is categorised as being of high vulnerability, with an additional 10% of extreme vulnerability. Approximately 8-10% is of moderate vulnerability; however, susceptibility assessment and sample collection were not undertaken in these areas. The aquifer in this area is entirely categorised as being Ll.

The total area of SA1 is 20.9 km^2, comprising a total of 50 sample sources, equating to a sample source density of 2.4 km^{-2} within this study area.

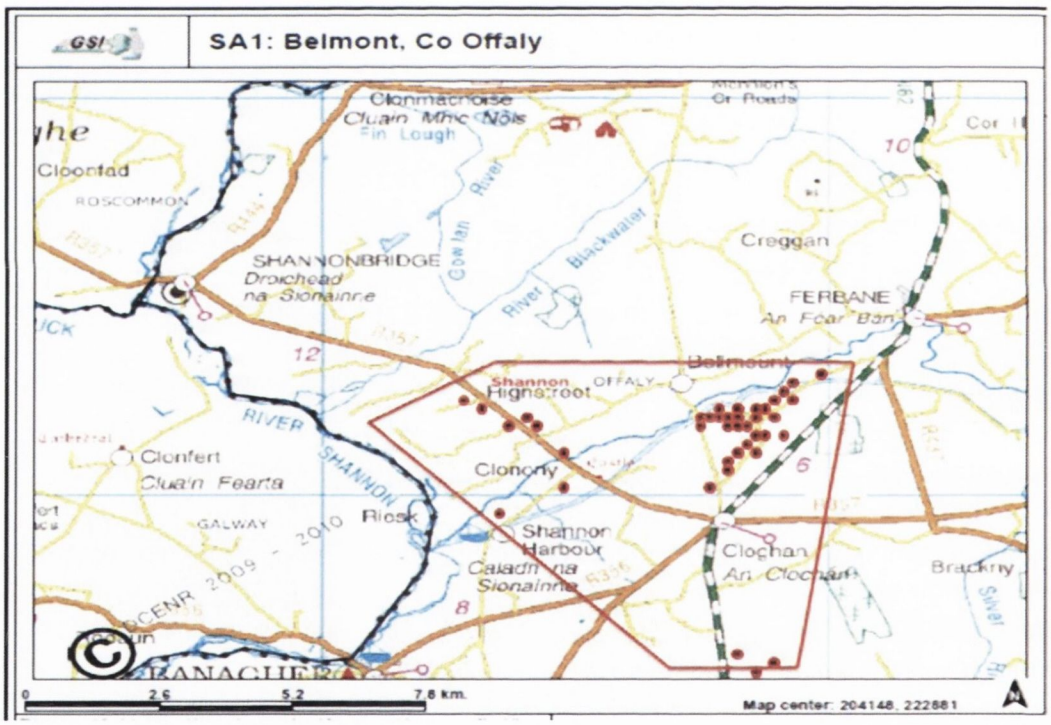


Figure 6.2 Study Area 1: Location Map (Belmont)

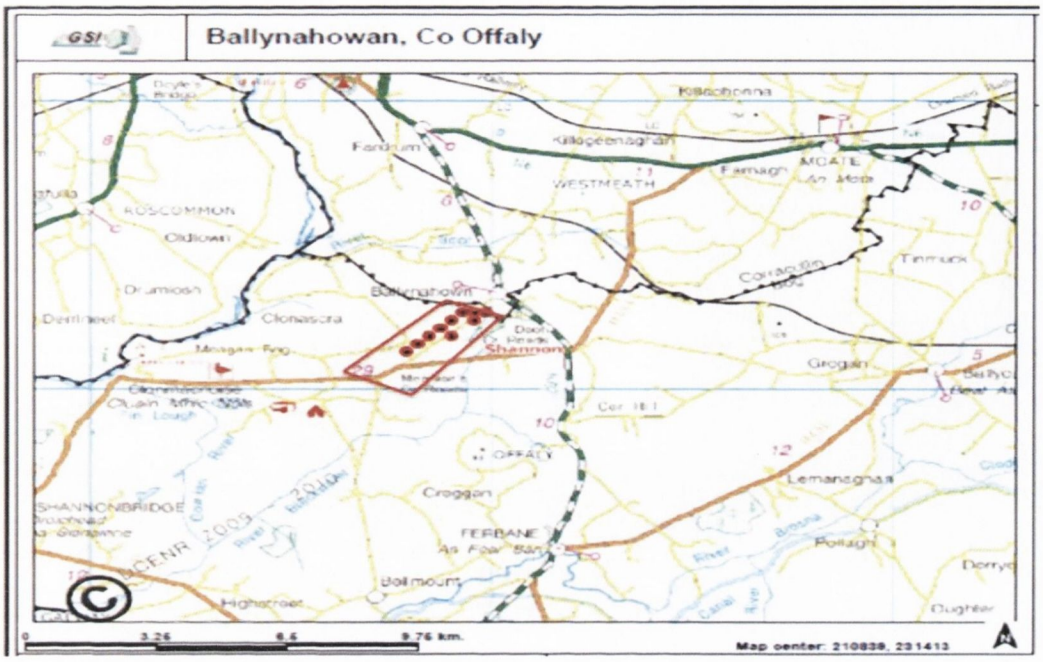


Figure 6.3 Study Area 1: Location Map (Ballynahowan)

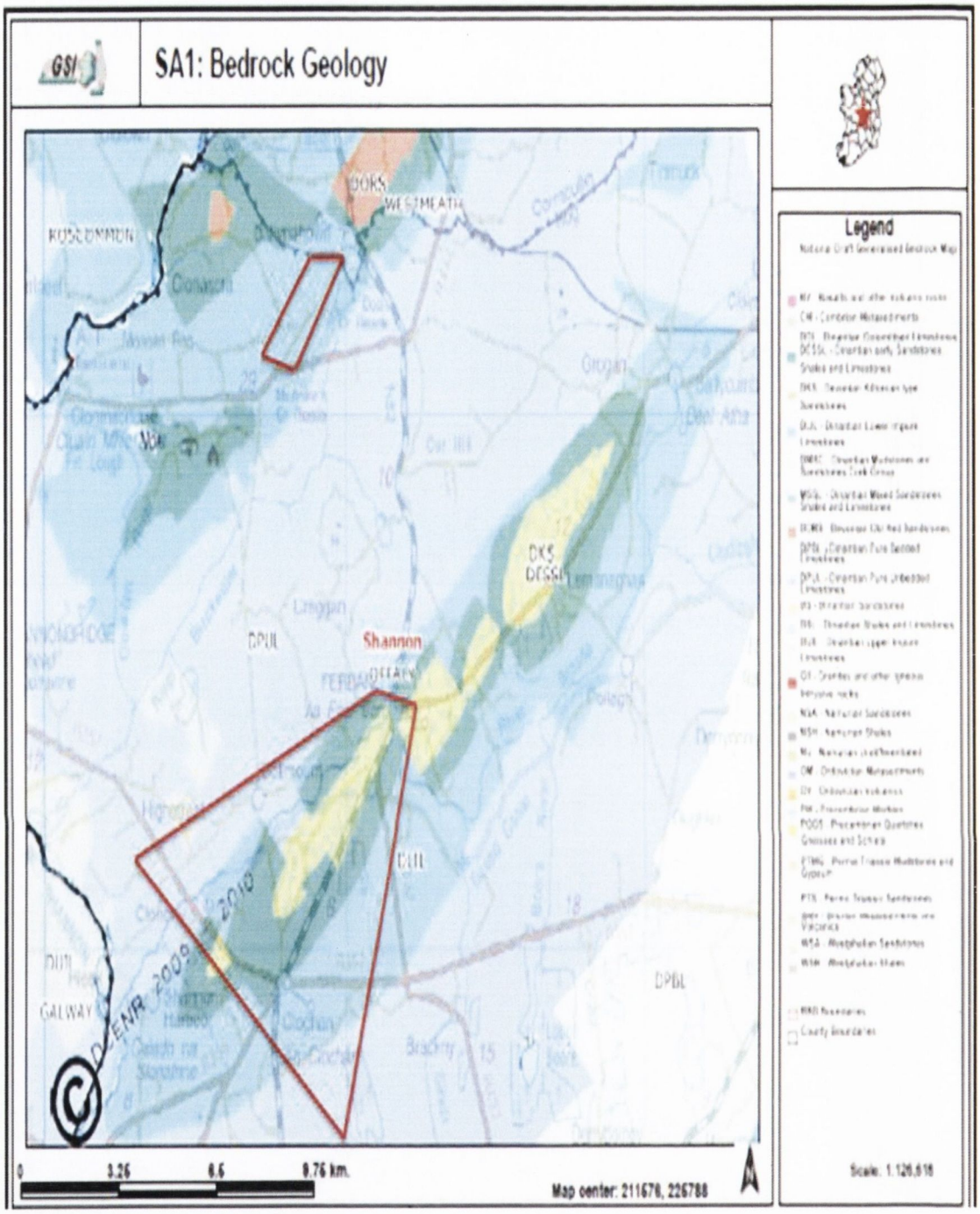


Figure 6.4 Study Area 1: Bedrock Geology Map

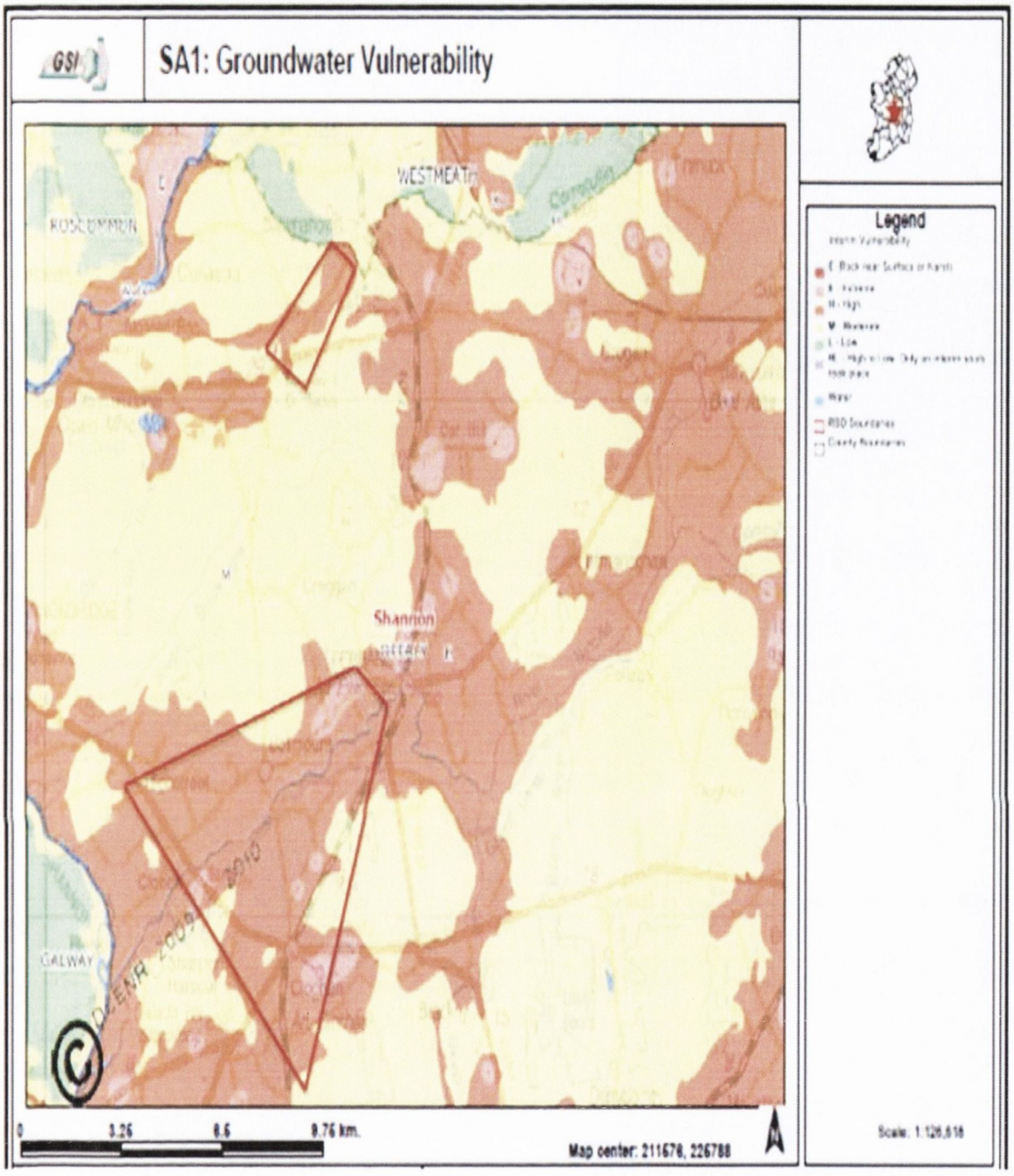


Figure 6.5 Study Area 1: Groundwater Vulnerability Map

Study Area 2: Crossbridge, Co Wicklow

SA2 is located in the southwest of Co Wicklow, directly adjacent to Co Carlow, approximately 90 km from Dublin City and is composed of two smaller study areas, namely Crossbridge (SA2a) and Kilquiggin (SA2b). The Crossbridge study area is approximately 35 km^2 (*Figure 6.6*), located approximately 3.1 km from Tinahely, Co Wicklow, with a central point at Bridgeland crossroads. The Kilquiggin study area is approximately 5.2 km^2 (*Figure 6.7*), located approximately 9.6 km from Tinahely, Co Wicklow, with a central point at Kilquiggin. The central points of the study areas are

located approximately 7.3 km apart. The two study areas are separated by a portion of Co Carlow, for which a GWPS has yet to be submitted and therefore was not suitable for study due to lack of data.

Subsoils in the area are dominated by granite tills (>80%), interspersed with pockets of outcrop and subcrop, granite sands and gravels and small veins of undifferentiated alluvium following a number of small rivers running through the study area, the main one being the Derry River. The study area geology is uniform throughout the area and characterised as granites and other igneous intrusive rocks (GII) of the Wicklow granite (*Figure 6.8*). A large area of Ordovician slates and grits is found directly east of the study area; however, this does not form any part of SA2.

SA2 is entirely characterised as being of *High* and *Extreme* vulnerability, with a significant proportion of the *Extreme* vulnerability deemed either rock outcrop or subcrop (*Figure 6.9*). Aquifers within the study area are characterised as poor with bedrock generally unproductive except in local zones (Pl) and locally important, with bedrock being moderately productive only in local zones (Ll), with Pl aquifers dominant overall.

The majority of the study area is located in an upland/highland, rural area and therefore remains unserviced in terms of public water supply and wastewater treatment. Accordingly, private groundwater is the primary household source of water in the areas, with Bored SPS, HD SPS and springs (Sp Wells) being used. The majority of residences employ individual septic tanks for wastewater treatment. Agriculture is the main activity in the area, with mixed grazing the mainstay activity, particularly sheep. Commercial forestry is also carried out in highland areas.

The area is situated within the South Eastern RBD. The total area of SA2 is 40.5 km^2, with a total of 50 sample sources. The sample source density in this area is 1.2 km^{-2}.

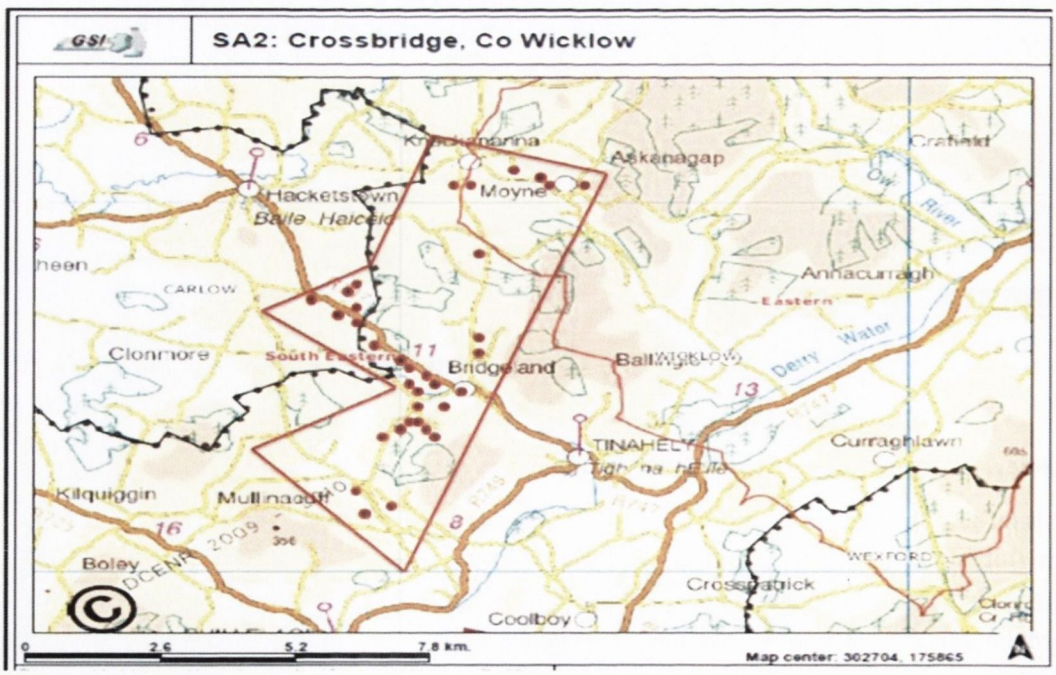


Figure 6.6 Study Area 2: Location Map (Crossbridge)

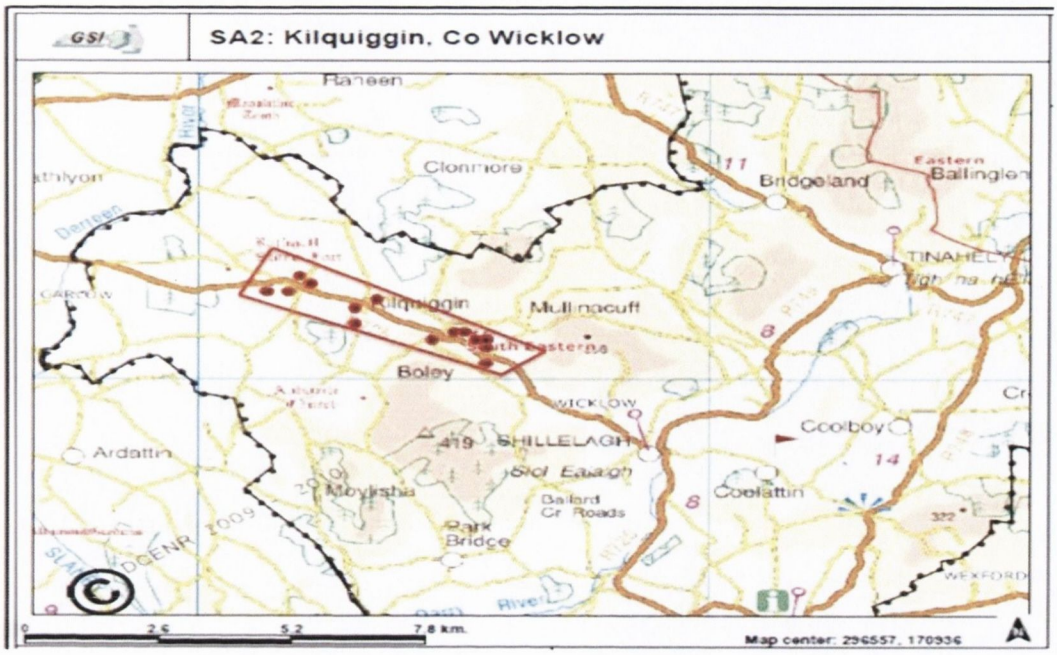


Figure 6.7 Study Area 2: Location Map (Kilquiggin)

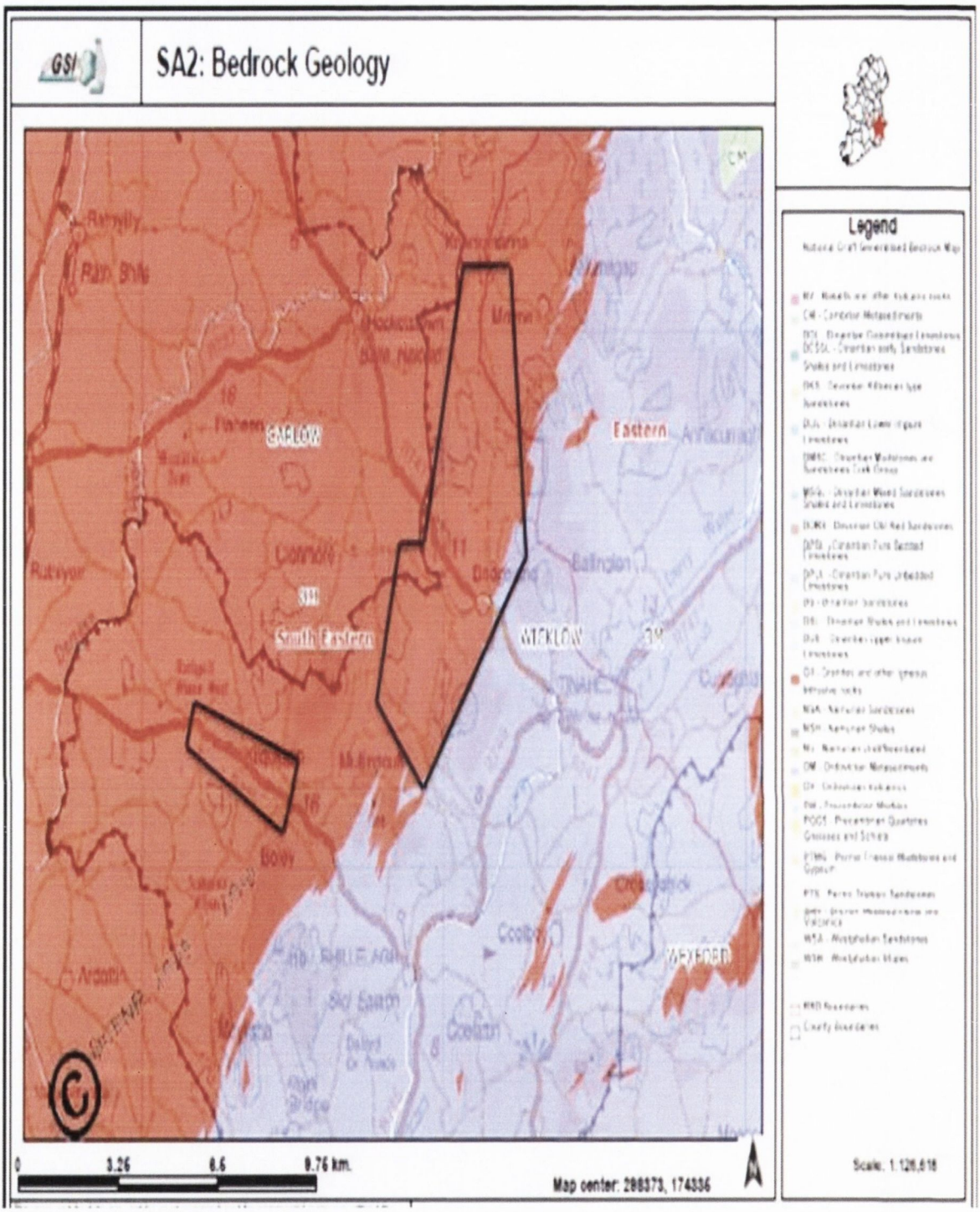


Figure 6.8 Study Area 2: Bedrock Geology Map

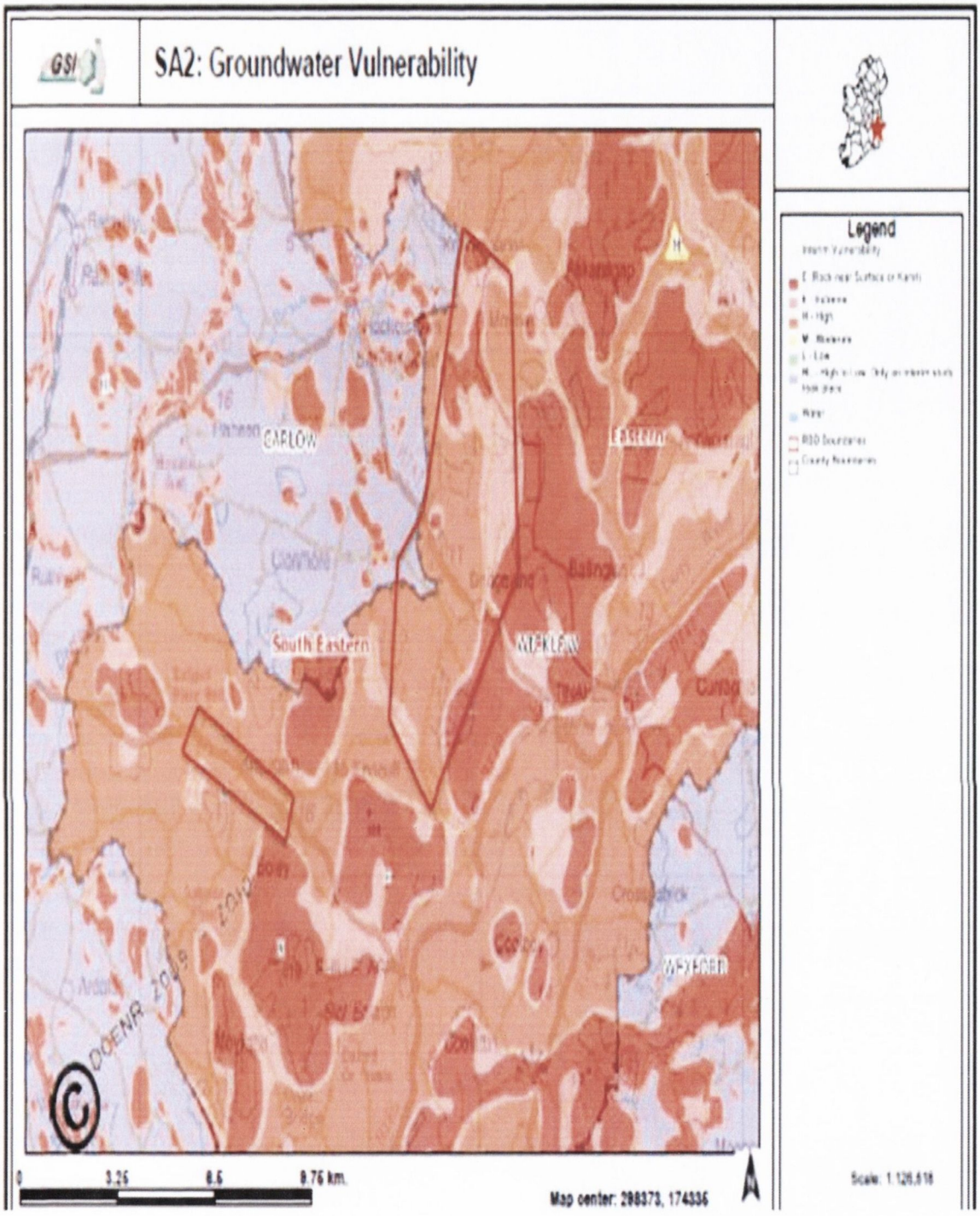


Figure 6.9 Study Area 2: Groundwater Vulnerability Map

Study Area 3: Paulstown, Co Kilkenny

SA3 is located in the northeast of Co Kilkenny, approximately 14.9 km from Kilkenny City and 106 km from Dublin City. SA3 covers an area of approximately 40.6 km^2 (*Figure 6.10*) with a central point at Castlewarren village, situated in the South Eastern RBD. The Co Carlow border runs directly adjacent to the northeastern border of the study area.

Subsoils in the study area are predominantly Namurian shales and sandstone tills with large pockets of interspersed bedrock outcrop and subcrop in the highlands. In the lower lying areas, Carboniferous limestone tills dominate. The bedrock geology throughout the study area is extremely diverse, occurring in relatively narrow strips of horizontal widths 0.25 – 0.7 km (as they appear on the ground surface) from highlands in the north, to wider bands of 2 – 4 km horizontal widths in the south. The bedrock consists of the following, from north to south; Westphalian sandstones (WSA), Westphalian shales (WSH), Namurian sandstones (NSA), Namurian shales (NSH), Dinantian pure bedded limestones (DPBL), Dinantian upper impure limestones (DUIL) and Dinantian dolomitised limestones (DDL) (*Figure 6.11*).

The entire northern half of the study area is characterised as *Extreme* vulnerability, with a significant proportion of this further characterised as outcrop/subcrop (Ex_{SUR}) (*Figure 6.12*). The southern half of SA3, although primarily of *High* and *Extreme* vulnerability, includes a lowland area surrounding the village of Paulstown, with a width ranging from 2.8 km in the northeast to 0.6 km in the southeast of the area. This *Low/Moderate* vulnerability area was avoided during fieldwork.

Aquifer maps indicate diverse aquifer characterisation in the highlands, due to the varied bedrock lithology, as previously outlined. There is a number of alternating narrow bands of generally unproductive poor aquifers (Pu) associated with sandstones (WSA and NSA) and moderately productive locally important aquifers (Lm) associated with shales (WSH and NSH). As the surface gradient decreases towards the middle of the study area, a 2.5 km wide layer of generally unproductive (except in local zones) poor aquifer (Pl) dominates. In the lowlands, a regionally important, diffuse karstified aquifer is present, associated with low and medium vulnerability. At the southern boundary of the study area, a locally important aquifer, moderately productive in local zones (Ll) is found. The majority of the study area is composed of poor and locally important aquifers.

The study area is located in an upland rural area with agriculture being the dominant activity, particularly beef cattle and sheep farming, with a small amount of commercial forestry also carried out. Although public mains/local authority supplies are in place to the west and south of the study area, the majority is un-serviced and therefore reliant on private groundwater sources. Households in this area typically employ individual septic tanks for domestic wastewater treatment.

As previously mentioned, the total area of SA3 is 40.6 km^2, with a total of 50 sample sources. The sample source density in this area is 1.2 km^{-2}.

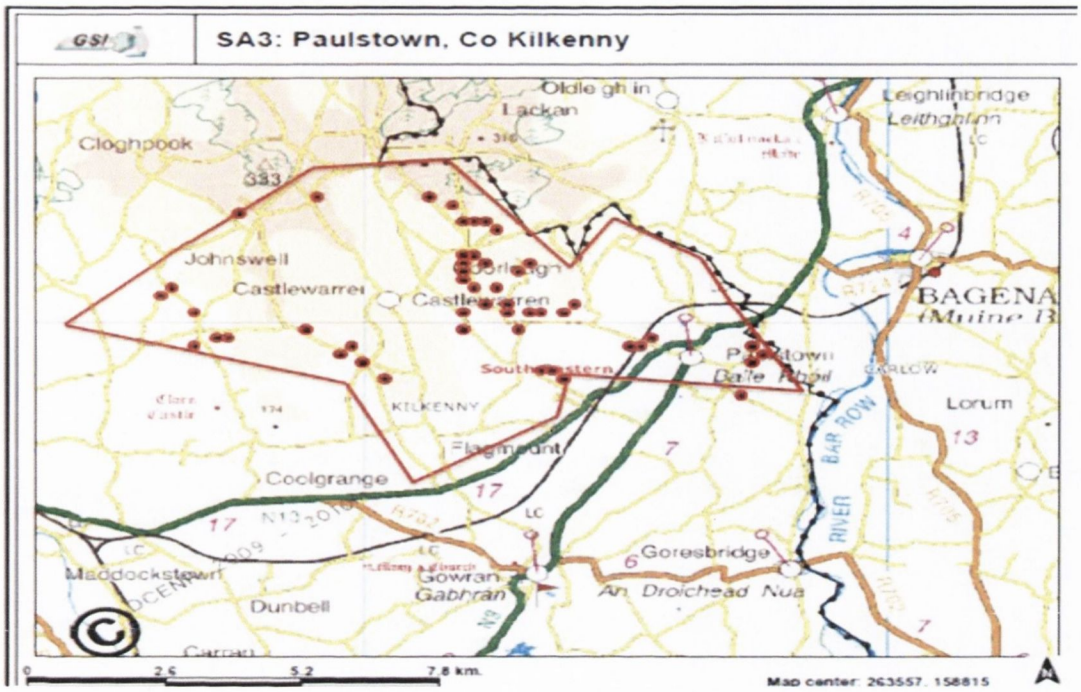


Figure 6.10 Study Area 3: Location Map

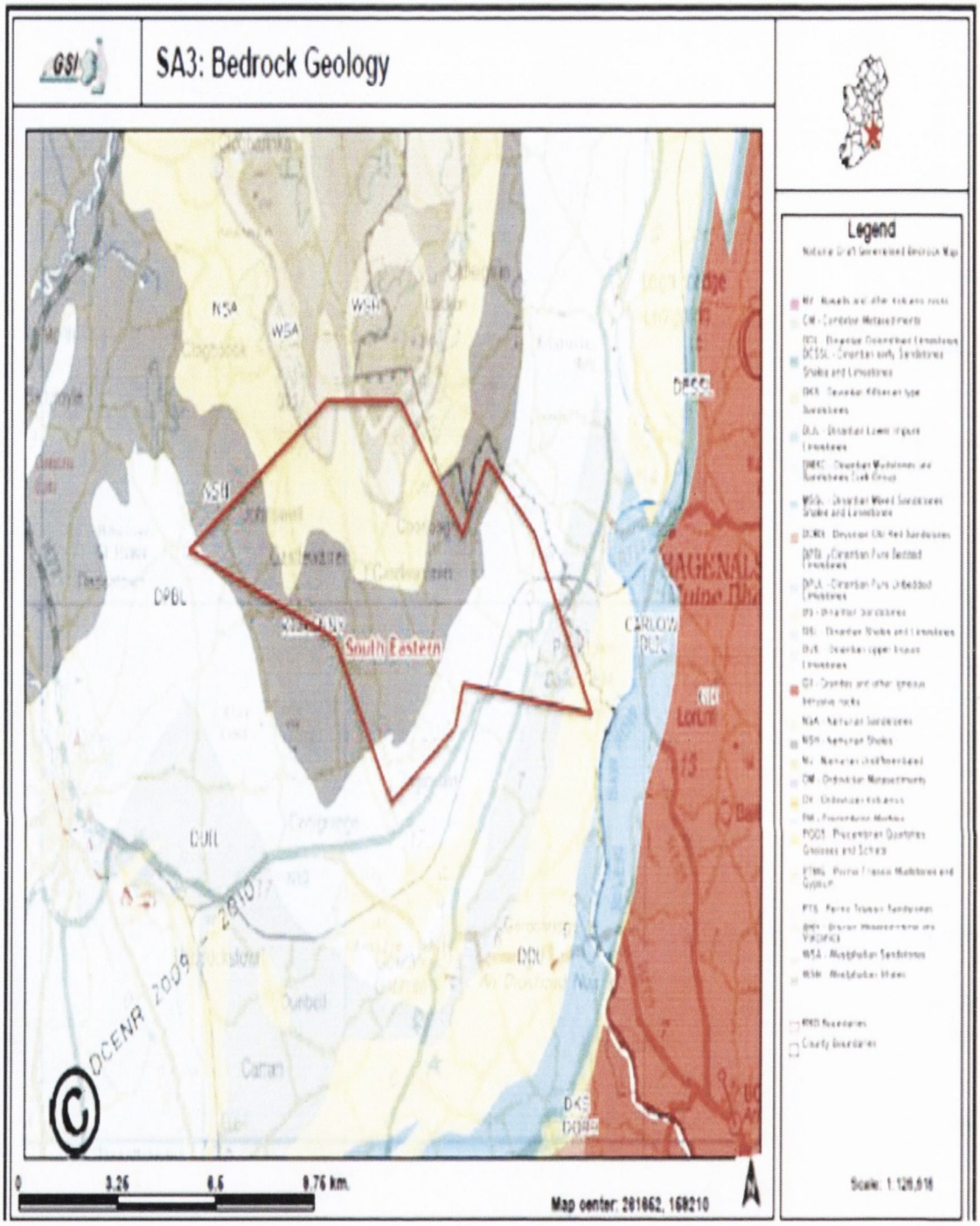


Figure 6.11 Study Area 3: Bedrock Geology Map



Figure 6.12 Study Area 3: Groundwater Vulnerability Map

Study Area 4: Kilclone, Co Meath

SA4 is located in the south of Co Meath, directly adjacent to the Meath-Kildare border. The study area covers an area of approximately 28.9 km^2 (*Figure 6.13*), with a central point at Kilclone village, situated approximately 30 km northeast of Dublin City and 6.6 km North of Maynooth, Co Kildare. The study area is located within the Eastern RBD.

Carboniferous limestone tills are the dominant subsoil in the area, with Namurian shale and sandstone tills (TNSSs) being found to the northwest of the area, with these TNSSs being found further southeast than the associated parent material. The area is also marginally interspersed with thin layers of undifferentiated alluvium, following a number of small rivers throughout the study area. Subsoil depths vary throughout the area, with well records indicating a subsoil depth range of 10-20 m. Dinantian Upper impure limestones (DUIL) dominate the bedrock geology of the study area, with a large area of undifferentiated, Namurian bedrock (NU), underlying approximately 15% of the study area, situated in the northwest of SA4 (*Figure 6.14*).

The majority of the area (>80%) is characterised as being of *Low* vulnerability, bordered by large areas of medium and high vulnerability to the south and west. Pockets of *Moderate* and *High* vulnerability within the area were avoided during fieldwork (*Figure 6.15*). The majority of the area (>90%) is dominated by a locally important aquifer, which is productive only in local zones (Ll), with the area underlain by Namurian shales and sandstones to the northwest characterised as being a generally poor aquifer, which is productive only in local zones (Pl).

The study area is generally described as being flat to very gently undulating, with agriculture being the predominant activity, in the form of beef and dairy farming. Although a public water supply is now available, this is relatively recent and a large proportion of households still use private groundwater supplies, either accompanying public supplies as part of a dual supply system or as the primary household supply. Dwellings are generally served by on-site septic tank systems. The area is considered "overdeveloped", owing to a high density of private dwellings and its rural setting (McCole, 1992).

In all, 52 sample sources were acquired in SA4, leading to a sample source density of 1.8 km^{-2}.

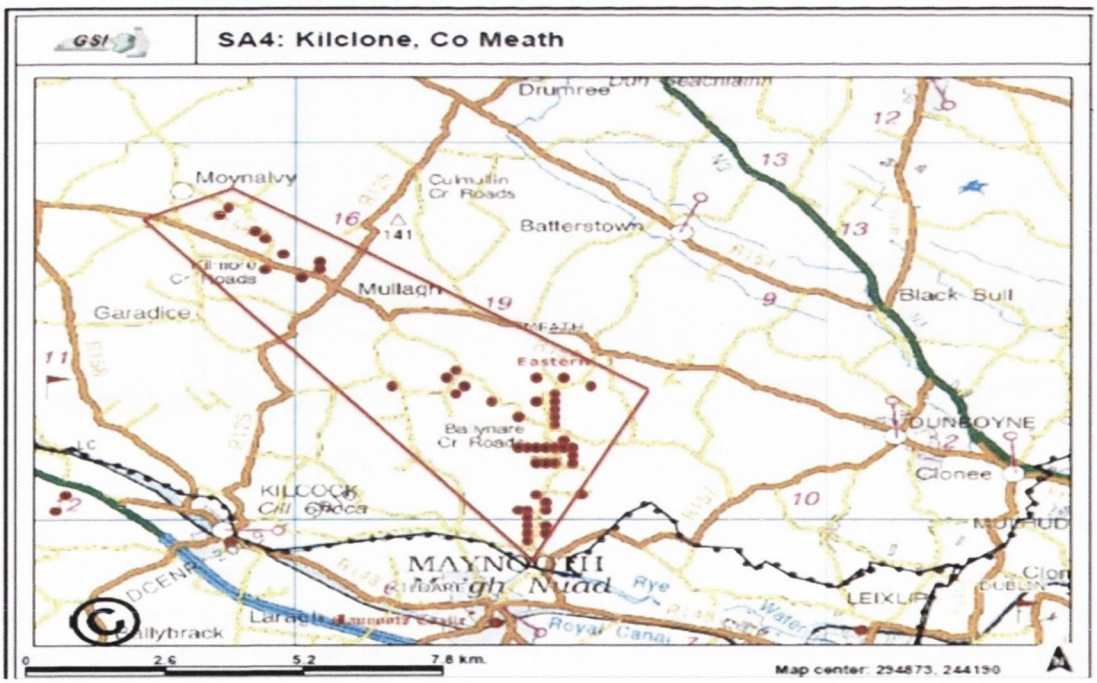


Figure 6.13 Study Area 4: Location Map

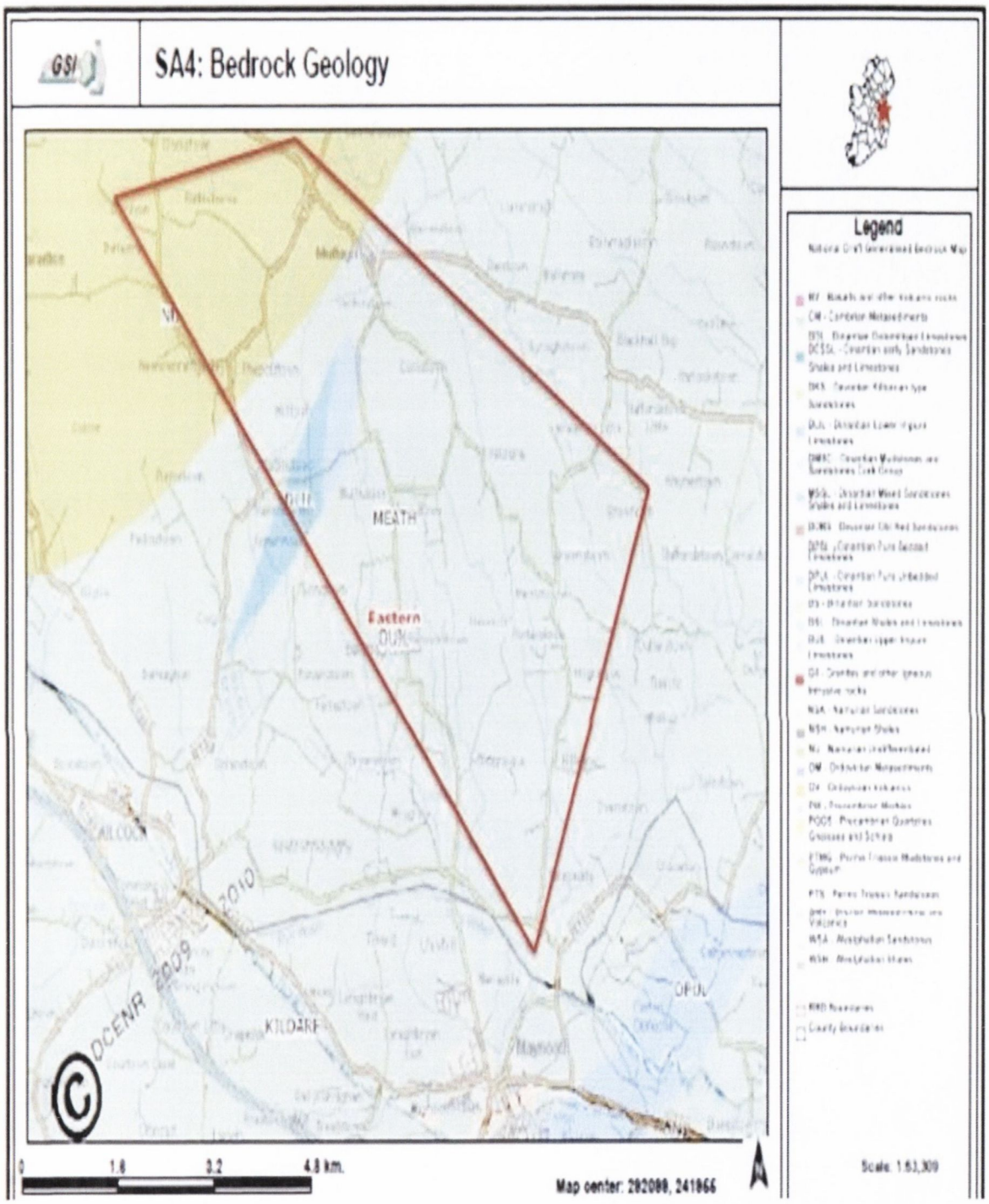


Figure 6.14 Study Area 4: Bedrock Geology Map

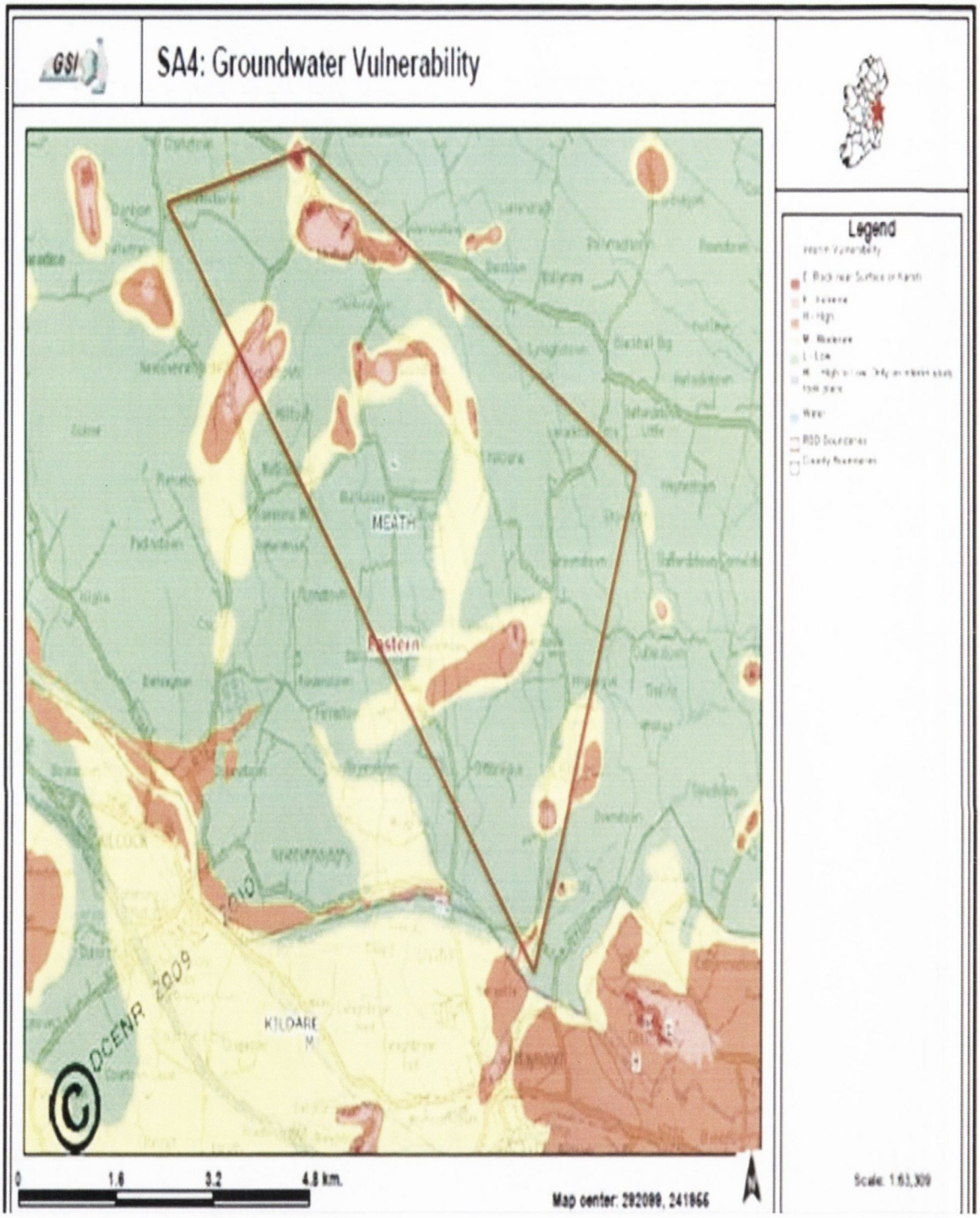


Figure 6.15 Study Area 4: Groundwater Vulnerability Map

Study Area 5: Birdhill, Co Tipperary

Although four study areas were considered sufficient for data collection, a limited volume of fieldwork was carried out in a fifth area, at Birdhill, Co Tipperary because suitable data were readily available (even though this study area was not initially selected as part of the study area selection procedure).

SA5 is located in the northwest of Co Tipperary. The study area covers approximately 10.6km^2 (*Figure 6.16*), with a central point of Birdhill, Co Tipperary, approximately 23 km east of Limerick city. The study area is located within the Shannon RBD.

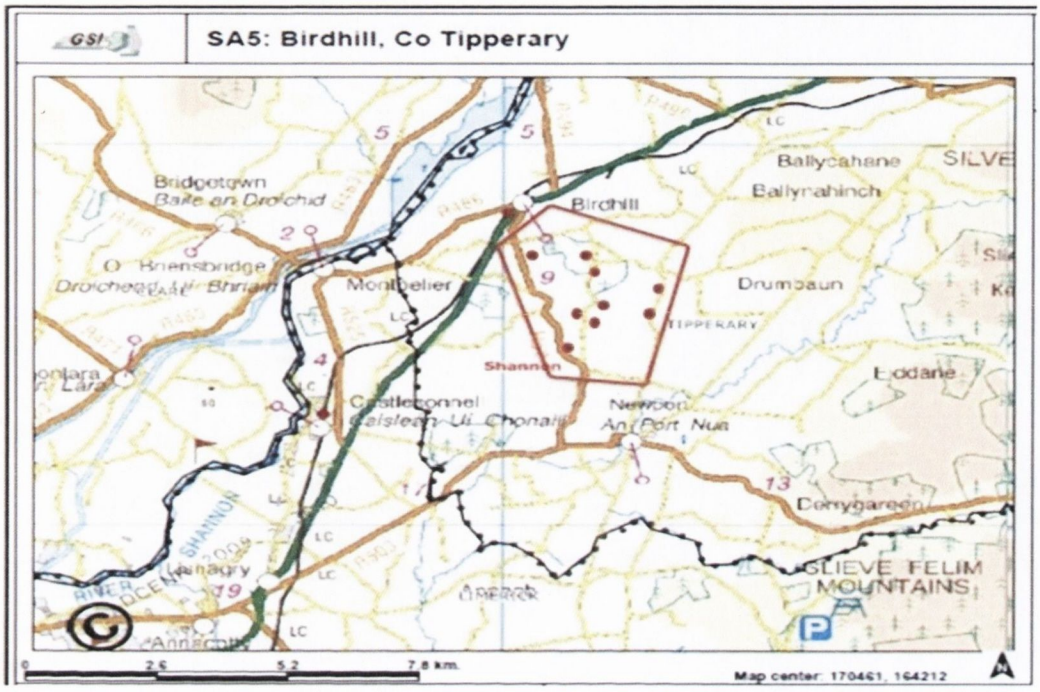


Figure 6.16 Study Area 5: Location Map

Lower Palaeozoic/Devonian sandstone tills are the dominant subsoil throughout the area, with some interspersed subcrop or outcrop. The area is also covered by small pockets of cutover peat. Bedrock within the area is diverse. The north of the area is underlain by Silurian Metasediments, the middle dominated by Devonian Old Red Sandstone (DORS) and the south characterised by Dinantian Lower Impure Limestones (DLIL) with a 300 m layer of Dinantian Early Limestones and Sandstones (DESSL) buffering the DORS and DLIL layers (*Figure 6.17*).

Although vulnerability maps are not currently available for Co Tipperary, as the county groundwater protection scheme remained incomplete during the study period , Teagasc subsoil maps characterise the majority of the area as *Extreme* vulnerability, with a high proportion of this being E_{sur} (rock near surface or karst) (*Figure 6.18*). The main aquifer in the area is characterised as locally important, with bedrock being productive in local zones (Ll), with some areas in the north and south of the area characterised as being poor aquifers, with bedrock being generally unproductive except in local zones (Pl).

The area may be described as relatively sparsely populated, with agriculture being the dominant landuse activity. Two PuGWS are in operation within the area, with the remainder of households using private groundwater supplies. In all, 9 sample sources were obtained, resulting in a sample source density of 0.8 km^{-2}.

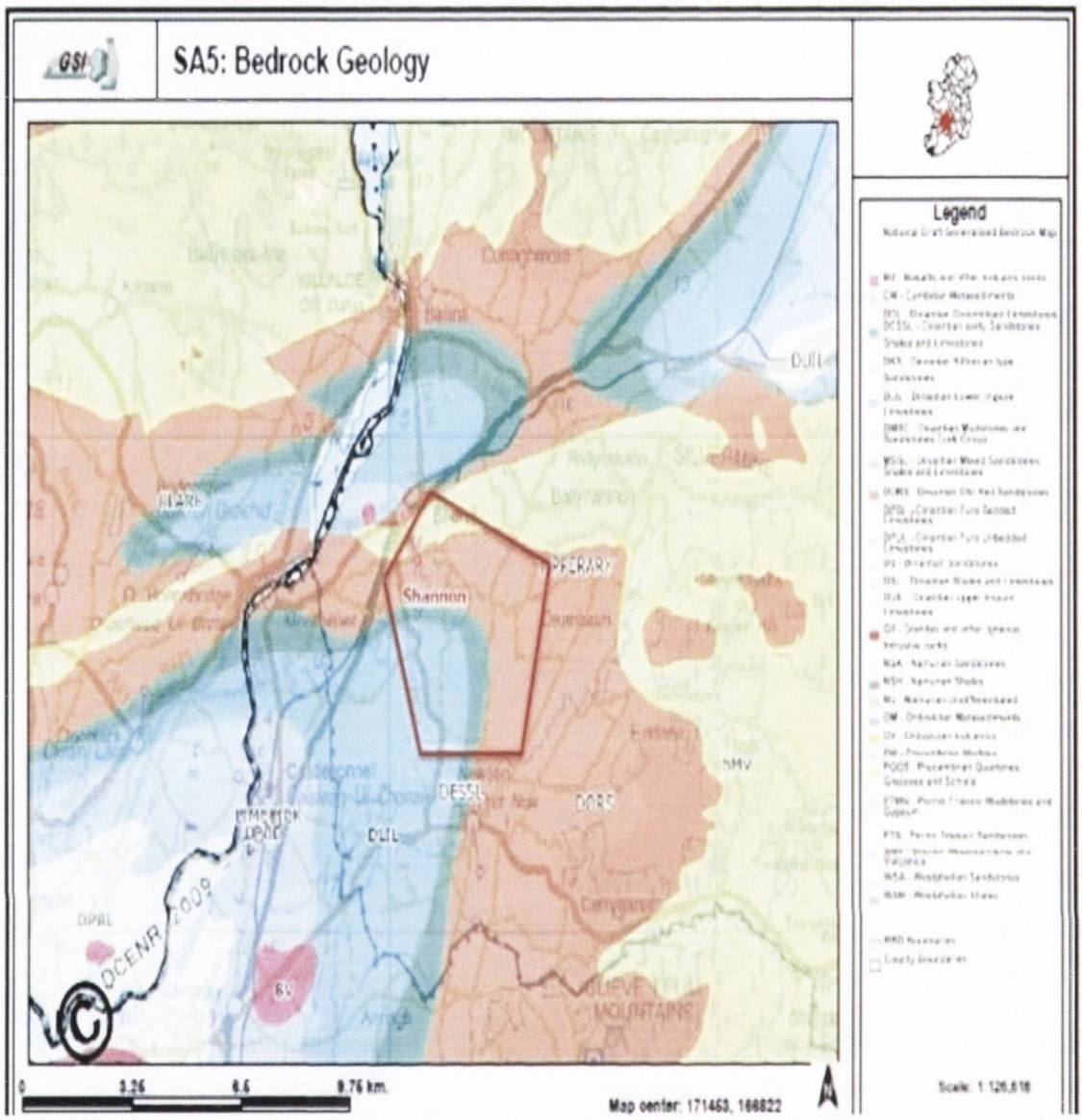


Figure 6.17 Study Area 5: Bedrock Geology Map

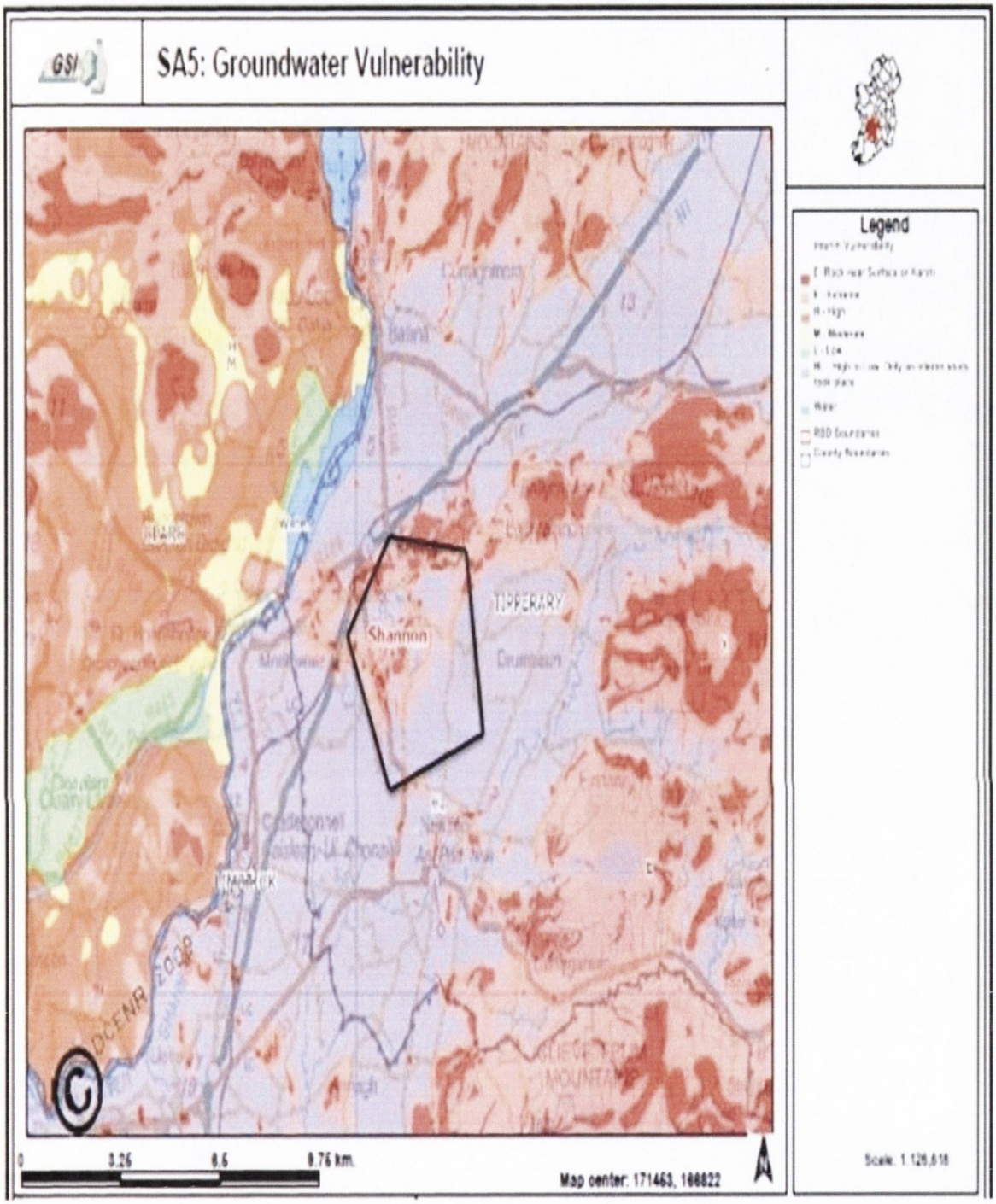


Figure 6.18 Study Area 5: Groundwater Vulnerability Map

6.3 Site Susceptibility Assessment

One of the key research goals of this study, outlined in Chapter 1 (*Section 1.2*), was the investigation of a quantified link between contamination risk (and hence health risk) and the groundwater vulnerability category. Groundwater vulnerability categorisation, although very useful for assessing potential contamination risk due to aquifer contamination, does not quantify the contamination risk posed by inappropriate location, design, construction and maintenance of a private groundwater supply. Therefore, it was considered that in order that both localised and generalised contamination risks could be assessed with a view to quantifying of the overall contamination risk associated with private groundwater sources in Ireland, a source susceptibility assessment protocol was developed.

This source susceptibility assessment was carried out concurrently with sample collection at all sample sources, in order that sample analysis results could be placed in context and any potential causes of a particular sample result occurring could be later assessed. Many of the parameters included in the site assessment will remain unchanged or change gradually over extended time periods, including hydraulic gradient, setback distance from roads, or the presence of permanent surface water features in the vicinity. However, other parameters are more transient over shorter periods of time and may have a direct effect upon water quality in the area including recent weather changes, changes in agricultural practices, construction in the area, new well construction etc. It was therefore considered essential that all parameters with the potential to affect water quality be documented.

The source susceptibility assessment comprised three main parts:

1. *Hydrogeological Characterisation* – A record of hydrogeological characteristics associated with the sample source, including vulnerability category, aquifer type, superficial and bedrock geology and bedrock faults in the vicinity.

2. *Site Characterisation* – A record of conditions that could have relevance to source contamination, although not directly associated with the source construction, design or maintenance, including potential point or non-point sources of contamination and associated pathways. For example, distance and gradient to potential contaminant sources, land use, drainage conditions, surface hydrology, ground condition and vegetation indicators.

3. *Source Characterisation* – A record of source construction, design and maintenance conditions that could influence source contamination. For example, source type, condition of well/wellhead, well design specifications, presence of waterproof barriers, presence of a secure boundary and wellhead construction finish.

This procedure was carried out at all groundwater sampling locations in order to collect and document as much useful data as possible for each site. The susceptibility assessment was divided into 6 distinct sections, as follows:

1. General Details
2. Desk Study

3. Visual Site Assessment
4. Potential Sources of Pollution
5. Wellhead Assessment
6. Water Sample Analysis (further divided into on-site analysis and laboratory analysis)

The completion of source susceptibility assessments allowed for comparative analysis, not only between study areas, but also between distinct sources within the same study area. A copy of the developed susceptibility assessment is included in Appendix D3.

6.4 Sample Collection and Fieldwork Protocols

6.4.1 Parameter Selection

The purpose of the monitoring programme in this study was the investigation of the link between contamination risk, and hence health risk, and the groundwater vulnerability category. As the main health concerns with regard to private groundwater sources are faecally derived pathogens (Gale, 1996; Gerba *et al.,* 1996a; Macler & Merkle, 2000; Abbaszadegan, 2003; Howard *et al.,* 2003; Gonzales, 2008), it was considered important that water quality data collection should focus on faecal pathogens and indicators of faecal contamination.

Additionally, physicochemical and chemical analysis was also carried out. The reasons for this were (Misstear *et al.,* 2006):

- These parameters dominate the water chemistry and may influence how other elements or species behave.
- Provision of a hydrochemical context for the groundwater i.e. hydrochemical changes, residence time, etc

The parameters chosen for analysis and the analytical protocols used are now outlined. These are separated into three sections:

1. Field Analysis;
2. Chemical Analysis; and
3. Microbial Analysis (further separated into bacteriological and protozoan analysis)

6.4.2 Sample Collection

A total of 50 sampling sites (wells) were targeted in each study area for groundwater sampling and source susceptibility assessment. All sampling sites were functional private water supply wells i.e. wells installed primarily for supply of private domestic households with a dedicated pump system in place. In order that all collected water quality data were comparable and sample integrity maintained, a groundwater sampling protocol was developed for use at all sampling sites. Several documents outline appropriate groundwater sampling procedures including Keith (1988); Horowitz *et al.* (1994), and Koterba *et al.* (1995). The sample collection protocol developed and used

throughout this study was based upon USGS (2005), as it was considered appropriately comprehensive and current.

All samples were collected in sterilised (autoclaved at 121°C for 20 minutes) sample bottles. Samples for bacterial analysis were collected and stored in 250 ml glass bottles, with samples for chemical analysis being collected and stored in 600 ml polyethylene sample bottles. Sampled wells were purged for a minimum of 5 minutes to clear stagnant groundwater from the distribution system, with the closest permanent fixture directly connected to the well being used for well purging and subsequent sample collection. Indoor fixtures were avoided where possible, in order that stagnant water from domestic storage tanks was not sampled. All forms of water treatment were avoided, with samples being taken pre-treatment where possible. It should be noted that, where bacteriocidal treatment systems, or systems capable of bacterial filtration (i.e. UV, reverse osmosis, nanofiltration, ultrafiltration) were in place and unavoidable i.e. no available pre-treatment source, the source was not sampled. After purging, sample bottles were filled and rinsed 3-4 times with the water being sampled, and the sample was then taken, with sample bottles being filled from bottom to overflowing in order to avoid sample aeration. Field measurements were immediately recorded (i.e. pH, EC, temperature and turbidity).The sample bottle was tightly sealed, labelled and stored in a cooler box to avoid excessive temperature change and light. All field measurement equipment was thoroughly rinsed with distilled water between sample sources. All samples were processed and analysed within six hours of collection.

6.4.3 On-Site Analysis

All on-site analysis was undertaken, in accordance with USGS National Field Manual for the Collection of Water Quality Data (2005).

Temperature

Temperature is a useful physicochemical parameter due to its ease and speed of measurement and may be used to distinguish "true" groundwater from surface water ingress. All sample temperatures were taken with a conventional thermometer and were recorded in °C.

Electrical Conductivity/Total Dissolved Solids

Electrical Conductivity (EC), like temperature, is a useful parameter due to ease of measurement, and may be used to indicate the concentration of dissolved ions in solution. All groundwater samples had EC measured in the field, immediately after sample collection. A calibrated EC meter (WTW ProfiLine 197i™ Portable Conductivity Meter) was used for all analysis and results recorded in µS cm^{-1} at 25 °C. As most of the dissolved content of natural groundwaters is in ionic form, EC may be empirically related to *total dissolved solids* (TDS). For a typical fresh groundwater, dominated by calcium and bicarbonate, with subsidiary sodium and chloride, the following approximate relationship exists at 25°C, and is valid to approximately 2000 µS cm^{-1} (Appelo & Postma, 1996; Misstear *et al.*, 2006):

$$\sum \text{anions (meq } l^{-1}) = \sum \text{cations (meq } l^{-1}) \approx EC/100$$

The following relationship is also valid:

$$TDS\ (mg\ L^{-1}) \approx EC\ (\mu S\ cm^{-1}) \times F$$

where the factor F = 0.55 for a water dominated by sodium chloride and F = 0.75 for a water dominated by calcium bicarbonate. F = 0.64 was used as an average conversion factor throughout this study.

pH

The pH (potential of hydrogen) is a measure of acidity (or alkalinity) of a solution and may be defined as:

$$-\log_{10}c$$

where c is the concentration of hydrogen ions in moles per unit decimetre. A more accurate approximation of pH is given by the Henderson-Hasselbalch equation;

$$pH = Pk_a + \log [A^-]/[HA]$$

The pH of water determines the solubility (amount that can be dissolved) and biological availability (amount that can be utilised by aquatic life) of chemical constituents such as nutrients (phosphorus, nitrogen, and carbon) and heavy metals (lead, copper, cadmium, etc.) (Covington *et al.,* 1985). A pH meter (Cyber-Scan Series 600™, Waterproof Portable Meter) was used for all pH analysis, with pH measurements taken immediately after sample collection. pH meter calibration was undertaken at the start of each day in the field at reference pH values 4 and 7.

Turbidity

Turbidity is a measure of the degree to which the water loses its transparency due to the presence of suspended particulates, including phytoplankton, sediments from erosion, re-suspended sediments, waste discharge, algae growth and urban runoff. All groundwater samples were measured for turbidity using an OXFAM-DelAgua turbidity tube (Oxfam-DelAgua, 2004), covering the range 5 – 2,000 TU.

6.5 Laboratory Protocols

6.5.1 Chemical Analysis

Throughout the duration of the site-work, all groundwater samples collected were analysed for ammonium (NH_4-N), nitrate (NO_3-N) and chloride (Cl) using a Merck Spectroquant Nova 60™ spectrophotometer and associated US EPA approved reagent kits. All kits and reagents used were deemed suitable for groundwater analysis (APHA/AWWA/WEF, 2005), with all groundwater samples falling within the required pH range for analysis. Chemical analysis of samples was undertaken in the Environmental Engineering Laboratory, Trinity College Dublin.

Ammonia (NH_4-N)

Ammonium nitrogen (NH_4-N) occurs partly in the form of ammonium ions and partly as ammonia. A pH dependent equilibrium exists between the two forms. In strongly alkaline solution ammonium nitrogen is present almost entirely as ammonia, which reacts with a chlorinating agent to form monochloramine. This in turn reacts with thymol to form a blue indophenol derivative that is determined photometrically. All groundwater samples were analysed in the range 0.1-3.0 mg L^{-1}. The method is analogous to EPA 350.1, APHA 4500-NH_4 D, and ISO 7150/1.

Chloride (Cl)

Chloride ions react with mercury (III) thiocyanate to form slightly dissociated mercury (II) chloride. The thiocyanate released in the process in turn reacts with iron (III) ions to form red iron (III) thiocyanate that is determined photometrically. All groundwater samples were analysed in the range 10-250 mg L^{-1}. The method is analogous to EPA 325.1 and US Standard Methods 4500-Cl^- E.

Nitrate (NO_3-N)

In sulphuric and phosphoric solution nitrate ions react with 2, 6-dimethylphenol (DMP) to form 4-nitro-2, 6-dimethylphenol that is determined photometrically. All groundwater samples were analysed in the range 0.1-25 mg L^{-1}. The method is analogous to ISO 7890/1.

6.5.2 Trace Element Analysis

Inductively coupled plasma-atomic emission spectrometry (ICP-AES) was used to determine trace elements, including metals, in solution. Numerous studies have used ICP-AES for the determination of trace elements in groundwater samples including Frapporti *et al.* (1996), Csuros & Csuros (2002) and Kousa *et al.* (2004). A plasma source is used to dissociate the sample into its constituent atoms or ions, exciting them to a higher energy level. They return to their ground state by emitting photons of a characteristic wavelength depending on the element present. This light is recorded by an optical spectrometer. When calibrated against standards the technique provides a quantitative analysis of the original sample. A useful review of ICP-AES principles and applications is presented in Fifield & Haines (2002).

A Varian Liberty AX Series II was used for analysis of the following elements:

- Aluminium (Al)
- Cadmium (Cd)
- Calcium (Ca)
- Cobalt (Co)
- Copper (Cu)
- Chromium (Cr)
- Iron (Fe)
- Lead (Pb)
- Magnesium (Mg)
- Manganese (Mn)
- Nickel (Ni)
- Phosphorus (P)

- Potassium (K)
- Sodium (Na)
- Zinc (Zn)

The protocol used throughout this study is analogous to EPA Method 6010B: 1996 and ISO 11885, both of which are outlined in van de Wiel (2003). The protocol used is now outlined.

Preparation of Samples

ICP-AES Analyses was carried out on a batch basis necessitating acidification of groundwater samples in the laboratory immediately after returning from sites. 10 ml of each sample was filtered through a 1.2 µm (Whatman Glass Microfibre) filter to ensure that particulate matter did not enter the analytical process, causing overestimation of the dissolved elemental concentrations. 1 ml of nitric acid (69%) was then added (chemical preservation) to prevent precipitation, sorption or biodegradation. Groundwater samples that have been pre-filtered and acidified did not need acid digestion, as with other media i.e. surface water, solid wastes, etc. Samples were stored in a refrigerator for subsequent batch analyses.

Preparation of Standards

Analyses were carried out for Total P and Multi-Element separately (as standard solutions were not available for multi element analysis which included phosphorus) and therefore separate standards were prepared for Total P and Multi Element.

10 ml standards for total phosphorus were made up using AccuStandard (AccuTrace Reference Standard Phosphorous 1000 µg ml^{-1}) to 0.25, 0.5, 1, 2, 5 and 10 mg L^{-1} by dilution using deionised water, and 1 ml of nitric acid added to each (to negate the initial acidification). A blank was also prepared using only deionised water and nitric acid. Calibration was carried out with a maximum percentage error of 10%.

10 ml standards for multi-element were made up using AccuStandard Custom (AccuTrace Reference Standard Al, As, B, Ba, Cd, Co, Cr, Cu, Fe, Mg, Mn, Mo, Ni, Pb, Se, Sr, Zn 5 µg ml^{-1}, and K 50 µg ml^{-1}) to 0.25, 0.5, 1 and 2 mg L^{-1} for all elements apart from K, for which standards were prepared to 2.5, 5, 10 and 20 mg L^{-1}. These standards were prepared similarly to Total P standard by dilution using deionised water, with 1 ml of nitric acid was added to each. A blank was also prepared using only deionised water and nitric acid. Calibration was carried out with a maximum percentage error of 10%.

ICP-AES Analysis

Stocked samples were allowed to reach room temperature before ICP-AES analysis. The Liberty Series II Hardware Operation Manual (Varian, 1997) was adhered to during set-up and calibration of the ICP-AES system. Upon successful calibration, prepared samples were analysed individually, with a system recalibration taking place after every 25 sample cycle ensuring drift control. The content of each element present was calculated by system software (Plasma 96 Version 1.12) and presented as an emission spectrum (plot of the radiation intensity (y-axis) versus the wavelength (x-axis)) and associated elemental concentration (mg/L).

The first batch of samples subjected to multi-element analysis were analysed for all elements as outlined above. This batch consisted of 81 filtered and acidified groundwater samples. Interpretation of the results led to the exclusion of a number of elements from future multi element analysis, due to negligible or undetectable elemental concentrations including; Cd, Co, Cu, Cr, Pb, Ni, Zn.

6.5.3 Bacteriological Analyses

The primary goal of this research was the assessment of human health impacts due to contamination of private groundwater sources in Ireland. Therefore it was considered that the isolation and enumeration of both faecal indicator bacteria and directly pathogenic organisms from private groundwater samples were the most important data associated with sample collection and analyses. These protocols are now outlined. All bacterial analyses took place within 6 hours of sample collection.

E. coli

The heterotrophic plate count (HPC) is used for estimating the number of live heterotrophic bacteria in water samples. The detection of these organisms is based upon their ability to ferment lactose, with fermentation detected by use of an indicator in the medium (Gleeson & Gray, 1997). Three HPC methods are commonly used for the analysis and enumeration of heterotrophic bacteria in water samples, namely:

- the pour plate method (9215B),
- the spread plate method (9215C), and
- the membrane filtration method (9215D)

Colonies may arise from pairs, chains, clusters or single cells all of which are included in the term colony forming units (CFU). Other, non-HPC methodologies which may be used include multiple tube fermentation, hybridisation and polymerase chain reaction (PCR) (Gray, 2004). The membrane filtration (MF) method was used to culture and subsequently enumerate coliform bacteria due to its being highly reproducible, suitability for testing relatively large, low solids, sample volumes and ability to yield numerical results more rapidly than many multiple-tube fermentation techniques. MF is also the preferred method for low count waters (<1 to 10 cfu/100ml), and was therefore considered the most suitable method for this research. The MF technique with subsequent growth on membrane lauryl sulphate broth (MLSB) is extremely useful in monitoring drinking water and a variety of natural waters (Alonso, 1996; Timms *et al.,* 1996; Alonso *et al.,* 1999; Sartory, 2005; APHA/AWWA/WEF, 2005), including groundwater (Gleeson & Gray, 1997; Howard *et al.,* 2003). MLSB gives a direct count of indole positive coliforms (presumptive *E. coli*) at 44°C without the need for sub-culturing (Gleeson & Gray, 1997). The enumeration of presumptive *E. coli* was favoured over total coliforms as *E. coli* is faecal specific, while other members of the coliform group such as *Citrobacter*, *Klebsiella* and *Enterobacter*, although all found in faeces, are also found in soil and water (Holt *et al.,* 1993; Gleeson & Gray, 1997).

Preparation of media

Membrane Lauryl Sulphate Broth (MLSB) was used as the culture media for all presumptive *E. coli* bacteria. The MLSB is buffered at pH 7.4 ± 0.2 at 25°C and was composed as follows:

Bacteriological peptone – 39.0 g L^{-1}
Yeast extract – 6.0 g L^{-1}
Lactose – 30.0 g L^{-1}
Phenol Red – 0.2 g L^{-1}
Sodium Lauryl Sulphate – 1.0 g L^{-1}

76.2 g of dehydrated culture media was dissolved in 1 litre of distilled water, applying gentle heat. The broth was then distributed into 100 ml screw top polypropylene bottles and autoclaved at 121°C (15 p.s.i.) for 15 minutes and allowed to cool. Prepared broth was stored at 4°C for no longer than 3-4 months (Maximum storage guideline – 6 months).

Preparation of E. coli

One absorbent pad was placed in each Petri dish. MLSB was allowed to return to ambient temperature prior to analyses and 2.5 ml was pipetted onto each pad. Tweezers were flamed and used to transfer a sterile membrane filter (white, grid marked, 47 mm diameter, 0.45 µm) onto the bronze disc filter support of a sterilised membrane filtration unit (OXFAM-DelAgua™). The filter funnel and collar were then screwed into position, insuring a watertight seal. The groundwater sample (of appropriate volume; see note below) was then carefully poured (to ensure filter membrane is not broken) into the filter funnel. A vacuum pump was then connected to the filtration base and a partial vacuum maintained in the filtration base. Upon complete filtration of the sample, a flame sterilised tweezers was used to transfer the membrane filter onto the absorbent pad. The Petri dish lid was then replaced and appropriately labelled. The filtration unit was thoroughly rinsed with distilled water after each sample had been filtered. One "blank" membrane filter was prepared and included in each sample batch (100 ml of distilled water filtered through filter) to ensure no cross-contamination occurred and analytical quality was maintained.

Note that appropriate sample volumes were assessed based on the sample turbidity and expected *E. coli* counts. Where there was reason to expect a low or moderate *E. coli* count and the sample exhibited low turbidity, a sample volume of 100 ml was analysed. Where high *E. coli* counts were expected or samples display high turbidity, 50% or 20% dilutions of the sample were prepared using distilled water as deemed appropriate. This was done to ensure membrane filter did not become blocked and to aid accurate colony counts (i.e. >100 cfu/100 ml result in smaller colonies due to competition for nutrients and make counting more difficult and subsequently less accurate).

Sample Incubation

A minimum of 1 hour and a maximum of 4 hours resuscitation was allowed after each batch of samples were processed before incubation. This was done to allow for "stresses" caused due to sample collection and transport (environmental exposure). Samples were incubated for 16-18 hours at a constant temperature of 44°C ± 0.5°C (i.e. guideline temperature and incubation period for thermotolerant coliforms (faecal coliforms) using MLSB).

E. coli Enumeration

Upon completion of the required incubation period, Petri dishes were removed from the incubator. Lids were removed and the surface of the membrane filter studied under good incident light, with

magnifying lens used where necessary to examine colonies. All yellow colonies with a diameter of 1-3mm were recorded as a colony forming unit (cfu). Where two or more colonies merged, the shape of the colony was closely examined and the number of merged colonies assessed. Each sub-colony was then recorded as a cfu. Where there were transparent, pink, or blue/grey colonies present, these were not counted as lactose fermenting bacterial colonies (i.e. not possible to identify without further study). Colony counts were then converted into cfu/100 ml and recorded as presumptive *E. coli*.

Faecal streptococci (FS)

A number of procedures are described for the enumeration of faecal streptococci (enterococci) in a variety of media (Niemi & Ahtiainen, 1995) including KF Streptococcus agar, Slanetz-Bartley agar (also referred to in clinical literature as M-Enterococcus (ME) agar) and 4-methylumbelliferyl-β-D-glucoside (MUST method).

During this study, faecal streptococci were isolated and enumerated by membrane filtration (MF) and subsequent growth on ME agar, a procedure which has been successfully used in previous studies (Slanetz & Bartley, 1957; Dutka & Kwan, 1978; Papapetropoulou & Pagonopoulou, 1998; Howard *et al.,* 2003). The following procedure was used for faecal streptococci enumeration throughout the study, as proposed by Dutka & Kwan (1978).

Preparation of media

The ME agar is buffered at pH 7.2 ± 0.1 at 25°C and was composed as follows:

Bacteriological tryptose – 20.0 g L^{-1}
Yeast extract – 5.0 g L^{-1}
Dipotassium Phosphate – 4.0 g L^{-1}
Glucose – 2.0 g L^{-1}
Sodium Azide – 0.4 g L^{-1}
Triphenyltetrazolium Chloride – 0.1 g L^{-1}
Bacteriological Agar – 10.0 g L^{-1}

42 g of dehydrated culture media was dissolved in 1 litre of distilled water and brought to the boil to dissolve. Excessive heat was avoided (i.e. brought to boil for 15-20 seconds and removed from heat). The solution was then dispensed to Petri dishes at a rate of 25 ml solution per dish and allowed to solidify at room temperature for 24 hours. Petri dishes were inverted and stored at 4°C for a maximum of 5 days or until signs of deterioration of contamination were evident. If not used within 5 days or if deterioration/contamination occurred, plates were disposed.

Preparation of FS

Prepared ME plates were allowed to return to ambient temperature prior to analyses. Appropriate sample volumes were filtered using the same technique as for faecal coliforms. Upon complete filtration of the sample, a flame sterilised tweezers was used to transfer the membrane filter onto the ME plate, ensuring that no air bubbles were present between agar and membrane filter (i.e. all faecal streptococci, if present, were in direct contact with nutrient source). The ME plate lid was then replaced, appropriately labelled and the plate inverted. The filtration unit was thoroughly

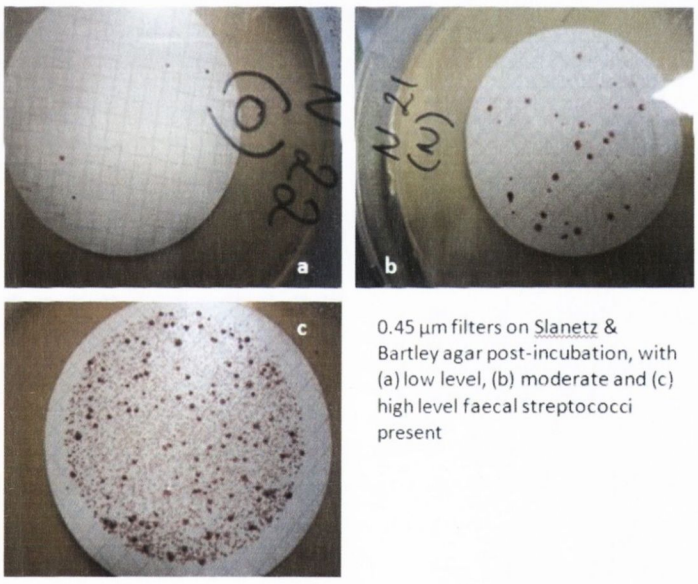
rinsed with distilled water after each sample had been filtered. One "blank" membrane filter was prepared and included in each sample batch (100 ml of distilled water filtered through filter) to ensure no cross-contamination occurred and analytical quality was maintained. Appropriate sample volumes were assessed based on similar protocols to those used for the enumeration of faecal coliforms.

Sample Incubation

Similar resuscitation times as those used for the enumeration of faecal coliforms were adhered to prior to incubation of ME plates. Plates were incubated for 44 ± 4 hours at a constant temperature of 37°C ± 1°C (i.e. guideline temperature and incubation period for faecal streptococci (*S. faecalis* ATCC 11700; *S. faecalis* ATCC 19433) using ME agar).

FS Enumeration

Upon completion of the required incubation period, ME plates were removed from the incubator. Lids were removed and the surface of the membrane filter studied under good incident light, with magnifying lens used where necessary to examine colonies. All red, dark red and brown colonies with a diameter of 1-3mm were recorded as a colony forming unit (cfu). Sub colonies were assessed and recorded in the same manner as those encountered in the enumeration of faecal coliforms. Colony counts were then converted into cfu/100 ml and recorded (*Figure 6.19*).



0.45 µm filters on Slanetz & Bartley agar post-incubation, with (a) low level, (b) moderate and (c) high level faecal streptococci present

Figure 6.19 FS colonies visible on 0.45µm filters

6.5.4 Protozoan Analysis

A number of potential procedures are available for the detection, isolation and subsequent enumeration of protozoan cysts and oocysts from raw water samples including immunomagnetic separation (IMS) (Bukhari *et al.,* 1998), phase-contrast microscopy (Suach, 1985), gene probe techniques (Abbaszadegan *et al.,* 1991) and flow cytometry (Vesey *et al.,* 1996; Barbosa *et al.,* 2007). It was decided that flow cytometry would be the method employed due to the availability of a suitable clinical microbiology laboratory and flow cytometer.

Flow cytometry is a technique used for counting, examining and sorting microscopic particles suspended in a stream of fluid. Using flow cytometry for the enumeration of protozoan cysts and oocysts involves multiparametric analysis of the physical attributes/characteristics of single cells flowing through an optical detection apparatus. A beam of laser light of a single wavelength is directed onto a hydro-dynamically focused stream of liquid, in this case untreated groundwater. Four detectors are aimed at the point where the stream passes through the light beam; one in line with the light beam (Forward Scatter (FSC)) and three perpendicular to it (Side Scatter (SSC) and one or more fluorescent detectors). Each suspended particle passing through the beam scatters the light in some way, and fluorescent chemicals attached to the particle are excited into emitting light at a lower frequency than the light source. This combination of scattered and fluorescent light is picked up by the detectors, and by analysing the fluctuations in brightness at each detector, it is possible to extrapolate regarding the physical structure of the cell. FSC correlates with the cell volume and SSC correlates with the inner complexity of the particle i.e. presence and shape of nucleus, type of cytoplasmic granules, membrane roughness, etc. In this manner, cytometry results, when correctly interpolated, can be used to discern between suspended dirt/detritus and protozoan cysts/oocysts in a groundwater sample (Vesey *et al.*, 1996; Barbosa *et al.*, 2007).

In this process, particulate matter from water samples is concentrated by filtration and re-suspended in a PBS solution using sonication and centrifugation. Protozoan cysts and oocysts are then stained with fluorescein isothiocyanate (FITC) – labelled monoclonal antibodies specific to the cyst wall. The process involves 5 steps, as follows:

1. Filtration
2. Suspension
3. Sonication
4. Centrifugation
5. Cytometry

Filtration

Upon completion of physiochemical, chemical and bacterial analysis of sample, the sample was filtered through a 0.22 µm Micropore™ filter. This was done by pouring 500 ml of raw sample into a filtration reservoir which has had a filter inserted using a sterilised tweezers. The filtration reservoir was then connected to a small vacuum (<400 mBar) pump, drawing the sample through the filter. When the sample has been filtered (15-25 minutes depending on the turbidity of the sample; high turbidity samples required more than one filter), the pump was switched off and the reservoir lid is carefully opened. The filter was then extracted using a sterilised tweezers and placed in a sterilised 50 ml polypropylene test tube.

Re-Suspension

Once the filter has been placed in the 50 ml test tube, 25 ml of phosphate buffered saline (PBS) solution was immediately transferred into the test tube (or enough to completely submerge the filter. The test tube is then tightly sealed using a screw top lid and shaken vigorously for 4-5 minutes (to re-suspend all particles from filter into PBS solution). The test tube was then refrigerated at 4°C and stored for future analyses, as cytometry was carried out on a batch basis.

Sonication

Sonication is the process of applying ultrasound energy to agitate particles in a sample, using an ultrasonic bath or ultrasonic probe. All samples were sonicated at 1 amp for 10 seconds using an ultrasonic probe to ensure that all protozoan particles (both viable and nonviable) have been re-suspended in the concentrated PBS solution. The ultrasonic probe was carefully cleaned and sterilised after each sample.

Centrifugation

After sonication, samples were centrifuged at 975 g (1g = 9.81 ms^{-1}) or 3,000 rpm for 10 minutes. After centrifugation, samples were placed in a waterbath at 37°C for 60 minutes for incubation/reanimation of cysts/oocysts.

Flow Cytometry

Concentrated, re-suspended samples were transferred to cuvettes and placed in cytometer one at a time. Graphical outputs were saved and interpolated.

6.5.5 Waterborne Viruses

At the outset of this research, it was hoped that a limited amount of viral sampling could be carried out, with particular emphasis placed on rotavirus as an effective analogue for waterborne enteric viruses. A number of potential techniques were investigated for potential use, including enzyme-linked immunosorbent assay (ELISA), epifluorescence microscopy and transmission electron microscopy (Gault & McClenaghan, 2009). However, due to likely low environmental concentrations and associated necessary large sample volumes (50-100 L), in concurrence with high laboratory costs, no viral work was subsequently undertaken.

6.6 Sample Analysis Quality Control

In order to assess the accuracy of sample analysis in the TCD laboratory, quality control measures were carried out. Two sets of groundwater samples, five in each batch, were delivered to City Analysts Limited (CAL) Laboratories, an accredited laboratory located in Dublin. Samples analysed ranged from relatively deep bored SPS to shallow spring well sources. The chemical and bacteriological results obtained from those externally analysed, were compared with duplicate analyses carried out in the TCD laboratory. As shown (*Table 6.3)*, individual parameters presented significantly linear relationships.

Table 6.3 Regression statistics from analyte comparisons

Parameter	R^2	Slope
NO_3-N	0.909	0.99
NH_4-N	0.925	1.12
Cl	0.981	0.91

Comparison of *E. coli* results between TCD and City Analysts Ltd analysis showed a good degree of correlation. Raw analyses returned regression statistics of $R^2 = 0.999$, Slope = 15.08x -18.4; however, one sample analysed at the TCD laboratory was deemed uncountable using MF techniques i.e. >300cfu/100ml. When this result was changed to coincide with the result obtained by City Analysts Ltd, using MPN techniques, the resulting regression statistics were $R^2 = 0.999$, Slope = 0.999.

No quality control measures were carried out for enumeration of faecal streptococci (MF) or protozoa (flow cytometry), however, in both cases, experimental controls were built into both protocols, by way of positive and negative standards and spike experiments, used during processing of each sample batch.

6.7 Acquisition and Analysis of Meteorological Data

Hourly values for rainfall, minimum and maximum temperature and MSL pressure were obtained from Met Eireann for the period April 2007 to April 2010. These data were obtained from the closest available synoptic or climatological station to the study areas, as follows;

SA1 (Belmont, Co Offaly); Slieve Bloom Climatological Station (Rainfall, Temperature) located approximately 35 km south of study area; Mullingar Synoptic Station (MSL Pressure) located approximately 50 km southwest of study area. 1362 mm of rainfall were recorded at Slieve Bloom during 2009, resulting in a daily average of 3.7 mm. Average minimum and maximum temperatures were 5°C and 12.7°C.

SA2 (Crossbridge, Co Wicklow); Glenealy Climatological Station (Rainfall, Temperature) located approximately 35 km northeast of study area. This area has the highest average annual precipitation rate, with 1,642 mm recorded during January-December 2009, equating to a 4.5 mm daily average. Average minimum and maximum temperatures over the same period were 7°C and 13°C respectively.

SA3 (Paulstown, Co Kilkenny); Oak Park Synoptic Station (Rainfall, Temperature) located 20 km north of the study area. Approximately 1,124 mm of rainfall was recorded during 2009, averaging 3.1 mm per day or 93.6 mm per month. Average minimum and maximum temperatures during the same period were 6.1°C and 13.5°C.

SA4 (Kilclone, Co Meath); Warrenstown Climatological Station located 6 km west of the study area. Approximately 1,053 mm rainfall was recorded during the 12-month period January – December 2009, averaging 87.8 mm per month and 2.9 mm per day. Average minimum and maximum daily temperatures were 6.3°C and 13.6°C respectively.

SA5 (Birdhill, Co Tipperary); Shannon Synoptic Station located approximately 30 km west of study area. Approximately 1,167 mm rainfall was recorded during the 12-month period January – December 2009, averaging 97.2 mm per month and 3.19 mm per day. Average minimum and maximum daily temperatures during the same period were 7.3°C and 13.9°C respectively.

6.7.1 Rainfall Data

In order to investigate the effect of rainfall (mm) on source susceptibility and contamination probability, rainfall data were summed for the following periods, 24 hours (1-d), 48 hours (2-d), 120

hours (5-d) and 30 days (monthly) preceding all spatial and temporal sampling events. These data were subsequently used to investigate the rapidity of specific source response to contamination and quantify the role played by rainfall in overall groundwater source susceptibility.

Rainfall distribution during the study period were not typically distributed as shown (*Figure 6.20*), with higher than normal rainfall during summer and autumn months.

6.7.2 Temperature Data

The average of daily maximum and minimum daily temperature (° C) were calculated and hereafter referred to as average daily temperature.

Temperature distributions during the study period were found to exhibit a relatively normal distribution, as shown in *Figure 6.20* below.

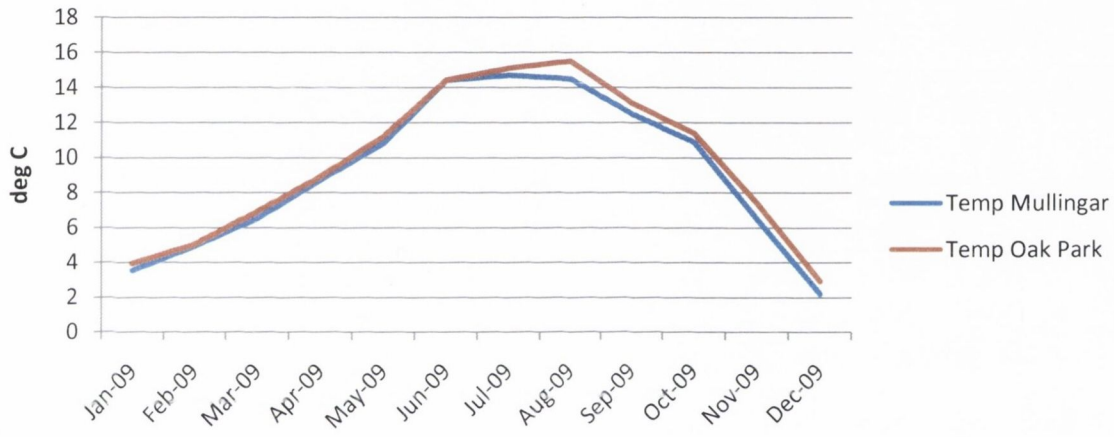


Figure 6.20 Average monthly recorded temperatures from two synoptic stations

6.8 Awareness Questionnaire Development

6.8.1 Aim and Objectives

The purpose of this part of the research was twofold:

1. Quantify the level of awareness among private groundwater users in Ireland (through the development of a generalised scoring methodology), present the findings and infer at a national level.
2. Collection of data to be used in the development of human health risk assessment of private groundwater consumption in Ireland (consumption frequency and volume, household composition, waterborne illness rates and potential waterborne illness rates).

6.8.2 Initial Background Study

A background study was carried out in order to ascertain if an awareness survey of private groundwater users had been previously carried out and if so, could existing protocols or results be

utilised to inform the present study i.e. study hypotheses, questionnaire design and administration. The background study found that no survey of this type had been carried out in Ireland and where groundwater awareness surveys had been carried out elsewhere, they had been carried out in different hydrogeological settings, with different survey objectives (Paul, 2004; Faruque Parvez *et al.,* 2006; Celik & Muhammetoglu, 2008). Therefore survey design, although based upon best available techniques, was undertaken "from scratch". Similarly, with no previous studies of this kind, study hypotheses were based on intuitive probabilities as opposed to available data.

6.8.3 Sample Size Determination

Sampling is the process of unit selection, used to represent a larger population for investigative research. Thompson (1999) suggests that the ability to infer the findings of the research study to the wider population is completely dependent on the adequacy of the sampling process. Sampling and sample size calculation is, therefore an essential component of research and requires careful consideration.

The required sample size depends on two key factors (De Vaus, 2002):

1. The degree of accuracy required for the sample;
2. The extent to which there is variation in the population in regard to the key characteristics of the study

These factors may be influenced by a number of factors, including the purpose of the study, population size, the risk of selecting a "bad" sample, and the allowable sampling error (Miaoulis & Michener, 1976; Israel, 2003).

Typically, surveys of this nature within published literature are based upon 95% levels of confidence and a 5% margin of error (C.I.), therefore, it was decided that these limits would be targeted for this study. As no survey of this type had been previously carried out, there were no existing data with regard to variation within the population in regard to key characteristics (i.e. no previous evidence of highly skewed distributions within this population). Although the researcher had developed hypotheses based on likely (theoretical) outcomes, it was decided that a response distribution (also known as *degree of variability or P)* of 50% would be used for sample size calculation, as the most conservative approach. The more heterogeneous a population, the larger the sample size required to obtain a given level of precision (Isreal, 2003). Conversely, a less variable (more homogeneous) population, results in a smaller necessary sample size. A proportion of 0.5 indicates the maximum variability in a population and therefore its use results in determination of conservative sample sizes i.e. sample size may be larger than if the true variability of the population attributes were known

The total population from which to select a sample for this study is defined as any person over 18 years, permanently residing in a household served by a private groundwater source, equating to approximately 720,000 people in the Irish Republic, as outlined below (*Table 6.4*). It should be noted that this is the total population served by private groundwater sources in Ireland on a daily basis, including those <18 years of age.

Table 6.4 Sources of domestic water supply in Ireland (after CSO, 2007; EPA, 2009)

Source	% (Total Population)	Number Supplied
Surface Water (Public)	73	3,095,089
Groundwater (Public)	9	381,586
SPS	12.9	546,940
PrGWS	4.1	173,833
Other/Unknown	1	42,398

NB: Groundwater (Public) refers to users of public mains and local authority group schemes

There are several approaches to determining the sample size. These include using a census for small populations, imitating a sample size of similar studies, using published tables, and applying formulas to calculate a sample size (Israel, 2003). Using a standard sample size calculation equation (Yamane, 1973), outlined below, which incorporates an assumption of 95% confidence level and P = 0.5, and a total population of approximately 720,000, a sample size of 400 surveys was calculated as being necessary to achieve a 95% level of confidence, with 5% C.I. This sample size calculation conforms to published sample size tables (see Appendix D4).

$$n = N/ (1+N (e)^2) \qquad \textit{Eqn. 6.1}$$

where,

n: Sample size,
N: Population size, and
e: probability of error (0.05)

therefore, $n = 720{,}000/ (1 + 720{,}000(0.05)^2) = 400$

As the larger the sample size, the more representative the research findings are of the population (Polit & Hungler, 1995), it was decided that the calculated necessary sample size (n = 400) would be treated as a target sample size. However, as the overall aim of the fieldwork was to assess and sample 200 private groundwater sources (see *Section 6.4.2*), it was calculated that in the event of a 100% response rate associated with these well users would result in a 6.93% C.I., which did not meet the target C.I. It was therefore decided to increase the number of completed surveys by initiating a number of self-administered (SA) group survey sessions as described in Section 6.8.4 below. Additionally, a supplementary study area was selected to increase the number of researcher administered (RA) surveys completed.

Over the duration of the awareness survey study period, a total of 545 surveys were completed by respondents >18 years of age whose daily drinking water supply derives from a private groundwater source in Ireland. In all, 245 RA surveys were administered, resulting in a 95% confidence level and an associated 6.26% C.I. However, due to differing survey completion modality, which is discussed in greater detail in *Chapter 8*, the total dataset is separated into two distinct datasets for analysis. Therefore, the RA survey (N = 245) results in a 95% level of confidence with 6.3% C.I.; with the SA dataset (N = 300) effecting a 95% level of confidence with 5.6% C.I.

6.8.4 Respondent Selection

A research population may be defined as the complete set of participants who meet the inclusion criteria for a research study (LoBiondo-Wood & Haber, 1998). The sample population for this study was defined as those members of the Irish population who own or are served by a private groundwater source. Any private source was deemed suitable including Bored SPS, HD SPS, Sp Well, and PrGWS. This population was mainly located in rural areas, outside the perimeter of towns and cities served by public/local authority water schemes. As noted above, two different modes of survey completion were used: a) face-to-face interviews or researcher-administered (RA) surveys, and b) group surveys, self-administered by the respondent in the presence of the researcher, referred to as self-administered (SA) surveys. As RA surveys were carried out in concurrence with site susceptibility assessments and water quality analysis, sample selection for these surveys was defined as "any person over 18 years, residing in the household on a permanent basis". In the case of SA surveys, the researcher provided a five minute introduction to the survey group, highlighting the purpose of the study and eligibility for completion of the questionnaire. All respondents were reminded that responses would be treated confidentially, with sequential numbering being used for data recording purposes. Each respondent completed one questionnaire, with the same questionnaire used for both the RA and SA survey modes.

6.8.5 Questionnaire Design

There are three basic designs for a study with more than one measurement occasion: cross-sectional design, longitudinal design, and time-series design. The study was designed as a cross-sectional survey in order to provide a snapshot of the current level of awareness of private groundwater users in Ireland of issues such as groundwater occurrence, groundwater contamination and the potential human health effects of groundwater contamination. Cross-sectional surveys use different units (respondents) at each of the measurement occasions, by drawing a new sample each time. The time intervals may be different between measurement occasions, but they are the same for all units (respondents).

Typically, when using questionnaires, it is difficult or impractical to return to respondents to collect additional data, therefore questionnaires need to be designed in such a way as to collect all the required data in relation to the overall investigation and study objectives. The most important stage is the preparation and design of the questionnaire itself (Cormack, 2000). De Vaus (2002) states there are a number of ways of working out which questions to include:

1. The research problem will affect which concepts need to be measured.
2. The indicators devised for these concepts are important in determining which questions to ask.
3. Hunches/hypotheses about the mechanism by which variables are linked or about factors which might explain relationships will require that certain questions be included.
4. The way data are to be analysed affects what data is needed.
5. The method by which the questionnaire is to be administered affects the type of questions which can be asked.

As no previous study has been carried out to assess the level of awareness of private groundwater users in Ireland, and no broadly similar study had been carried out at a national or international

level, the researcher developed a general set of theoretical hypotheses (below) based on likely outcomes and using the five points outlined (above) and the previously outlined survey objectives, developed a draft questionnaire.

The main objective of the questionnaire used to investigate levels of awareness among Irish groundwater users, as part of this study, (as opposed to the overall survey) was subsequently defined as "the presentation of a range of questions in order to evaluate the present state of factual knowledge of private groundwater users in Ireland of groundwater contamination and the subsequent potential human health impacts". A secondary objective was defined as "the collection of data concerning relevant opinions and patterns of private groundwater users in Ireland to groundwater contamination and the subsequent potential human health impacts". A third objective was defined as "the collation of data and information which may be used in the quantification of the magnitude of risk to health from contamination of private groundwater supplies in Ireland". In order to address the study objectives and theoretical hypotheses, the questionnaire was separated into four distinct sections:

1. Well/Source Awareness
2. Water Quality Testing
3. Groundwater Contamination
4. Human Health

Questionnaire sections were developed individually, in order that all data required could be effectively collected with the minimum of difficultly and also provide the required data i.e. the levels of awareness of groundwater contamination among groundwater users in Ireland. Closed format questions were favoured to minimise the time taken to complete the survey and in doing so, prevent response patterns emerging or respondents becoming disinterested, which could negatively impact the reliability of collected data (De Vaus, 2002). Also, this would increase the response rate and overall reliability of data by making completion as easy as possible for respondents. The use of closed format questions would also make subsequent analysis easier, as categories had already been assigned to responses. When using closed format questions, it was important that an adequate range of response alternatives are provided, and a "don't know" or "undecided" category was included where possible, to avoid artificial creation of opinions or knowledge (Converse & Presser, 1986). Particularly in the case of this study, the "don't know" response alternative was an important inclusion, as the focus of the research was the evaluation of awareness.

Where possible, response categories were limited to Yes/No/Don't know or dichotomous Yes/No responses preceded by a Know/Don't know filter question. This approach left little room for misinterpretation on the respondent's behalf. However, some questions necessitated "interval" responses i.e. age of private source. One Likert scale question response (i.e. an ordinal rating scale) was included, to which respondents indicated their level of satisfaction.

As the questionnaire was to be administered using two different methods, albeit both of which involved the researchers presence, care was taken to minimise bias by excluding overly probing or sensitive questions. The presence of the researcher during questionnaire completion also meant

that an introductory letter and/or completion guidelines did not have to accompany the questionnaire.

Questionnaire sections varied in length depending on the volume of data the researcher needed to collect, with the longest section comprising 13 questions (Questionnaire Section 1) and the shortest section comprising 4 questions (Questionnaire Section 2).

6.8.6 Pilot Study

A preliminary study was carried out after development of the pilot questionnaire had been completed in order to test the questionnaire's adequacy for purpose i.e. specific questions, reliability of responses, etc, and provide estimates of general response rates and times for calculation of necessary sample and subsequent timeframe for study completion.

In all, 27 pilot questionnaires were completed by the researcher as part of the pilot study, over a 4-week period. The questionnaire was administered with well users from Moate Co Westmeath, on a face-to-face interview basis, in the same manner as was proposed for the main study. Upon completion of the questionnaire, respondents were asked for feedback including identification of ambiguities and difficult questions. Based on this and the researcher's own experience of questionnaire completion with respondents, a number of questions were shortened/revised/re-worded and one question was discarded. Pilot study interviews were timed and it was found that the completion time ranged from approximately 9-21 minutes, depending on respondent, with a mean completion time of 13 minutes. Data collected during the pilot study were not included in the final analysis dataset as the data obtained during the pilot study are not comparable to the overall survey (Teijlingen *et al.,* 2001).Respondents taking part in the pilot study were not re-surveyed as knowledge of the previous survey could have introduced bias.

6.8.7 Survey Administration

As previously outlined, two differing survey completion modes were used throughout this study, namely; researcher-administered (RA) completion and self-administered (SA) group completion. In all, 40.2% of completed questionnaires were conducted using the RA completion mode, with the remainder (59.8%) completed using the SA completion mode.

Face-to-face interviews and self-administered surveys in the presence of an interviewer traditionally result in higher response rates than other administration types (Hox & de Leeuw, 1994; de Leeuw & Collins, 1997). De Leeuw & Collins (1997) reviewed response rates from 45 studies using face-to-face, telephone and postal administration methods, concluding that face-to-face interviews had the highest average response rates (70%), followed by telephone surveys (67%) and postal surveys (61%). De Leeuw & Collins (1997) and Dillman (2000) also found however, that more anonymous internet and postal methods may achieve higher response rates for sensitive topics where anonymity is important. Furthermore, face-to-face interviews are the most costly of the survey methods. For these reasons, Dillman (2000) advocates the use of multi-mode methods of administration, as used in this study. However, while this multi-modal strategy may recruit more representative samples it introduces a new problem, with questionnaire responses being subject to modal effects (De Vaus, 2002).

Another potentially important point which should be highlighted is the differing respondent demographic across the two completion modes. RA interviews were carried out with any private groundwater user, over 18 years of age, who was willing to participate in the study. SA surveys were in the most part carried out at a number of undergraduate agricultural institutions throughout Ireland, meaning that although they offered a spatially diverse respondent base, which is preferable, the majority of these respondents were male, between the ages of 20-30 and from an agricultural background. Therefore, this respondent database may not be considered a nationally representative sample, which may be considered a limitation of this database.

6.9 Survey Analysis

Responses were numerically coded and analysed using the Statistical Package for Social Sciences (SPSS, 16.0).

6.9.1 Data Coding

The quantitative analysis of survey data requires that question responses be converted into numeric datapoints. Many variables also require that data be categorised, a process known as *datacoding*. De Vaus (2002) presents a six-step coding protocol, used by the researcher, as follows:

1. Classifying responses
2. Allocating codes to each variable
3. Allocating column numbers to each variable
4. Producing a codebook
5. Checking for coding errors
6. Entering data

Each variable was assigned a code resulting in 67 variable codes from 29 questions. Each variable has at least 2 categories (possible responses) with each respondent belonging to one and only one category. Each category was assigned as simple a numerical code as possible, with dummy variables being created in the case of a number of variables to adhere to a minimum number of categories per variable (2), known as the *multiple dichotomy method* (De Vaus, 2002; Lyberg *et al.,* 1997). An example of dummy variable creation and the multiple dichotomy method is shown below:

Q7. Do you currently possess records of the drilling/well construction process?

Yes	( )
No	( )
Don't know	( )

Variable:	*WellRecAwareness*
Categories:	Not aware – 0
	Aware – 1

Variable:	WellRecYN
Categories:	No – 0
	Yes – 1

The same method was used in the case of questions with multiple parts i.e. each part is broken up into a dichotomous category and subsequently coded. This methodology was used as once variables and their categories had been selected the coding itself is simple. Additionally, no loss of detail is incurred if variables and their categories are carefully written and coded.

Numerical Data

Where a response was already in numerical form i.e. age of respondent, age of groundwater source, depth of groundwater source; it was not considered necessary to convert data into code and so the exact answer was coded and input as a scaled data point. In cases where respondents were asked to select a particular category i.e. average consumption per day, the category was assigned a code representing the category midpoint i.e. 1-2 L day^{-1} was coded as 1.5.

Missing Data

Each variable for each respondent should be assigned a code. In the case of missing data, these were assigned a missing data code and were coded in the same manner as normal, valid responses (De Vaus, 2002; Saldana, 2009). There are a number of reasons why data may be missing from a case; previous questions may exclude a respondent from answering subsequent questions, a respondent may not understand a question or may not know the answer. All missing data were coded with "-1", which was not used for any other variable.

Codebook

As utilised in various other studies and widely regarded as best practice (De Vaus, 2002; McQueen *et al.*, 2008), a codebook was produced to provide a systematic record of all decisions made. The methodology put forward by De Vaus (2002) was followed, with the codebook including all of the following:

- Exact wording and number of question
- Name by which variable is referred to in data entry and analysis software
- Type of data used for each variable (ordinal, nominal, scale)
- First and last row numbers in which variable is located
- Valid codes for each question
- Any special coding instructions

The codebook is presented in Appendix D5.

6.9.2 Coding Error Checks

Coding errors, if undetected, can render valid data and subsequent analysis useless, therefore a system of checks must be put in place to minimize these coding errors. Common types of coding error are miscoding, variable and/or rows moving into adjacent variables or rows and variables and rows being inadvertently switched. The most common form of error is miscoding during data collection, date entry or data coding (De Vaus, 2002). Three coding errors checks were used for this study (Lyberg *et al.*, 1997; Brace *et al.*, 2009):

1. Random sampling and checking of raw data source (completed questionnaire) in both spreadsheets (EXCEL) and analysis software (SPSS 16.0).
2. Valid range checks; where a variable has two valid categories i.e. 0 and 1; cross-tabulations of the variable against any other random variable will locate cases where 0 or 1 have not been input i.e. non-valid range
3. Filter checks; particular questions should only be answered by respondents who provide filter answers to previous questions. Cross-tabulations were used to locate that this was the case.

6.9.3 Data Entry

There are three common methods of data entry (Brace *et al.,* 2009):

1. Manual Entry
2. Scanning
3. Automatic Data Entry

Manual data entry was used for all data in this study, with raw data being entered into Excel 2007 spreadsheets and coded by the researcher.

6.9.4 Data Analysis

The level of measurement of a variable (dichotomous, nominal, ordinal, interval) is fundamental in deciding what analytical methods may be appropriately used (Blalock Jr., 1979; Polit & Hungler, 1999; De Vaus, 2002). Although interval data allow for use of a wider range of statistical techniques, they also necessitate a greater level of data collection, equating to increased questionnaire time and subsequent probability of loss of interest and/or development of response patterns on the respondents behalf. The majority of data collected were ordinal and nominal with subsequent analysis reflecting this. Upon completion of data coding, all coded data were imported from Excel 2007 into SPSS 16.0.

Chapter 7: Groundwater Quality Trends and Patterns

7.1 Introduction

As discussed in *Chapter* 6 (*Section 6.2.1*), four study areas were chosen for inclusion in an integrated spatial ("one-off") and monthly fieldwork programme, including three *High/Extreme* vulnerability study areas (SA1-3) and one low vulnerability study area (SA4). An additional small-scale study area was added later in the study due to the availability of suitable data (SA5). This chapter presents the results of the fieldwork programme. Firstly, an overview of private groundwater sources encountered, assessed and sampled is presented, including typical construction and design specifications, geological setting and hazard source locations and proximity in relation to the private source location and setting. Secondly, the results of chemical, bacteriological and protozoan groundwater analyses are presented and discussed in the context of source type, construction, geological setting and hazard proximity. Finally, the results of a 21-month monthly sampling programme are presented, also in the context of source type, construction, geological setting and hazard proximity, with meteorological data also included and analysed with respect to groundwater quality. All statistical analyses presented in this chapter were carried out in SPSS 16.0, with significance determined by a p-value less than 0.05 as per convention (Agresti, 1996).

A number of photographs have been included in Appendix E1, to aid understanding of terms used throughout the chapter, including well chamber, well type, liner type (material and configuration), flange clearance, wellhead cracking, ground condition, liner cap, wellhead cover and wellhead finish. It is worth noting that typically, it was only possible to assess the survey features of the well, although, GSI well records and the completion of awareness surveys did result in data collection pertaining to subsurface well features, as this dataset was incomplete, it is not discussed here. All statistical tests used in this chapter are bivariate, and are based on Blalock Jr. (1979), as presented in Appendix F2.

7.2 Design and Construction of Source Types

Altogether, 211 private groundwater sources were assessed and sampled on a "one-off" (unrepeated) basis, 80.4% (n = 169) of which were small-diameter boreholes, with the remaining 19.6% (n = 42) comprising large-diameter, shallow hand-dug wells.

7.2.1 Boreholes

The overall mean measured depth of boreholes was found to be 44.7 m (from ground surface), with minimum and maximum recorded depths of 8 m and 150 m, respectively. A mean source age of 18 years was recorded, with minimum and maximum source ages of 2 months and 58 years respectively. Overall, these well types comprised a domestic water supply for an average of 3.5 consumers per source, or 588 consumers in total, with a mean of 1.04 households supplied per source and a maximum of 3 households supplied.

Just under 74% of boreholes (n = 124) had a well chamber present (see *Section 3.7*), with a significant difference found between the well liner finish (*Figure 7.1*) and chamber presence (χ^2 (2) = 80.664, p <0.001), as shown in *Table 7.1*.

Table 7.1 Chamber presence and well liner finish (Boreholes; n = 168)

Well liner finish	Chamber presence		
	Yes	No	Total
Above ground level	20 (46.5)	23 (53.5)	43
At ground level	4 (18.2)	18 (81.8)	22
Below ground level	100 (97.1)	3 (2.9)	103

(Percentages in parentheses)

Approximately 87% (n = 146) of boreholes were constructed using a combination of steel well liner and concrete wellhead (and where applicable, well chamber), with other construction materials being steel (6%), PVC (5.4%) and concrete (1.8%). Where constructed, well chambers were typically constructed using cemented 100 mm solid concrete blocks (84.8%, n = 106) or solid mass concrete (8%, n = 10). Of the remainder, 4.8% were moulded PVC and 2.4% were comprised of stone. A large range of chamber areas and volumes were recorded, with a mean chamber volume of 0.55 m^3 (range 0.04-8.00 m^3), and a mean chamber depth of 0.45 m. Above ground chambers, where present, were not found to have significantly larger volumes (1.09 m^3 compared with 0.42 m^3). Likewise, chamber depths were not found to be significantly different between over ground and underground chambers. An inspection of chamber floors (where applicable) for an appropriate concrete base, with the floor level falling towards a chamber drain found that only 8.8% of well chambers were of an appropriate standard (n = 11) to ensure chamber ponding did not occur.

Typically, steel was the most commonly used liner material, and was recorded at 96.4% of boreholes (n = 162). Other liner materials were PVC (2.4%, n = 4) and concrete (1.2%, n = 2). Boreholes with concrete liners were shallow (<10 m depth) in both cases. Typical liner diameters were 145-230 mm, (median = 170 mm). There was no significant difference between steel and PVC liner diameters, however, a significant difference was found when comparing concrete liners and both steel and PVC liners, with both concrete lined boreholes having inner diameters of 1000 mm. Although both of these sources were bored (8 m and 9.2 m), they were found to have more in common with traditional shallow, hand-dug wells. Approximately 34% of boreholes had a liner cap in place (n = 57), which were composed of a variety of materials including steel (35.1%), PVC/HDPE (28.1%), and concrete (5.1%). The remaining (31.6%) installed well caps were classified as being composed of "other materials", including polystyrene, fibreglass and various plastics.

As set out by SEPA (2009) and SEPA (2010), a minimum 300 mm clearance between the liner flange and chamber floor (or pumphouse floor) is advised, to ensure that surface water ponding and/or runoff, does not ingress the borehole. It was found however that only 3.6% of wells conformed to this 300 mm clearance (n = 6), with a mean flange to concrete clearance of 89.8 mm (n = 165), ranging from 0-520 mm. As shown in *Figure 7.1*, all "at ground level" and all but one "below ground level" wells had liner clearances of <300 mm.

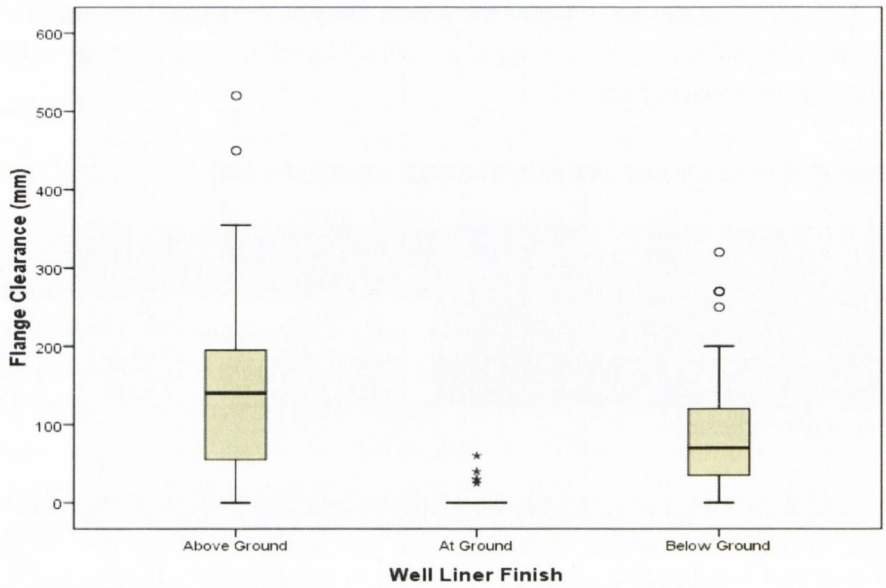


Figure 7.1 Summary results showing average (centre line) 25-75% (box) and minimum and maximum (error bars) of flange clearance distances (Boreholes)

Just under 76% of boreholes (n = 122) had a wellhead cover in place. A significant difference was found between wells without an adequate wellhead cover and well liner finish ($\chi^2(2) = 57.23$, $p < 0.001$) i.e. uncovered wells tended to have decreased or absent well liner clearance. Overall, 93% (n = 96) of underground wellheads, 45.4% (n = 10) of ground level wellheads and 37.2% (n = 16) of above ground level wellheads were covered. As with liner caps, wellhead covers were composed of a variety of materials, including steel (53.3%), concrete (24.6%) and PVC (9.8%).

In order that the influence of an appropriately designed and constructed well chamber could be investigated, a number of chamber characteristics were recorded, including chamber ponding, evidence of cracking, waterproofing of chamber and ingress of surface water runoff and/or shallow groundwater into the well chamber or borehole itself. In all, 23.8% of boreholes had either chamber flooding or visible surface water ponding within 10 m of the wellhead, with chamber ponding ranging from 1 – 50 mm depth. No significant difference was found between the well liner finish and the presence of visible ponding. Similarly, there was no significant mean difference between the presence of ponding and flange clearance. Approximately 41% (n = 69) of boreholes had evidence of cracking either in the well chamber, or where no chamber was present, in the concrete surface pad surrounding the wellhead. Howard *et al.* (2003) noted that cracking at or directly adjacent to wellheads was found to be a risk factor with regard to microbial contamination of communal wells in Uganda. No significant difference was found between the presence of cracking and the liner finish (i.e. above ground level, at ground level, below ground level). The presence of a well chamber was found to be significantly associated with cracking ($\chi^2(1) = 4.86$, $p = 0.033$), with those wells having well chambers present found to be more likely to exhibit cracking around the wellhead (Appendix E1). It was found however, that the chamber construction material was not significantly associated with cracking nor was chamber area or chamber depth. Those wells which were appropriately covered were found to be significantly more likely to exhibit cracking, than those without ($\chi^2(1) = 4.464$, $p = 0.037$). Source age was not found to be a significant factor with regard to cracking,

however, both source depth (t = 2.01, p = 0.047) and liner flange clearance (t = 1.984, p = 0.049) were found to be related to the instance of cracking. It was found that wells with evident cracking had a mean flange clearance of approximately 71 mm, whereas those wells without evident cracking had a mean flange clearance of 99 mm. This would suggest that where ingress to the wellhead was possible as a result of cracking, there was also a lower level of protection by flange clearance. Wells with visible cracking were found to have a mean depth of approximately 38 m, as opposed to 49 m for those without evident cracking.

Overall, 69% of boreholes (n = 116) were adjudged to be inadequately waterproofed, as determined by visible evidence of ponding within the chamber or directly adjacent to the wellhead (10 m radius), or wetting/dampness within the chamber or well liner. Further analysis found that wellhead finish exhibited a statistically significant relationship with waterproofing ($\chi^2(2) = 7.234$, $p = 0.027$), with those wellheads finished above ground level displaying a higher level of appropriate waterproofing than either at ground level or below ground level wellheads (*Table 7.2*).

Table 7.2 Wellhead finish and associated waterproofing (Boreholes)

Waterproof	Wellhead Finish			
	Above Ground	At Ground	Below Ground	Total
Yes	20	4	28	52
No	23	18	75	116
Total	43	22	103	168

The presence of a well chamber, the well chamber construction material, liner material and the presence of an adequate wellhead cover were all found to have no significant relationship with waterproofing. However, a highly significant relationship was found between the presence of a liner cap and waterproofing (χ^2 (1) = 15.426, $p < 0.001$), with 50.8% of those boreholes with a liner cap being adjudged to be adequately waterproofed (n = 57), whereas 79% of those without a liner cap were not appropriately waterproofed. The source age and depth, chamber area and depth, liner diameter, and sample watertable depth were not found to have any relationship with waterproofing of the wellhead area, however, the liner flange clearance was found to be significantly associated (t = -3.197, p = 0.002), with those wells adjudged to be inadequately waterproofed exhibiting a lower mean flange clearance (75 mm) as opposed to those with adequate waterproofing in place (121 mm). Overall, 76% (n = 128) of boreholes were fitted with a submersible well pump, with the remainder using surface pumps.

The final phase of the wellhead assessment and survey was an assessment of the overall ground condition adjacent to the wellhead and on-site in general, in addition to the presence of a secure boundary and, where present, the boundary area. It was found in the case of boreholes that 26.2% (n = 44) had a secure boundary present, with these boundaries having a mean area of 10.03 m^2 and a range of 1.21-63.0 m^2. A five-point classification system was then developed and used to qualify the overall ground condition within a 10 m radius of the wellhead and the site on which the well was located, with classification ranging from good to poor. Poor ground condition was noted where surface ponding, detritus and/or animal faeces were evident and there was little or no evidence of general maintenance. As shown (*Figures 7.2 & 7.3*), medium to good ground condition was typically associated with both 10m radius and general site conditions, with both showing similar distributions.

Further analysis confirmed a significant rank order correlation between the two parameters ($r = 0.721$, $p < 0.001$) i.e. where the 10 m well radius ground condition was adjudged to be "good", typically, so to was the general site condition.

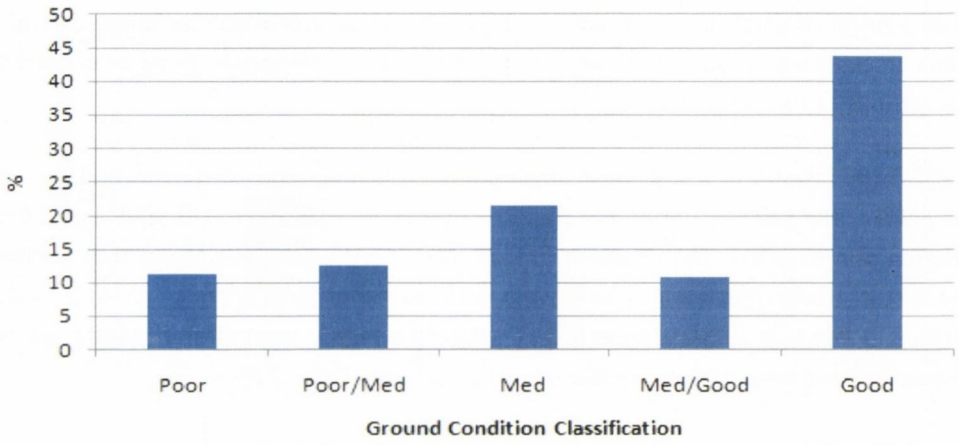


Figure 7.2 Wellhead (Radius 10 m) Ground Condition Classification (Boreholes)

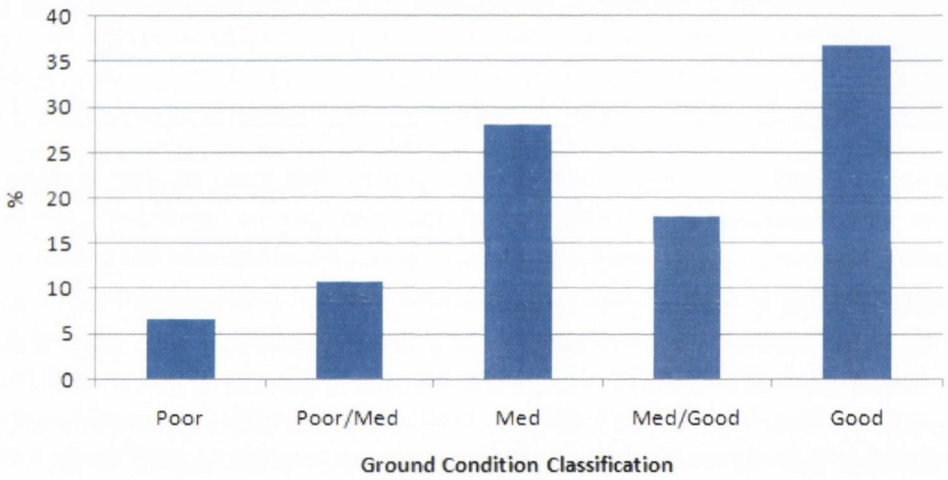


Figure 7.3 General On-Site Ground Condition Classification (Boreholes)

7.2.2 Hand-dug Wells

The mean depth of hand-dug wells was found to be 4.5 m (from ground surface), with minimum and maximum recorded depths of 0.6 m and 9 m, respectively. Well depths could not be measured for 24.4% (n = 10) of hand-dug wells due to an inability to access the well (i.e. sealed well cover/cap). A mean source age of 49.6 years was recorded, with minimum and maximum source ages of 4 years and >300 years respectively. Only 4.8% (n = 2) of hand-dug wells were found to be <10 years, which was expected. It is important to note that well ages were all self-reported by the well users and therefore approximate. Overall, hand-dug wells assessed as part of this project supplied a total of 127 consumers, resulting in a mean of 3.17 consumers per household and a range of 1-20 consumers/source.

Just under 88% (n = 36) of hand-dug wells had no chamber present, with the majority of hand-dug wells assessed simply constructed, with little or no wellhead finish. This was duly reflected when the liner and wellhead finish were considered, with the majority of hand-dug wells finished at ground level (75.6%), with only 17.1% and 7.3% finished below and above ground level, respectively. It is therefore considered that the ingress of surface runoff is more likely to occur at these well types. Well chambers, where present (n = 10), were primarily constructed of concrete, with 100 mm concrete block-work used in 70% (n = 7) of these cases, and the remainder (n = 3) comprising solid mass concrete. The mean chamber volume was found to be 0.96 m^3, with a range of 0.17-4.0 m^3, and a mean chamber depth of 0.25 m. Therefore, where present, well chambers associated with hand-dug wells were typically larger than those associated with boreholes, which was due to the greater liner diameters associated with hand-dug wells. Chamber floor inspections found that none of the well chambers associated with hand-dug wells were appropriately constructed, with none offering adequate drainage.

Overall, 85.3% (n = 35) of hand-dug wells were constructed wholly of concrete with 9.8% (n = 4) using stone as the primary construction material, and 4.8% (n = 2) constructed using a steel liner and concrete surround/wellhead. Concrete was the most commonly used liner material in the case of hand-dug wells, with interlocking 1000 mm deep concrete liners typically used. Concrete liners were used to line 80.5% of assessed hand-dug wells (n = 33), with moulded concrete used in 4.9% of cases (n = 2). Just under 10% (n = 4) of assessed hand-dug wells had no manufactured liner in place, but were constructed in the traditional manner, using un-cemented stone lining. The remaining 5% of hand-dug wells (n = 2) were lined with steel. Typically, standard concrete liners were 1000 mm in diameter, which is reflected in the mean overall hand-dug well diameter of 1,067 mm, with a range of 150-2,750 mm. When two outliers (stone lined wells of 2 m and 2.75 m diameter) were removed, a mean hand-dug well diameter of 1,023 mm resulted. Due to the typically large diameters of hand-dug well liners, and the subsequent unavailability of commercially manufactured liner caps, only 4.8% (n = 2) of hand-dug wells had a liner cap in place, one of which was steel, and the other being PVC.

Approximately 70% of hand-dug wells (n = 29), had no liner flange clearance, and were therefore considered unprotected from surface water ingress (Appendix E1). Of the remainder (n = 12), the mean flange clearance was found to be 99 mm above ground/chamber floor level, which is significantly less than the aforementioned SEPA (2010) guideline minimum of 300 mm. Overall, the range of flange clearance was 0-1,250 mm, with only 14.6% (n = 6) of hand-dug wells exhibiting >300 mm clearance. As shown in *Figure 7.4*, there was a significant mean difference found between wellhead finish and liner flange clearance ($F(2) = 9.187$, $p = 0.001$), with those hand-dug wells finished above ground displaying a mean flange clearance of 371.4 mm, while those hand-dug wells finished at ground level and below ground level had mean liner flange clearances of 34.5 and 133.3 mm, respectively. As previously shown in *Figure 7.1*, a similar pattern was evidenced among boreholes.

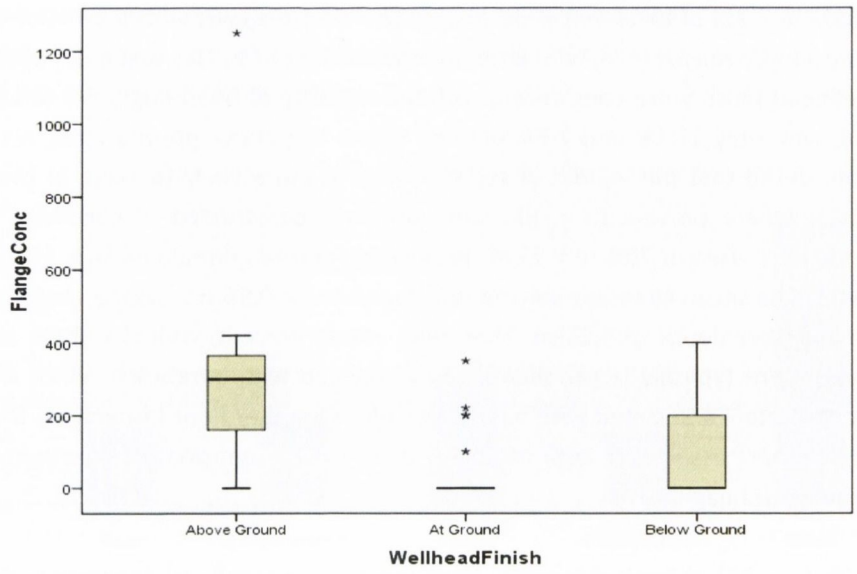


Figure 7.4 Summary results showing average (centre line) 25-75% (box) and minimum and maximum (error bars) of flange clearance distances (Hand-dug wells)

Approximately 90% of hand-dug wells (n = 37) had a wellhead cover, with the most commonly used cover materials being concrete (62.2%) and steel (24.3%). The remaining covers were classified as being comprised of "other materials" and included wood, PVC and polystyrene. The high proportion of covered wellheads associated with assessed hand-dug wells was likely due to the aforementioned unavailability of large diameter concrete liner caps. No significant relationship was found between wellhead finish and wellhead cover, with findings presented in *Table 7.3*.

Table 7.3 Wellhead finish and associated wellhead cover (Hand-dug wells)

Wellhead Cover	Wellhead Finish			
	Above Ground	At Ground	Below Ground	Total
Yes	6	28	3	37
No	1	3	0	4
Total	7	31	3	41

The wellhead survey carried out at hand-dug wells revealed that well ponding (chamber and/or wellhead radius) was observed at 24.4% (n = 10) of assessed hand-dug wells, which was comparable to the rate of ponding observed at boreholes. Evidence of ponding and the presence of a wellhead cover, liner cap, or well chamber, were not significantly related.

An investigation was carried out at all hand-dug wells of cracking of the well chamber (where present), walls and floor, and where no chamber was present, of the concrete base surrounding the wellhead. In addition, the visible interior surface of the concrete liner was also examined where possible. Approximately 63% (n = 26) of assessed hand-dug wells were recorded as having visible cracking evident. A chi-square test of independence suggested no significant relationship between the presence of cracking and the overall wellhead finish (*Table 7.4*), nor was any significant association found between chamber presence and the presence of cracking (as for boreholes). However, an association was found at a 90% level between chamber presence and cracking (χ^2 (1) =

4.626, p <0.051), with 69.4% of those hand-dug wells without a well chamber (n = 36) having visible cracking, while this figure was 20% (n = 5) for those wells with a well chamber.

Table 7.4 Wellhead finish and associated wellhead cracking (Hand-dug wells)

Wellhead Finish	Visible Cracking		
	Yes	No	Total
Above ground	5	2	7
At ground level	19	12	31
Below ground	2	1	3

As was the case for boreholes, it was found that neither well chamber construction material nor chamber dimensions were significantly associated with the presence of cracking. Conversely, while the presence of an appropriate well cover was found to be associated with cracking among assessed boreholes, no similar relationship was found among hand-dug wells. No mean difference was found between source age and visible cracking, however, both source depth and liner flange clearance were found to have significant mean differences with regard to cracking, although neither at a 95% level. Wells with cracking in evidence were found to be of greater mean depth and have a greater liner flange clearance than those without visible cracking.

Just under 83% of hand-dug wells (n = 34) had visible evidence of some surface ingress and so were adjudged to be inadequately waterproofed. As shown in *Table 7.5*, approximately 90% (28/31) of hand-dug wells with a ground level wellhead finish were adjudged to be inadequately waterproofed, with this association found to be statistically significant ($\chi^2(2) = 7.063$, p =0.029), as was also the case for boreholes. In the case of hand-dug wells, which are generally considered to be more susceptible to contamination due to their reduced depth and construction methods, appropriate waterproofing is of paramount importance and therefore, this finding represents a concern.

Table 7.5 Wellhead finish and associated waterproofing (Hand-dug wells)

Waterproof	Wellhead Finish			
	Above Ground	At Ground	Below Ground	Total
Yes	2	3	2	7
No	5	28	1	34
Total	7	31	3	41

The presence of a wellhead cover or liner cap was not found to be significantly associated with waterproofing, nor was the presence of a well chamber or chamber composition. Liner composition was however found to be significantly associated (χ^2 (3) = 11.036, p =0.012), with all (n = 4) traditionally stone-lined wells found to be inadequately waterproofed. As shown (*Table 7.6)*, a 100% rate of inadequate waterproofing was also observed in those wells lined using poured concrete, although as for stone lined wells, this subset was very small (n = 2) and therefore inference cannot be made. Almost 85% of those hand-dug wells lined using concrete liners were adjudged to be inadequately waterproofed.

Table 7.6 Well liner material and associated waterproofing (Hand-dug wells)

Waterproof	Liner Material				
	Precast Concrete	Stone	Steel	Poured Concrete	Total
Yes	5	0	2	0	7
No	28	4	0	2	34
Total	33	4	2	0	41

The source age, source depth, liner diameter and liner flange clearance were all found to have no significant relationship with waterproofing, however, the watertable at the time of assessment was found to be significantly associated with waterproofing ($t = -2.116$, $p = 0.046$), with those wells adjudged to be inadequately waterproofed found to have a higher mean measured watertable (1.17 m from ground level) compared to those with adequate waterproofing in place (2.81 m).

Overall, 39% of hand-dug wells (n = 16) had a submersible pump installed, with surface pumps being used at 51.2% of wells (n = 21). The remaining four assessed hand-dug wells (spring wells) (9.8%), had no pump in place, with gravity feed being used. A relationship was found as expected between pump type and source depth, with submersible pumps being typically used in deeper hand-dug wells. This relationship was not found to be statistically significant at a 95% level ($F(3) = 3.107$, $p = 0.056$).

On the whole, 31.7% (n = 13) of assessed hand-dug wells were constructed within a secure boundary, with a mean boundary area of 8 m^2. Secure boundary areas, where present, were in the range 1-25 m^2. Using the same five-point classification system to quantify ground condition within a 10 m wellhead radius and generally on-site as before, it was found that although the majority were classified as being good, a higher proportion of poor ground condition was recorded pertaining to hand-dug wells than that observed in boreholes (*Figures 7.5 & 7.6*). As observed for boreholes, a high rank order correlation also resulted between the wellhead radius ground condition and on-site ground condition from site assessment of hand-dug wells ($r = 0.867$, $p < 0.001$).

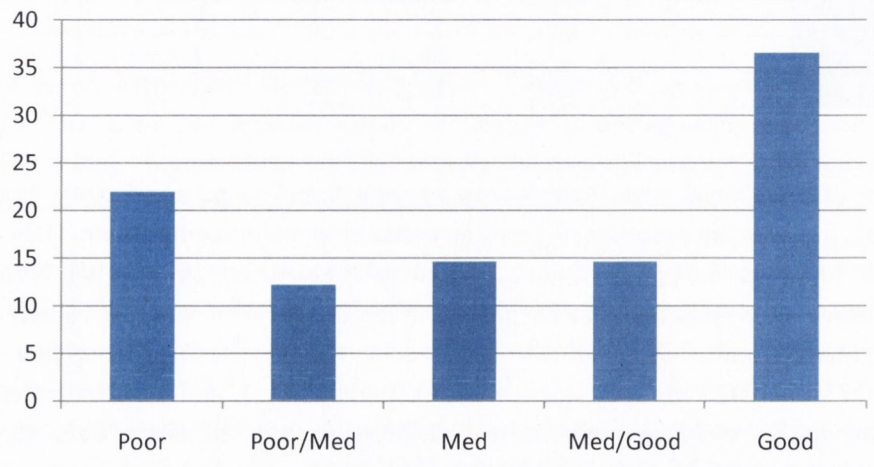


Figure 7.5 Wellhead Radius (10 m) Ground Condition Classification (Hand-dug wells)

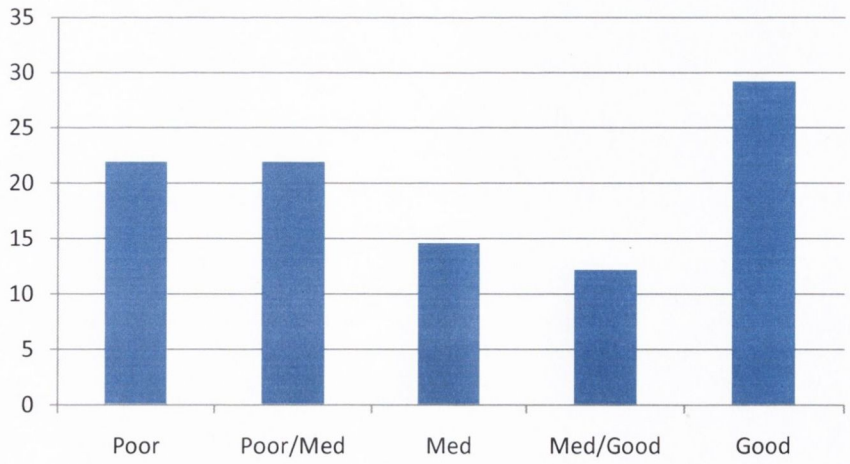


Figure 7.6 On-Site Ground Condition Classification (Hand-dug wells)

7.3 Effect of geological setting on well design and construction

A chi-square test of independence suggested a significant association between well type and the generalised local bedrock type (χ^2 (5) = 12.063, p = 0.034). As shown in *Table 7.7*, both local limestone and granite bedrocks were associated with relatively low proportions of hand-dug wells, with a slightly higher proportion observed in sandstone areas. Significantly higher proportions of hand-dug wells were recorded in areas with undifferentiated Namurian, shale and volcanic bedrocks, however, these data should be treated with caution due to small sample sizes.

Table 7.7 Bedrock types associated with source types

Local Bedrock Type	Supply Type	
	Hand-dug	Bored
Limestone	13 (16%)	68 (84%)
Sandstone	10 (20.4%)	39 (79.6%)
Granite	6 (12.2%)	43 (87.8%)
Undifferentiated Namurian	5 (41.6%)	7 (58.4%)
Shale	5 (33.3%)	10 (66.7%)
Silurian Volcanic	2 (66.6%)	1 (33.3%)

No significant association was implied by statistical analysis of the dominant local subsoil type or the subsoil thickness range and the associated supply type. Further analysis, however, found a significant association between subsoil thickness >3 m and supply type (χ^2 (1) = 13.243, p <0.001), with approximately 45% (n = 28) of private sources in areas with >3 m subsoil (regardless of subsoil type) found to be hand-dug wells, whereas, this figure was 10.9% (n = 13) in areas with <3 m of overlying subsoils. This was adjudged to be due to a statistically significant difference between the generalised bedrock and associated mean sample watertable (F (5) = 2.448, p = 0.036). For example, wells located in granite had a mean sample watertable of 3.27 m (n = 40), while wells located in shales had a mean sample watertable of 9.38 m (n = 13) (see Appendix E2). No statistical association

was found with regard to supply type and either the local groundwater vulnerability and aquifer importance classification.

Due to their inherently different design and construction configurations, hand-dug wells and boreholes were analysed separately with regard to geological/hydrogeological influence upon overall well design. It was found that local bedrock type had no statistically significant influence upon overall well design i.e. source depth, wellhead configuration, source protection, for either bored or hand-dug wells. Likewise, local bedrock was not found to have any relationship with the overall wellhead condition/maintenance i.e. cracking, waterproofing, well ponding.

In the case of hand-dug wells, the local dominant subsoil type was not found to have any significant influence on recorded well design or construction parameters. However, a relationship was found to exist between dominant local subsoil type and the presence of surface ponding directly adjacent to the wellhead. Approximately 45% of hand-dug wells associated with Namurian tills (n = 11) were found to have visible wellhead ponding, compared with 24.3% recorded among all hand-dug wells (n = 41). Statistical analysis found no existing relationship between soil type and either bored well design or wellhead condition. Likewise, no significant relationships were found to exist between groundwater vulnerability category and hand-dug or bored wellhead design, construction or condition. The aquifer importance category was not found to have any influence upon hand-dug well design or condition. In the case of boreholes, those wells constructed in regionally important karstic aquifers (Rk) were found to be significantly shallower than wells associated with other aquifer importance categories. This mean difference was found to be significant ($F(4) = 3.794$, $p = 0.007$). As outlined in Chapter 2 (*Section 2.5.2*) limestones typically comprise Ireland's most productive aquifers and hence shallower wells will normally be adequate to achieve the required yield, therefore, this result was expected.

7.4 Groundwater Hazard Sources

As part of the overall site and wellhead susceptibility assessment, potential contamination sources were recorded (setback distance and gradient), including on-site wastewater treatment, grazing animals, point agricultural sources, roads and surface water sources, in addition to adjacent operational and/or abandoned wells, landfill sites and other potential sources. The following section outlines these findings, firstly with an overview on all study areas, followed by a more detailed area-by-area investigation.

7.4.1 On-Site Wastewater Treatment

Overall, 95.2% (n =199) of wells had ≥1 septic tank in operation within 100 m, with a median septic tank setback distance of 48 m (mean 77.6 m) and a range of 14.6-100 m. The mean septic tank setback distance associated with boreholes was found to be 77.1 m, compared to 79.5 m among hand dug wells. A median of 1 septic tank within a 100 m radius (mean 1.54) and a range of 1-5 were found within the remaining 95% (*Table 7.8*). Again, no mean difference was found to exist with regard to well type and number of septic tanks occurring within a 100 m radius of the wellhead.

Table 7.8 Number of septic tanks systems within a 100 m radius of assessed wells

Septic Tank Number (<100m)	Frequency	Percent
0	10	4.8
1	110	52.6
2	65	31.1
3	17	8.1
4	4	1.9
5	3	1.4

As shown in *Table 7.9*, over 67% of septic tanks were found to be down gradient of the wellhead, as recommended (EPA, 2009), however, over 18% of wells had a septic tank operating up gradient of the wellhead within 100 m, which is cause for concern. Overall, wells located down gradient of a septic tank had a mean setback distance of 57.7 m, while wells located "on-gradient" and up gradient of septic tanks had mean setback distances of 44.5 m and 57.5 m, respectively. A Bonferroni post-hoc multiple comparisons test found no significant mean difference between septic tank setback gradient and setback distance (Appendix E3), which also presents a concern, as wells located down gradient of a septic tank should have a greater setback distance, where possible.

Table 7.9 Setback gradient of septic tanks systems within a 100 m radius of assessed wells (n= 202)

Septic Tank Gradient	Frequency	Percent
Down gradient of wellhead	136	67.3
Level with wellhead	29	14.3
Up gradient of wellhead	37	18.3

Hand-dug wells were more frequently associated with up gradient septic tanks within 100 m of the wellhead, with almost 31% (12/39) of hand-dug wells having a septic tank operating up gradient, while the corresponding figure was 15.3% (25/163) in the case of boreholes, although this was not significant at a 95% level (χ^2 (2) = 5.827, p = 0.054).Hand-dug wells were, however, associated with a significantly greater mean setback distance of 150.5 m, compared with a 82.4 m setback distance associated with boreholes (F = 7.179, p = 0.043).

As set out in the most recent "*Wastewater Treatment and Disposal Systems Serving Single Houses*" Code of Practice (EPA, 2010d), the recommended minimum septic tank setback distances from wells and springs for down gradient domestic wells or where flow direction is unknown is 30 m. This study employed both the recommended 30 m setback distance and a 100 m setback distance, referred to by Deakin (2000) as a "Generalised Inner Protection Zone", which corresponds with an approximate 100 day time of travel. Overall, 20.8% (n = 44) of wells had a septic tank within a 30m radius, with figures of 11.9% and 23% associated with hand-dug wells and boreholes, respectively. Just under 30% (n = 11) of wells located down-gradient of a septic tank had a setback distance <30 m, with figures of 34.5% (n = 10) and 16.9% (n = 23) associated with wells located "on-gradient" and up-gradient of septic tanks, respectively.

Domestic wastewater treatment systems excluding conventional septic tank systems (i.e. package plants such as bioCycle™ etc.), were in operation within a 100 m radius of 11.5% (n = 24) of assessed wells, with a median setback distance of 40.5 m, and a setback range of 14 - 100 m. The average

mean setback distances were found to be 38.1 m and 80.3 m for boreholes and hand-dug wells, respectively, with this mean difference found to be significant (t = 3.72, p = 0.001). As shown in *Table 7.10*, approximately 54% of treatment systems were located down gradient of the wellhead, with the remainder being located either level with or up gradient of the well. No statistical significance was found with regard to the well type and the treatment system gradient.

Table 7.10 Number of onsite wastewater systems (excl septic tank systems) within a 100 m radius of assessed wells

Treatment system gradient	Frequency	Percent
Down gradient of wellhead	13	54.1
Level with wellhead	3	12.5
Up gradient of wellhead	8	33.3

7.4.2 Agricultural and Commercial Activities

An overall assessment of predominant land-use in the immediate vicinity of the wellhead (100 m) found that 17.7% (n = 37) of wells were located in predominantly agricultural areas, 16.3% (n = 34) were located in predominantly residential sites, with a majority of 63.6% (n = 133) situated on mixed land-use sites i.e. with both residential and agricultural activities occurring within 100 m. Of the remainder, 1.4% (n = 3) were located on commercial sites and 1% (n = 2) were located on sites dominated by forestry.

All of the assessed wells (n = 211) had grazing animals recorded within a 100 m radius of the wellhead, which was expected due to the predominantly rural setting associated with private groundwater sources in Ireland. The median grazing setback distance was found to be 15 m (mean 20.7 m), with mixed grazing the most commonly encountered grazing type (*Table 7.11*). No significant mean difference was found in regard to grazing setback distance or grazing type and the well type. Overall, the majority of assessed wells (n = 109) were on a level gradient with the closest grazing animals (*Table 7.12*), with no significant difference found between grazing animal gradient and well type.

Table 7.11 Grazing animal type within a 100 m radius of assessed wells

Animal Type	Frequency	Percent
Beef Cattle	71	34
Mixed (Sheep & Beef)	128	61.2
Dairy Cattle	6	2.9
Horses	4	1.9

Table 7.12 Setback gradient of grazing animals within a 100 m radius of assessed wells

Grazing animal gradient	Frequency	Percent
Down gradient of wellhead	47	22.5
Level with wellhead	109	52.2
Up gradient of wellhead	53	25.4

An assessment of point agricultural sources i.e. animal housing, animal feed/manure storage, farmyards, in the vicinity of assessed wells found that approximately 27% of wells (n= 57) had an operational farmyard in the vicinity of the wellhead, with a further 6.7% (n = 14) and 1.9% (n = 4) having animal housing and silage pits respectively within a 100 m radius of the wellhead. The median setback distance of point agricultural activities from assessed wellheads was 45.1 m, with no significant mean difference found pertaining to well type. Additionally, 7.7% (n = 16) and 5.3% (n = 11) of assessed wells had a domestic garage and commercial activity, such as a commercial garage, greenhouse or business, situated within a 100 m radius of the wellhead. In all, 12.4% (n = 26) of agricultural or commercial activities were situated up gradient of the wellhead, with 17.7% (n = 37) and 18.7% (n = 39) situated level with and up gradient of assessed wellheads, respectively.

A one-way ANOVA test found that there was, as expected, a strong relationship between wells categorised as being located in agricultural areas and grazing animal setback distances ($F(4) = 9.363$, $p < 0.001$), with a Bonferroni multiple comparisons test finding that these wells were, on average, 11.5m closer to grazing animals than mixed land-use wells and 20.5 m closer than residential wells. A similar pattern was found in relation to agricultural activities ($F(4) = 9.304$, $p < 0.001$).

7.4.3 Roads

Approximately 98% (n = 206) of wells were located within 100 m of a roadway, with a median setback distance of 25 m (mean 34.4 m; range of 2-301 m). Boreholes had a mean road setback of 30 m, while hand-dug wells had a mean setback of 53.2 m, with this mean difference found to be statistically significant ($t = 2.03$, $p = 0.049$).

Overall, the most frequently recorded road type was tertiary/link, as expected due to the typically rural settings (*Table 7.13*), with no significant association found between road type and well type. On the whole, road setback gradients were found to be relatively evenly distributed, with 40.2% (n = 84), 23.4% (n= 49) and 36.4% (n = 76) of wellheads located down gradient of, level with and up gradient of adjacent roadways, respectively.

Table 7.13 Road type within a 100 m radius of assessed wells

Road Type	Frequency	Percent
Primary	1	0.5
Secondary	70	33.5
Tertiary/Link	138	66

7.4.4 Surface Water Features

Approximately 30% (n = 63) and 29% (n = 61) of wells had rivers and surface water drainage courses in the general vicinity, respectively, with a median setback distance of 100.1 m (range 1-500 m). Additionally, 7.1% (n = 15) had a surface spring within 100 m of the wellhead. In all, 50.7% of assessed wells (n = 106) had a surface water feature occurring within 100 m of the wellhead. Further analysis found that that the occurrence of surface water features within 100 m of the wellhead were significantly associated with well type ($\chi^2(1) = 8.174$, $p = 0.005$), with 70.7% of hand-dug wells (n =

29) having surface water within 100 m, whereas this figure was 45.8% (n = 77) in the case of boreholes. This finding was expected as high watertables and surface springs are typically associated with adjacent surface water features.

7.4.5 Existing Wells

Both operational and abandoned wells within 100 m of each wellhead were surveyed. Overall, 17.2% (n = 36) of wells had a working well within 100 m (range 5-100 m), and 13.9% (n = 29) of wells were situated within a 100 m radius of an abandoned wells, over a similar setback range. No significant relationship was found pertaining to the occurrence of either an operational or abandoned well within the aforementioned radius and the well type.

7.4.6 Potential Drainage Indicators

The assessment of overall drainage in the vicinity of the wellhead was undertaken through the identification of vegetation indicators of poor drainage including *Phragmites* and mosses. By necessity, this was done visually, with extensive volumes of visible of *Phragmites* and/or moss within a 100 m radius of the wellhead categorised as "potential drainage indicators present". This approach has previously been employed by Lee (1999). Approximately 30% of wells (n = 63) had vegetation indicators of poor drainage occurring within a 100 m radius. No association was found between the presence of a vegetation indicator and well type, however, significant associations were found between the presence of a vegetation indicator and on-site surface water ponding ($\chi^2(1) = 17.764$, $p < 0.001$), land-use category ($\chi^2(4) = 20.906$, $p < 0.001$) and on-site ground condition ($\chi^2(4) = 38.559$, $p < 0.001$). Additionally, a relationship between groundwater vulnerability category and the presence of vegetation indicators was borne out by statistical analysis, albeit contrary to expectation. As shown (*Table 7.14*), *High* and *Extreme* vulnerability areas had increased evidence of potentially poor drainage than the *Low* vulnerability area (χ^2 (4) = 23.537, $p < 0.001$), which was unexpected due to the low permeability subsoils prevalent in *Low* vulnerability areas.

Table 7.14 Presence of potential indicators of poor drainage and groundwater vulnerability

Vulnerability	Vegetation Indicators Present		
	Yes	No	Total
Low	10 (20.4)	39 (79.6)	49
Moderate	0 (0)	4 (100)	4
High	17 (19.5)	70 (80.5)	87
Extreme	24 (48)	26 (52)	50
Extreme (surface)	12 (57.1)	9 (42.9)	21

(percentages in parentheses)

7.5 Groundwater Hazard Sources in individual Study Areas

7.5.1 Study Area 1: Belmont, Co Offaly

The primary method of on-site wastewater treatment utilised in SA1 was conventional septic tank systems, with 98% (n = 49) of wells having at least one septic tank situated within a 100 m wellhead radius (*Table 7.15*).

Table 7.15 Study Area 1: Number of septic tank systems within a 100 m radius of assessed wells

Number of septic tanks within 100m	Frequency	Percent
0	1	2
1	32	64
2	12	24
3	3	6
4	1	2
5	1	2

The median septic tank setback distance in SA1 was 43.5 m (mean 48.9 m) with a setback distance range of 19-180 m. No significant mean setback difference was found between hand-dug wells and boreholes. Overall, 44% (n = 22) of septic tanks were located down gradient of the wellhead, however, 38% (n = 19) and 18% (n = 9) of septic tanks were situated up gradient of and level with assessed wells, respectively. No association was noted between the septic tank gradient and well type. Only 6% (n = 3) of wells had an on-site wastewater treatment system excluding septic tanks, within 100 m of the wellhead. These had a mean setback distance of 54.1 m, and none occurred up gradient of the wellhead.

Since SA1 is situated in a predominantly rural area, most of the wells (74%) were assessed as being situated in an area (100 m radius) associated with agricultural activities. A chi-square test of independence found that all of the 26% of wells (n = 13) associated with residential sites were boreholes, suggesting a statistically significant association between hand-dug wells and agricultural activities (χ^2 (2) = 7.357, p = 0.025).

All of the assessed wells in SA1 were situated within 100m of grazing animals, with grazing type categorised as beef (72%) and mixed (28%). The median grazing setback distance was 20 m (Mean 24.1 m), with a setback range of 1-62.1 m. As shown in *Table 7.16*, the majority of grazing lands were located up gradient of or level with the assessed wellhead, reflecting the overall topography of the area, as much of the area down gradient is comprised of peats. No mean difference was found to exist between well type and grazing setback distance, grazing type or grazing setback gradient.

Table 7.16 Study Area 1: Setback gradient of septic tank systems within a 100 m radius of assessed wells

Grazing Gradient	Frequency	Percent
Up gradient of wellhead	16	32
Level gradient with wellhead	23	46
Down gradient of wellhead	11	22

32% of wells in SA1 had a point agricultural activity occurring within 100 m of the wellhead, comprising farmyards (20%), animal housing (6%) and feed storage/silage pit (6%), with a median setback distance of 25 m (Mean 49.4 m) and a setback range of 5-220 m. No significant mean difference was noted between hand-dug wells and boreholes. The majority of agricultural activities took place up gradient of (33%) or level with (42%) the wellhead, which, when coupled with the relatively low median setback distance may be considered a cause for concern. Another potential

hazard source situated within SA1 was a quarry, abstracting esker sand and gravel deposits. Although this was not located within 100 m of any assessed wells, five wells were situated within 500 m of the operation, all of which were down gradient.

Overall, 36% (n = 18) of assessed wells had a surface water feature within the general wellhead vicinity, comprising 22% (n = 11) surface water drains, 10% (n = 5) rivers (Grand Canal) and 4% (n = 2) surface springs. The median surface water setback distance was 89.5 m (Mean 98.2 m). Although the mean setback distances were substantially greater for boreholes (139 m) than hand-dug wells (66 m), the mean difference was not found to be statistically significant.

Only 10% of assessed wells (n = 5) had one or more operating wells within a 100 m radius, accurately reflecting the relatively low population density within this study area. Abandoned wells occurred within 100 m of 16% (n= 8) of assessed wells. Potential vegetative indicators of poor drainage (*Phragmites* and mosses) were recorded at 16% (n = 8) of sites.

7.5.2 Study Area 2: Crossbridge, Co Wicklow

As for SA1, conventional septic tank systems were the favoured on-site waste management system used in SA2, with only one recorded occurrence of a "non-septic tank" treatment system within 100 m of an assessed well. Only 4% (n = 2) of wells did not have any septic tanks situated within a 100 m radius of the wellhead. A median figure of one septic tank per wellhead radius was recorded (Mean 1.34), with a maximum of three within a 100 m wellhead radius. No significant difference was observed between well type and septic tank number. A median septic tank setback distance of 52.7 m (Mean 74.8 m) was found, with a setback range of 15-300 m. An independent samples (Students) t-test found a significant mean difference between septic tank setback and well type ($t = 3.52$, $p = 0.012$), with an average borehole setback distance of approximately 20.3 m, compared with an average hand-dug well setback distance of 105.8 m. The majority of septic tanks (66%) were situated down gradient of the wellhead, with only 10% situated up gradient.

Over 90% (n = 46) of wells were associated with agricultural or mixed agricultural land-use, with all assessed hand-dug wells categorised as being located within exclusively agricultural land. This was found to be statistically significant ($\chi^2 (2) = 18.992$, $p < 0.001$). This point may be used to explain the aforementioned significance pertaining to well type and septic tank setback, as typically well location where greater areas of land are available, allow for greater setback distances, whereas well location on residential sites is confined to the available site area. As expected, all of the assessed wells had grazing animals within a 100 m wellhead radius, with mixed (78%) and beef grazing (16%) comprising the majority of grazing types. The remaining 6% (n = 3) were in closest proximity to dairy cattle. The median grazing setback distance in SA2 was found to be 15 m (Mean 19.7 m), occurring within a range of 3-50 m. Again, hand-dug wells were found to be significantly associated with agriculture ($t = -5.494$, $p < 0.001$), borne out by a significantly lower mean grazing setback distance (7.5 m), when compared with boreholes (21.7 m). Additionally, 34% (n = 17) of wells assessed within SA2 were located within 100 m of a point agricultural activity, including 22% (n = 11), 10% (n = 5) and 2% (n = 1) within close proximity to operating farmyards, animal housing units and silage pits, respectively. No statistical significance was noted with regard to the occurrence of agricultural activities within 100 m and the well type.

Due to the predominantly upland topography associated with SA2, there were a greater number of surface water features recorded in this study area than others, with 14% (n = 7) of wells having a nearby surface spring, 48% (n = 24) of wells located in the vicinity of a river and 18% associated with surface water drainage ditches. Over 71% (n = 5) of hand-dug wells were situated within 100 m of a surface spring, which was found to be significant (χ^2 (1) = 22.296, p <0.001). A median surface water setback distance of 99 m (Mean 119 m) was found, within the range 9-380 m. Overall, 22% (n = 11) of wells were located within 100 m of an abandoned well, which was higher than expected. Vegetative indicators of poor drainage were recorded within 100 m of wellheads in 32% (n = 16) of cases.

7.5.3 Study Area 3: Paulstown, Co Kilkenny

Due to a higher number of "non-septic tank" wastewater treatment systems in operation throughout this study area than observed elsewhere, a lower level of septic tank occurrence within a 100 m wellhead radius was recorded in SA3 than encountered in SA1 and SA2. Overall, 86% of assessed wells had at least one septic situated within 100 m of the wellhead, with only 50% of hand-dug wells having a septic tank occurring within this radius (*Table 7.17*).

The median septic tank setback distance was 54 m (Mean 54.4 m), within a setback range of 20-101 m, with the majority (72%) situated down gradient of the wellhead. The median setback distance for "non-septic tank" wastewater treatment systems was 30.8 m (Mean 38.7 m), within the range 15.9-101 m. Overall, 16% of wells (n = 8) had one of these systems located within 100 m, with 50% (n = 4) situated up gradient of the well, and the remaining 50% (n = 4) installed down gradient.

Table 7.17 Study Area 3: Number of septic tank systems within a 100 m radius of assessed wells

Well Type	Number of septic tanks within 100m wellhead radius			
	0	1	2	3
Hand-dug	3 (50%)	1 (17%)	1 (17%)	1 (17%)
Bored	4 (9%)	27 (61%)	13 (29%)	3 (7%)

This study area was again dominated by agriculture, with 88% (n = 44) of wells designated as being located on or within 100 m of agricultural grazing land. Only 6% of wells were categorised as being situated on a predominantly residential site, with all of these being boreholes, which was found to be statistically significant (χ^2(4) = 28.147, p <0.001). Overall, mixed grazing and beef grazing were associated with 68% (n = 34) and 22% (n = 11) of assessed wells, respectively, with the remainder comprising 6% (n = 3) dairy grazing and 2% (n = 4) horses. The median animal grazing setback distance was 8.75 m (Mean 11.5 m), which was less than that found over other study areas. The grazing setback range was 1-45 m. The mean grazing setback for hand-dug wells was 21 m, whereas this figure was 10 m for boreholes, however, due to the small hand-dug well sample size and the resulting unequal variance, this mean difference was not found to be significant. The majority of animal grazing was found to occur approximately level with the wellhead (60%), with 24% (n = 12) and 16% (n = 8) occurring up gradient and down gradient, respectively.

Point agricultural activities were found to occur within 100 m of 36% of wells in SA3, comprising 30% (n =15) farmyards and 6% (n = 3) animal housing. The median setback distance relating to these

activities was 43 m (mean 48.2 m), occurring over a range of 1-110 m. Additionally, 8% of wells (n = 4) were located in the vicinity of an operational commercial activity.

Overall, 62% of wells were constructed in the vicinity of a surface water feature, comprising 22% (n = 11) rivers, 30% (n = 15) drainage ditches and 10% (n = 5) surface springs, with these features having a median setback distance of 40 m (mean 122 m). Due to the relatively sparsely populated nature of the study area, there was a low instance of operating wells within 100 m, with only 14% of assessed wells having another well in the vicinity. Similarly, only 6% (n = 3) of assessed wells had an abandoned well within a 100 m radius. Additionally, 50% of assessed wells (n = 25) had *Phragmites* growth evident within 100 m of the wellhead.

7.5.4 Study Area 4: Kilclone, Co Meath

Previous work (McCole, 1992) had identified SA4 as an "overdeveloped" rural area and it was therefore expected that high numbers of on-site wastewater treatment systems would be in operation in this study area. This expectation was realised, as 100% of assessed wells (n = 52) were found to have at least one conventional septic tank system in operation within a 100 m radius. As shown in *Table 7.18*, approximately 75% of both hand-dug and boreholes had two or more septic tanks within a 100 m radius.

Table 7.18 Study Area 4: Number of septic tank systems within a 100 m radius of assessed wells

Well Type	Number of septic tanks within 100m				
	1	2	3	4	5
Hand-dug	3 (25%)	7 (58%)	1 (8%)	0	1 (8%)
Bored	11 (27%)	18 (45%)	7 (17%)	3 (7%)	1 (2%)

The median septic tank setback distance was found to be 40.9 m (Mean 44.6 m), occurring over an overall range of 14.6-90 m. The majority of wells (65.4%) were located, as recommended, up gradient of the septic tank system, with 15.4% of wells situated down gradient of the septic tank. Additionally, it was also found that where wells were sited down gradient of existing septic tanks, they also presented with a lower mean setback distance (30.5 m), when compared to wells finished level with (38.1 m) and up gradient of (49.9 m) septic tanks within 100 m. This result was found to be statistically significant ($F(2) = 4.771$, $p = 0.013$). "Non-septic tank" on-site wastewater treatment systems were recorded within 100 m of at 23% (n = 12) of assessed wells, with a median setback distance of 44.4 m (Mean 44.5 m) and a range of 14 - 99 m.

The "overdeveloped" description of this area was reflected in the overall land-use categorisation, with approximately 27% of wells (n = 14) classified as being situated on predominantly residential land, which is notably higher than the observed mean over all study areas (16.3%).

As found throughout the fieldwork programme, mixed grazing (77%) and beef grazing (19%) were the dominant grazing types within SA4, with the remaining 4% (n = 2) of recorded grazing type being attributed to horses. The median grazing setback distance was 23.2 m (Mean 26.4 m), occurring over a range of 2-90 m. The majority of grazing activities took place either down gradient of (36.5%) or level with (46.2%) assessed wellheads. Additionally, 32.7% of assessed wells (n = 17) had an

agricultural activity excluding animal grazing within a 100 m radius, comprising farmyards (n = 14) and animal housing (n = 3), situated at a median setback distance of 49.5 m (Mean 46.6 m).

Overall, 15.4% of wells (n = 8) were situated within 100 m of an abandoned well, while approximately 27% of assessed wells had a functioning well within the same radius. Just over 19% (n = 10) of wells had a vegetative indicator of potential poor drainage within the 100 m wellhead radius. As this study area was primarily categorised as being *Low* vulnerability, therefore a greater prevalence of vegetation associated with poor drainage was expected, however as noted discussed (*Section 7.4.6*), this expectation was not realised.

7.5.5 Study Area 5: Birdhill Co. Tipperary

Due to the low sample number associated with SA5, no statistically significant relationships regarding hazard sources/susceptibility were found. All of the assessed wells in SA5 (n = 9) had at least one septic tank within a 100 m radius of the wellhead, and a maximum of two (n = 2). The median septic tank setback distance was 45 m (Mean 60.5 m), with a range 30-101 m. No "non-septic tank" wastewater treatment systems were recorded.

All wells were categorised as being within 100 m of agricultural land, with beef (66%) and mixed (33%) recorded. Overall, the median animal grazing setback distance was 15 m (Mean 21.3 m), with a range of 9-40 m, and the majority (66.7%) of grazing took place on a level gradient with the wellhead. Excluding animal grazing, 78% (n = 7) of assessed wells had an operational farmyard in the vicinity, with a median setback distance of 40 m (Mean 50 m). In all but one of these cases, the farmyard was located down gradient of the wellhead.

7.6 "One-Off" Chemical and Microbial Sampling

7.6.1 Introduction

The following section presents the groundwater analysis results associated with the "one-off" sampling and assessment of 211 private groundwater sources over five study areas. The results from the individual study areas have been amalgamated and analysed with regard to well type and overall susceptibility and their observed influence on perceived groundwater quality. The focus is placed on microbial quality, with a number of chemical parameters also investigated as potential indicators of contaminant ingress. Additionally, the potential influence of groundwater source characteristics on groundwater quality, including geological setting, source location, design and construction, as outlined earlier in this chapter, are investigated. It should be noted that ICP-AES analysis was carried out for Cadmium (Cd), Cobalt (Co), Chromium (Cr), Nickel (Ni) and Lead (Pb), but as all samples were found to be under the limit of detection they are not discussed.

7.6.2 Sample Sources

The "one-off" sampling programme took place during the 28-month period January 2008 to April 2010, inclusive. Across the five study areas, 80.1% (n = 169) of sampled wells were boreholes, with the remaining 19.9% (n = 42) comprising shallow (<10 m), large diameter hand-dug wells. Approximately 56% (n = 119) of groundwater samples were taken from the closest outdoor tap to

the well, with 35.5% (n = 75) sampled from an indoor tap where no treatment systems were in place. The remaining 8% (n = 17), all of which were large diameter hand-dug wells, had samples taken directly from the well using drop sampling techniques. An overview of sample sources pertaining to specific study areas is presented in Appendix E4.

The mean depth of boreholes was found to be 30.5 m, while the mean depth of hand-dug wells was 3.7 m. This mean difference was statistically significant ($t = -12.164$, $p < 0.001$). The mean measured watertable over the duration of the sampling programme was 5.82 m, with minimum and maximum measured watertables of 0 m (i.e. surface springs, measured at both SA2 and SA3) and 48.66 m (measured at SA3). Boreholes had a mean measured watertable depth of 6.28 m, while hand-dug wells had a shallow measured mean watertable depth of 1.41 m, a statistically significant difference ($t = -4.419$, $p < 0.001$), as expected. An overview of measured watertables relating to specific study areas and source types is presented in Appendix E5.

7.6.3 Physicochemical Parameters

The mean measured pH was slightly acidic at 6.71, occurring over the range 4.51 – 7.97. As shown (*Table 7.19*), the mean sample pH in SA2 was the only study area in which the mean deviated significantly from neutral, with a mean measured pH of 5.8. This is expected in areas of acidic intrusive igneous and metamorphic rocks with only minor fractions of buffering carbonate minerals (Lavapuro *et al.*, 2008). A recent report (OCM, 2007) cites an expected groundwater pH of 5.6 in granite areas in Ireland. Likewise, the OCM report (2007) cites expected groundwater pH values of 7.3 and 7 in predominantly limestone and sandstone areas, respectively, both of which are closely reflected in *Table 7.19*. Both limestones and sandstones have larger fractions of carbonate minerals, which serve to buffer slightly acidic recharge (with the natural pH of rainwater being approximately 5.7).

No relationship was found to exist between watertable depth and sample pH, however, a significant, albeit weak, correlation existed between sample temperature and sample pH ($r = 0.2$, $p = 0.005$). This would suggest some slight seasonal pH variation in SA1, which is investigated later in this chapter.

Table 7.19 Area by Area Mean Groundwater pH

Study Area	Mean pH	pH Range	Dominant Bedrock
SA1: Belmont, Co Offaly	7.14	6.4 – 7.8	Sandstone/Limestone
SA2: Crossbridge, Co Wicklow	5.80	4.5 – 6.8	Granite
SA3: Paulstown, Co Kilkenny	6.70	5.2 – 7.4	Sandstone/Shale/Limestone
SA4: Kilclone, Co Meath	7.06	6.8 – 7.7	Limestone
Birdhill, Co Tipperary	7.20	6.4 – 7.9	Sandstone/Limestone

The mean measured sample electrical conductivity (EC) was 553 $\mu S\ cm^{-1}$, ranging from 71 $\mu S\ cm^{-1}$ to 2.47 $mS\ cm^{-1}$. As shown in *Table 7.20*, typically groundwater from granite bedrock had a lower mean EC, with groundwater from limestone and sandstone areas typically having higher EC values, as expected. This is comparable with a previous large-scale study of bedrock influenced parameters in Irish groundwater (see Appendix C2) (OCM, 2007).

No significant correlation was found between watertable depth and sample EC within the amalgamated dataset (n = 211), however, an exception was noted in SA2, where a positive correlation was found (r = 0.466, p = 0.002) i.e. higher EC values associated with increased watertable depths. This is possibly due to longer subsurface residence times associated with increased watertable depths, leading to increased mineralisation of granite bedrock in this study area and subsequently increased groundwater EC (and pH).

Table 7.20 Area by Area Mean Groundwater EC

Study Area	Mean EC ($\mu S\ cm^{-1}$)	Range($\mu S\ cm^{-1}$)	Dominant Bedrock
SA1: Belmont, Co Offaly	737	245 - 958	Sandstone/Limestone
SA2: Crossbridge, Co Wicklow	289	71 – 2,470	Granite
SA3: Paulstown, Co Kilkenny	440	86 – 1,460	Sandstone/Shale/Limestone
SA4: Kilclone, Co Meath	756	264 – 2,080	Limestone
Birdhill, Co Tipperary	452	304 - 849	Sandstone/Limestone

Significant relationships existed between both sample EC and sample temperature (r = 0.417, p < 0.001) and sample EC and sample pH (r = 0.509, p < 0.001). The correlation between sample EC and sample temperature would suggest a seasonal groundwater EC variation or an EC variation linked to source type i.e. boreholes/hand-dug wells. An independent samples t-test found no significant mean EC difference with respect to source type, therefore it is probable that the groundwater temperature/EC correlation was seasonal, which is investigated later in this chapter.

As recharge water tends to be more acidic than groundwater, pH will tend to increase as mineralisation (EC) increases, and therefore a positive correlation between groundwater pH and groundwater EC is typical (Hem, 1992; Kelly & Wilson, 2008). However, a significant positive pH/EC relationship was not observed in SA1, where an inverse correlation was found between sample pH and sample EC (r = -0.667, p > 0.001). As outlined later in this chapter (*Section 7.8.2*), monthly sampling at one borehole (BM1) found a highly variable water table, following a diurnal pattern. This constant wetting/drying effect may have initiated reducing conditions within the borehole and lead to colonization of iron-reducing bacteria, which in turn may initiate corrosion of the steel well liner and consequently reduce the pH. ICP-AES analysis carried out on a sample from this borehole (May 2009) had an iron concentration of 730 $\mu g\ L^{-1}$, which is well above the natural expected background level of approximately 25 $\mu g\ L^{-1}$ (OCM, 2007). Furthermore, an independent analysis carried out in March 2006 (results acquired from well owner in May 2009) found an iron concentration of approximately 1,800 $\mu g\ L^{-1}$. These results support the hypothesis that corrosion was occurring at this well, however, it is not clear why an inverse correlation was found between sample pH and sample EC at SA1 in general.

No significant mean differences were found between hand-dug wells and boreholes and the primary physicochemical parameters (sample temperature, sample pH and sample EC). A correlation matrix pertaining to the main physicochemical parameters is presented in Appendix E6.

7.6.4 Major Ions

Chloride (Cl)

The mean measured groundwater Cl in the amalgamated dataset was found to be 27 mg L^{-1} (range 3-230 mg L^{-1}), with individual study area mean concentrations ranging from 13.4 – 35.8 mg L^{-1} (*Table 7.21*). These mean values are comparable with previous studies in Co Offaly (SA1), with mean Cl concentrations found to be 21.5 mg L^{-1} at boreholes and 33.6mg/L at spring/hand-dug wells (Spain, 2000) and Co Meath (SA4) with mean Cl concentrations found to be 24.5 mg L^{-1} at boreholes and 36 mg L^{-1} at hand-dug wells (McCole, 1992). As shown in *Figure 7.7*, almost 75% (n = 157) of one-off samples had a Cl concentration <30 mg L^{-1} (proposed as a guideline value in Irish groundwaters, an exceedance of which may be indicative of contamination), and approximately 48% (n = 102) had a Cl concentration <20 mg L^{-1} which is comparable with national figures i.e. a previous large-scale study reports a national background median of 18 mg L^{-1} (OCM, 2007). Measured groundwater Cl ranges for individual study areas are presented in Appendix E7, with all study areas displaying a similar Cl range as observed in the amalgamated dataset. In all, no wells exceeded the EU MAC of 250 mg L^{-1} Cl, although two wells did exceed 200 mg L^{-1} Cl.

Table 7.21 Area by Area Mean Groundwater Chloride

Study Area	Mean (mg L^{-1})	Range (mg L^{-1})	Borehole Mean (mg L^{-1})	Hand-dug Mean (mg L^{-1})
SA1: *Belmont*, Co Offaly	26	7 – 73	26.5	26
SA2: Crossbridge, Co Wicklow	20.5	3 – 163	22	12
SA3: Paulstown, Co Kilkenny	28.1	8 – 230	29.2	20
SA4: Kilclone, Co Meath	35.8	3 – 201	35	38
Birdhill, Co Tipperary	13.4	3 – 59	16.8	6.6

Statistical analysis found no significant correlations between groundwater Cl concentration and watertable depth or sample temperature, and therefore no seasonal Cl trend was suggested. Neither was any significant relationship found between groundwater Cl and groundwater pH in the amalgamated dataset. A notable exception was found in SA1, where groundwater pH and Cl had a significant inverse correlation ($r = -0.472$, $p = 0.001$). Similar inverse groundwater pH/Cl correlations have been previously reported by Helena *et al* (2000) and Shymala (2008), with Helena *et al.* (2000) stating that negative and weak correlations typically exhibit a "higher aggressiveness of acidic media towards rock and soil hosts, increasing the concentrations of other ions".

Significant correlations were found to exist in all individual study areas between Cl and sample EC, with this also reflected in the amalgamated dataset, as shown in *Figure 7.8* ($r = 0.688$, $p < 0.001$). This geochemical relationship is expected, with similar correlations having been found in a number of previous studies (Ballukraya, 1999; Raja & Venkatesan, 2010), as chloride is typically the dominant anion in more highly mineralised groundwaters. Individual study area EC/Cl correlations ranged from $r = 0.755$, $p < 0.001$ in SA4 (*Low* vulnerability, Limestone bedrock) to $r = 0.465$, $p = 0.001$ in SA3 (*High* vulnerability, Sandstone/Shale/Limestone bedrock), with the lower correlation coefficient observed between these parameters in SA3 than other study areas possibly due to the relatively diverse bedrock lithology in this area. The remaining three study areas (SA1, SA2, SA5), all exhibited EC/Cl correlation coefficients in the range $r = 0.65 – 0.7$.

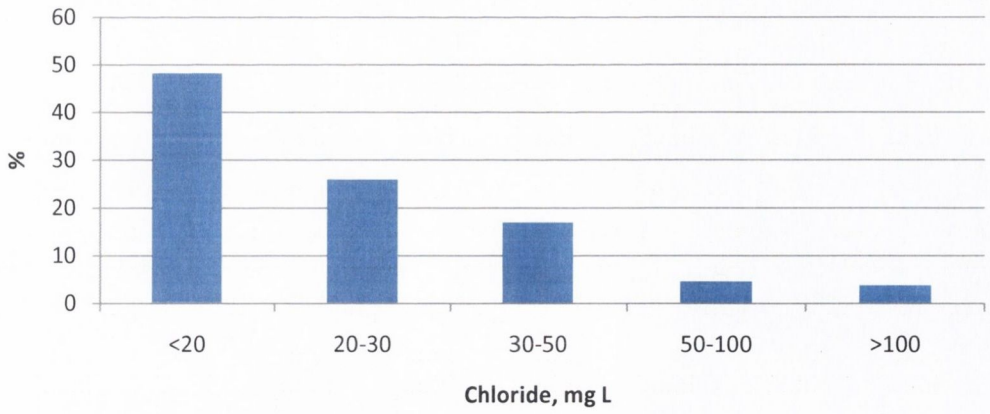


Figure 7.7 Percentage of wells with mean Chloride concentrations in the ranges indicated

As shown in both *Figures 7.7 & 7.8*, only 3.8% of groundwater samples (n = 8) exceeded 100 mg L^{-1} Cl. These exceedances were probably due to increased precipitation i.e. increased recharge, and/or small-scale contamination events. These potentialities are further explored later in this chapter.

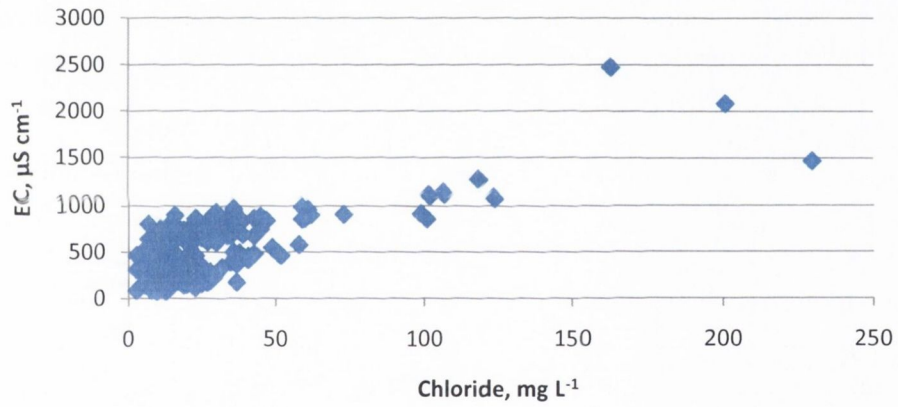


Figure 7.8 Amalgamated Dataset Groundwater EC/Cl relationship (n = 211)

Sodium (Na)

As ICP-AES facilities only became available after the start of the one-off groundwater sampling and analysis programme, not all groundwater samples were analysed for sodium (Na) concentrations. In all, just under 48% of wells (n = 100) were analysed for groundwater sodium. The mean measured groundwater sodium concentration was 19.8 mg L^{-1}, ranging from 4.3 (SA2) to 72.7 (SA4) mg L^{-1} Na. As previously outlined (*Chapter 4, Section 4.8.7*), normal background sodium levels in uncontaminated groundwater in Ireland are 5-15 mg L^{-1} Na (Spain, 2000), with significant departures from this being a potential indicator of contamination. OCM (2007) have reported groundwater Na as being a bedrock influenced parameter, and cite normal background levels in the range 9 mg L^{-1} Na in Devonian bedrocks to 19 mg L^{-1} Na in Lower Palaeozoic and confined sandstone bedrocks (*Table 7.22*). Therefore, the mean measured groundwater sodium concentration falls within the expected range (*Figure 7.9*), with 53% of samples (n = 53) adhering to the expected background sodium levels

of <15 mg L^{-1}, however, 16% of wells (n = 16) had sodium levels >25 mg L^{-1}, suggesting a decline in groundwater quality.

Table 7.22 Area by Area Mean Groundwater Sodium

Study Area	Mean (mg L^{-1})	Range (mg L^{-1})	Dominant Bedrock
SA1: *Belmont*, Co Offaly	N/A	N/A	Sandstone/Limestone
SA2: Crossbridge, Co Wicklow	9.9	4.3 – 15.6	Granite
SA3: Paulstown, Co Kilkenny	17	5.6 – 60.1	Sandstone/Shale/Limestone
SA4: Kilclone, Co Meath	7.5	5.1 – 72.7	Limestone
SA5: Birdhill, Co Tipperary	11.2	7.6 – 24.2	Sandstone/Limestone

A number of correlations were noted with regard to groundwater sodium concentration, including with watertable level ($r = 0.308$, $p = 0.007$), groundwater temperature ($r = 0.222$, $p = 0.026$), groundwater EC ($r = 0.536$, $p < 0.001$) and groundwater Cl ($r = 0.499$, $p < 0.001$). The relationship between sodium and both watertable level and groundwater temperature suggests a seasonal trend possibly due to ionic exchange (Ca and/or Mg) occurring with decreasing watertables. Ramkumar *et al* (2010) have proposed similar findings, with increasing groundwater sodium during drier months. Also, as noted by Scheytt (1997), as sodium concentrations in rainfall are typically ten-fold lower than in groundwater, a significant level of sodium dilution may occur during wetter months, particularly in shallow groundwaters. Additionally, Hem (1992) reports that sodium sulphate solubility is strongly influenced by temperature, with solubility typically decreasing in concurrence with decreasing temperatures.

The significant correlation between groundwater sodium and both groundwater EC and groundwater Cl would suggest that sodium is a dominant element in the aquifer matrix (Lloyd & Heathcote, 1985; Kim & Yun, 2005), which is expected in the majority of Irish aquifers.

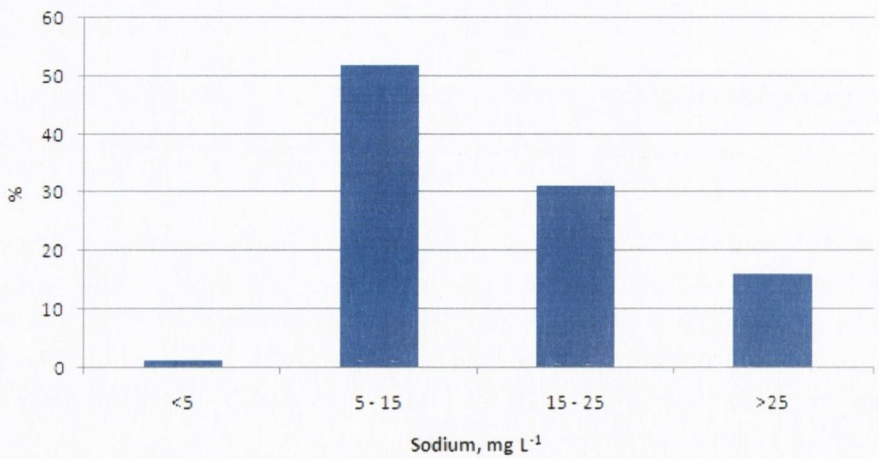


Figure 7.9 Percentage of wells with mean Sodium concentrations in the ranges indicated (n = 100)

Potassium (K)

A mean potassium concentration of 3.83 mg L^{-1} K was found (n = 134), with minimum and maximum values of 0.23 mg L^{-1} (SA2) and 68.6 mg L^{-1} (SA4), respectively (*Table 7.23*; *Figure 7.10*). Kiely (1997) proposes that background potassium levels in most areas of Ireland are typically <3.0 mg L^{-1}, while OCM (2007) reports potassium as a bedrock influenced parameter, with natural background levels in the range 1.1 mg L^{-1} (Devonian bedrock) to 1.9 mg L^{-1} (confined limestone bedrock). Therefore, while the recorded mean is higher than expected, a number of large outliers were present, with just under 65% (n = 88) of samples having a potassium concentration <3 mg L^{-1}. Furthermore, higher groundwater potassium concentrations are occasionally found in Ireland, particularly in granite or sandstone areas, as elevated concentrations may be due to increased leaching from these potassium rich geological materials.

Table 7.23 Area by Area Mean Groundwater Potassium

Study Area	Mean (mg L^{-1})	Range (mg L^{-1})	Dominant Bedrock
SA1: Belmont, Co Offaly	N/A	N/A	Sandstone/Limestone
SA2: Crossbridge, Co Wicklow	1.99	0.23 – 6.27	Granite
SA3: Paulstown, Co Kilkenny	2.43	0.36 – 16.2	Sandstone/Shale/Limestone
SA4: Kilclone, Co Meath	4.4	0.48 – 68.6	Limestone
SA5: Birdhill, Co Tipperary	3.92	1.06 – 13.07	Sandstone/Limestone

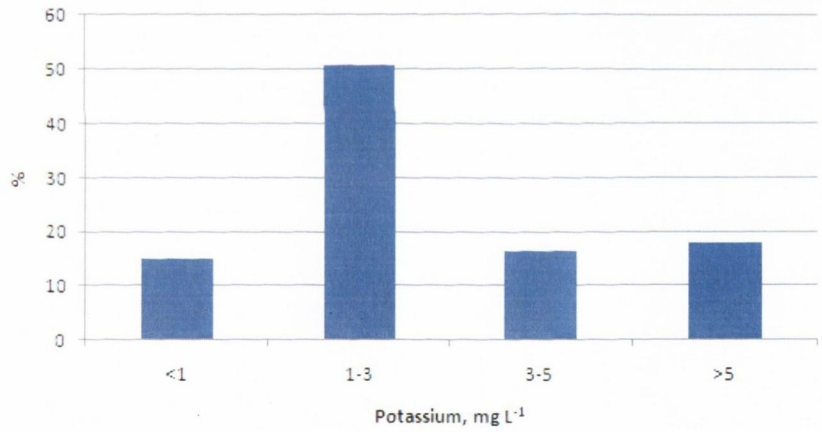


Figure 7.10 Percentage of wells with mean Potassium concentrations in the ranges indicated (n = 134)

Significant associations were found with regard to groundwater potassium concentrations were positive correlations with both groundwater chloride (r = 0.24, p = 0.005) and groundwater EC (r = 0.41, p < 0.001). This may be a result of over fertilisation and/or wastewater ingress (Trauth & Xanthopoulos, 1997), with the magnitude of correlation between chloride and potassium similar to that reported by Helena *et al.* (2000). Aris *et al.* (2007) state that this correlation may be due to the interaction of groundwater with the aquifer matrix, while Appelo & Postma (2005) note that a positive and significant correlation between groundwater Cl and K may be interpreted as light seawater intrusion, but, as this is not a coastal aquifer, this is not a possible cause here. Therefore, it is likely that ingress of organic or inorganic contaminants including silage effluents, farmyard runoff

or inorganic fertilisers are at least partly responsible. Statistical analysis found that a significant correlation also existed between groundwater potassium concentrations and sample temperature ($r = 0.256$, $p = 0.003$), which would suggest some level of seasonality and perhaps point to spreading of fertilisers as a potential source.

Potassium/Sodium Ratio (K:Na)

As illustrated in Chapter 4 (*Section 4.8.8*), both Daly & Daly (1982) and Thorn & Hanna (1989) proposed utilising the K:Na ratio, whereby potassium levels of greater than 5.0mg L^{-1} coupled with K:Na ratios greater than 0.3 may indicate local, vegetable based contamination i.e. contamination by plant organic matter (e.g. farmyards, landfills), with this threshold later increased to 0.4 (Daly, 1994b; Spain, 2000). Overall, just under 48% of samples (n = 100) were analysed for both K and Na and therefore had a calculated K:Na ratio. A mean K:Na ratio of 0.29 was found (n = 100), with minimum and maximum ratios of 0.008 and 3.6, respectively, with 17% of samples (n = 17) exceeding a ratio of 0.4. Of those samples exceeding a 0.4 K:Na ratio, eight samples had a potassium concentration exceeding 5 mg L^{-1} K.

Magnesium (Mg)

Groundwater magnesium is a bedrock influenced geochemical parameter, with OCM (2007) reporting expected natural levels within the range 3.3 mg L^{-1} in Devonian bedrocks to 23 mg L^{-1} in confined limestones. The measured mean groundwater Mg was found to be 7.3 mg L^{-1} and therefore within the expected range in Irish aquifers. Minimum and maximum measured concentrations of 0.01 and 20.5 mg L^{-1} Mg were recorded, respectively. There were no measured exceedances of the EU MAC 50 mg L^{-1} Mg. As shown (*Figure 7.11*), the majority of samples (n = 132, 97%) had Mg concentrations < 15 mg L^{-1} Mg.

A significant correlation was noted between groundwater Mg and EC ($r = 0.29$, $p = 0.001$), which as mentioned in Chapter 4 (Section 4.8.4), is an expected relationship as magnesium is an abundant cation in groundwater. Helena *et al.* (2000) have reported similar correlation magnitudes. Additionally, correlations were noted between measured groundwater Mg concentrations and chloride ($r = 0.203$, $p = 0.018$), sodium ($r = 0.204$, $p = 0.041$) and nitrate ($r = 0.173$, $p = 0.044$). Correlations between Mg, Cl and Na are expected, for the same reasons as the relationship between Mg and groundwater EC. The relationship between groundwater Mg and nitrate, which was particularly strong in SA2 ($r = 0.562$, $p < 0.001$), have been reported in recent studies (Baalousha, 2008; Anandakumar *et al.*, 2009), with Baalousha (2008) stating that this is indicate of a non-point agricultural source of nitrate.

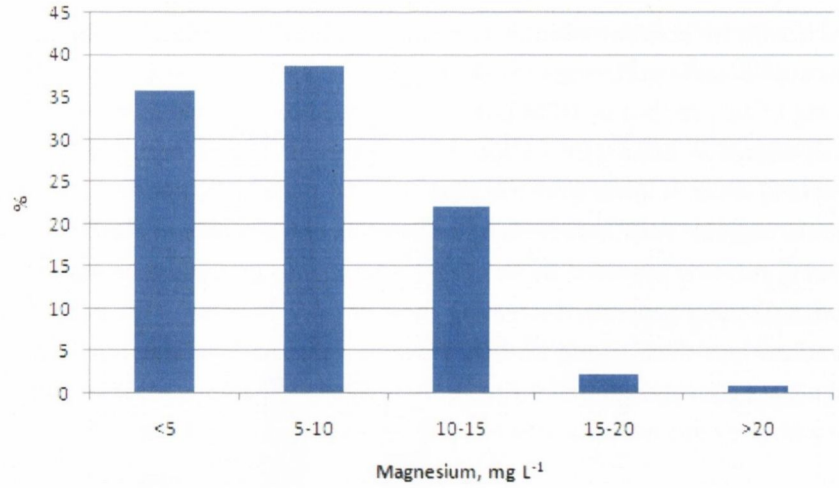


Figure 7.11 Percentage of wells with mean Magnesium concentrations in the ranges indicated (n = 136)

7.6.5 Nutrients

Ammonia (NH_4)

An overall mean of 0.17 mg L^{-1} NH^4 was found with minimum and maximum concentrations of 0.01 mg L^{-1} and 3.01 mg L^{-1}, respectively. Measured concentrations bore a similar pattern to recently published national monitoring figures (EPA, 2010c), as shown in *Figure 7.12*. Ammonia was negatively correlated with both watertable depth and groundwater nitrate concentration, although neither correlation was statistically significant, due to the small sample number (n = 58).

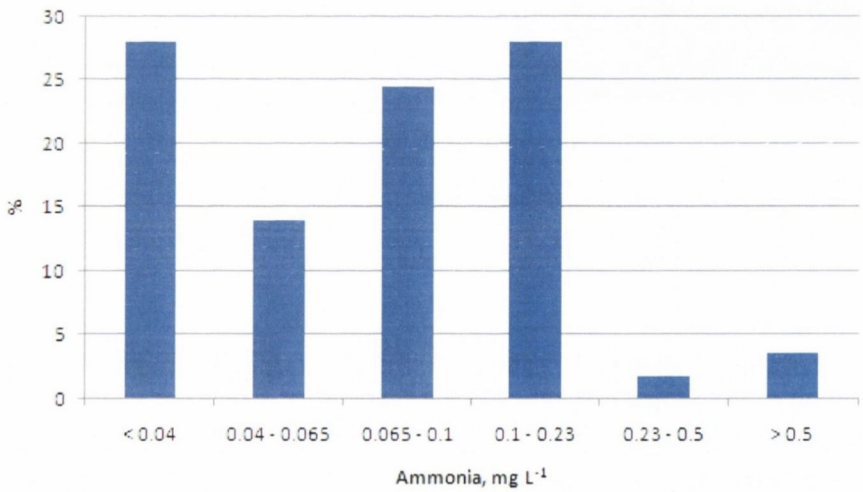


Figure 7.12 Percentage of wells with mean ammonia concentrations in the ranges indicated (n = 58)

Nitrate (NO_3)

A mean measured nitrate concentration of 12.8 mg L^{-1} NO_3 was found among the 211 sampled wells in the amalgamated dataset occurring over the range 0.1 – 110 mg L^{-1} NO_3. This mean value is lower than the 14.6 mg L^{-1} NO_3 set out by OCM (2007) as a typical background level in Irish groundwaters. Furthermore, as shown in *Table 7.24*, a mean concentration of 15.5 mg L^{-1} NO_3 was found in SA1 (*Belmont*, Co Offaly) which is lower than the reported 21.5 mg L^{-1} NO_3reported by Spain (2000) in Co Offaly. This would suggest recently decreasing nitrate concentrations in Irish groundwaters, with this pattern also being recently reported by the EPA (2010c), with groundwater nitrate concentrations over the period 2007-2009 generally decreasing and a notable increase in the percentage of samples with concentrations less than 10 mg L^{-1} NO_3. However, this pattern was not in evidence at SA4, where a mean nitrate concentration of 9.7 mg L^{-1} NO_3 was found among assessed wells in SA4, which is comparable with previous work done in the same area (Mean – 9.11 mg L^{-1} NO_3, n = 39) (McCole, 1992).

The occurrence of groundwater nitrate within a number of ranges is presented in *Figure 7.13.* Similar figures pertaining to individual study areas are provided in Appendix E8. As shown, only one exceedance of the EU MAC (50 mg L^{-1} NO_3) was noted, with 13.7% of samples (n = 29) exceeding the 25 mg L^{-1} NO_3set out by Daly (1994b) as being indicative of a nearby organic waste source. A similar nitrate profile was exhibited in all study areas, with exceedances of the 25 mg L^{-1} NO_3 guide limit typically occurring at 10-20% of wells.

Table 7.24 Area by Area Mean Groundwater Nitrate

Study Area	Mean (mg L^{-1})	Range (mg L^{-1})	Borehole Mean (mg L^{-1})	Hand-dug Mean (mg L^{-1})
SA1: Belmont, Co Offaly	15.5	0.9 – 35.2	14.6	17.2
SA2: Crossbridge, Co Wicklow	15.3	0.5 – 41.1	16.3	8
SA3: Paulstown, Co Kilkenny	10.8	0.9 – 49	11.1	8.4
SA4: Kilclone, Co Meath	9.7	0.4 - 110	7.1	18.1
SA5: Birdhill, Co Tipperary	12.3	5.7 – 19.9	14.1	8.4

An inverse correlation was found to exist in the total dataset between nitrate concentration and sample pH ($r = -0.158$, $p = 0.022$). A similar correlation pattern and magnitude has been reported in a number of previous studies (Helena *et al.,* 2000; Babiker *et al.,* 2004; Kundu *et al.,* 2008) and may indicate entry of low pH, organic effluent.

A series of correlations found a positive relationship between nitrate concentration and watertable depth, temperature and EC, which, although not statistically significant within the total dataset, were significant within individual study areas e.g. SA2: watertable depth ($r = 0.351$, $p = 0.024$); groundwater temperature ($r = 0.612$, $p < 0.001$); EC ($r = 0.542$, $p < 0.001$). These suggest a seasonal nitrate pattern, with higher nitrate concentrations during warmer months, with a significant positive correlation between sample EC and nitrate suggestive of organic waste ingress, probably agricultural contamination i.e. grazing animal faeces. Furthermore, a significant positive correlation was found between groundwater nitrate and potassium concentrations in SA4 ($r = 0.46$, $p = 0.001$), which is also suggestive of organic waste ingress and particularly fertiliser runoff. As this is a *Low* vulnerability area i.e. low permeability subsoil's with consequently low recharge rates, this may imply direct

ingress of fertiliser runoff at the wellhead as a primary contamination mechanism. Also, it should be noted that nitrifying bacteria are highly temperature sensitive and therefore this may provide further evidence of surface ingress of agricultural effluents (Lipponen *et al.*, 2002).

As stated in Chapter 4 (*Section 4.8.8*) Pionke *et al.* (1990) found the K:Na ratio to have a good correlation with NO_3 in groundwaters, particularly at the higher end of the range, correctly identifying 62.5% of high nitrate groundwater wells. This potential relationship was not found in this study (see Appendix E9).

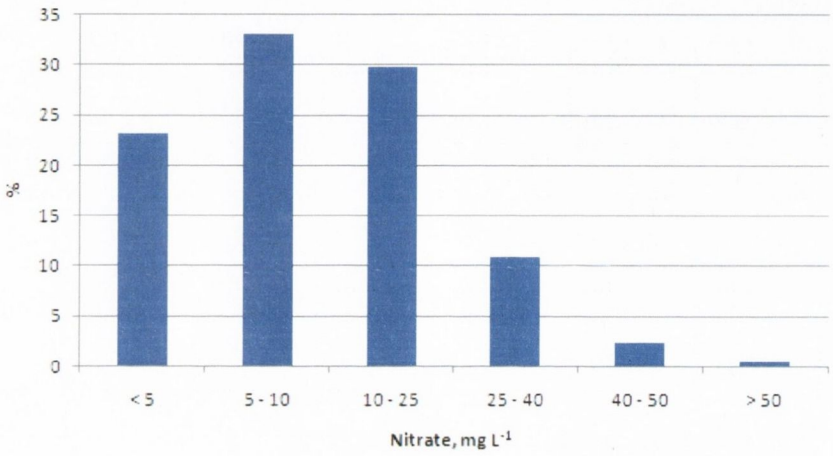


Figure 7.13 Percentage of wells with mean Nitrate concentrations in the ranges indicated (n = 211)

Phosphorus (Total P)

Due to difficulties encountered in analysis, only 31 samples from two areas, SA1 and SA2, were analysed for total P. A mean measured total P of 0.057 mg L^{-1}, occurring over a range of 0.6 µg to 0.18 mg/L was recorded. As shown in *Figure 7.14*, just under half of analysed samples (45.1%) exceeded 0.05 mg L^{-1} total P. Notably, three samples exhibiting the highest total P concentrations in SA1 all had a distinct yellow/brown colouring, indicative of rapid recharge of acidic surface water/shallow groundwater from nearby peatlands or ingress of heavily contaminated runoff. A similar relationship was not noted in SA2. Both SA1 and SA2 are *High/Extreme* vulnerability areas (i.e. shallow high permeability subsoils), which may act as an additional nutrient enrichment pathway for receptors such as lakes, rivers and wetlands i.e. potentially rapid contaminant pathway (EPA, 2010c).

Statistical analysis found a discernable correlation between groundwater P and both groundwater temperature and pH in both SA1 and SA2, with no link found between total P and nitrate concentration. In SA1, a significant inverse correlation between sample temperature and total P concentration ($r = -0.456$, $p = 0.038$) suggests a seasonal pattern i.e. higher total P concentrations during periods of low temperature, with a similar correlation (albeit, statistically insignificant due to small sample size) found in SA2. These are similar to findings by Chen *et al.* (2010) and suggest increased recharge of surface water and/or contamination. Furthermore, inverse correlations found between sample pH and groundwater phosphorus in both study areas, particularly in SA2 ($r = -0.701$, $p = 0.024$), also indicate rapid recharge of acidic surface water or contaminants.

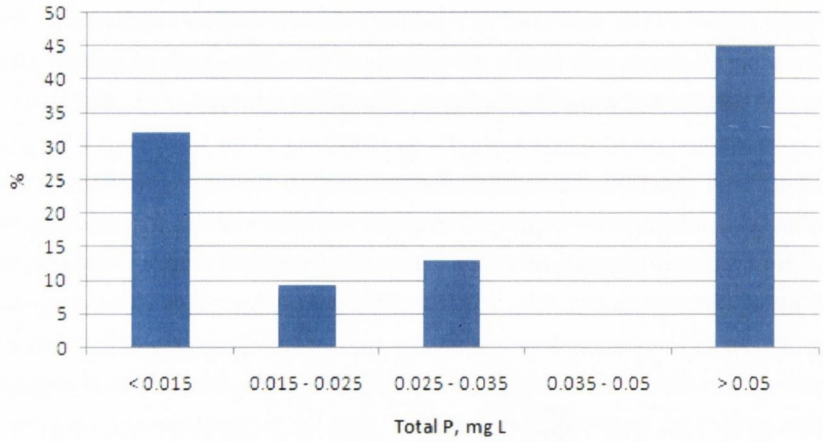


Figure 7.14 Percentage of wells with mean Phosphorus concentrations in the ranges indicated (n = 31)

7.6.6 Trace Metals

Aluminium (Al)

In all, 40.3% (n = 85) of collected 'one-off' samples were analysed for aluminium, with 12.9% (n = 11) of these having detectable levels. An overall mean measured Al value of 0.011 mg L^{-1} was recorded with minimum and maximum values of 0.2 µg L^{-1} and 0.08 mg L^{-1}, respectively. A high correlation was found between Al level and sample pH ($r = 0.949$, $p = 0.051$), in SA1, however, due to the very small number of positive samples, it is unclear as to whether this may be a significant relationship, particularly as this correlation was not found in SA2. The only other resulting correlation relating to groundwater Al concentration was in relation to measured K concentrations in SA2 ($r = 0.893$, $p = 0.007$), however, again due to a very small sample number (n = 7), this is an inconclusive association.

Copper (Cu)

As for aluminium, a small number of samples were found to have detectable levels of copper (Cu) present (n = 9). The mean measured Cu concentration, where present, was 0.038 mg L^{-1} with minimum and maximum concentrations of 0.5 µg L^{-1} and 0.11 mg L^{-1}, respectively. This is significantly higher than the expected natural background level of approximately 2.5 µg L^{-1} cited by OCM (2007). No significant relationships or trends were noted in relation to groundwater Cu concentrations and other variables.

Manganese (Mn)

Only 16% of analysed samples had detectable levels of groundwater manganese (5/50 in SA1; 11/50 in SA2), with a mean concentration of 0.4 mg L^{-1}. As this mean value is considered high, and far exceeds the current EU MAC level of 0.05 mg L^{-1}, samples were re-analysed. This resulted in the exclusion of four notable outliers, and a new mean value of 8 µg L^{-1} Mn, which is reflective of expected natural background levels of 8 µg L^{-1} as cited by OCM (2007). All outliers were identified using the IQR x 1.5 rule, as set out by Schwertman *et al.* (2004). No physicochemical or chemical correlations were noted in relation to Mn concentrations.

Zinc (Zn)

Overall, 35 samples from SA2 were analysed for zinc of which, just under 26% of samples (n = 9) had zinc levels above the limit of detection, resulting in a mean concentration of 0.06 mg/L, and minimum and maximum values of 0.01 mg/L and 0.24 mg/L respectively. These levels are significantly higher than expected background levels of 8 $\mu g\ L^{-1}$. No significant correlations were found to exist between groundwater zinc concentration and any other measured groundwater parameters. This may be due to the small sample size associated with detectable levels of zinc. Helena *et al.* (2000) also found that no significant correlations existed between groundwater zinc and 16 measured physicochemical and chemical variables.

7.6.7. Microbial Quality

E. coli

Overall, 29.4% of wells (n = 62) tested positive for presumptive *E. coli* during the 'one-off' sampling programme, of which 67.7% (n = 42) were boreholes, with the remaining 32.3% (n = 20) being shallow hand-dug wells. Within individual study areas, *E. coli* contamination rates ranged from 45% in SA4 to 11% in SA5 (*Table 7.25*). As reported by EPA (2010c), during the period 2007 – 2009, a total of 2,718 samples were analysed for faecal coliforms at 211 groundwater monitoring stations. Positive faecal coliform counts were detected in 945 (34.8%) samples. Therefore, the overall figure of 29.4% found during this study is closely comparable. EPA (2010c) also reported that exceedances of 100 cfu/100ml occurred in 6.7% of samples, while this figure was found to be 6.1% (n = 13) in this study (*Figure 7.15*), while figures exhibiting 1-10 cfu/100ml and 11-100 cfu/100ml were 18.4% (n = 39) and 4.7%, respectively.

Table 7.25 E. coli presence, vulnerability category and source type (n = 211)

Study Area	Vulnerability	*E. coli* Present (%)	Source of *E. coli*	
			Boreholes (%)	Hand-dug wells (%)
SA1	*High*	44	59	41
SA2	*Extreme/High*	14	29	71
SA3	*High*	18	89	11
SA4	*Low*	45	65	35
SA5	*High*	11	100	0

Overall, the level of bacterial contamination encountered in SA2, SA3 and SA5 was lower than expected in areas categorised as being predominantly of *High* and *Extreme* groundwater vulnerability. Conversely, a much higher than expected level of bacterial contamination was encountered at SA4, particularly as this area is predominantly categorised as being *Low* vulnerability due to relatively thick (>10 m), low permeability subsoils. A chi-square test of independence found a significant relationship between groundwater vulnerability and *E. coli* presence (χ^2 (4) = 10.686, p = 0.03), with just under 47% of wells located in a *Low* vulnerability area (n = 23) having *E. coli* present, while figures of 24% and 26% were observed in *High* and *Extreme* vulnerability areas, respectively. Where present, a mean *E. coli* concentration of 16 cfu/100ml was found (Range 1-301 cfu/100ml).

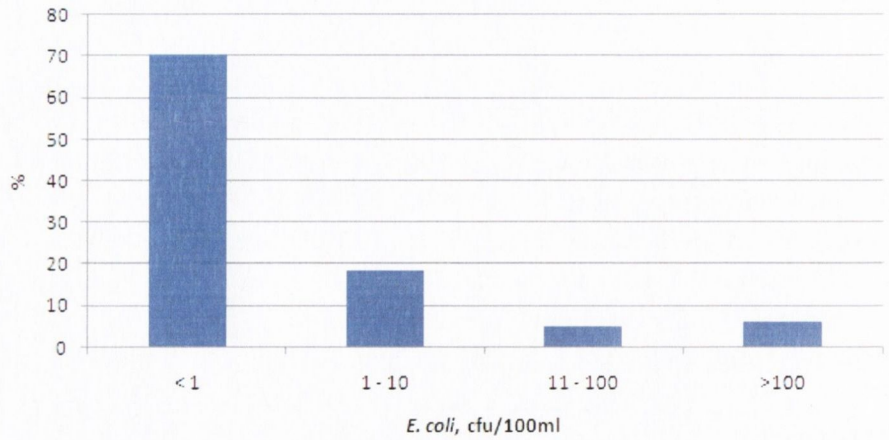


Figure 7.15 Percentage of wells with E. coli concentrations in the ranges indicated (n = 211)

No significant difference was found in relation to study area and source type (see Appendix E11) therefore, this was not adjudged to be the primary explanatory factor regarding differing levels of *E. coli* presence among study areas. Where *E. coli* were present, an ANOVA test did not find any significant difference regarding *E. coli* magnitude and study area.

As shown (*Table 7.26*), there was a significant mean difference between the presence of *E. coli* and the measured watertable depth at the time of sampling, with a mean watertable depth of 3.82 m where *E. coli* were present, compared with 5.57 m where absent ($t = 1.914$, $p = 0.05$). Furthermore, a significant mean difference was found with regard to sample temperature ($t = -2.264$, $p = 0.025$), with bacterially contaminated samples found to have a higher mean temperature (10.9°C) than those which were not contaminated (9.7°C). This is reflective of higher levels of contamination found in shallow hand-dug wells than boreholes (see Section 7.6.8). However, as independent samples t-tests did not suggest any significant difference between groundwater EC and pH, with both EC ($t = -2.567$, $p = 0.011$) and pH ($t = -2.571$, $p = 0.011$) significantly higher where *E. coli* were present, it would suggest that seasonality was an additional explanatory factor. Furthermore, the concurrence of elevated EC and pH where *E. coli* were present are suggestive of ingress of organic wastes

Although nitrates were found to be elevated where *E. coli* was present, this elevation was not found to be significant within the amalgamated dataset. However, significant differences were observed at individual study areas including SA1 and SA4, with significantly higher mean NO_3 concentrations measured in wells with *E. coli* present.

Table 7.26 Study Area 3, Groundwater parameter mean differences with respect to E. coli presence

	E. coli Presence	
	Present	**Absent**
WT Depth (m)	3.82	5.57
Sample Temperature (°C)	10.9	9.7
Sample pH	6.9	6.6
EC (µS cm^{-1})	641	517
Cl (mg L^{-1})	27.1	27.2
NO_3 (mg L^{-1})	3	2.7
K (mg L^{-1})	4.8	3.5
Mg (mg L^{-1})	7.4	7.3
Na (mg L^{-1})	15.8	21.3
NH_4 (mg L^{-1})	0.3	0.1

Additionally, an inverse correlation was found between groundwater sodium concentrations and *E. coli* concentrations (where present) ($r = -0.326$, $p = 0.035$), which is probably associated with watertable fluctuations and associated levels of cation exchange in the aquifer matrix, as opposed to being a direct relationship i.e. lower groundwater Na concentrations in concurrence with shallower watertables, indicating less protection by unsaturated subsoils and therefore potentially higher levels of *E. coli* presence and magnitude. This potential pattern was also in evidence within the amalgamated dataset, however, it was not significant ($t = 1.707$, $p = 0.09$).

A statistically significant difference was found to exist with regard to the presence of *E. coli* and the K:Na ratio magnitude ($t = -2.915$, $p = 0.004$), with a mean K:Na ratio of 0.22 in wells with no *E. coli* present, while a mean ratio of 0.48 was found where *E. coli* were present. This suggests an agricultural contamination source, as K:Na ratios in septic tank effluents are typically <0.3 (Spain, 2000). Furthermore, it would seem to provide further evidence that a K:Na ratio >0.4 may be used as an effective indicator of local, vegetable based contamination i.e. contamination by plant organic matter (e.g. farmyards, landfills).

A number of significant, albeit weak non-parametric correlations were found between *E. coli* magnitude (where present), and groundwater parameters including watertable depth ($r = -0.191$, $p = 0.014$), groundwater temperature ($r = 0.203$, $p = 0.003$), groundwater pH ($r = 0.156$, $p = 0.023$) and groundwater EC ($r = 0.247$, $p < 0.001$). The probable reasons for these relationships have already been outlined. An additional significant correlation was found between *E. coli* magnitude and groundwater K concentration ($r = 0.221$, $p = 0.01$). As previously outlined in Chapter 4 (*Section 4.8.6*), a number of studies (Daly *et al.*, 1989; Whitehead *et al.*, 1999; Krapac *et al.*, 2002), have used elevated potassium levels in private groundwater sources as being indicative of organic waste ingress in general, and farmyard and/or animal waste in particular.

Faecal *streptococci*

A limited number of one-off samples (n = 52) from SA3 and SA4 were cultured for the presence of faecal *streptococci*, with 57.7% of samples testing positive (n = 30), with a mean magnitude (where present) of 29 cfu/100ml. As shown in *Figure 7.16*, typically where faecal *streptococci* were present, they were present in relatively low concentrations (<10 cfu/100ml).

A series of independent samples t-tests found that, where *streptococci* were present in SA3 (*High* vulnerability), there was a significantly increased groundwater nitrate concentration (t = -2.995, p = 0.006), which may be an indicator of adjacent organic waste sources as a causative factor in relation to faecal *streptococci*. Additionally, statistical analysis suggests some relationship between watertable depth, sample temperature and *streptococci* presence, however, due to the small sample size, these were not significant.

No significant relationship was found to exist between *E. coli* and faecal *streptococci* presence, nor was any significant correlation found between *E. coli* and faecal *streptococci* concentration, where present. It should be noted however, that as previously outlined in *Chapter* 4 (*Section 4.6.2*), *E. faecalis* var *liquefaciens* is ubiquitous in nature with a higher persistence in water than similarly exposed faecal coliforms frequently forming a substantial proportion of the low faecal *streptococci* densities common to good quality waters (Gray, 2004). This is a potential reason for the unexpected lack of significance between faecal streptococci presence and well types i.e. higher levels of *E. coli* in hand-dug wells pointing to higher levels of contamination, whereas insignificant differences regarding faecal *streptococci* and different well types suggest a different bacterial source i.e. soils, etc.

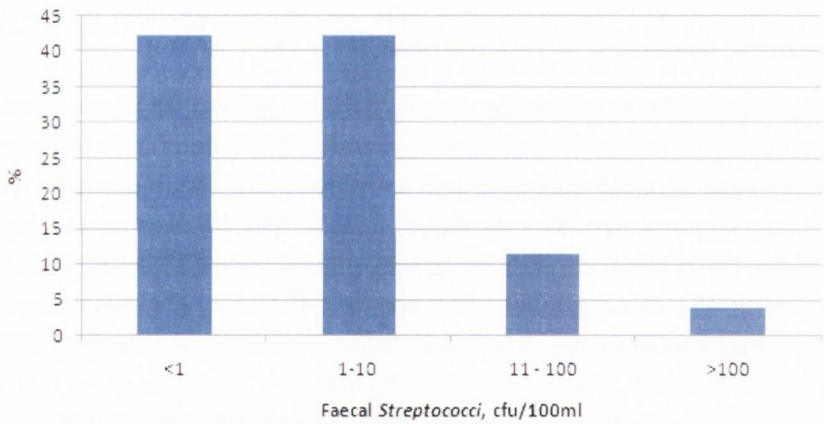


Figure 7.16 Percentage of wells with faecal Streptococci concentrations in the ranges indicated (n = 52)

Protozoa

Overall, flow cytometric analysis for the presence of *Cryptosporidium* and *Giardia* were carried out on samples from 81 wells, resulting in 4.9% of samples (n = 4) testing positive for *Cryptosporidium* spp. and no samples testing positive for *Giardia* spp. This *Cryptosporidium* spp. contamination rate is generally consistent with previous studies, for example, Hancock *et al* (1998) reported *Cryptosporidium* spp. in 5% of vertical wells, with this result reported as being consistent with earlier findings from Hibler (1988) and Rose *et al* (1991). Furthermore, Smith *et al* (1995) found a 7% rate of *Cryptosporidium* spp. presence in wells in Scotland, with Wallis *et al* (1996) finding a 6.4% rate of *Cryptosporidium* presence among 1,215 raw and treated drinking water samples.

Three positive samples were from SA1, two of which were taken from boreholes, with the remaining one from a hand-dug well. One sample from SA2 tested positive for *Cryptosporidium* spp., and was

taken from a hand-dug well. All four positive samples had evidence of medium (>10 cfu/100ml) to high (>100 cfu/100ml) *E. coli* contamination, however, due to the very small number of positive samples, no further statistical analysis was carried out.

7.6.8 Source type and groundwater quality

A number of independent samples t-tests were carried out in order to examine if measured mean chemical concentrations (where n = >100 samples) were significantly different among source types. The results are presented in *Table 7.27*.

Table 7.27 Groundwater chemical parameter means and associated source type

Well Type	Cl (mg L^{-1})	K (mg L^{-1})	Na (mg L^{-1})	NO_3 (mg L^{-1})	NH_4 (mg L^{-1})
Bored	27.8	3.2	15.3	12.4	0.18
Hand-dug	24.6	6.5	21	14.2	0.09

None of the mean chemical parameters were significantly different between source types at a 95% level, however, the observed mean differences are suggestive. Higher mean nitrate and potassium levels among hand-dug wells are suggestive of rapid recharge of agricultural/animal wastes. This is likely, not only due to the inherently higher susceptibility posed by hand-dug source design and construction, but also, as outlined earlier in this chapter (*Section 7.5.2*), hand-dug wells are typically located in agricultural land-use areas and were found to have lower grazing animal setback distances. As shown (*Figure 7.17*), nitrate concentrations from hand-dug wells typically covered a wider range than that observed at boreholes. Similar observations have been made as a result of previous studies in SA1 (Spain, 2000) and SA4 (McCole, 1992). Higher chloride concentrations among boreholes are probably due to the deeper nature of these sources and the consequently "older" nature of recharge due to longer subsurface residence (see Appendix E10).

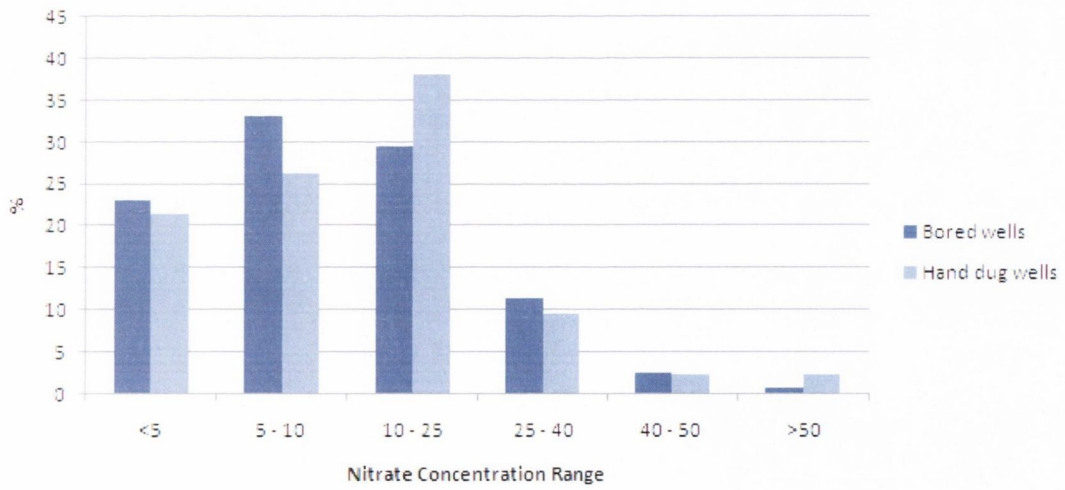


Figure 7.17 Mean nitrate concentration ranges and associated source types

The level of *E. coli* presence among individual study areas (*Table 7.25*) was significantly different (χ^2 (4) = 20.946, p < 0.001). Furthermore, the level of *E. coli* presence among boreholes and hand-dug wells was also significantly different (χ^2 (1) = 8.403, p = 0.007), with *E. coli* found in 24.8% and 47.6%

of boreholes and hand-dug wells, respectively, which would suggest that shallow, hand-dug wells are more susceptible to microbial ingress and therefore pose a greater overall human impact than deeper boreholes, as expected. Furthermore, where *E. coli* were found to be present, a higher mean colony count was observed in samples obtained from hand-dug wells, with a mean concentration of 26 cfu/100ml. A mean colony count of 13 cfu/100ml was found among samples obtained from boreholes. Furthermore, it was found that where *E. coli* colony counts were >100 cfu/100ml, the sample originated from a hand-dug well in 30% (n = 6) of cases, with a figure of 16% (n = 7) attributable to boreholes.

No association was found between well type and *streptococci* presence (see Appendix E12). Furthermore, where faecal *streptococci* were present, no significant mean concentration difference was found.

7.7 Physicochemical Cluster Analysis

Analysis was carried out to determine whether definable source subgroups existed within the entire sample (n = 211), with regard to physicochemical parameters and source type. Schwartz Bayesian Inference Criterion (BIC) two-step clustering was used for this purpose (Martin-Lopez *et al.,* 2007; Lindberg *et al.,* 2010). The three primary physicochemical parameters (groundwater temperature, pH, and EC) were used as cluster inputs, as it was considered that these would account for source type, seasonality and geological setting.

Results indicated three distinct physicochemical clusters. Results of cluster centroid analysis, simultaneous cluster mean variation and cluster-wise parameter importance using Bonferroni and Students t-test adjustment are presented in Appendix E13. Cluster 1 was characterised by low sample temperature, low sample pH and low sample EC. Cluster 2 was characterised by high sample temperature, neutral sample pH and high sample EC, while Cluster 3 was typified as being almost directly centred between Clusters 1 and 2 for all three physicochemical parameters (*Table 7.28*). A chi-square test of independence found that clusters had a significant association with well type (χ^2 (2) = 208.0, $p < 0.001$), with subsequent examination and qualification of clusters finding that Cluster 3 was comprised entirely of hand-dug wells (n = 42). Subsequent analysis (Chi-square tests) found that no discernable relationship existed between membership of Cluster 3 (i.e. hand-dug wells) and either vulnerability category or geological setting, therefore suggesting that Cluster 3 (hand-dug wells) was primarily based upon inherent design and construction characteristics.

Membership of Clusters 1 and 2 was more difficult to interpret, beyond well type. No significant associations were found between cluster membership and a number of borehole design/construction characteristics (i.e. wellhead finish, wellhead clearance, flange clearance, wellhead cover, site conditions, chamber presence), implying that borehole design was not an explanatory factor. A number of significant relationships were found regarding cluster membership and geological/hydrogeological setting, including vulnerability category (χ^2 (4) = 52.06, $p < 0.001$),bedrock type (χ^2 (13) = 140.9, $p < 0.001$), subsoil type (χ^2 (5) = 118.6, $p < 0.001$) and aquifer importance (χ^2 (4) = 53.33, $p < 0.001$). Sources in Cluster 1 were predominantly located in limestone tills (n = 71), while boreholes in Cluster 2 were typically located in Namurian tills (n = 34) and granite tills (n = 39). All boreholes located in sands and gravels (n = 4) were categorised as being in Cluster 1 (see Appendix E13). The association between cluster membership and predominant bedrock also

reflected these findings i.e. Cluster 1 associated with limestone bedrock, Cluster 2 associated with granites and Namurian rocks i.e. "non-limestones" (see Appendix E13). Although vulnerability category has a statistical association with cluster membership, with 97.3% of sources categorised as being *Low* vulnerability classified within Cluster 1 (n = 37), this was considered due to the *Low* vulnerability area (SA4) being predominated by limestone and therefore due to colinearity i.e. only 42% of Cluster 1 were *Low* vulnerability, with a further 40% and 16% being *High* and *Extreme* vulnerability, respectively. Therefore, Clusters 1 and 2 were qualified as "limestone/sandstone boreholes" and "granite/Namurian boreholes" (or "non-limestone" boreholes), respectively.

Table 7.28 BIC Cluster Characteristics

Cluster	Mean Temperature	Mean pH	Mean EC	Boreholes (n)	Hand-dug wells (n)
1	8.11	6.29	310	81	0
2	11.71	7.12	756	88	0
3	10.22	6.66	546	0	42

A chi-square test of independence found that there was statistically significant association between source clusters and *E. coli* presence (χ^2 (2) = 14.895, p = 0.001) (*Table 7.29*), suggesting that these clusters not only appropriately describe the source type and geology, but may also be used to appropriately describe source susceptibility (*Table 7.24*). No statistical link was found between *E. coli* magnitude (where present) and cluster number.

Therefore, boreholes in limestone were found to be more susceptible to bacterial contamination than boreholes in "non-limestone" areas, with no significance assigned to vulnerability category.

Table 7.29 E. coli presence among clusters

E. coli Presence/Absence	Cluster		
	Cluster 1	Cluster 2	Cluster 3
Presence	30 (34.1%)	12 (15.4%)	20 (47.6%)
Absence	58 (65.9%)	66 (84.6%)	22 (52.4%)
Total	88 (100%)	78 (100%)	42 (100%)

Furthermore, there was also a marked mean difference between clusters with regard to mean chloride concentration, with Cluster 2 ("granite/Namurian" boreholes) characterised by elevated mean chloride (Mean 35 mg L^{-1} Cl), Cluster 1 (limestone boreholes) characterised by relatively low mean chloride and Cluster 3 (hand-dug wells) approximately centred between Clusters 1 and 2 (see Appendix E13). An ANOVA test found this was significant (F = 6.672, p = 0.002). Although nitrate concentrations did not vary as much between clusters, a series of independent t-tests found that both Clusters 1 and 3 varied significantly from Cluster 2 (t = 2.094, p = 0.038). Mean well depth was found to be significantly associated with cluster membership (F (2) = 26.275, p < 0.001), however, this was expected due to inherently shallow source depths associated with Cluster 3. An independent samples t-test found a significantly different mean source depth between Clusters 1 and 2 (t = -2.224, p = 0.027), with mean measured depths of 32.6 m and 44.9 m, respectively. This is

probably due to higher levels of aquifer permeability and transmissivity within predominantly limestone aquifers, and consequently lower necessary borehole depths.

7.8 Water Quality Temporal Monitoring

7.8.1 Introduction

In order that monthly/seasonal trends relating to the chemical and microbial quality of private groundwater sources in Ireland could be investigated, fifteen domestic sources in SA1, SA2 and SA4 were selected for monthly sampling and analysis. Additionally, due to the inherent limitations of one-off spatial sampling, monthly findings could be used to predict likely contamination patterns within study areas. Monthly sampling was not carried out in SA3 (Paulstown, Co Kilkenny), as fieldwork was initiated in this study area five months before the end of the sampling programme.

Source selection for monthly sampling was based upon hydrogeological location, previous water quality results obtained during spatial (one-off) sampling, and well design characteristics. It was considered that monthly sampling should include SA4 in order that monthly variation within both high and low vulnerability areas could be investigated and compared. All sources were chosen to be geologically (bedrock, subsoil, aquifer importance) representative of their associated study areas, with all wells in SA1 and SA2 located in *High* or *Extreme* vulnerability areas, and all wells in SA4 located in *Low* vulnerability areas. Of the fifteen private groundwater sources chosen for monthly sampling, six (40%) sources had shown evidence of presumptive microbial contamination i.e. *E. coli*>0 cfu/100ml and were selected in order to investigate the persistence of microbial contamination of private groundwater sources (BM3, BM24, CB3, Mea15, Mea18, Mea21). A range of source types were chosen in order that source type as a potential causative factor of monthly contamination could be examined. In this regard, ten (67%) sources were boreholes, three (20%) were hand-dug wells, one (6.6%) was a spring well and one (6.6%) was a combined hand-dug/bored well (*Table 7.30*).

Table 7.30 Monthly sampled source selection characteristics

Well Reference	Well Type	Vulnerability Category	Bedrock Type	*E. coli* Presence (cfu/100ml)
BM1 (SA1)	Borehole	*High*	Limestone	No
BM2 (SA1)	Borehole	*High*	Limestone	No
BM3 (SA1)	Hand-dug	*High*	Sandstone	Yes (1)
BM11 (SA1)	Borehole	*High*	Sandstone	No
BM24 (SA1)	Hand-dug	*Extreme*	Sandstone	Yes (6)
CB2 (SA2)	Borehole	*High*	Granite	No
CB3 (SA2)	Borehole	*High*	Granite	Yes (1)
CB5 (SA2)	Borehole	*Extreme*	Granite	No
CB8 (SA2)	Borehole	*High*	Granite	No
CB10 (SA2)	Spring	*High*	Granite	No
Mea4 (SA4)	Borehole	*Low*	Limestone	No
Mea12 (SA4)	Borehole	*Low*	Limestone	No
Mea15 (SA4)	Combined	*Low*	Limestone	Yes (2)
Mea18 (SA4)	Hand-dug	*Low*	Limestone	Yes (4)
Mea21 (SA4)	Borehole	*Low*	Limestone	Yes (3)

Variable meteorological characteristics such as rainfall and temperature, and their effect on chemical and microbial quality of private groundwater sources were also examined using Met Eireann data, as previously outlined (Chapter 6, *Section 6.7*). All fieldwork, sample collection, processing and analysis associated with monthly sampling was carried out in the same manner as spatial sampling, as outlined in Chapter 6 (*Section 6.4*).

7.8.2 Study Area 1

Monthly sampling took place in SA1 (*High* vulnerability; limestone/sandstone bedrock) during the 20-month period August 2008 – April 2010, excluding October 2008. *Tables 7.31 & 7.32* present 20-month mean values for the primary parameters and a correlation matrix pertaining to overall parameter correlations. Non-parametric Spearmans correlations have been used, as these are commonly applied within the water resources field and show no loss of power compared to parametric tests (Helsel & Hirsch, 2002; Giannoulis *et al.,* 2005). All parameter ranges and standard deviations are provided in Appendix E14.

Table 7.31 Study Area 1, 20-month mean values

Well Number	Watertable Depth (m)	Sample Temp (^{o}C)	pH	EC ($\mu S\ cm^{-1}$)	NO_3 ($mg\ L^{-1}$)	Cl ($mg\ L^{-1}$)	NH_4 ($mg\ L^{-1}$)	Na ($mg\ L^{-1}$)	K ($mg\ L^{-1}$)
BM1	9.41	8.9	7.32	643	1.3	52	0.12	43.7	3.2
BM2	10.01	9.2	7.12	710	5.2	22.3	0.12	61.8	3
BM3	2.42	8.8	7.15	739	3.2	22.2	0.07	10.6	4.2
BM11	10.33	9.4	6.96	877	8.4	20.5	0.03	13.8	11.4
BM24	2.03	8.4	6.89	865	4.7	22.5	0.03	10.5	5.8

Table 7.32 Study Area 1, Correlation matrix of main groundwater parameters

	Water Table	Temperature	pH	EC	NO_3	Cl	NH_4	Na	K
Water Table	1.000								
Temperature	0.753**	1.000							
pH	0.295	0.281	1.000						
EC	0.161	0.147	-0.361	1.000					
NO_3	0.537*	0.564**	0.59**	-0.102	1.000				
Cl	0.575*	0.605**	0.657**	0.062	0.77**	1.000			
NH_4	0.371	0.543	-0.029	-0.086	-0.429	-0.029	1.000		
Na	0.261	-0.006	-0.345	0.442	0.17	-0.31	0.3	1.000	
K	-0.261	-0.152	-0.333	-0.455	-0.073	-0.049	-0.1	-0.479	1.000

Watertable

Monthly watertable variations are presented in *Figure 7.18*. As shown, the mean watertable level fluctuated more than expected over the sampling period, primarily due to the high level of fluctuation measured at BM1. All maximum WT depths i.e. greatest distance between ground surface and watertable, were found to occur during the period June-August apart from BM1, conversely, all minimum depths i.e. least distance between ground surface and watertable, were found to occur during the period November-December. *Figure 7.18* also clearly differentiates the

shallow hand-dug wells (BM3, BM24) from deeper boreholes (BM1, BM2, BM11). Rainfall was atypically distributed over the sampling programme duration, with particularly high precipitation during summer months (e.g. 161.3 mm during July 2009).

Due to the large level of watertable fluctuation observed at BM1 and non-conformity with expected trends i.e. lowest measured watertable depth (13.34 m below ground level) during December 2008, a small-scale pressure transducer (Diver™) study was carried out at this borehole in order to investigate watertable fluctuations at a higher (i.e. hourly) resolution and approximate the effect of pumping patterns on results. This was undertaken during the four-month period December 2009 – March 2010. An example is presented in Appendix E15, whereby over the course of the 24-hour period 2nd January 2010, although a mean measured watertable of 10.21 m was recorded, minimum and maximum values of 8.24 m and 10.69 m were recorded, with minimum watertable depths recorded during the daytime. This is suggestive of a diurnal pattern, whereby pumping to storage is occurring during the night, to avail of lower power (electricity) costs, with the watertable gradually recovering during the day. Therefore the time of day at which the watertable depth was measured was a probable source of variation.

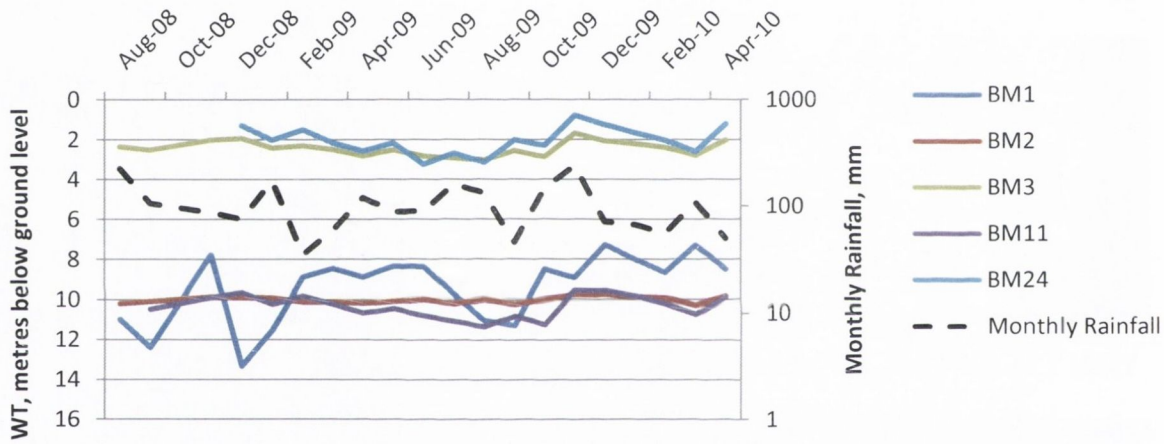


Figure 7.18 Study Area 1, Monthly measured watertable

Physicochemical Parameters

Figures 7.19, 7.20 & 7.21 below outline monthly trends found in SA1 for sample temperature, pH, and electrical conductivity (EC), respectively. As shown in *Figure 7.19*, sample temperatures followed an expected pattern, with highest temperatures occurring during the warmer summer months and lower temperatures evident during cooler winter months. These temperatures generally mirrored average daily air temperatures ($r = 0.79$; $p < 0.001$), with expected buffering of peaks occurring at extreme temperatures i.e. warmer than average air temperature during winter and cooler than average air temperature during summer. Overall an average annual groundwater temperature of 8.3°C was recorded (May 2009 – April 2010). For groundwater deeper than 15-20 m, seasonal changes are generally less than 1°C and temperature variations do not play a significant role in groundwater composition. For shallow groundwater, however, larger seasonal variations, related to warming of or cooling at the surface are common, and may be on the order of 5 to 10 degrees. Another source of temperature change in shallow groundwater is the introduction of water from the surface during high-recharge periods (Nelson, 2002; from Heath, 1989).

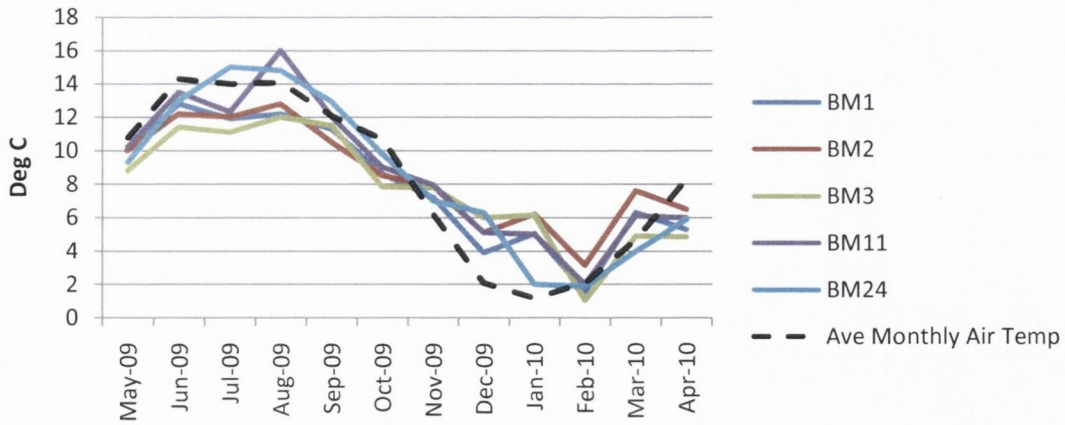


Figure 7.19 Study Area 1, Monthly groundwater temperature variation during 12-month period May 2009 – April 2010

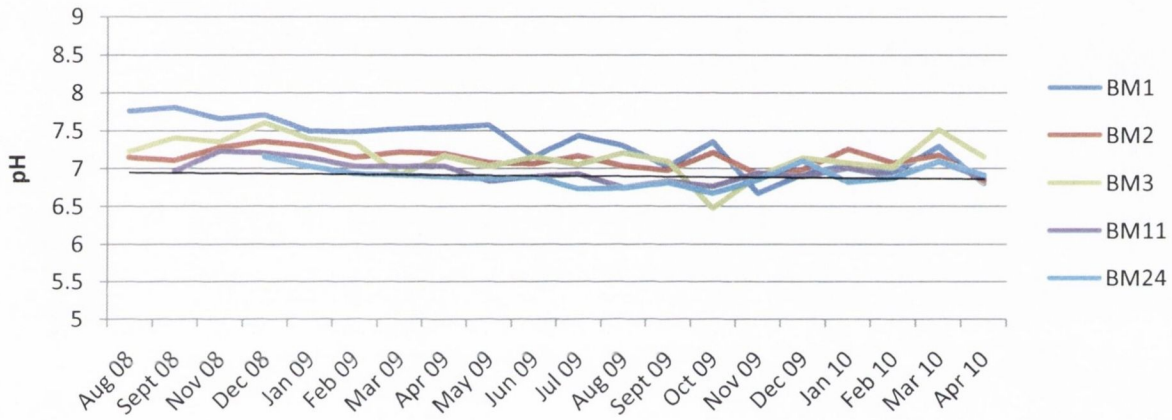


Figure 7.20 Study Area 1, Monthly groundwater pH

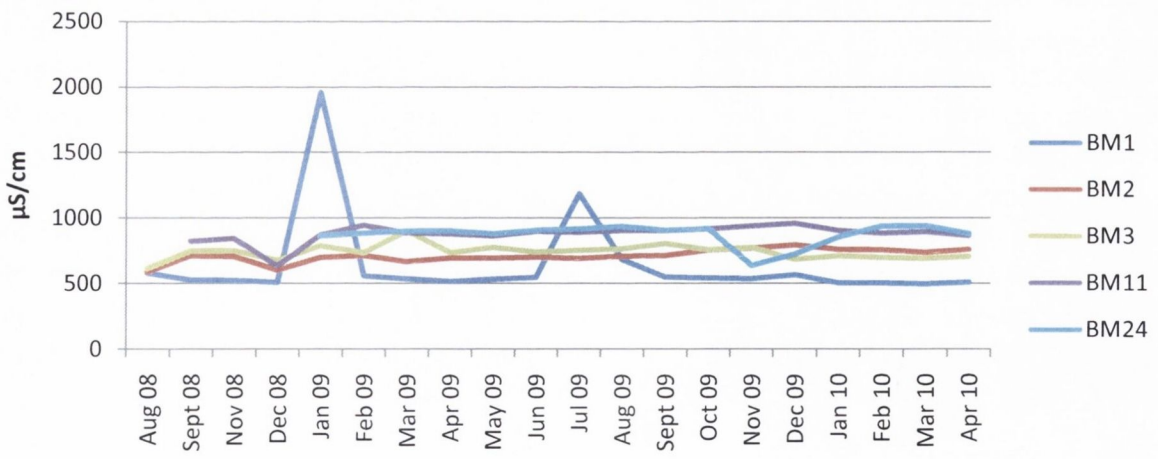


Figure 7.21 Study Area 1, Monthly groundwater EC

As shown (*Figure 7.20*), pH was stable over the sampling period with average pH values for all wells were in the range 6.8 -7.3, which is expected in limestone/sandstone areas (OCM, 2007). Groundwater EC was also found to be relatively stable over the sampling period, with mean EC values found to be significantly higher in those wells situated in glaciofluvial sands and gravels (BM3, BM11, BM24), than those situated in tills. The overall mean EC over the sampling period was 769 µS cm^{-1}, with those wells located in glaciofluvial sands and gravels averaging 830 µS cm^{-1} and those in tills averaging 677 µS cm^{-1}. From *Figure 7.21*, the most apparent deviations from the mean are those measured at BM1 during January and July 2009, which are discussed later in concurrence with measured groundwater chloride.

Chemical Parameters

All monthly samples were analysed for chloride (Cl) and nitrates (NO_3-N), with ICP-AES analysis carried out on 55% (11/20) of monthly samples. Ammonia (NH_4-N) analysis was carried out on 30% (6/20) of samples. Monthly chloride results are presented in *Figure 7.22*.

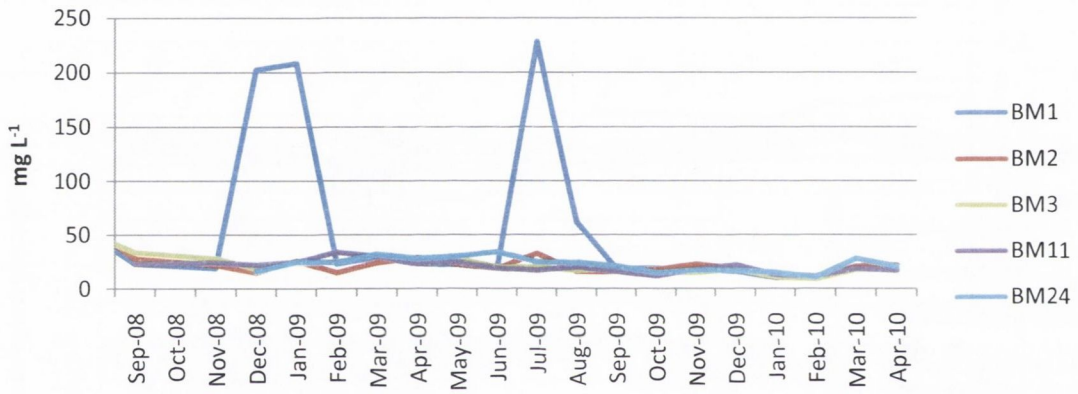


Figure 7.22 Study Area 1, Monthly groundwater Cl concentration

As shown, chloride was generally found to be stable during the sampling period with an average of 28.5 mg L^{-1} Cl over the sampling period (22.8 mg L^{-1} Cl when large peaks at BM1 excluded). This compares well with previous research, carried out by Spain (2000) at five boreholes in Co Offaly, which resulted in a mean chloride concentration of 21 mg L^{-1} Cl. The peaks observed at BM1 coincided with peaks in groundwater EC (see Figure 7.21 and Appendix E16), resulting in a statistically significant relationship between these two parameters at this sample source ($r = 0.73$, $p < 0.001$), however, this expected high positive correlation (Helena *et al.*, 2000; Aris *et al.*, 2007; Hajalou & Khaleghi, 2009) was not found at all sample sources. It is unclear why the expected EC/Cl correlation was not generally found in SA1. One reason may be a disproportionate increase in or predominance of other cations such as bicarbonate (HCO_3^-) coupled with no change in Cl, however, as no data were collected for HCO_3^-, it was not possible to test this hypothesis.

A correlation was found between chloride, water table depth and groundwater temperature, which is suggestive of seasonal chloride variation(*Table 7.32*),with a correlation between monthly pH and chloride concentration probably influenced by study area geology, recharge residence time and the resultant buffering capacity.

As previously outlined in Chapter 4 (*Section 4.8.3*), a previous study (Banks *et al.*, 1998) has found that, groundwater Cl levels tended to be spatially constant in the absence of anthropogenic activities, with Cl peaks highly suggestive of contamination by human or animal sewage. It is therefore probable that the Cl concentration peaks observed at BM1 during January 2009 and again in July 2009 are associated with contaminant ingress, with this hypothesis further strengthened by the aforementioned concurrence between monthly groundwater EC and Cl concentration peaks, which are again suggestive of contamination as opposed to analytical or sampling error. As shown (*Table 7.32*), it would appear that as the water table depth increased, so too did the average chloride concentration, which is suggestive of increasing groundwater chloride concentration due to increased groundwater residence, in concurrence with decreased dilution of groundwater chloride concentrations during drier periods. Similar patterns have been proposed in a number of previous studies (Scheytt, 1997; Vetrimurugan & Elango, 2007).

An autocorrelation function was subsequently carried out on natural log transformed average chloride concentrations, in order that an overall pattern might be found. As shown in *Figure 7.23*, in addition to the aforementioned seasonal variation, there was a slightly decreasing chloride concentration implied, although this was not found to be significant i.e. outside confidence limits.

Nitrate was found to have a greater range than chloride and greater potential for fluctuation over short periods (*Figure 7.24*). Monthly NO_3 sampling in SA1 resulted in a total sample average of 18.5 mg L^{-1} NO_3. This agrees with EPA groundwater monitoring data, with approximately 45% of groundwater samples in the range 10 – 25 mg L^{-1} NO_3 over the period 1995-2009 (EPA, 2010c). Additionally, a previous study (Spain, 2000) reported a mean NO_3 concentration of 21.2 mg L^{-1} from boreholes in Co Offaly. Significant spatial variation was seen to exist in nitrate levels, with BM1 and BM2 in particular (two sources located approximately 120 m apart) displaying monthly averages of 5.4 and 21.9 mg L^{-1} NO_3, respectively, indicating localised sources of nitrate (Pacheco & Cabrera, 1997).

As shown in *Table 7.32*, a significant correlation was found between average monthly nitrate and chloride concentrations ($r = 0.77$, $p < 0.001$), similar to that found by Spain (2000), Tang *et al* (2004a) and Tang *et al* (2004b). This implies that groundwater nitrate concentration is probably, like chloride, associated with effective recharge rates. As for chloride, nitrate concentrations were also found to be significantly correlated with watertable depth, groundwater temperature and groundwater pH (*Table 7.32*), which strengthen this assertion. It is important to note that during the 20-month sampling period, only three samples (BM11, December 2008; BM11 February 2009; CB24, July 2009) had nitrate concentrations >50 mg $L^{-1}NO_3$, which would suggest that although nitrates may be used as a potential indicator of groundwater contamination by surface water ingress, it is unlikely to present a public health concern within this study area at this time. A mean concentration of 32.7 mg $L^{-1}NO_3$ was measured at BM11 over the 20-month sampling period, which as illustrated in Chapter 4 (*Section 4.8.2*) is suggestive of an organic waste source nearby i.e. farmyard, grazing animals, septic tank, etc.

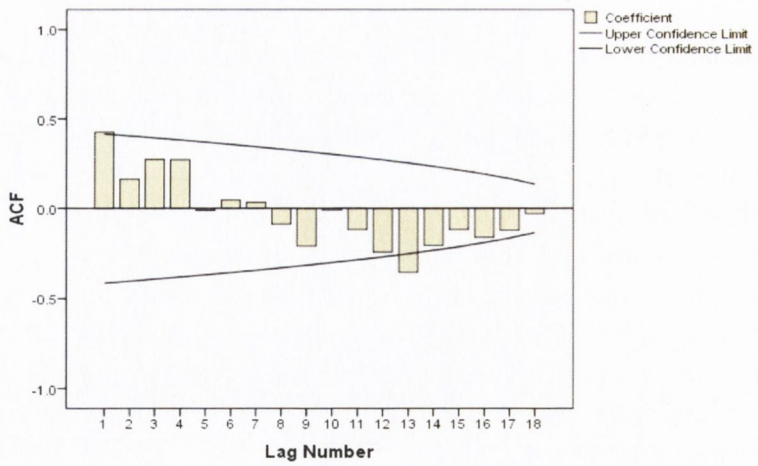


Figure 7.23 Study Area 1, Autocorrelation function of transformed (ln) mean monthly Cl concentration

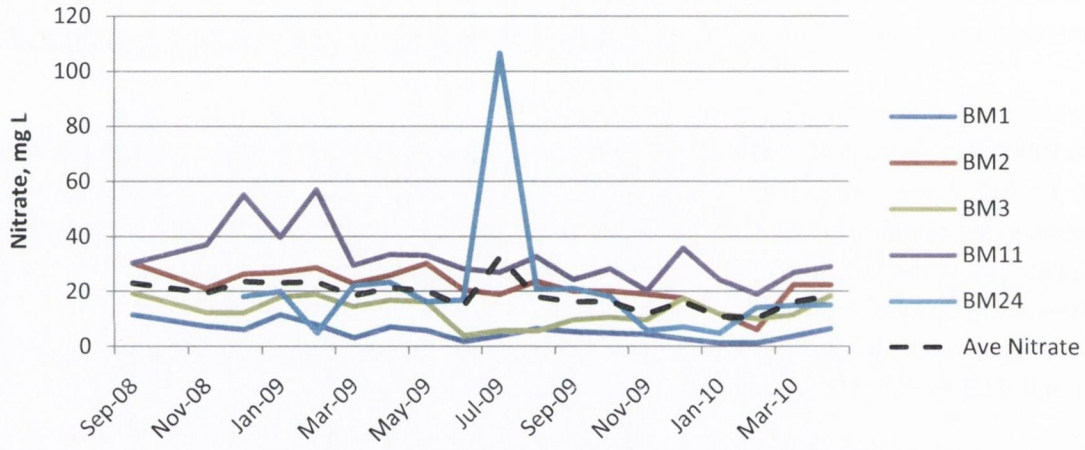


Figure 7.24 Study Area 1, Monthly groundwater NO_3 concentration

As shown (*Figure 7.24)*, although there was a slight decreasing trend overall, agreeing with the most recent national groundwater monitoring data and analysis (EPA, 2010c), no significant seasonal trend was apparent, although the overall mean NO_3 concentration was seen to a exhibit a slight increase during the period November-December, during both sampling years. This peak in nitrates in during early/mid winter is typical, due to higher total (and effective) precipitation and lower vegetative uptake of nitrogen (Schulte *et al.*, 2006). The significant mean nitrate peak during July 2009 was due to the large peak measured at BM24.

Microbiological Parameters

All monthly samples from SA1 were analysed for *E. coli* using methods previously outlined in Chapter 6 (*Section 6.5.3*). Additionally, faecal *Streptococci* analysis was undertaken on all samples during the period January-April 2010. Outline statistics relating to presumptive *E. coli* are presented in *Table 7.33*.

Table 7.33 Study Area 1, Monthly E. coli statistics

Well Number	Months present	Percent present (samples)	Mean, when present (cfu/100ml)
BM1	4/20	20	23.5
BM2	2/20	10	10.5
BM3	11/20	55	53.3
BM11	3/19	15.8	22.6
BM24	13/17	76.5	47.3

Figure 7.25 outlines the frequency and concentrations of presumptive *E. coli* found at SA1 during monthly monitoring. Notably, *E. coli* were found in all wells at least once during the study period. The shallow hand-dug groundwater sources (BM3 and BM24) showed the highest percentage of faecal contamination, with *E. coli* being enumerated in 55% and 76.5% of monthly analyses, respectively. Deeper boreholes (BM1, BM2, BM11) were less frequently contaminated, with 20%, 10% and 15.8% respectively. All boreholes showed a median contamination frequency of zero i.e. *E. coli* were absent more often than not, however BM3 and BM11 (hand-dug wells) had median contamination frequencies of 1 and 6 respectively, and were therefore found to be contaminated with *E. coli* more often than not. Contamination was generally evidenced throughout the year, however, rates tended to be lowest during late winter and early spring with peaks during late summer/early autumn and early/mid winter. As no significant association was found between (total) monthly precipitation, which is suggestive that apparent seasonality is driven by agricultural cycles. Moreover, as a number of individual measured K:Na ratios in SA1 exceeded FC:FS 0.4, with an overall mean ratio of 0.42, this provided evidence that contamination, where present, is of agricultural origin, particularly at BM3, BM11 and BM24, where mean K:Na ratios of 0.5, 0.82 and 0.56 were measured, respectively (see *Section 4.8.8*).

The significant nitrate concentration increase observed at BM24 during July 2009 was accompanied by measured *E. coli* presence (4 cfu/100ml), which is suggestive of contamination by waste organic matter, particularly as this source was a shallow, hand-dug well. However, it should be noted that, as shown in *Table 7.33*, *E. coli* were present at BM24 in over 75% of monthly samples, and therefore inference is difficult. Furthermore, contrary to expectation, no correlation was found between nitrate concentration and *E. coli* (cfu/100ml) within the total SA1 monthly dataset, nor was there a significant mean nitrate difference suggested where *E. coli* were present.

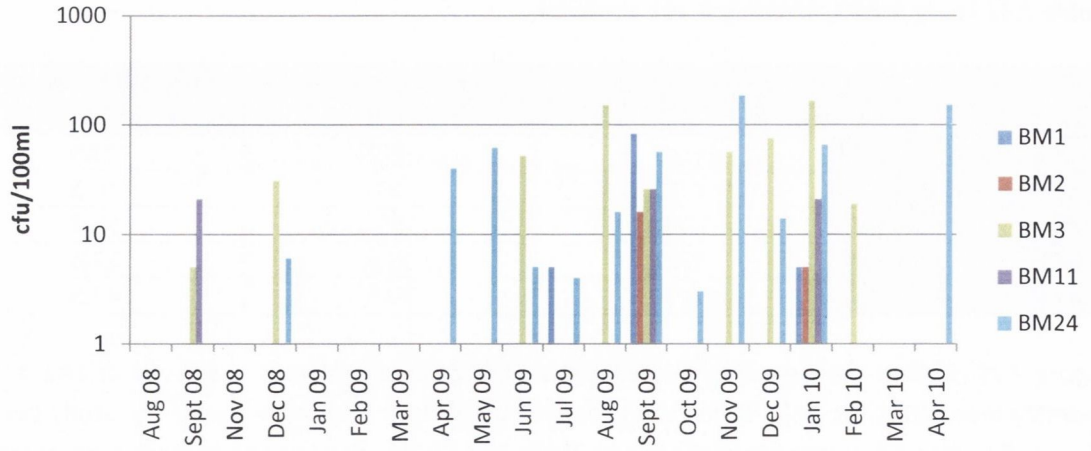


Figure 7.25 Study Area 1, 20-month E. coli frequency and concentration

Likewise, no significant associations were found between mean chloride concentrations and *E. coli* presence or magnitude, although the notable chloride concentration peak measured at BM1 during July 2009 was accompanied by *E. coli* presence (5 cfu/100ml). As outlined in Chapter 4 (*Section 4.8.3*), human and animal sewage contains large concentrations of chlorides (Hudak, 2003b) and this is therefore suggestive of sewage ingress, particularly from agricultural sources during late summer/early autumn (i.e. animals grazing).

Overall, faecal *Streptococci* (FS) were found in three out of five wells during the 4-month period January-April 2010. These positive wells were BM1, BM2 and BM3. Only one of these (BM1) showed evidence of FS on more than one occasion. Both BM1 and BM2 showed evidence of FS in January 2010, resulting in FC/FS ratios of 1.66 and 2.5, respectively. As outlined in Chapter 4 (*Section 4.7.1*), this may imply a human faecal source of contamination.

Influence of precipitation

Meteorological data collected from Met Eireann during the temporal sampling period were used to examine the relationship between groundwater contamination and precipitation in the 24-hour, 48-hour, 120-hour and 30-day period prior to sampling (*Figure 7.26*). The rainfall periods chosen were based upon a previous study by Howard *et al* (2003).

No significant correlations were found between average monthly chloride or average monthly nitrate concentrations and total precipitation rates. This was also found to be the case when individual chloride and nitrate concentrations and the respective total precipitation rates were analysed. Examination of *E. coli* data showed that while no correlations were evident in the case of *E. coli* concentration and precipitation period, both hand-dug wells (BM3 and BM24) exhibited positive correlations between *E. coli* concentration and the monthly precipitation rate, although neither of these were statistically significant (BM3: $r = 0.43$, $p = 0.059$; BM24: $r = 0.364$, $p = 0.15$). This suggests an association between *E. coli* contamination and 30-day precipitation rates at these two wells i.e. contamination occurring due to subsurface ingress as opposed to direct rapid ingress at the surface.

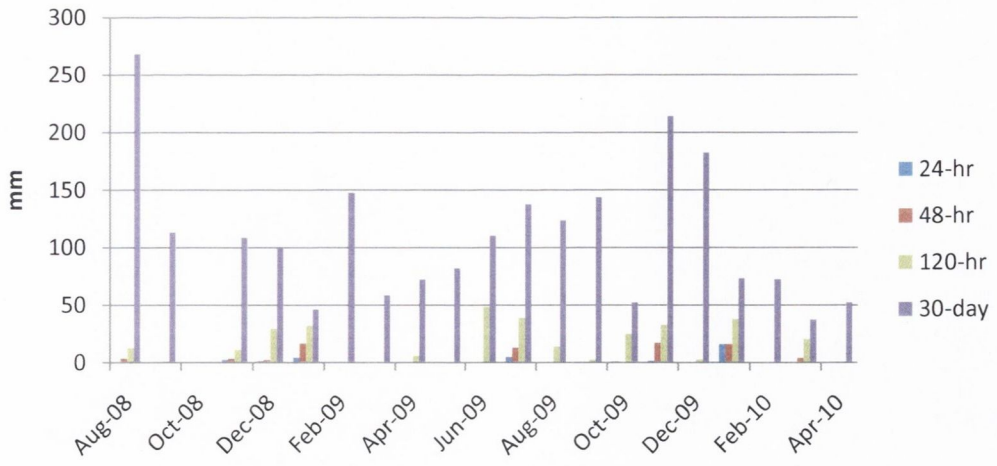


Figure 7.26 Study Area 1, Precipitation depth in the 24-hour, 48-hour, 120-hour and 30-day period prior to sampling

Additionally, data pertaining to evapotranspiration rates and soil moisture deficits were collected from the closest Met Eireann synoptic stations to the study area, so that effective precipitation (recharge) rates could be calculated. Effective precipitation calculations and monthly rates over the sampling period are presented in Appendix E17. As expected, monthly effective precipitation was comparable with total precipitation during winter months, when soil moisture deficits and evapotranspiration rates were typically low, with most recharge occurring during these periods. A series of correlations were undertaken between 30-day effective precipitation and watertable depth, nitrate concentration, chloride concentration and *E. coli* magnitude. Although, as expected there was a negative correlation between watertable depth and effective precipitation rate i.e. increased recharge resulting in decreased watertable depth, this correlation was not significant. Likewise, no significant correlation was found between effective precipitation and either nitrate or chloride. One source (BM24; hand-dug well) had a significant correlation between effective precipitation and *E. coli* magnitude ($r = 0.515$, $p = 0.034$), suggesting that *E. coli* contamination was taking place due to rapid recharge at this shallow hand-dug source. However, correlations between effective precipitation and *E. coli* magnitude were lower at the remaining four monitored sources, than correlations between total precipitation and *E. coli* magnitude. This may suggest direct surface ingress and/or rapid recharge at wellhead (due to poor design or construction i.e. lack of sanitary seal) being the primary mode of contaminant transport at these sources i.e. *E. coli* magnitude had a higher association with total rainfall than recharge.

7.8.3 SA2: Crossbridge, Co Wicklow

Monthly sampling took place in SA2 (*High/Extreme* vulnerability; Granite bedrock) on a monthly basis during the 21-month period July 2008 – April 2010, with samples taken every month excluding October 2008, resulting in 21 samples from each source overall. Four boreholes and one shallow spring well were chosen for monthly sampling. Sampling did not take place at the spring well (CB10) during November 2009 as due to low temperatures this shallow groundwater source had frozen and therefore a sample could not be taken. It should be noted that due to the nature of this source (i.e. spring well in highland area), there is expected to be a high level of surface water – groundwater

interaction present. Mean parameter values and correlations are presented in *Tables 7.34 & 7.35*, respectively.

Table 7.34 Study Area 2, 21-month mean values

Well Number	Watertable Depth (m)	Sample Temp (°C)	pH	EC ($\mu S\ cm^{-1}$)	NO_3 ($mg\ L^{-1}$)	Cl ($mg\ L^{-1}$)	NH_4 ($mg\ L^{-1}$)	Na ($mg\ L^{-1}$)	K ($mg\ L^{-1}$)
CB2	2.62	8.87	5.47	249	4.6	20	0.1	11.2	7.67
CB3	2.89	8.69	5.76	263	3.5	26.4	0.15	14.5	17.1
CB5	1.19	9.05	5.73	156	2.3	16.7	0.05	8.87	2.77
CB8	1.36	9.03	5.94	302	3.7	23.6	0.12	8.66	4.11
CB10	0.6	8.53	5.96	75.4	1.3	12.7	0.04	6.79	1.78

Table 7.35 Study Area 2, Correlation matrix of main physicochemical parameters

	Water Table	Temperature	pH	EC	NO_3	Cl	NH_4	Na	K
Water Table	1.000								
Temperature	0.523*	1.000							
pH	-0.224	-0.715**	1.000						
EC	0.165	0.141	-0.149	1.000					
NO_3	0.008	0.45*	-0.108	0.204	1.000				
Cl	0.251	0.163	0.099	0.844**	0.386	1.000			
NH_4	0.3	0.2	0.2	-0.1	-0.3	-0.2	1.000		
Na	-0.943	-0.029	-0.2	0.257	-0.14	-0.029	0.5	1.000	
K	-0.164	0.474	-0.491	-0.127	0.14	0.018	-1**	0.643	1.000

Watertable

As shown (*Figure 7.27*), water table depths over the period, although highly variable, occurred within a relatively narrow range (see Appendix E14), with a maximum depth of 4.38 m at CB3 in August 2009, compared with a maximum water table depth at SA1 of over 13 m. This high level of variability was likely due to domestic usage i.e. pumping to storage (previously discussed for BM1), and is reflected in the lack of correlation between mean watertable depth and monthly rainfall. The average watertable depth in SA2 followed a similar expected pattern as that evidenced in SA1 i.e. lower watertable during drier summer months with recovery beginning in mid to late autumn reaching a peak in early/mid winter (see Appendix E18), although, as for SA1, rainfall was atypically distributed during the monitoring period e.g. mean monthly rainfall of 180 mm during the period July – September 2008.

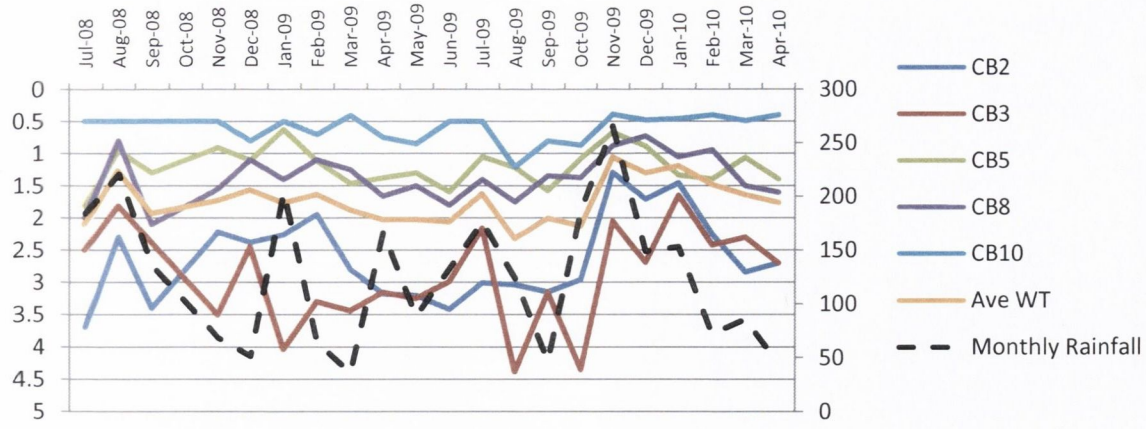


Figure 7.27 Study Area 2, Monthly measured watertable

Physicochemical Parameters

Monthly patterns for sample temperature, pH and EC are shown below in *Figures 7.28, 7.29 & 7.30.*

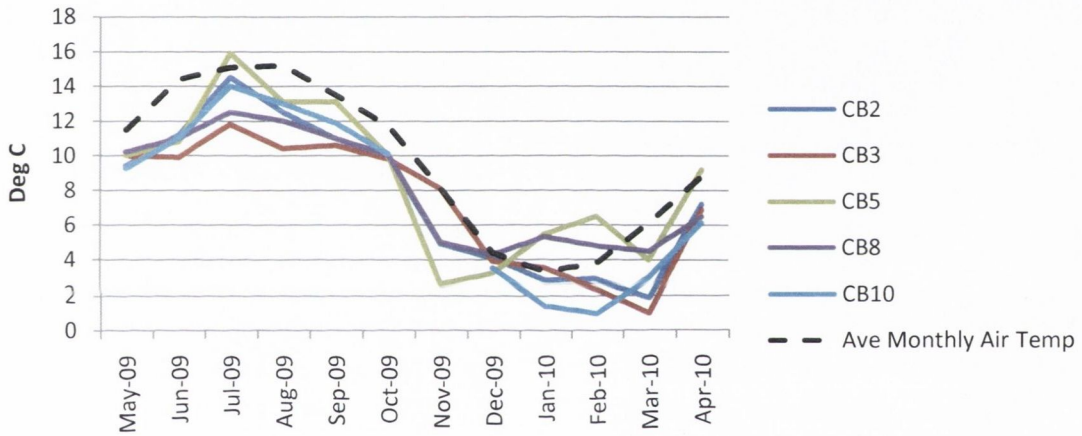


Figure 7.28 Study Area 2, Monthly groundwater temperature (May 2009 – April 2010)

As shown (*Figure 7.28*), sample temperatures measured at SA2 throughout the monthly sampling period followed a similar pattern to that encountered in SA1, with peak temperatures recorded during summer months, and lowest recorded during winter months. Significant correlations were found between average monthly sample temperature and average monthly air temperature ($r= 0.96$, $p < 0.001$) (*Table 7.35*) and average monthly sample temperature and average monthly water table depth ($r = 0.54$, $p = 0.012$). A 12-month (May 2009 – April 2010) average groundwater temperature of 7.85°C was recorded.

pH was found to be relatively stable throughout the sampling period, with a mean pH of 5.8 and an overall range of 5 - 6.8 (*Figure 7.29).* This pH mean and range are lower than those for SA1 and generally fall within expectations (OCM, 2007), due to a lack of buffering and tend to therefore remain around pH 5.6 – 6.7 (natural pH of contaminant free rain and snow i.e. principal groundwater source). Furthermore, due to relatively high water table depths in SA2, even less buffering (i.e. decreased mineralisation at shallower depths) is likely to occur due to low residence time of

groundwater in the subsurface (Hill & Neal, 1997). A general trend of increasing pH during November – March with pH decrease starting to occur during April/May was observed, in concurrence with a significant correlation between pH and groundwater temperature ($r = -0.715$, $p < 0.001$), suggesting an annual pattern (see Appendix E18). Moreover, an inverse correlation was found between mean monthly pH and total monthly rainfall ($r = -0.475$, $p = 0.03$). Likewise, Siegel *et al.* (1995) found that increased pH (<4.5 to 6.0) occurred during periods of decreased rainfall, due to decreased recharge.

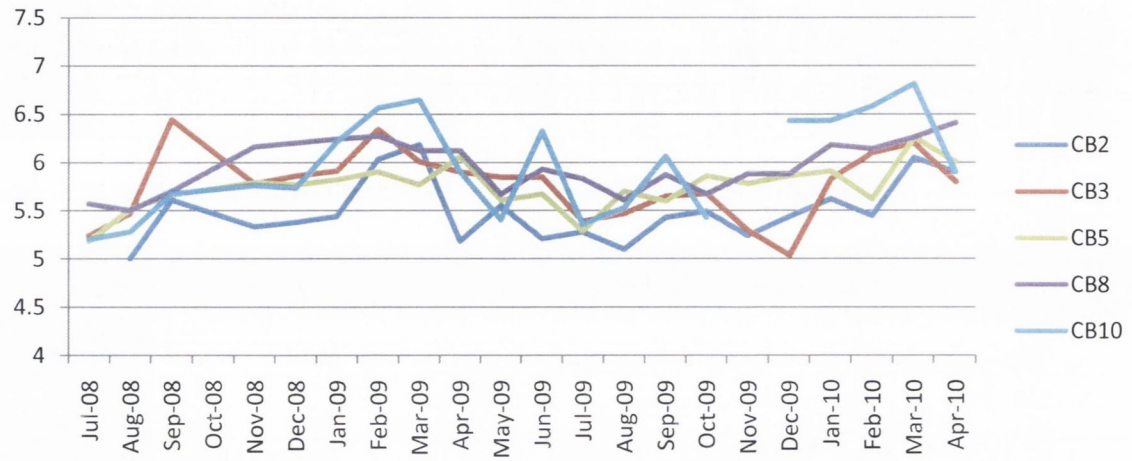


Figure 7.29 Study Area 2, Monthly groundwater pH

The average overall EC over the monthly sampling period was 220 µS cm^{-1} and generally EC was seen to be relatively stable on a well by well basis, apart from the obvious peaks encountered at three of the five wells during December 2009 (*Figure 7.30*). As these three wells (CB5, CB8, CB10) were within a close proximity relative to the remaining two (CB2, CB3), both of which showed little or no deviation from background levels, equipment error was deemed unlikely. Low mean EC values in SA2 were expected, as a recent OCM (2007) report cites low median groundwater EC values in granite bedrocks. Moreover, high average watertables and consequent lower subsurface residence times may also contribute to low EC values. Correlations between pH and EC were found to be insignificant, with similarly insignificant correlations found between EC and sample temperature.

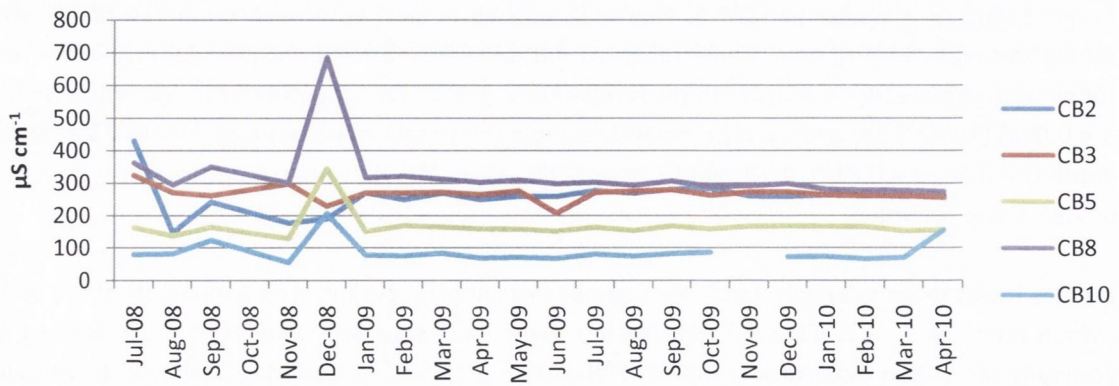


Figure 7.30 Study Area 2, Monthly groundwater EC

Chemical Parameters

As for SA1, the main chemical analysis routinely carried out were NO_3 and Cl with all monthly samples analysed for these parameters (21 samples from each source excluding November 2009 at CB10). NH_4-N analysis was carried out on all monthly samples during the period October 2009 – April 2010. ICP-AES analysis was carried out on 13 monthly samples at each sample source. Outline statistics for individual monitored sources are presented in Appendix E14.

Monthly chloride results are presented in *Figure 7.31*. The mean groundwater Cl concentration found in SA2 was 19.7 mg L^{-1}. A significantly higher average overall concentration of 26.4 mg L^{-1} was observed at CB3, and when excluded, resulted in a mean concentration of 17.8 mg L^{-1}. As shown (*Figure 7.31)*, CB10 samples returned a consistently low Cl concentration (16 mg L^{-1}) which was expected due to its shallow nature and likely high surface water/groundwater interactions at this source.

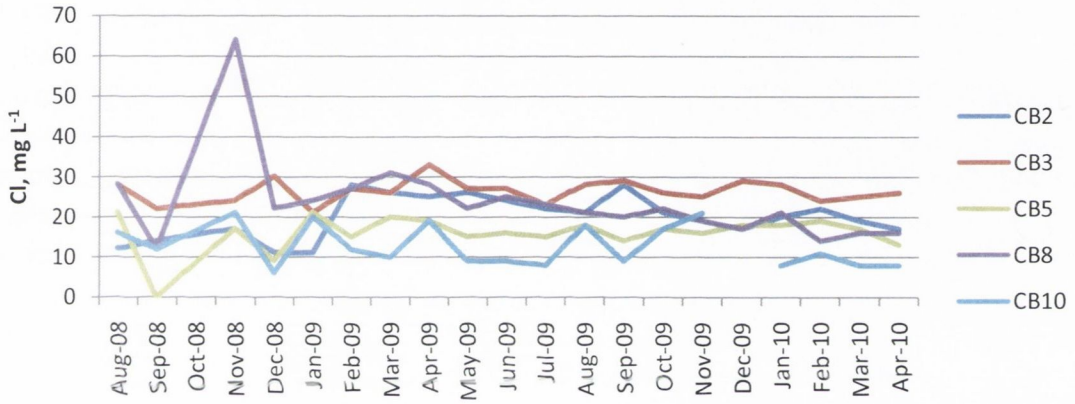


Figure 7.31 Study Area 2, Monthly groundwater Cl

An autocorrelation function found that no significant seasonal trend was apparent with regard to groundwater chloride (see Appendix E18), however, lower average values were measured during the mid to late summer period, with higher values occurring during late winter and early spring, probably due to increased total and effective precipitation as a major source, particularly in high watertables, as typically observed in SA2 (Hem, 1992). However, unlike SA1, no significant correlations were found between average chloride concentration and average water table depth, average sample temperature or average sample pH. Although individual sources exhibited higher correlations between chloride concentrations and the primary physicochemical parameters, none were found to be statistically significant.

Nitrate concentrations were found to have a high degree of variability in SA2, coupled with a greater overall potential for fluctuation. A total average nitrate concentration of 13.7 mg L^{-1} NO_3 was found in SA2, with CB10 found to have a significantly lower average concentration than other monitored sources (*Figure 7.32*). It is considered that this is likely due to its shallow construction, with a lower overall nitrate concentration in SA2, compared with SA1, probably a result of less intensive agriculture in the area and a lower overall septic tank density. The results are again in agreement with recent national groundwater monitoring figures, with the majority of samples (n = 72) in the

range 10 – 25 mg L^{-1} NO_3 (EPA, 2010c). A 12-month average groundwater nitrate concentration of 12.4 mg L^{-1} NO_3 was recorded (May 2009 – April 2010).

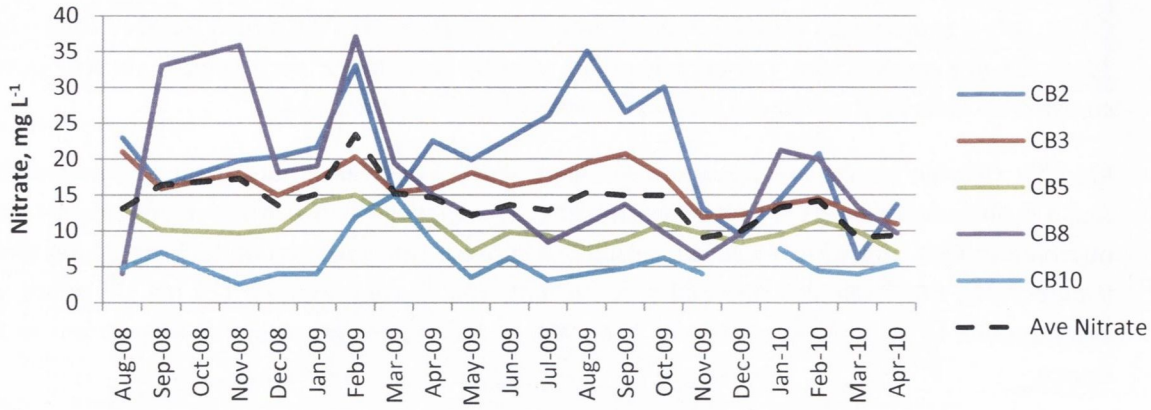


Figure 7.32 Study Area 2, Monthly groundwater NO_3

Some level of potentially significant spatial variation was noted in SA2 with regard to groundwater nitrate concentrations, for example CB5, CB8 and CB10 were all located within approximately 500 m of each other, however, average monthly nitrate concentrations of 10.2 mg L^{-1} NO_3, 16.3 mg L^{-1} NO_3 and 5.7 mg L^{-1} NO_3 were found at these sources, respectively. The low average nitrate concentration found at CB10 is due to its being a shallow spring well. Furthermore, as CB8 is located within 20 m of an operating farmyard, while CB5 is located on a residential property (although surrounded by low stocking rate grazing land), it is therefore possible that some farmyard runoff ingress is occurring at CB8 and therefore the cause of higher average nitrate concentrations.

As shown in *Table 7.35*, a correlation was found between average monthly nitrate and chloride concentrations, although not found to be significant. Nitrate concentrations were found to be significantly correlated with groundwater sample temperature ($r = 0.45$, $p = 0.04$), which would suggest a seasonal nitrate pattern, with nitrate concentration typically decreasing during winter months and gradually increasing during mid to late spring and peaking during mid to late summer. Unlike that found in SA1, there was no apparent relationship between nitrate and water table depth in SA2 (*Table 7.35*). A peak was noted in February 2009, however, as shown in the next section, this did not occur in concurrence with bacterial contamination, it is therefore probable that this nitrate peak may be due to spreading of fertilisers (straight/compound/complex) during or directly before high rainfall conditions. Overall, a slight downward trend was noted, which is in agreement with recent findings (EPA, 2010c). During the 20-month sampling period, no samples had a nitrate concentration >50 mg/L NO_3, implying that nitrates are unlikely to present a current public health concern within this study area.

Microbiological Parameters

As shown in *Table 7.36*, four of the five monitored wells had evidence of microbial contamination at least once during the 21-month sampling period, with CB5 being the only source which did not exhibit microbial contamination. Overall, of those wells exhibiting microbial contamination, a mean

E. coli presence of 40% (i.e. 8 months out of 20) was recorded, with an overall mean *E. coli* concentration of 6.75 cfu/100ml.

Figure 7.33 outlines the frequency and concentrations of presumptive *E. coli* found at SA1 during the monthly study. As shown in both *Table 7.36* and *Figure 7.33*, CB3 exhibited both the highest frequency and highest mean concentration of *E. coli* contamination. It should be noted that CB3, although not situated directly adjacent to a potential hazard source, was poorly located, designed and constructed, with no wellhead protection in place. An autocorrelation function found that no obvious seasonal pattern existed with regard to microbial groundwater contamination in this area (see Appendix E18), although, contamination seemed to be more frequent during summer months, possibly due to grazing animals. However, it was noted that precipitation levels were atypical during the sampling programme duration, with wetter than average summers and dryer than average winters. A significant correlation between mean *E. coli* concentrations and monthly rainfall in SA2 ($r = 0.503$, $p = 0.017$), suggests than rainfall is a primary contamination driver. A series of independent samples t-tests reinforced this apparent trend, with higher mean watertables measured where *E. coli* were present at CB2, CB3 and CB8.

No significant non-parametric correlation was found between mean monthly nitrate concentration and average *E. coli* (cfu/100ml) ($r = 0.386$, $p = 0.092$), however, individual sources did exhibit significant mean nitrate differences where *E. coli* were present, with a higher mean nitrate concentration typically found where *E. coli* were present ($t = -2.587$, $p = 0.019$). All sources (excluding CB5) exhibited positive correlation coefficients between *E. coli* concentration and nitrate concentration, however none of these were significant at a 95% level. Likewise, no statistically significant relationship was found between *E. coli* concentration and chloride concentration at any sources, although all sources (excluding CB5) exhibited positive correlation coefficients.

Table 7.36 Study Area 2, Monthly E. coli presence statistics

Well Number	Months present	Percent present (samples)	Mean, when present (cfu/100ml)
CB2	3/21	14.3	1
CB3	12/21	57.1	20
CB5	0/21	0	0
CB8	8/21	38.1	4
CB10	9/20	45	2

As shown (Figure 7.33), four out of five monitored sources exhibited evidence of bacterial contamination during the monitoring period, with elevated K:Na ratios noted at all four sources when *E. coli* were present (see Appendix E19), which is suggestive of agricultural contamination sources, as expected in a predominantly rural area such as SA2. However, as noted in Chapter 4 (*Section 4.8.8*), caution should be used for the interpretation of K:Na ratios in SA2, due to the predominance of granite bedrocks in this area and associated high K levels in these bedrock types, for example, the mean measured K:Na ratio in SA2 was 0.499, which exceeds the 0.4 proposed by Daly (1994b) as being indicative of local, vegetable based contamination.

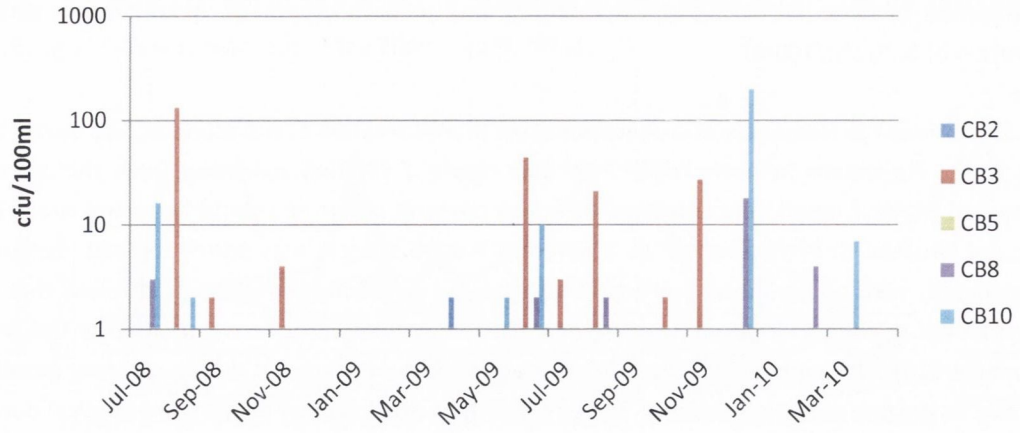


Figure 7.33 Study Area 2, 21-month E. coli frequency and concentration

Faecal *Streptococci* (FS) were found in all five wells during the 4-month period January-April 2010, with three of the five wells (CB5, CB8, CB10) testing positive for FS on more than one occasion. Measured FC:FS ratios were typically <2, which is evidence of the predominance of animal wastes (Section 4.7.1), however, an FC:FS ratio of 2.3 was measured at CB11 in March 2010, which is suggestive of the predominance of human wastes (Gleeson & Gray, 1997).

Influence of precipitation

As shown in *Figure 7.34*, precipitation rates in the four outlined periods prior to monthly sampling were similar to those observed at SA1. No significant correlations were found between average monthly chloride or average monthly nitrate concentrations and precipitation rates (24-h, 48-h, 120-h, 30-d), although correlation coefficients exhibited increasing correlations with increasing rainfall period duration i.e. increased ingress over increased recharge times. Likewise, no significant correlations were noted with regard to total precipitation over the precipitation periods and mean *E. coli* concentration. One source (CB3), exhibited a significant correlation between *E. coli* concentration and the 30-day precipitation depth ($r = 0.566$, $p = 0.009$), with the remaining four sources indicating no notable effect. In relation to *E. coli* presence, a series of independent t-tests found that, while no significant associations were found, typically higher rainfall during the 120-hour and 30-day periods prior to sampling was found where *E. coli* were present.

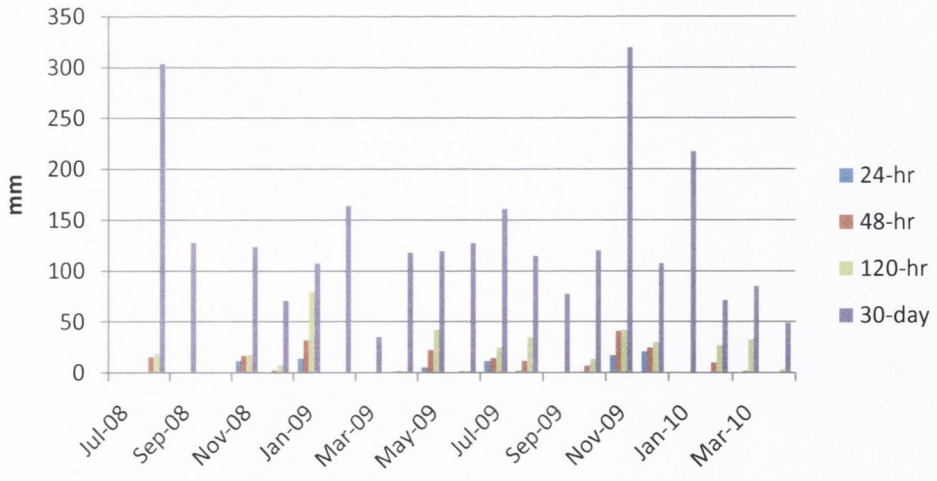


Figure 7.34 Study Area 2, Precipitation depth in the 24-hour, 48-hour, 120-hour and 30-day period prior to sampling

As for SA1, effective precipitation (potential recharge) rates were calculated on a monthly basis for SA2. A series of analysis found that the average watertable among boreholes was significantly associated with effective precipitation ($r = -0.616$, $p = 0.005$), however, this was not the case at the shallow hand-dug source, which may be indicative of significant groundwater/surface water interaction at this source. No significant correlations were found between effective precipitation and either chloride or nitrate concentrations at boreholes or the hand-dug source. With regard to bacterial contamination, a significant inverse correlation was noted between *E. coli* magnitude and effective precipitation at CB2 (borehole) ($r = -0.506$, $p = 0.023$). No significant correlations were noted between *E. coli* magnitude and effective precipitation at the other sources.

7.8.4 SA4 Kilclone, Co Meath

Monthly sampling took place in SA4 (*Low* vulnerability, Limestone bedrocks) on a monthly basis during the 8-month period September 2009 – April 2010, with groundwater samples collected every month during this period. *Tables 7.37 & 7.38* present 8-month mean values for the primary monthly parameters and a correlation matrix pertaining to overall parameter correlations. As for SA1 and SA2, all parameter ranges and standard deviations are provided in Appendix E14.

Table 7.37 Study Area 4, 8-Month mean values

Well Number	Watertable Depth (m)	Sample Temp (^{o}C)	pH	EC ($\mu S\ cm^{-1}$)	NO_3 ($mg\ L^{-1}$)	Cl ($mg\ L^{-1}$)	NH_4 ($mg\ L^{-1}$)	Na ($mg\ L^{-1}$)	K ($mg\ L^{-1}$)
Mea4	9.74	7.5	7.16	633	1.13	26.2	0.07	17.3	4.08
Mea12	2.27	8.2	7.06	679	1.51	27.5	0.14	15.8	1.67
Mea15	2.49	7.91	7.09	686	2.26	19.2	0.1	12.3	2.9
Mea18	1.17	7.93	7.04	770	3.01	28	0.04	15.8	19.9
Mea21	1.44	6.88	7.15	670	1.28	24	0.05	14.7	2.14

Table 7.38 Study Area 4, Correlation matrix of main physicochemical parameters

	Water Table	Temperature	pH	EC	NO_3	Cl	NH_4	Na	K
Water Table	1.000								
Temperature	-0.262	1.000							
pH	-0.19	-0.095	1.000						
EC	0.595	0.333	0.5	1.000					
NO_3	-0.452	-0.452	0.167	-0.048	1.000				
Cl	0.252	-0.228	0.527	0.743*	0.084	1.000			
NH_4	-0.143	0.357	0.821*	0.286	-0.393	0.679	1.000		
Na	0.886*	0.029	-0.086	0.6	-0.6	0.543	0.6	1.000	
K	0.771	0.2	0.029	0.429	-0.657	0.314	0.5	0.943**	1.000

Watertable

As shown in *Figure 7.35*,watertable depths were both relatively shallow and constant in SA4, apart from Mea4, which was significantly deeper and exhibited greater overall variation. A significant watertable depth increase was measured at Mea4 during January 2010, however, as a comparable increase was not measured at any other source at SA4 during this period, this is probably attributable to domestic usage i.e. pumping for storage, etc., similar to that evidenced in SA1 (BM1). This is particularly likely due to relatively thick (>10 m), low permeability subsoils in this area and typically slow watertable recession rates during this period (late winter). In all, however, an expected trend was observed, with decreasing watertable depths measured over the period September-December, reaching a water table peak during December/January, followed by an overall watertable drop from approximately February onwards. The inverse correlation between average watertable and total monthly rainfall was significant, as expected (r = -0.857, p = 0.007) (*Figure 7.36*).

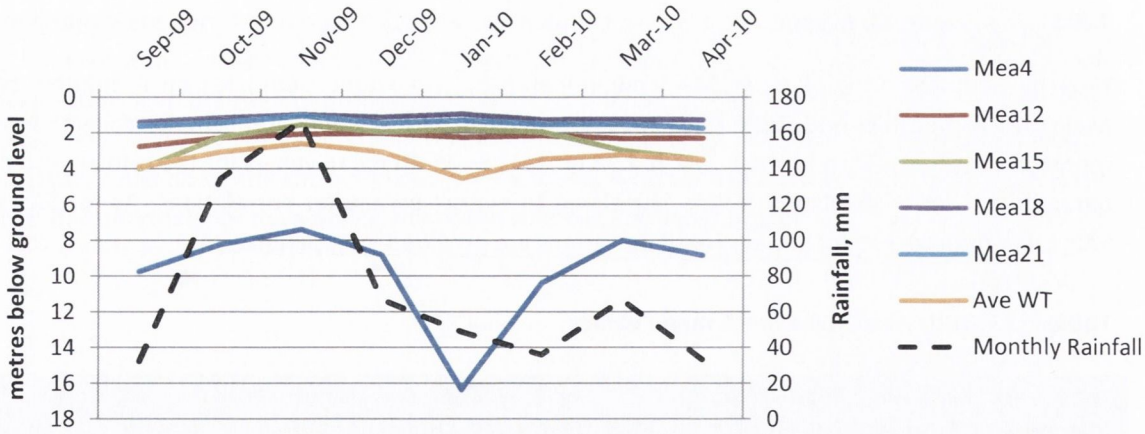


Figure 7.35 Study Area 4, Monthly watertable depths

Physicochemical Parameters

Monthly patterns for sample temperature, pH and EC are presented in *Figures 7.36 – 7.38*. The overall groundwater temperature profile was as expected and similar to those seen at both SA1 and SA2, with decreasing temperatures reaching their lowest during January-February 2010, followed by

an increase during March and April 2010. The relationship between average groundwater temperature and average monthly air temperature was significant, as expected ($r = 0.929$, $p = 0.001$) (*Figure 7.36*).

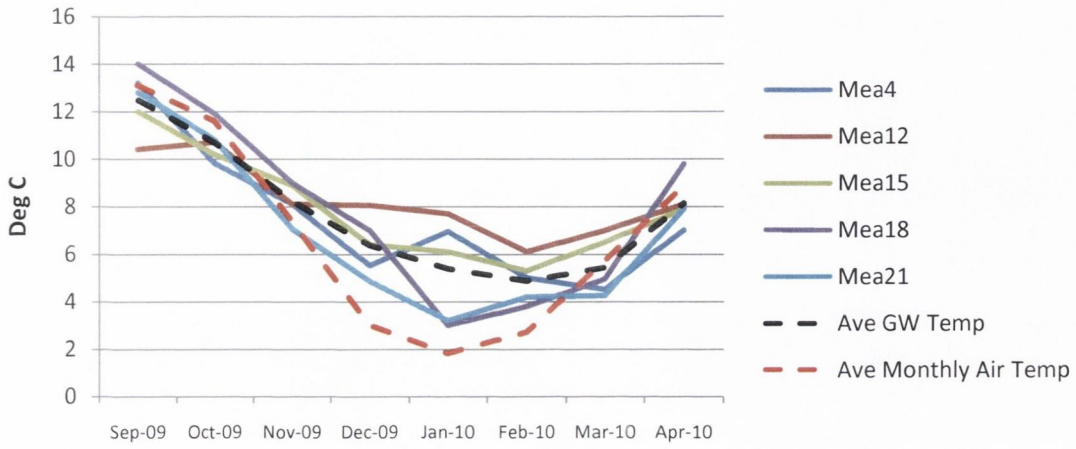


Figure 7.36 Study Area 4, Monthly groundwater temperature

As shown (*Figure 7.37*), pH was found to vary somewhat over the 8-month sampling period, although this variation occurred within a relatively narrow range (6.8-7.4). Previous work done by McCole (1992) in SA4 found a similar pH range (7-7.6) during winter sampling. This pH range is similar to that found in SA1 and falls within expectations, with pH in limestone areas e.g. SA1 being buffered by $CaCO_3$ and subsequently exhibiting pH ranges centred around neutral (OCM, 2007; EPA, 2010c).

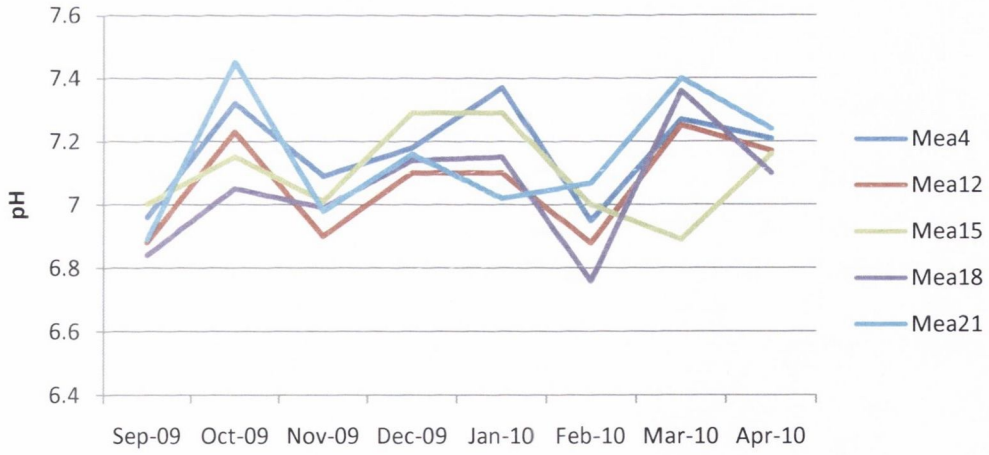


Figure 7.37 Study Area 4, Monthly groundwater pH

The average overall EC over the 8-month sampling period was 688 µS cm^{-1}, which is very similar to the overall mean EC of 681.5 µS cm^{-1} previously recorded by McCole (1992) in the same study area. EC was found to be generally stable among boreholes in SA4, with Mea4 exhibiting one significant departure from the mean EC value during November 2009 (*Figure 7.38*). The monitored hand-dug

well (Mea18) showed a high degree of variation with regard to conductivity, exhibiting an overall range from 517-954 µS cm^{-1} during the 8-month sampling period. As Mea18 was located in close proximity to both a busy road and agricultural land, in addition to being stone-lined and poorly covered, this variation may be due to a number of factors including ingress of road de-icing materials or agricultural effluents. No significant correlation was found between sample temperature and chloride concentration at Mea18 however, which would suggest that ingress of road de-icing materials did not occur. Likewise, no significant correlation existed between EC and nitrate at this source, which would be indicative of agricultural effluent. However, positive correlations between EC and both watertable depth and groundwater temperature may point to seasonal EC variation (*Table 7.37*), and therefore an agricultural source. Due to the short sampling duration, inference is not possible.

Although a significant level of variation was evidenced over the 8-month sampling period at SA4, a series of autocorrelation functions found that no discernable trends were evident with regard to water table depth, groundwater temperature, groundwater pH and groundwater conductivity, due to the short monitoring duration.

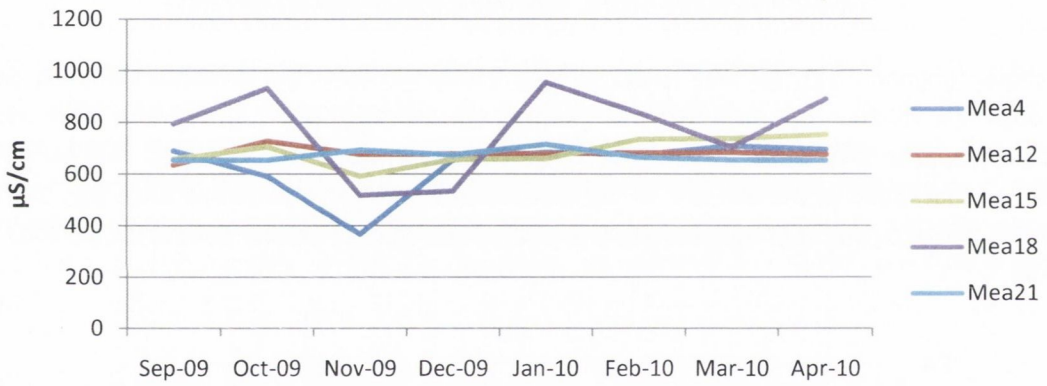


Figure 7.38 Study Area 4, Monthly groundwater EC

Chemical Parameters

As for SA1 and SA2, the main chemical analysis routinely carried out were NO_3 and Cl with all monthly samples analysed for these parameters (8 samples from each source). Additionally, NH_4-N analysis was carried out on all monthly samples during this period. Outline statistics pertaining to individual sources in SA4 are presented in Appendix E14.

Monthly chloride results are presented in *Figure 7.39*. The mean groundwater Cl concentration found in SA4 was 25 mg L^{-1}.Although Cl concentrations were variable at all individual sources during the sampling period, greatest variation was recorded at Mea12 and Mea18, with 8-month ranges of 17-44 mg L^{-1} and 15-43 mg L^{-1}, respectively. At Mea12, an inverse, although statistically insignificant, correlation was found between the water table depth and chloride concentration ($r = -0.57$, $p = 0.13$), signifying that rising water table levels at this well occurred concurrently with increasing chloride and therefore may be due to surface water ingress (*Table 7.38*). However, this correlation was not found at Mea18 as water table levels were relatively constant compared with chloride concentrations. Further analysis found however, that a similar correlation ($r = 0.571$, $p = 0.14$) was

apparent between precipitation in the 24 hours prior to sampling at Mea18 and chloride concentrations at that source, suggesting that similar surface ingress may have occurred at this source, resulting in chloride increases.

As for SA1 and SA2, an autocorrelation function carried out on natural log transformed average overall chloride values found that no discernable trend was apparent (see Appendix E20), although a slight increase in chloride concentrations with increased precipitation was noted i.e. late winter/early spring, possibly attributable to surface ingress of organic wastes.

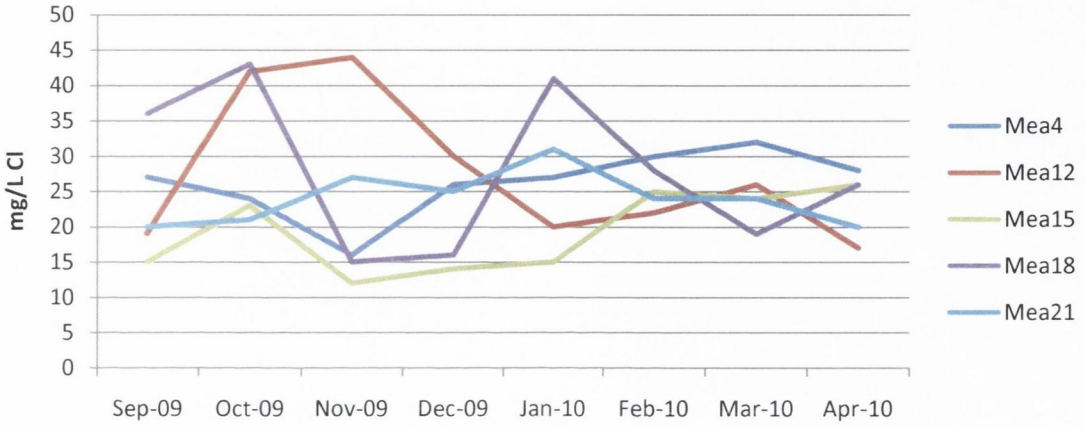


Figure 7.39 Study Area 4, Monthly groundwater Cl

A total average nitrate concentration of 8.1 mg L^{-1} NO_3 was found in SA4, with no individual source found to exceed 50 mg L^{-1} NO_3 during the 8-month sampling period, suggesting that groundwater nitrates are unlikely to present a significant direct health burden in SA4. As shown in *Figure 7.40*, significant variation was exhibited by individual sources, occurring within a range of 0.9 – 34.9 mg L^{-1} NO_3, with the highest peak occurring at Mea18 during November 2009. As previously discussed, a peak in nitrates in November-December is very common, due to higher total (and effective) rainfall, and lower rates of vegetative uptake (Schulte *et al.*, 2006). While, the mean groundwater nitrate concentration was found to increase during this period, this was primarily due to the large peak measured at Mea18.

Nitrate concentrations were not found to be significantly correlated with water table depth or groundwater temperature. Likewise, no relationship was found between mean nitrate and total precipitation rates. Further analysis found that individual sources typically had a positive correlation between precipitation and nitrate concentration, which is expected and may indicate surface water ingress from agricultural sources. However, a significant inverse correlation ($r = -0.766$, $p = 0.027$) was found between 24-hour precipitation and nitrate concentration at Mea18. This source is distinct due to its design and construction (stone-lined, shallow hand-dug well), with direct runoff perhaps causing dilution and consequently decreasing nitrate concentrations.

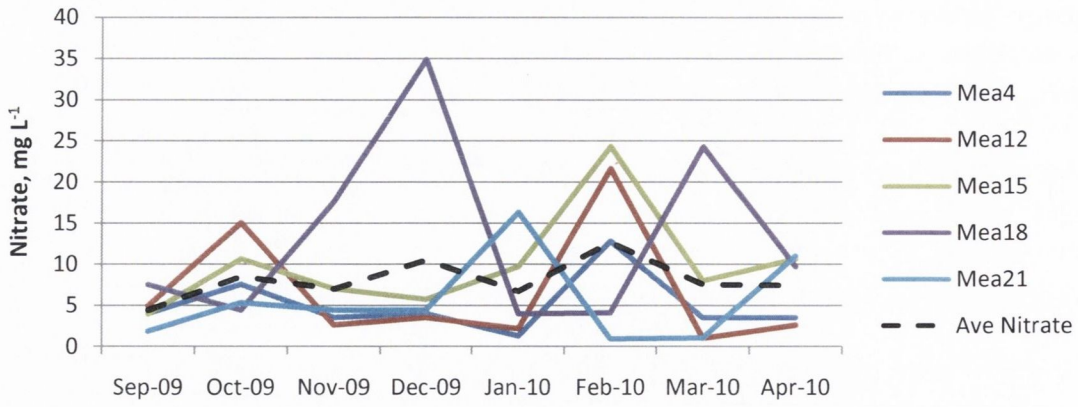


Figure 7.40 Study Area 4, Monthly groundwater NO_3

Microbiological Parameters

Table 7.39 outlines the frequency and mean concentration (where present) of *E. coli* from monthly samples at SA4 over the 8-month period September 2009-April 2010.As shown, all of the monthly sampled wells showed evidence of bacterial contamination at least once during the 8-month sampling period. Where *E. coli* were found to be present, a mean concentration of 15.8 cfu/100ml was found. *Figure 7.41* outlines the frequency and concentrations of presumptive *E. coli* found at SA4 during the sampling period. This level of contamination frequency was unexpected as SA4 was almost exclusively *Low* vulnerability and is therefore suggestive of source characteristics (i.e. well design, construction and location) being an important contamination risk factor in this area.

Table 7.39 Study Area 4, Monthly E. coli presence statistics

Well Number	Months present	Percent present (samples)	Mean, when present (cfu/100ml)
Mea4	4/8	50	21
Mea12	3/8	37.5	10
Mea15	2/8	25	3
Mea18	3/8	37.5	23
Mea21	5/8	62.5	22

As shown in *Figure 7.41*, peak contamination rates (frequency and concentration) occurred during October and November 2009, which may suggest bacterial contamination was not due to grazing animals or other agricultural activities in this area, as typically animals were being wintered during this period i.e. animals taken inside. A series of independent samples t-tests found that mean precipitation rates (24-h, 48-h, 120-h and 30-d) were higher at all wells when *E. coli* were present; however, they were not significant. This would suggest that bacterial contamination of groundwater in this area may be associated with point sources of contamination and both increased precipitation rates and low permeability subsoils in the area, leading to direct ingress of contaminants at the wellhead. A series of independent t-tests found that, typically *E. coli* concentrations increased with a decreasing watertable depth (e.g. Mea21 $r = -0.884$, $p = 0.004$). However, this was not found to be the case with Mea18, which had a positive, albeit insignificant, correlation with watertable depth. This was further reinforced at Mea18 by a positive correlation between groundwater temperature

and *E. coli* concentration (r = 0.655, r = 0.078), while all other sources exhibited a negative correlation between *E. coli* and temperature. Although there was an expected positive correlation between nitrate concentration and *E. coli* concentration at all sources, excluding Mea18, these were not found to be significant.

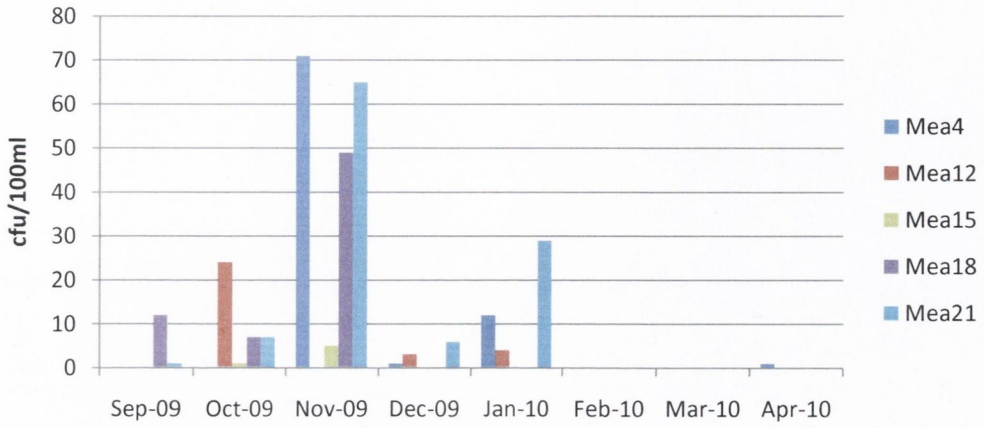


Figure 7.41 Study Area 4, 8-month E. coli frequency and concentration

The nitrate peak measured at Mea18 during November-December 2009 occurred in concurrence with *E. coli* presence (49 cfu/100ml) during November 2009. This is suggestive of a contamination incident i.e. ingress of organic wastes from point source, with more rapid bacterial die-off than nitrate dispersion a potential reason for the absence of *E. coli* during December 2009.

Faecal *Streptococci* (FS) were found in all five monthly wells during the 4-month period January-April 2010, with three of the five wells (Mea12, Mea18, Mea21) testing positive for FS on more than one occasion. Measured FC:FS ratios were typically <2, ranging from 0 – 1.33 which is suggestive of an agricultural source of pollution, however, one source (Mea 4) had a FC:FS ratio of 12 during January 2010, which is strongly suggestive of ingress of human faecal waste. Subsequent analysis of the K:Na ratio did not find any notable results with regard to potential sources of bacterial contamination (see Appendix E21). An overall K:Na ratio of 0.4 was recorded in SA4, with a particularly high mean (1.24) noted at Mea18, due to high monthly K values.

Influence of precipitation

Precipitation patterns relating to the 24-hour, 48-hour, 120-hour and 30-day periods prior to monthly sampling at SA4 are presented in *Figure 7.42*.

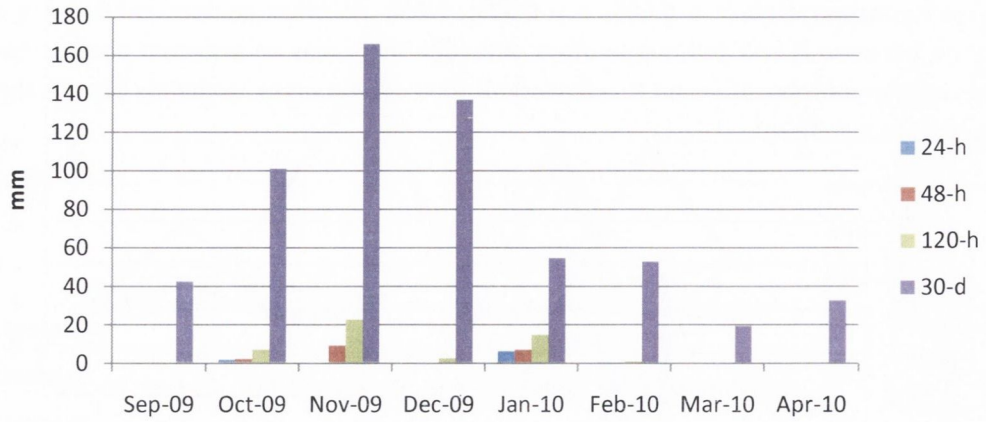


Figure 7.42 Study Area 4, Precipitation depth in the 24-hour, 48-hour, 120-hour and 30-day period prior to sampling

Overall, it was found that the average watertable depth had a negative correlation with precipitation depth, with this correlation being most significant (although not statistically significant) for the 30-day precipitation period. This implies that effective recharge to the source as a direct result of precipitation occurred over a period >30 days, which is expected in an area with relatively thick (>10 m) low conductivity subsoils, such as SA4. Further analysis suggests however, that while this was true over the entire study area, individual sources exhibited higher associations with the 120-hour precipitation period and watertable depth at sampling. The Geological Survey of Ireland (GSI) borehole log database for this area contains historical evidence of a number of sources where bedrock was encountered at <10 m (e.g. 2923SWW160 – 4.9 m depth to bedrock; 2923NWW065 – 2.4 m depth to bedrock; 2923NWW057 – 9.1 m depth to bedrock).Therefore the likely occurrence of pockets of shallower subsoils (<10 m) may account for higher associations with the 120-hour precipitation period at some sources.

No significant correlations were found between average monthly nitrate concentrations and precipitation rates (24-h, 48-h, 120-h, 30-d),although correlation coefficients showed an increasing trend of correlation with increasing rainfall as was expected. A significant correlation was found to exist between average monthly chloride concentrations at SA4 and precipitation rates 24 hours prior to sampling ($r = 0.727$, $p = 0.041$).This would seem to indicate an unexpectedly rapid precipitation response among these wells. As shown in *Table 7.40*, further analysis found however, that individual sources and their respective chloride concentrations responded very differently to precipitation. As shown, both Mea4 and Mea15 responded negatively to precipitation, with higher levels of rainfall resulting in lower levels of chloride, potentially due to localised rapid recharge (although this area is Low vulnerability and therefore dominated by thick layers of low permeability subsoil) and subsequent dilution. Direct surface water ingress is also a potential dilution mechanism at these sources. However, at Mea12, Mea18 and Mea21, increased rainfall resulted in increased chloride concentrations i.e. rainfall a primary source of chloride. In the case of Mea12 and Mea21, this occurred more significantly over a greater duration, however, in the case of Mea18, likely due to its construction i.e. stone-lined, shallow hand-dug well, chloride concentration seemed to respond almost immediately i.e. after 24 hours.

Table 7.40 Study Area 4, Precipitation response and Cl concentration correlation coefficients

Well Number	24-hour	48-hour	120-hour	30-day
Mea4	-0.287	-0.555	-0.749	-0.886
Mea12	0.254	0.547	0.659	0.714
Mea15	-0.083	-0.402	-0.65	-0.766
Mea18	0.571	0.14	-0.024	-0.19
Mea21	0.481	0.623	0.765	0.578

Similar analysis relating to *E. coli* concentrations and precipitation durations found a significant correlation between *E. coli* and the level of precipitation in the 120-hour period prior to sampling ($r = 0.785$, $p = 0.021$), implying a relatively rapid ingress of contaminants to these sources as a direct result of precipitation. Due to the typically slower response with regard to chloride concentration and precipitation (excluding Mea18), this may suggest contaminant ingress directly at the wellhead as opposed to in the subsurface.

No significant correlations were noted between effective precipitation and watertable depth (boreholes or hand-dug), chloride concentration or nitrate concentration at 4 of 5 temporally sampled sources in SA4. A correlation was found between effective precipitation and *E. coli* magnitude at Mea4 (60 m deep borehole), suggesting that when *E. coli* were present at this source, a proportion of the ingress occurred due to recharge, however, this correlation was not significant ($r = 0.677$, $p = 0.065$).

7.9 Summary

Design and Construction of Source Types

Overall, a greater level of variability was observed with regard to borehole design and construction than that among assessed hand-dug wells. A typical borehole employed a 170-180 mm diameter steel liner, with a liner cap and/or a wellhead cover present. A mean measured borehole depth of approximately 45 m was found, with boreholes generally finished either below ground in an underground chamber (61%) or above ground level (25%). Hand-dug wells were typically older, with a narrower depth range (mean 4.5 m) and generally employed a simpler overall design and construction, with 1,000 mm diameter concrete liners used in the majority of cases (80.5%). Additionally, the level of wellhead finish associated with hand-dug wells was simpler, with the majority finished at ground level (>75%), with no well chamber present (88%). Boreholes employed a greater overall range of construction materials (steel, PVC, HDPE, etc.) and as a consequence, a greater variety of design and construction configurations, including wellhead finish and wellhead protective measures. Generally, hand-dug wells exhibited more visible evidence of potential susceptibility to contamination, particularly ingress of contaminants at the wellhead, either due to direct ingress (i.e. down liner interior)or rapid recharge (i.e. down liner exterior). Furthermore, higher levels of waterproofing and lower levels of cracking were associated with boreholes.

Effect of geological setting on well design and construction

Limestone and granite bedrocks were associated with lower proportions of hand-dug wells (due to greater mean watertable depths in these areas), however no significant associations were found

with regard predominant local subsoil type or thickness range, groundwater vulnerability category or aquifer importance and either source type or source design.

Groundwater Hazard Sources

Overall, 95.2% (n = 199) of wells had ≥1 septic tank in operation within 100 m, with a median septic tank setback distance of 48 m (mean 77.6 m), a range of 14.6-100 m and a median of one septic tank within a 100 m radius (mean 1.54). No significant difference was found between septic tank setback distance and source type. As shown in *Table 7.9*, over 67% of septic tanks were found to be down gradient of the wellhead, as recommended. However, over 18% of wells had a septic tank operating up gradient of the wellhead within 100 m, which is cause for concern. No significant mean difference between septic tank setback gradient and setback distance was found, which also presents a concern, as wells located down gradient of a septic tank should have a greater setback distance, where possible. Overall, hand-dug wells were approximately twice as likely to have up gradient septic tanks within 100 m of the wellhead. Overall, 20.8% (n = 44) of wells had a septic tank within a 30m radius (the recommended minimum septic tank setback distances from wells and springs for down gradient domestic wells or where flow direction is unknown), with figures of 11.9% and 23% associated with hand-dug and boreholes, respectively. Domestic wastewater treatment systems excluding conventional septic tank systems (i.e. package plants such as bioCycle™ etc.), were in operation within a 100 m radius of 11.5% (n = 24) of assessed wells. The median setback distance was found to be 40.5 m, with a setback distance range of 14.2-100 m. The average mean setback distances were found to be 38.1 m and 80.3 m for boreholes and hand-dug wells, respectively.

Just under 18% of wells were located in predominantly agricultural areas, 16.3% (n = 34) were located in predominantly residential sites, with a majority of 63.6% (n = 133) situated on mixed land-use sites i.e. with both residential and agricultural activities occurring within 100 m. All of the assessed wells (n = 211) had grazing animals recorded within a 100 m radius of the wellhead, which was expected due to the predominantly rural setting associated with private groundwater sources in Ireland. The median grazing setback distance was found to be 15 m (mean 20.7 m), with mixed grazing the most commonly encountered grazing type. Approximately 27% of wells (n = 57) had an operational farmyard in the vicinity of the wellhead, with a further 6.7% (n = 14) and 1.9% (n = 4) having animal housing and silage pits respectively within a 100 m radius of the wellhead. The median setback distance of point agricultural activities from assessed wellheads was 45.1 m. A strong relationship was found between wells categorised as being located in agricultural areas and grazing animal setback distances, with agricultural wells, on average, 11.5m closer to grazing animals than mixed land-use wells and 20.5 m closer than residential wells. A similar pattern was found in relation to agricultural activities.

Approximately 98% (n = 206) of wells were located within 100 m of a roadway, with a median setback distance of 25 m (mean 34.4 m; range of 2-301 m). The most frequently recorded road type was tertiary/link, as expected due to the typically rural setting (*Table 7.13*), with no significant association found between road type and well type. Approximately 30% (n = 63) and 29% (n = 61) of wells had rivers and surface water drainage courses in the general vicinity, respectively, with a median setback distance of 100.1 m, with a further 7.1% of wells (n = 15) having a surface spring within 100 m of the wellhead.

Overall, therefore, the most frequently encountered potential hazard sources were septic tank systems, grazing animals and point agricultural sources (e.g. farmyards, animal housing, feed storage, silage pits, etc.), with typically no significant differences found between hazard source proximity and either source type or source design. Furthermore, no significant differences were found between hazard source type and proximity among assessed wells in individual study areas, although there was a higher over septic tank density found in SA4 (*Low* vulnerability, "overdeveloped" rural area). This assertion is therefore extended to include the groundwater vulnerability category i.e. no significant association was found between the groundwater vulnerability category and hazard source type or proximity.

"One-Off" Chemical and Microbial Sampling

Typically, measured watertable depths and physicochemical parameters were in agreement with expected ranges, in the context of source type and geological/hydrogeological setting. The mean measured groundwater Cl in the amalgamated dataset was found to be 27 mg L^{-1} (range 3-230 mg L^{-1}), with individual study area mean concentrations ranging from 13.4 – 35.8 mg L^{-1}. Almost 75% (n = 157) of one-off samples had a Cl concentration <30 mg L^{-1} (proposed as a guideline value in Irish groundwaters, an exceedance of which may be indicative of contamination), and approximately 48% (n = 102) had a Cl concentration <20 mg L^{-1} which is comparable with national figures i.e. previous large-scale study reports a national background median of 18 mg L^{-1} (OCM, 2007). In all, no wells exceeded the EU MAC of 250 mg L^{-1} Cl, although two wells did exceed 200 mg L^{-1} Cl.

A mean measured nitrate concentration of 12.8 mg L^{-1} NO_3 was found among the 211 sampled wells in the amalgamated dataset occurring over the range 0.1 – 110 mg L^{-1} NO_3. This mean value is slightly lower than the 14.6 mg L^{-1} NO_3 set out by OCM (2007) as a typical background level in Irish groundwaters. Furthermore, a mean concentration of 15.5 mg L^{-1} NO_3 was found in SA1 (*Belmont*, Co Offaly) which is lower than the 21.5 mg L^{-1} NO_3reported previously by Spain (2000) in Co Offaly. This would suggest recently decreasing nitrate concentrations in Irish groundwaters, with this pattern also being recently reported by the EPA (2010c).Only one exceedance of the EU MAC (50 mg L^{-1} NO_3) was noted, with 13.7% of samples (n = 29) exceeding the 25 mg L^{-1} NO_3set out by Daly (1994b) as being indicative of a nearby organic waste source.

In all, 29.4% of wells (n = 62) tested positive for presumptive *E. coli,* of which 67.7% (n = 42) were boreholes, with the remaining 32.3% (n = 20) being shallow hand-dug wells. As it is estimated that <10% of private groundwater sources are of the shallow hand-dug type, this would suggest that shallow, hand-dug wells are more susceptible to microbial ingress and therefore pose a greater overall human impact than deeper boreholes, as expected. Within individual study areas, *E. coli* contamination rates ranged from 45% in SA4 to 11% in SA5 (*Table 7.26*). Overall, the level of bacterial contamination encountered in SA2, SA3 and SA5 was lower than expected in areas categorised as being predominantly of *High* and *Extreme* groundwater vulnerability. Conversely, a much higher than expected level of bacterial contamination was encountered at SA4, particularly as this area is predominantly categorised as being *Low* vulnerability due to relatively thick (>10 m), low permeability subsoils. This is suggestive of source design and/or construction characteristics being a primary risk factor with regard *E. coli* contamination. A statistically significant difference was found to exist with regard to the presence of *E. coli* and the K:Na ratio magnitude (t = -2.915, p = 0.004),

with a mean K:Na ratio of 0.22 in wells with no *E. coli* present, while a mean ratio of 0.48 was found where *E. coli* were present. This suggests an agricultural contamination source, as K:Na ratios in septic tank effluents are typically <0.3 (Spain, 2000).

Just under 5% of samples (n = 4) tested positive for *Cryptosporidium* spp., with no samples testing positive for *Giardia* spp. This *Cryptosporidium* spp. contamination rate is generally consistent with previous studies elsewhere (Rose *et al*, 1991; Wallis *et al*, 1996; Hancock *et al*, 1998).

Physicochemical Cluster Analysis

Schwartz's Bayesian Inference Criterion (BIC) two-step clustering resulted in three distinct physicochemical clusters, based on "one-off" spatial groundwater sampling and analysis. Further analysis qualified these clusters as boreholes located in predominantly limestone or sandstone areas (Cluster 1), boreholes located in predominantly granite or Namurian areas (Cluster 2) and hand-dug wells (Cluster 3). A chi-square test of independence found that there was statistically significant association between source clusters and *E. coli* presence, suggesting that these clusters not only appropriately describe the source type and geology, but may also be used to appropriately describe source susceptibility. Therefore, these clusters were considered suitable for use in source susceptibility modelling (Chapter 9), as *E. coli* presence was the primary outcome variable. Notably, no association existed between cluster membership and groundwater vulnerability category, which reinforces the hypothesis that source susceptibility (particularly susceptibility to microbial contamination) and groundwater vulnerability category are independent i.e. increased groundwater vulnerability is not related to increased private source contamination rates, but may be associated with contamination mechanisms, where present. No statistical link was found between *E. coli* concentration (where present) and cluster membership.

Water Quality Monitoring (monthly temporal sampling)

The majority (93%) of monitored wells (14/15) showed evidence of *E. coli* contamination at least once during the monitoring period. However, only 40% of monitored sources showed evidence of *E. coli* contamination upon initial "one-off" sampling, therefore, it would appear that for microbial analysis, seasonal sampling was revealing, with a high level of variability and seasonality presented. However, no significant correlations or variability were noted with regard to a number of other measured parameters, including groundwater ammonia, potassium and sodium concentrations, with similar mean concentration values found between "one-off" and monthly samples for these parameters i.e. "one-off" results were reflective of seasonal results. It may be concluded therefore that for some parameters, a "one-off" sampling result was adequate.

Groundwater nitrate concentration means of 18.5 mg L^{-1}, 13.7 mg L^{-1} and 8.1 mg L^{-1} were measured in SA1, SA2 and SA4, respectively. Only one measured exceedance of the current EU MAC (50 mg L^{-1}) was recorded during the monitoring period, therefore, it may be concluded that groundwater nitrate is unlikely to present a current public health concern in these areas. Over the monitoring period, a slight mean decrease was noted, possibly due to improving agricultural practices and/or environmental legislation. This is reflective of current national trends (EPA, 2010c).

Typically, *E. coli* presence and magnitude (where present), was increased during mid to late summer and early to mid winter at all monitored sources. The increased presence and magnitude during mid

to late summer is suggestive of contamination from non-point agricultural sources i.e. grazing animals, and is therefore expected due to the predominantly rural nature of the study areas. The increase during early to mid winter is suggestive of agricultural point sources of contamination as animals are typically housed indoors (wintered) during this period. Additionally, septic tanks are typically considered a "year round" source of contamination, although increased rainfall and resulting higher watertables typically increase the contamination risk potentiality attributed to these contamination sources.

E. coli contamination rates of 34.3%, 30.4% and 42.5% were recorded at SA1, SA2 and SA4, respectively. Therefore, as previously outlined in regard to "one-off" sampling, a higher than expected rate of seasonal *E. coli* contamination was encountered at the *Low* groundwater vulnerability area, with both High/Extreme vulnerability areas both having lower rates of monthly contamination. Likewise, hand-dug wells typically had higher levels of seasonal *E. coli* contamination than boreholes, with mean contamination rates of 53.5% and 30% attributed to hand-dug wells and boreholes, respectively.

Four "cumulative precipitation periods" prior to sampling (24-hour, 48-hour, 120-hour, 30-day) and their respective association with bacterial contamination were analysed. In relation to *E. coli* presence, a series of independent t-tests found that, while no general (i.e. amalgamated dataset) associations were found, typically higher rainfall during the 120-hour and 30-day periods prior to sampling was found where *E. coli* were present, with hand-dug wells found to have higher correlations. Furthermore, at SA4 (*Low* vulnerability), similar analysis relating to *E. coli* concentrations and total precipitation durations found a significant correlation between *E. coli* and the level of precipitation in the 120-hour period prior to sampling ($r = 0.785$, $p = 0.021$), implying a relatively rapid ingress of contaminants to these sources as a direct result of total precipitation. Correlations between *E. coli* presence and effective precipitation rates (recharge rates) were less than measured associations between *E. coli* presence and total precipitation rates. This may be suggestive of rapid ingress due to poor wellhead design or lack of sanitary seal (i.e. preferential flow at well interface), as opposed to relatively slow contaminant recharge through subsoil layers. A correlation was found between effective precipitation and *E. coli* magnitude at Mea4 (60 m deep borehole), suggesting that when *E. coli* were present at this source, a proportion of the ingress was occurring due to recharge, however, this correlation was not significant ($r = 0.677$, $p = 0.065$). No other notable correlations were found. Precipitation patterns during the monitoring period were atypical, with relatively dry winter periods and wet summers (*Figure 7.26, 7.34, 7.42*).

Chapter 8: Groundwater source and contamination awareness among private well users in Ireland

8.1 Introduction

As previously set out (*Section 6.8*), the survey questionnaire was devised to examine the overall level of awareness amongst private well users with regard to a variety of contamination issues including specific knowledge of their own private source i.e. design and construction details, source age, use and importance of water treatment processes and maintenance of sources. Respondents were surveyed on their knowledge of potential groundwater contaminants and potential health effects of these contaminants in addition to potential sources of contaminants and the presence of these hazard sources in relation to their own well. Finally, to aid in the completion of a health risk assessment (*Chapter 10*), groundwater consumption patterns and historical health patterns with regard to gastrointestinal illnesses were examined. The questionnaire was developed so that all respondents would be presented with the same questions in the same order to ensure reliability and validity i.e. serial survey. The study presented is a cross-sectional survey as different units (respondents) were used for the collection of data.

This chapter presents results from the awareness survey carried out by the researcher on private groundwater users in Ireland. *Sections 8.2* and *8.3* focus on presentation of results on a question by question basis, placing particular emphasis on perceived knowledge gaps exhibited by respondents. As two separate questionnaire completion methodologies were used, survey modality was investigated (see Appendix F1). Consequently, it was considered appropriate to present survey results separately (RA and SA). *Section 8.5* presents overall source and contamination awareness shown by respondents and how and why this awareness may vary within the population. An awareness scoring methodology is presented and applied in *Section 8.6*, with awareness scores then analysed and investigated. Additionally, in order to quantify the probable effects of awareness (or lack thereof), on overall consumer susceptibility, a susceptibility model has been developed, comprising primary susceptibility (groundwater contamination) and secondary susceptibility (consumer controls), and the effects of awareness on susceptibility is assessed. All statistical tests used in this chapter are bivariate, and are based on Blalock Jr. (1979), as presented in Appendix F2.

8.2 Researcher Administered Survey Results

In all, 245 surveys were completed on a researcher-administered (RA) basis, across 6 study areas. As shown in *Table 8.1*, just over half of total RA respondents were male (55.1%). This slight majority was found in all study areas apart from SA4 (Kilclone, Co. Meath), where over half of respondents (56.7%) were female.

Table 8.1 Total Researcher Administered (RA) Surveys

Source	Female	Male	Total
Hand-dug wells	26	36	61
Bored SPS	80	97	177
PrGWS	1	0	1
Don't know	3	2	5
Total	110	135	245

8.2.1 Well Awareness

8.2.1.1 Respondent Source Awareness

Approximately 72% of surveys were completed with users of boreholes (bored SPS), with 24.9% being completed with users of hand-dug or spring wells. There are no data available regarding the relative proportion of hand-dug wells to bored SPS in Ireland; however, it is unlikely to exceed 10% and therefore this represents an important limitation of this dataset with an oversampling of hand-dug wells users. SA1 and the Westmeath study areas included 35% and 34% of total respondents respectively as hand-dug wells users. These high proportions are probably due to specific hydrogeological characteristics of the specific study areas i.e. SA1 had a high proportion of both glaciofluvial (carboniferous limestone) and esker sands and gravels (*Section 6.2.3*). The Westmeath study area, although predominantly of (carboniferous) limestone tills, is an area that is unserviced by a public supply with a high proportion of these wells being >50 years old. Therefore, the large percentage of hand-dug wells in this area may be as a result of lack of adequate drilling technology at the time of construction. The hand-dug wells in this area were found to be of a generally greater depth than others encountered during the study.

As expected, a majority of respondents were users of bored SPS, as these source types represent the majority of private groundwater sources in Ireland. Notably, a very low proportion of PrGWS users were included in the survey sample (0.4%), which may have been due to a misunderstanding among the general public as to the precise definition of a PrGWS. PrGWS may be defined as *"committee or shareholder run schemes serving fewer than 50 people, or supplying <10m^3/day"*, however, the researcher has concluded that a significant proportion of PrGWS in Ireland are supplying 2-3 households (5-15 people) with little or no organisation and these users consider themselves bored or hand-dug wells users as opposed to PrGWS users. Spain (2000) demarcates groundwater reliance in Co Offaly as: 22% public supplies, 18.6% PrGWS and 30.5% SPS respectively (with an overall groundwater reliance of 72% in Co Offaly). More recent figures at a national level suggest 3.2% of total households (4.1% of population) use PrGWS as primary household supply (CSO, 2007; EPA, 2007; EPA, 2009). However, it is not known what proportion of these are groundwater sources.

8.2.1.2. Residency

Just under 70% of respondents (n = 171) were living at the address during the well construction period.

8.2.1.3. Source Age

87.3% of respondents (n = 214) exhibited an awareness of the age of their source, although it is important to note that in many cases, responses provided were estimates, and therefore include a level of uncertainty. Respondents that indicated residency when the well was constructed (n =171), showed a high awareness (96%) of the source age. Of those respondents who were not living at the address at time of source construction (n = 74), 67.5% were aware of source age. Hand-dug wells were found to have a mean age of 38.8 years, while the mean age for bored SPS was 18.9 years.

8.2.1.4. Construction Involvement

An investigation of construction involvement awareness (i.e. knowledge of drilling contractor, local authority, water diviner or hydrogeologist involvement) found that 63% of respondents (n = 154) were aware of the contractor involved in source construction. This comprised 80% of hand-dug wells users and 58.6% of bored SPS users, a difference which was found to be significant($\chi^2(3) = 17.956$, $p < 0.001$).This difference is not unexpected however, as the majority of hand-dug wells are dug by residents, etc., with external contractors rarely being employed.

Approximately 66% of respondents (n = 162) were aware of the involvement of either a hydrogeologist, water diviner or local authority during source construction (*Figure 8.1).* Again, hand-dug wells were found to have a higher awareness of construction involvement, with 81.7% and 62% of hand-dug wells and bored SPS users respectively exhibiting an awareness of construction involvement. This was found to be a statistically significant difference ($\chi^2(3) = 13.083$, $p = 0.004$). No association was found between source type and utilisation of either a hydrogeologist or water diviner at the well construction stage, however, a chi-square test of independence found significant association between source type and local authority involvement($\chi^2(1) = 5.229$, $p = 0.021$).While 20.5% of bored SPS users acknowledged the involvement of a local authority in the design and/or construction of their source, this was found to be 6.1% among hand-dug wells users.

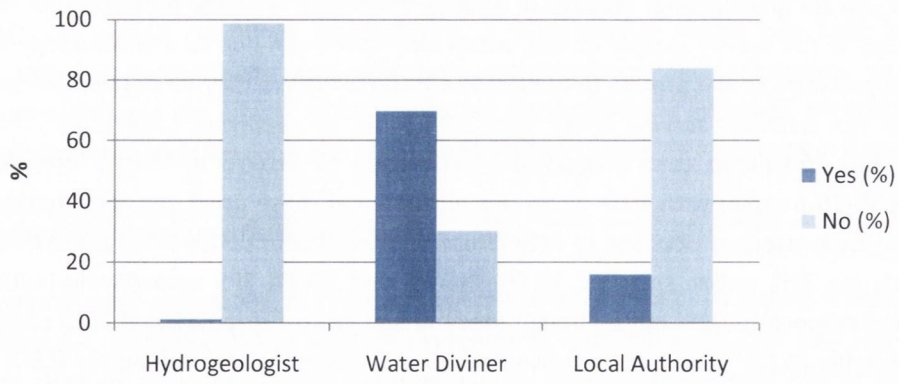


Figure 8.1 Level of construction involvement among RA survey respondents (n = 245)

As shown (*Figure 8.1*), there is a comparatively high level of involvement associated with water diviners (69.8%), which, when coupled with the very low involvement of hydrogeologists (1.2%) and local authorities (16%), may be cause for concern with regard to source susceptibility.

8.2.1.5. Design/Construction Records

84.1% (n = 206) of respondents were aware of whether or not they had records of the drilling process. Of this, 22.8% of respondents indicated they had records of drilling/construction. The majority of hand-dug wells (96.2%) were found to have no associated design or construction records (n = 53). This result was expected, due to the low involvement of contractors and local authorities. However, only 30% of Bored SPS users had design or drilling documentation (n = 150). This was unexpected, as the majority of boreholes are designed and constructed by contractors, who will typically supply documentation upon completion of a borehole. The difference between source type and awareness of well records was found to be statistically significant ($\chi^2(3) = 16.196$, $p = 0.001$), as were source type and well record awareness ($\chi^2(3) = 8.27$, $p = 0.041$).

8.2.1.6. Source Component Details

As shown in *Figure 8.2*, there was variable awareness of a range of well design details among respondents. There was a relatively high overall awareness of well depth, with 63.7% of respondents (n = 156) exhibiting an awareness of this parameter, probably due to the relationship between well depth and well cost, with the majority of drilling contractors in Ireland charging for drilling operations per unit depth. Awareness of other well parameters were 49.4% (well diameter), 33.9% (average watertable depth), 60.8% (use of a liner) and 58.8% (pump type) (*Table 8.2*). The figure of most significance is the low level of awareness associated with use of waterproof sealing of the well (20.8%). As discussed previously (*Section 3.4.5*), adequate sealing of wells is necessary for the prevention of source contamination. Users of bored SPS in particular, displayed a very low awareness of sealing of their source, with 81.5% of these respondents (n = 179) having no awareness of sealing or grouting of their source. This was found to be a significant difference when compared with other sources ($\chi^2(3) = 12.317$, $p = 0.006$).

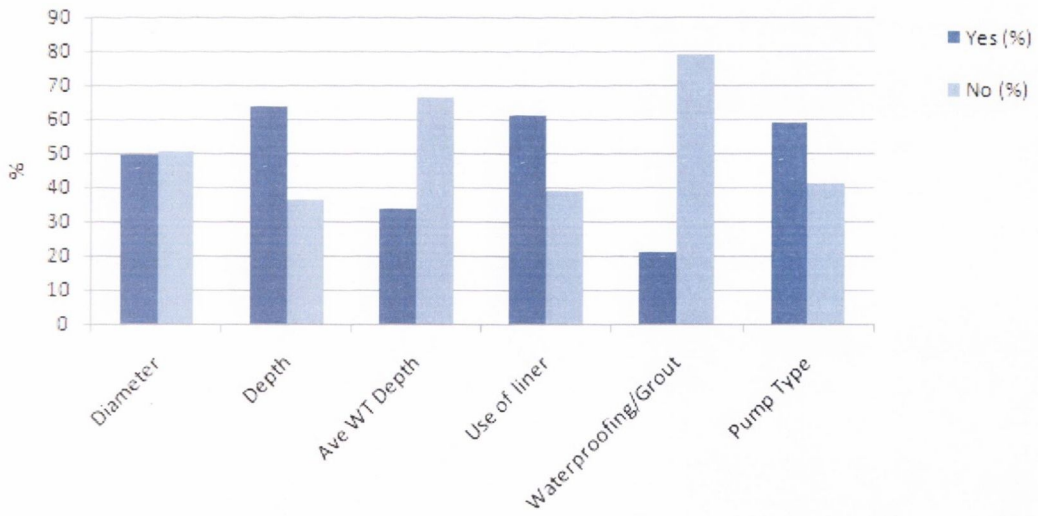


Figure 8.2 Overall Source Detail Awareness (n = 245)

Table 8.2 Groundwater Source Component Awareness (n = 245)

	Well Diameter		Well Depth		Average Watertable		Well Liner		Sanitary Seal		Pump Type	
	Yes	No	Yes	No	Yes	No	Yes	No	Yes	No	Yes	No
Hand-dug wells	37 (62)	23 (38)	40 (66)	20 (33)	30 (50)	30 (50)	38 (63)	22 (37)	22 (37)	38 (63)	35 (58)	25 (42)
Bored SPS	82 (46)	97 (54)	114 (64)	65 (36)	52 (29)	127 (71)	108 (60)	71 (40)	28 (16)	151 (84)	107 (60)	72 (40)
Unknown	1 (20)	4 (80)	2 (40)	3 (60)	1 (20)	4 (80)	2 (40)	3 (60)	1 (20)	4 (80)	1 (20)	4 (80)
PrGWS	1 (100)	0	0	1 (100)	0	1 (100)	1 (100)	0	0	1 (100)	1 (100)	0
Total	121 (49)	124 (51)	156 (64)	89 (36)	83 (34)	162 (66)	149 (61)	96 (39)	51 (21)	194 (79)	144 (59)	101 (41)

(Percentages in parentheses)

Of the six well details investigated, respondents were aware of a median of three. An independent samples t-test was carried out and suggested a significant mean difference between source type and source detail awareness ($t = 1.966$, $p = 0.05$), with a greater mean level of source detail awareness exhibited by hand-dug wells users.

Although there was little difference between source types at the lower end of the well detail awareness spectrum (i.e. similar proportions exhibiting little or no awareness), a significant difference was found at the higher end (i.e. those with a high level of awareness). 48.3% of hand-dug wells users were aware of ≥5 well details, with the equivalent figure for bored SPS being 24%.

8.2.1.7. Water Treatment

99.2% of respondents were aware of whether or not their water source was being treated (n = 243), with approximately 32% of these (n = 79) indicating that their source was currently being treated, i.e. almost 68% of private groundwater sources did not currently use a water treatment system. Further analysis showed a significant difference between water treatment rates across source types ($\chi^2 = 25.040$, $p < 0.001$) with 11.6% of surveyed hand-dug wells having a treatment system in place, compared with 41.3% of bored SPS. As previously discussed, it is probable that bored SPS currently constitute at least 90% of private groundwater supplies in Ireland, and these results infer that there is currently treatment in place at less than half of these sources. It is important to note, however, that 58% of respondents used domestic ion-exchange water softeners as their primary treatment process, with previous studies (Flemming, 1987; Parsons, 2000) having found that domestic water softeners have no treatment efficacy with regard to pathogenic organisms and in some cases ion exchange resins contained in these systems may even support microbial regeneration and growth. Therefore, in the context of microbial contamination, just over 18% of respondents (n = 45) had an appropriate water treatment system in place. This is discussed in greater detail in chapter 10.

8.2.1.8. Source Maintenance

82.9% of respondents (n = 203) were aware whether or not some form of maintenance had been carried out on their private source at some time (pumping, disinfection, wellhead maintenance, etc). Of those that were aware, 35% (n = 71) indicated that some maintenance had been carried out, with no previous maintenance having been carried out on the remainder. Further analysis indicates that source type had no significant association with awareness of previous source maintenance, with 83.3% and 84.3% of hand-dug wells and bored SPS users, respectively, exhibiting awareness. There was however a significant difference found pertaining to actual maintenance being carried out, with 46% and 22.5% of hand-dug wells and bored SPS users indicating previous maintenance had been previously carried out ($\chi^2(2) = 10.564$, $p = 0.005$).

8.2.1.9. Important Factors in Well Design and Construction

Figure 8.4 illustrates the factors considered important by respondents (n = 242) when having a new well designed and constructed. As shown, water quality and well yield are considered the two most important, with 95.5% and 89.3% of respondents respectively indicating these as important issues. Perhaps of greater importance however, is the low overall appreciation of location as an important factor in well design and construction, with 41.3% of respondents indicating this as being a major factor in well design and construction. As previously discussed (*Section 3.3*), correct well location is an imperative part of the well design process, with adjacent hazard sources being of primary concern with regard to source safety/susceptibility, particularly with regard to hand-dug wells, as these are typically sited by the well owner. No significant difference was found with regard to location as an important factor in well design/construction and source type.

31% of respondents indicated that well depth was an important factor. Further investigation found a relationship between well depth and well yield among a proportion of respondents, with many respondents indicating that they would favour increased well depth in order to increase well capacity and storage.

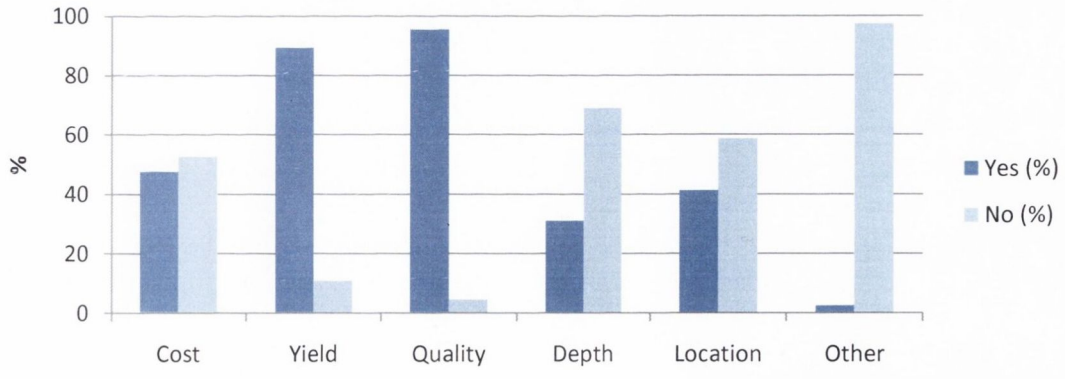


Figure 8.3 Factors Considered Important RA Users (n = 242)

Respondents were asked to state which factor they would consider most important when getting a new well constructed. *Figure 8.5* summarises these responses. The majority of respondents (71%) considered water quality to be the issue of most importance with regard to well construction. Yield, cost, depth and location were selected as being the most important factor in well construction by

22.3%, 4.2%, 1.7% and 0.8% of respondents, respectively. There was no significant difference found between source type when investigating the most important factor with reference to source construction/design.

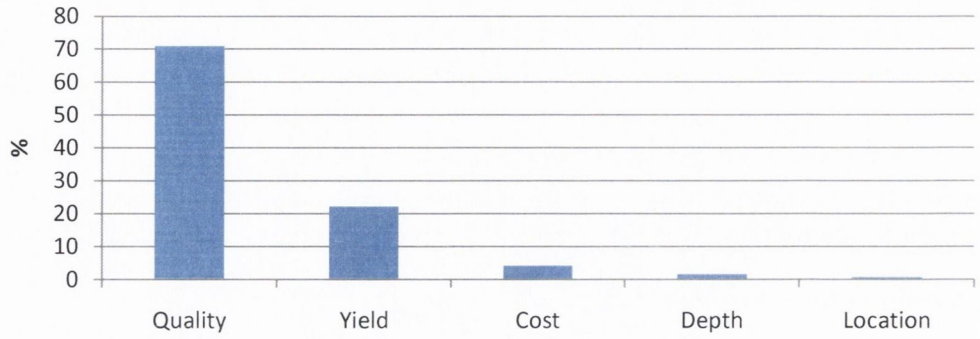


Figure 8.5 Most important factor in well construction (n = 242)

8.2.2 Water Quality Analysis

8.2.2.1. Water Quality Analysis Awareness

93.5% of respondents (n = 229) were aware whether or not their source had previously been tested. Of these, 59.8% (n = 137) indicated that their water had been quality tested, with 81.2% (n = 95) of these respondents aware of the water quality analysis results. There was no significance found between respondents' knowledge of previous water quality testing and respondents source type. 63.8% of bored SPS had been previously tested, inferring that on a national scale, over 35% of those served by bored SPS (excl. PrGWS) are using a source which has not been previously tested for chemical or microbial quality. This equates to over 225,000 people (>5.3% of national population).

8.2.2.2. Water Testing Importance

Respondents were asked to rate the importance they placed on regular water quality testing of their source and subsequently asked if they would consider paying for regular testing. As shown (*Figure 8.6*), 29.4%, 50.8% and 17.6% rated regular water quality testing of their source as high, medium and low respectively. A significant difference was found ($\chi^2 = 20.076$, $p = 0.017$) between hand-dug and bored SPS users, with bored SPS users generally placing a higher level of importance on regular water quality analysis i.e. 34% of bored SPS users (n = 179) assigned a high level of importance to regular water quality analysis, compared with 18.3% of hand-dug wells users (n = 60).

61.1% of total respondents (n = 149) indicated that they would be prepared to pay for regular water testing, with suggested frequencies ranging from once per month to once every five years. As expected, a significant difference was found between respondents perceived level of importance associated with regular water analysis and their willingness to pay for this analysis ($\chi^2 = 82.296$, $p < 0.001$), with users who assigned a low level of importance to regular testing much less likely to pay for this testing. This difference also translated to source type ($\chi^2 = 14.644$, $p = 0.002$), with a much lower willingness to pay for regular quality analysis among hand-dug wells users, who had previously been shown to attribute a lower overall level of importance to regular testing.

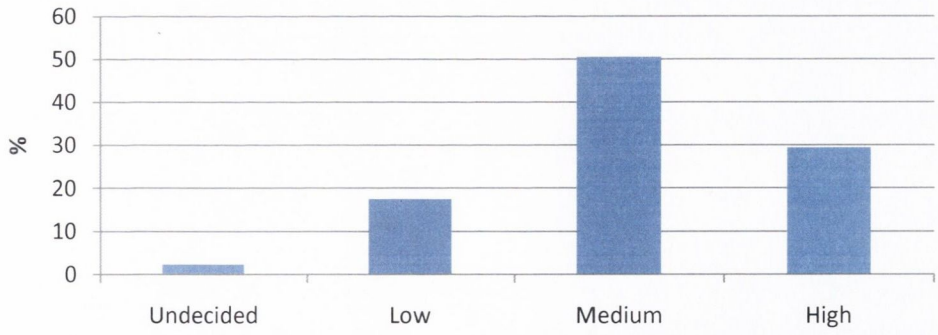


Figure 8.6 Water quality analysis importance (n = 245)

8.2.3 Groundwater Contamination

8.2.3.1. Contaminant Recognition

Respondents' knowledge of a number of groundwater contaminants, both microbial and chemical, was investigated. As shown (*Figure 8.7*), there were generally high levels of awareness of *E. coli* (93.9%), *Cryptosporidium* (73.1%), calcium carbonate/lime (96.7%), nitrate (76.7%) and fluoride (74.3%).The remainder of the potential contaminants exhibited relatively low associated degrees of awareness, including rotavirus (25.7%), *Giardia* (5.7%), manganese (32.7%) and arsenic (24.1%). Of particular concern are the low levels of awareness associated with rotavirus and *Giardia*, as both of these microbial contaminants are causative agents of waterborne gastroenteric illness (see *Chapter 4*).The relatively high level of awareness associated with *Cryptosporidium* spp. may be due to a highly publicised recent outbreak in the west of Ireland (FSAI, 2008; The Irish Times, 2008; Zintl *et al.*, 2009).

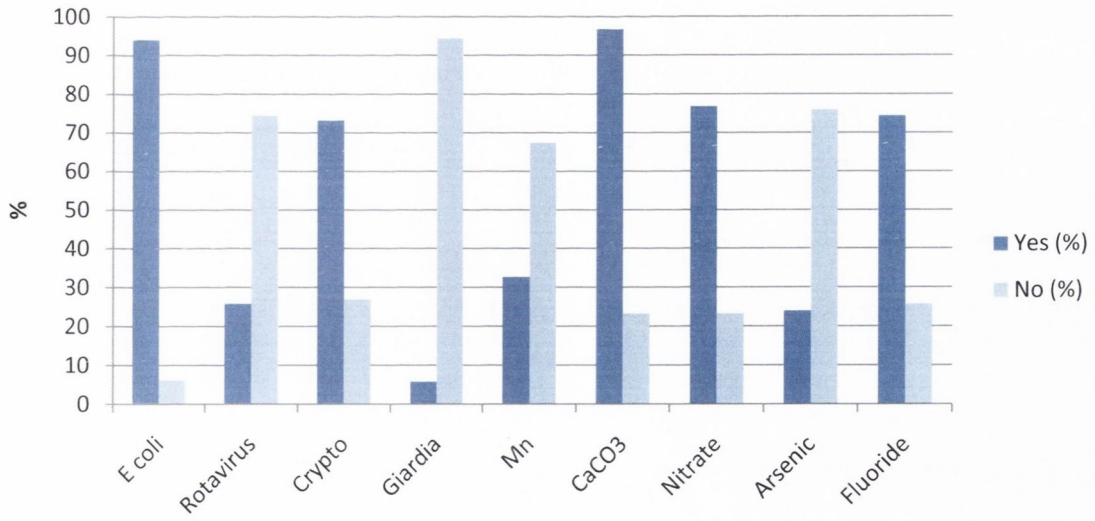


Figure 8.7 Potential Groundwater Contaminant Awareness (n = 245)

It was postulated that a relationship may exist between awareness of specific groundwater contaminants and respondents employed in agricultural activities, particularly regarding *E. coli*,

rotavirus, *Cryptosporidium* and nitrates as these are contaminants with agricultural connotations i.e. REPS, Nitrates Directive, zoonotic characteristics. However, no statistical differences were found with regard to specific contaminant knowledge and the respondent source. No significant mean difference was found between the total number of contaminants recognised by respondents and their source type.

8.2.3.2. Potential Health Effects

Respondents' awareness of potential health problems caused by groundwater contamination (illnesses and/or symptoms) showed that 69.8% of total respondents (n = 171) exhibiting an awareness of at least one illness or symptom caused by consumption of contaminated groundwater. The most frequently recalled symptoms were nausea, vomiting and diarrhoea (the foremost symptoms of gastroenteritis/AGI; see *Chapter 4*).

There was the expected difference found between respondents who had initiated previous water quality analysis and apparent health hazard awareness (χ^2=16.004, p = 0.000). Consequently, those respondents who assigned a greater level of importance to regular water quality analysis were found to have a greater overall knowledge of health hazards associated with groundwater contamination (χ^2=10.062, p = 0.018).

An investigation of potential relationships associated with health hazard awareness of groundwater contaminants and specific contaminant awareness found that all potential groundwater contaminants, with the exception of calcium carbonate (lime/hardness) and *Giardia*, had statistically significant relationships (*Table 8.3*),with total contaminant number awareness showing a strong statistical association with health hazard awareness(t (243) = -6.566, p < 0.001). Respondents with increased awareness of potential groundwater contaminants were found to be more likely aware of potential health hazards arising from contaminated groundwater consumption. There was no significant difference found between respondent source and health hazard awareness.

Table 8.3 Results of Chi-square analysis of health hazard awareness and contaminant awareness

	χ^2	df	p
E. coli	24.163	1	<0.001
Rotavirus	6.533	1	0.011
***Cryptosporidium* spp.**	16.792	1	<0.001
***Giardia* spp.**	3.746	1	0.07
Manganese	9.094	1	0.003
Lime/Hardness	1.537	1	0.248
Nitrate	23.703	1	<0.001
Arsenic	6.477	1	0.014
Fluoride	17.054	1	<0.001

8.2.3.3. Activities potentially affecting water quality

56.3% (n = 138) of respondents were aware of a potential hazard source in the immediate area of their source, with the most frequent potential hazard sources cited as being adjacent farmyards, roads and land spreading. Respondents were subsequently asked whether any of a number of

specific activities were ongoing within 100m (a generalised inner protection zone) of their source, with the findings summarised in *Figure 8.8*.

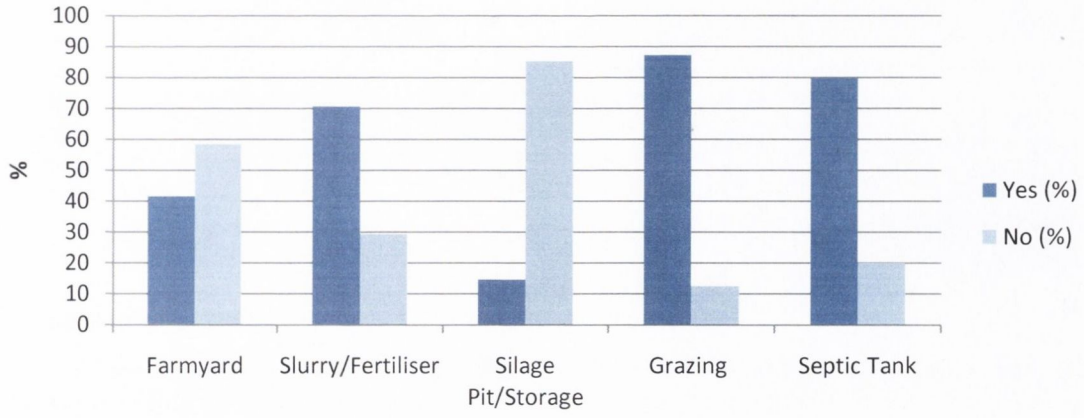


Figure 8.8 Activities within 100 m of respondent's well (n = 245)

As mentioned, 56.3% of respondents were aware of at least one potential contaminant hazard in the vicinity of their well. From this, it may be inferred that any specific contaminant hazard with a proportion in excess of 56.3% in *Figure 8.9* represents a lack of awareness with regard to the potential of that particular activity for contamination of private groundwater sources. This would imply that at least 14.3%, 31% and 23.7% of respondents are unaware of the potential threat of slurry/fertiliser spreading, grazing animals and septic tanks, respectively. This finding presents a concern, as the majority of private groundwater sources are located in rural areas and these three contaminant sources in particular are ubiquitous throughout rural Ireland.

8.2.4 Human Health

8.2.4.1. Household Composition

An investigation of the average composition of a household supplied by a private groundwater source in the study areas was carried out, with subsequent analysis on whether or not household size and/or composition differ relative to source type. The data collected are discussed in greater detail in *Chapter 10*, where the data are used in the context of vulnerable sub-populations and potential health effects of groundwater contamination in Ireland. As only 1 respondent was a PrGWS user, this has been omitted at this stage of the analysis and discussed further when amalgamated with self-administered (SA) survey results later in this chapter. *Table 8.4* summarises typical household composition, which indicates how many households have at least one occupant in each subset i.e. 27.1% of households supplied by hand-dug wells have at least one child (6-18) in residence.

An interesting finding was the obvious disparity between hand-dug wells and bored SPS with reference to elderly occupants, (found to be significant; $\chi^2 = 21.321$, $p < 0.001$), with households supplied by hand-dug wells significantly more likely to comprise elderly residents than those supplied by bored SPS.A probable reason for this finding is a decreasing reliance on hand-dug wells over the past 2-3 decades due to improved drilling methods; as borne out by earlier findings suggesting that the average age of hand-dug wells is approximately 20 years greater than bored SPS

i.e. the average age of SPS user may increase with the average source age. This issue is further investigated in *Chapter 10*, particularly with reference to sub-populations with higher vulnerability to waterborne pathogens.

Table 8.4 Typical household composition (n = 238)

Household Source Type	Infants (<1) (%)	Young children (1-5) (%)	Children (6-18) (%)	Adults (18-65) (%)	Elderly (>65) (%)
Hand-dug wells	5.1	5.1	27.1	69.6	49.1
Bored SPS	3.9	15.6	36.8	89.3	19

The overall average household size involved in the survey was 3.73 (St. Dev. 1.79, n = 242), with minimum and maximum households of 1 and 14, respectively. This is significantly larger household mean than both the national rural household average of 2.95 and the average rural household size of 3.03 for Leinster (CSO, 2007), in which approximately 95.7% of RA surveys were administered. When further fragmented, average household sizes for hand-dug wells and bored SPS households were 3.1 and 3.96 respectively. It is unclear why the average bored SPS household size differs from expected, however, this may be connected to the decreased average bored SPS consumer age, with these source types more frequently serving younger families of increasing number. Additionally, boreholes typically have higher yields than hand-dug sources, and are therefore capable of supplying greater numbers of consumers.

8.2.4.2. Consumption Source

The majority of respondents (91.8%) indicated the primary household consumption point being as household tap (treated or untreated). The majority of those respondents that indicated this as not being the case (8.2%) used bottled water. There was no significant difference found with regard to drinking water source and source type or gender. However, it was found that those residents living at the address at the time of well construction were more likely to use the household tap as opposed to other sources ($\chi^2 = 9.171$, $p = 0.004$). When individual respondents were asked if they themselves consumed from a household tap (treated or untreated), 90.2% stated that they did (n = 221).

8.2.4.3. Consumption Patterns

A median consumption frequency figure of 2-3 times per day was obtained (n = 226), with no significant difference found between consumption frequency and gender or source type. These data pertain to "raw" water, straight from the household tap, not water used for preparation of food or hot beverages i.e. tea, coffee, etc.

Regarding daily average consumption, a median range of 0.5-1 litre/day was found (N = 233), with an unsurprising significant relationship evident between consumption frequency and consumption volume ($r = 0.694$, $p < 0.001$).No significant difference was found between daily consumption volume and gender or source type. Further analyses are presented in *Chapter 10*.

8.2.4.4. Water-related Illness

Only 0.8% (2/245) of respondents reported previous (past 12 months) instances of themselves or another person in the household having been diagnosed by a medical practitioner with a water-related illness. Both were reported as having been associated with Bored SPS. No significant difference could be found between waterborne illness and any of the aforementioned awareness factors, due to the low level of diagnosed waterborne illness.

Just under 10% (9.87%) of respondents (N = 24) reported themselves or another person in the household having shown potential symptoms of gastroenteritis/AGI in the past 12 months. It should be noted however; that these cases, unlike positively diagnosed cases of water-related illness, pertain to potential cases of general gastrointestinal illness and it is therefore probable that a proportion of these were foodborne or related to recreational water as opposed to being a direct result of water consumption.

A number of significant differences were noted with regard to potential illness and other factors, particularly those relating to water quality testing and household composition as shown in *Table 8.5*.

Table 8.5 Statistical outputs from analysis of potential waterborne illnesses and other factors

Factor	Test statistic	df	P
Water Quality Analysis Awareness	$\chi^2 = 8.79$	1	0.013
Preparedness to pay for water testing	$\chi^2 = 5.701$	1	0.016
Awareness of *Giardia*	$\chi^2 = 5.883$	1	0.038
Household comprising infant residents	$\chi^2 = 16.29$	1	0.002
Household comprising young children (1-5)	$\chi^2 = 18.784$	1	<0.001
Household comprising elderly residents	$\chi^2 = 4.493$	1	0.048
Total household size	t = -3.314	238	0.001

Those respondents reporting previous potential waterborne illnesses in their household were found to exhibit a higher awareness of previous water quality analysis, with a similar pattern evident a propos preparedness to pay for regular water quality analysis - 83.3% of those reporting potential previous illness stating they would be prepared to pay for testing, while this figure was found to be 58.2% in the remainder of respondents.

Potential illnesses were found to be more frequent in households with both infants (20.8%) and young children between 1-5 years of age (41.6%), than in households without these vulnerable sub-populations. Conversely, households with elderly residents (>65years), were found to be associated with fewer previous instances of potential waterborne illness than expected (8.3%). This may be due to smaller average household sizes associated with elderly residents, as an independent samples t-test found a significant mean difference between households reporting potential previous illness and those not. Approximately 43.5% of reported potential previous illnesses were associated with

households comprising four residents, with this percentage rising to 69.5% for households comprising 4-6 residents. No single resident households reported previous potential illness.

8.2.4.5. Source Satisfaction

Respondents were subsequently asked to rate their source satisfaction from 1 (very dissatisfied) to 5 (very satisfied). Findings are presented in *Figure 8.9*, which suggested no significant mean difference between source satisfaction and source type, with a high overall level of satisfaction of their source found among private groundwater users in general.

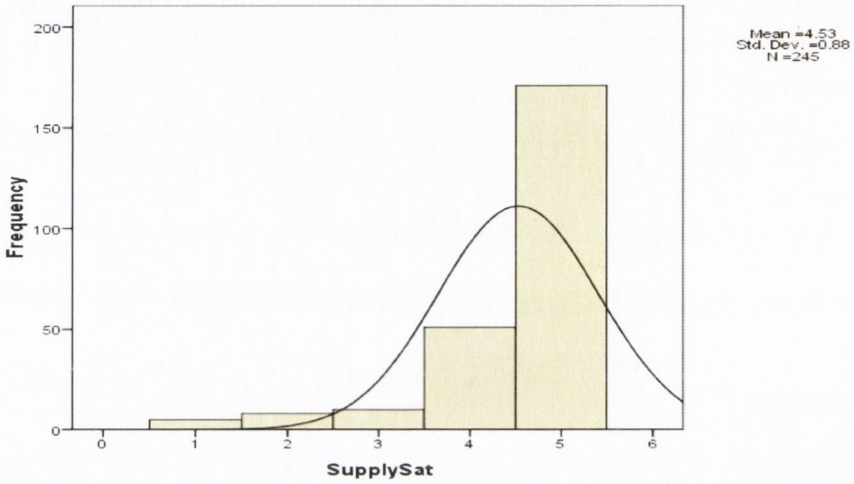


Figure 8.9 Source Satisfaction overlain with Normal (Gaussian) Curve (n = 245)

When source satisfaction was cross-correlated with a number of parameters, a number of trends were suggested, as shown in *Table 8.6*. Analysis found that gender was an explanatory factor in relation to source satisfaction, with female respondents generally exhibiting a lower overall satisfaction, particularly in households with young children (1-5 years) and children (6-18 years). Overall satisfaction typically declined as household number increased. It seems a higher level of awareness results in a lower overall level of satisfaction, particularly with regard to potential health hazard awareness and knowledge of potential contaminants. Respondents residing in households in which a household tap was not the primary daily consumption source showed a lower level of satisfaction with their supply ($p = 0.001$) with individual respondents not using the household tap exhibiting an even more significant lack of satisfaction ($p < 0.001$). This lack of satisfaction correlated with significantly lower daily consumption volumes ($r = 0.176$, $p = 0.006$).

Table 8.6 Statistically significant outputs from analysis of source satisfaction and other factors

Factor	Test statistic	df	P
Gender	U = 6472.5	1	0.04
Involvement Awareness	U = 5858.5	1	0.041
Cost as an important construction factor	U = 6345.5	1	0.03
Health hazard awareness	U = 4588	1	<0.001

Household comprising young children (1-5)	U = 2788	1	0.023
Household comprising children (6-18)	U = 5779	1	0.001
Use of household tap	U = 1432.5	1	0.001
Respondents use of household tap	U = 1573	1	<0.001
Contaminant recognition	r = -0.167	n/a	0.009
Household total	r = -0.132	n/a	0.04
Consumption volume	r = 0.176	n/a	0.006

8.3 Self-Administered Surveys

In all, 365 surveys were completed on a self-administered (SA) basis, across 6 study areas. These surveys were carried out, for the most part, with a younger cross-section of private groundwater users, with the majority of these being 20-30 years old and from a rural and/or agricultural background. The outline statistics for each study area and the amalgamated total are set out in *Table 8.7.*

Table 8.7 Total Self-Administered Awareness Surveys

Source Type	Female	Male	Total
Hand-dug wells	2	59	61
Bored SPS	5	180	185
PrGWS	4	41	45
PuGWS	5	59	64
Other	2	5	7
Don't Know	0	3	3
Total	18	347	365

As shown, approximately 95% of total RA respondents were male, which although considered a low level of female representation, it was considered representative of the institutions being surveyed (i.e. agricultural/horticultural colleges with significant majority of male students). Obviously, this is not representative of private groundwater users on a national basis.

8.3.1 Well Awareness

8.3.1.1. Respondent Source Awareness

Figure 8.10 shows that just over half of respondents (50.7%) were bored SPS users. Although lower than RA respondents, it again indicates these source types represent the majority of private groundwater sources in Ireland. Hand-dug wells respondents represent 16.7% of total SA respondents, which is yet again considered high relative to probable national importance of these supply types, although this proportion is closer to the likely national figure (≈10%).

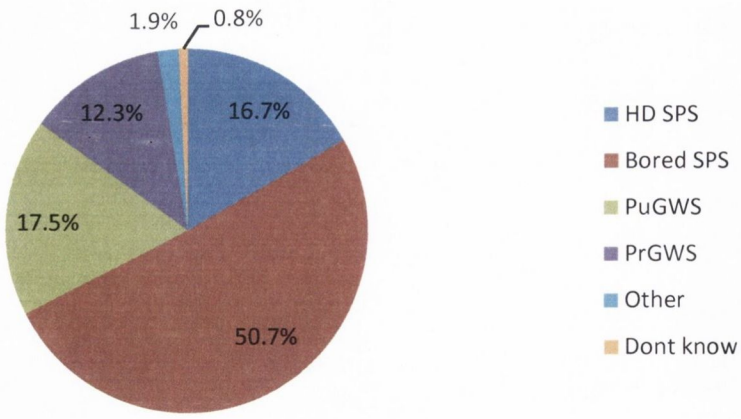


Figure 8.10 Total SA Groundwater Source Representation (n = 365)

PrGWS respondents accounted for a much higher proportion of SA respondents than RA respondents', however, it is important to note that PrGWS respondents were not targeted as part of this study. Spain (2000) has stated that approximately 18.6% of total population of Co. Offaly is supplied by PrGWS, although nationally it has been estimated that approximately 6% of households reply upon PrGWS for domestic supply (Mannix *et al.,* 2005). PuGWS respondents accounted for 17.5% of respondents. As the focus of this research is private groundwater sources and their respective users, PuGWS respondents are excluded from subsequent analysis. Outline statistics for PuGWS respondents are included in Appendix F3. Therefore, the following discussion is based on results attained from private groundwater users only, as shown in *Figure 8.11*.

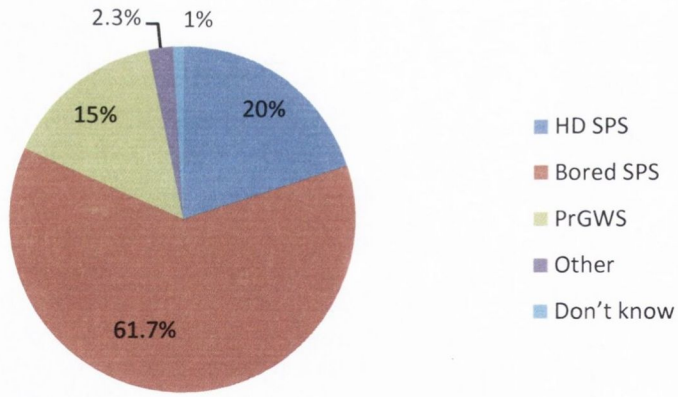


Figure 8.11 SA Private Groundwater Source Representation (n = 300)

8.3.1.2. Residency

53.7% of respondents' were living at the address when the well was constructed (n=298), compared with RA respondents' residency rate of 69.8%. This difference is probably due to the generally lower age of SA respondents, as previously discussed. A significantly higher proportion of hand-dug SPS respondents were not living at their address before or during well design/construction compared with both bored SPS and PrGWS (χ^2 (4)=23.576, p<0.001). This is to be expected owing to the generally greater age of hand-dug wells, coupled with the generally younger demographic of this

dataset, such that many hand-dug wells would be expected to have been constructed prior to the respondent's birth. This is expected to have an effect on source awareness particularly with reference to well design details/specifications. The RA study found 70% and 72% of hand-dug wells and Bored SPS respondents respectively were in residence at the time of source construction, while the SA study found these proportions to be 34.4% and 63.2% respectively. The large difference found between RA and SA respondents supplied by hand-dug wells is probably indicative of decreasing frequency of these source types.

8.3.1.3. Source Age

46.8% of respondents were aware of the age of their source (n=140). Significant differences were found between source type and knowledge of well age(χ^2 (4)=25.349, p<0.001), with 80% of PrGWS respondents unaware of well age, compared with 55% and 53% of hand-dug wells and bored SPS users respectively. This is possibly due to the lower average age of SA respondents and lower levels of consumer involvement in group water schemes (public and private), compared with private individual wells.

8.3.1.4. Construction Involvement

22.7% of respondents were aware of the contractor involved in source construction (n=68), which is significantly lower than RA surveys at 63%.This finding is again probably attributable to lower average respondent age levels of residency during the source design and construction phase. There was a significant difference found between contractor awareness and source type (χ^2 (4) =11.556, p= 0.021), with 25%, 27% and 4.4% of hand-dug wells, bored SPS and PrGWS users respectively exhibiting an awareness of contractor involvement. Only 30% of respondents' were aware of the involvement of a hydrogeologist, water diviner or local authority (n = 90), a marked reduction on the 66.1% of RA respondents. A significant difference was not found between source type and overall involvement awareness. *Figure 8.12* illustrates the frequency of this involvement, with the overall trend of involvement being broadly similar to that shown in the analysis of RA surveys.

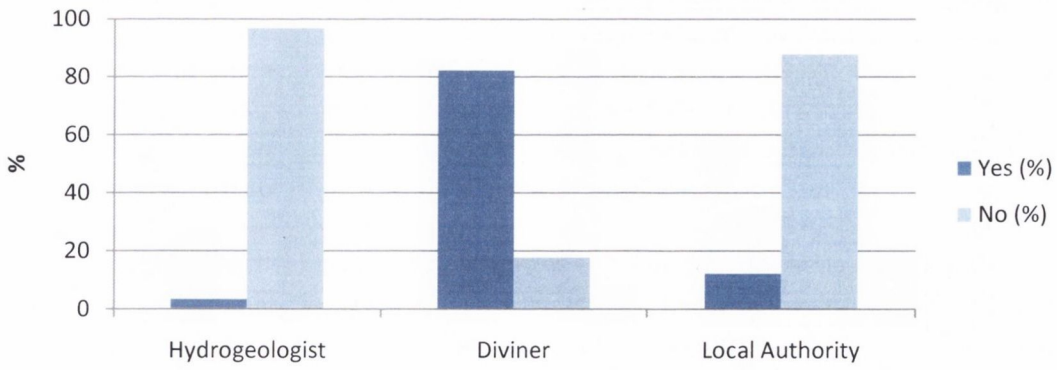


Figure 8.12 Involvement in Source Construction (n = 90)

A significant difference was found between water diviner involvement and source type (χ^2 (4)= 17.335; p = 0.001), with 85% and 88.5% of hand-dug wells and bored SPS users respectively having utilised the services of a water diviner during the source design phase, whereas 37.5% of PrGWS utilised a water diviner (see Appendix F4).Similarly, the involvement of a local/planning authority in

well design/construction was found to differ significantly with regard to source type ($\chi^2(4)= 40.575$; $p < 0.001$), with 5% and 4.9% of hand-dug wells and bored SPS users, respectively, acknowledging local authority involvement, while this figure stood at 75% for PrGWS users. This difference was expected.

8.3.1.5. Design/Construction Records

Approximately 40% of respondents (n = 164) were aware of whether or not they had records of the drilling process. Of these, 19.8% of respondents (n = 33) indicated they had records of drilling/construction, which is relatively similar to results found during RA analysis (22.8%). A significant difference was found between source type and construction record awareness ($\chi^2(4)= 11.675$; $p < 0.02$), with 55%, 37.5% and 33.3% of hand-dug wells, bored SPS and PrGWS users respectively being aware of well records. Similarly, there was found to be a significant difference between the retention of construction records and source type ($\chi^2 = 18.009$; $p = 0.001$), with none of those respondents supplied by hand-dug wells (n = 34) retaining any records of source design/construction. This finding is unsurprising due to the nature of hand-dug wells design/construction. 32.8% of bored SPS users had borehole logs/receipts/etc, comparable to 30% found during RA analysis. Only 6.6% of PrGWS users had design/construction records (see Appendix F5).

8.3.1.6. Source Component Details

As illustrated in *Figure 8.13*, there was a low overall awareness of the investigated well design components, with no single component associated with an awareness level greater than 40%. Similar to RA findings, the presence of a sanitary seal was the component with which the lowest level of awareness was associated, with only 6.1% of respondents exhibiting an awareness of this necessary component. There was no significant difference associated with sanitary seal awareness and source type, however, a number of other components did show significant differences in this regard, as shown in *Tables 8.8 & 8.9*.

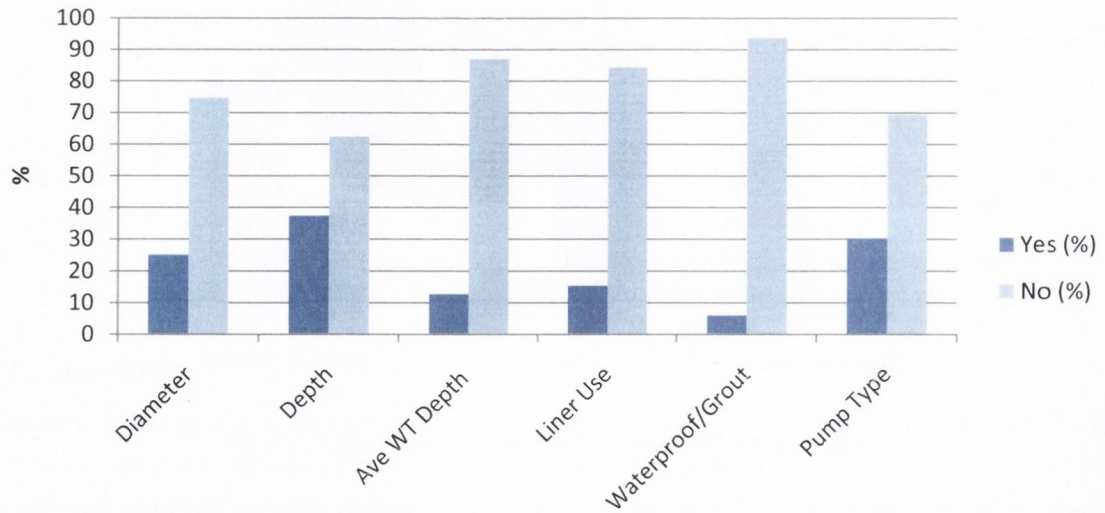


Figure 8.13 Overall Source Detail Awareness (n = 299)

Table 8.8 Groundwater Source Component Awareness (RA) (n = 299)

	Well Diameter		Well Depth		Average Watertable		Well Liner		Sanitary Seal		Pump Type	
	Yes	No	Yes	No	Yes	No	Yes	No	Yes	No	Yes	No
Hand-dug wells	21 (35)	39 (65)	26 (43)	34 (57)	13 (22)	47 (78)	9 (15)	51 (85)	4 (7)	56 (93)	20 (34)	40 (66)
Bored SPS	50 (27)	135 (73)	79 (43)	106 (57)	23 (12)	162 (88)	34 (18)	151 (82)	13 (7)	172 (93)	61 (33)	124 (67)
Unknown	0 (0)	9 (100)	1 (11)	8 (89)	0 (0)	9 (100)	0 (0)	9 (100)	0 (0)	9 (100)	2 (22)	7 (78)
PrGWS	4 (9)	41 (81)	6 (13)	39 (87)	2 (4)	43 (96)	3 (7)	42 (93)	1 (2)	44 (98)	8 (18)	37 (82)
Total	75 (25)	224 (75)	112 (37)	187 (63)	38 (13)	261 (87)	46 (15)	253 (85)	18 (6)	281 (94)	91 (30)	208 (70)

Percentages in parentheses

Table 8.9 Results of Chi-square analysis of source type and well component awareness (n = 299)

Well Component	X^2	*df*	*P*
Well Diameter	*12.805*	*4*	*0.012*
Well Depth	*18.562*	*4*	*0.001*
Ave. Watertable	*8.433*	*4*	*0.077*
Liner Presence	*5.544*	*4*	*0.236*
Seal Presence	*2.1*	*4*	*0.717*
Pump Type	*5.092*	*4*	*0.278*

The statistically significant differences illustrated in *Table 8.10* are probably due to a low overall awareness of source components by PrGWS and "unknown source" users, with levels of awareness among hand-dug wells and bored SPS users being broadly comparable. In the case of well diameter awareness, awareness rates displayed by hand-dug wells, bored SPS and PrGWS users were 35%, 27% and 8.8% respectively, with figures of 43.3%, 42.7% and 13.3% respectively in connection with well depth. Bored SPS users were also found to display low rates of awareness, with corresponding awareness figures of 23.6%, 12.4% and 4.4% awareness. It is considered that a lower level of involvement of PrGWS users leads to this perceived lower level of source awareness. The median number of well components recognised by both hand-dug wells and bored SPS was one out of six, while this figure was zero out of six among PrGWS and "unknown source" users.

Results of an independent samples t-test suggest that while there was no significant difference between overall well component awareness on the part of hand-dug wells and bored SPS, there were significant differences between both hand-dug wells and PrGWS ($t = 4.32$, $p < 0.001$) and bored SPS and PrGWS ($t = 4.986$, $p < 0.001$), with PrGWS respondents exhibiting significantly lower awareness in both cases.

8.3.1.7. Water Treatment

86.8% of respondents were aware of whether or not their water source was being treated (n = 258), with 30.2% of these (n = 78) indicating that their source was currently being treated. This is comparable with RA survey analysis results. Further analysis shows that there is a significant

difference between water treatment rates across the source types (χ^2 (3)= 36.819, $p < 0.001$)with 18.9% and 25% of surveyed hand-dug wells and bored SPS users aware of a treatment system being in place, while 64.5% of PrGWS users are aware of a treatment system. 100% of those users describing their supply as "Other" were aware of treatment being in place.

8.3.1.8. Source Maintenance

74.2% of respondents (n = 222) were aware whether or not some form of maintenance had previously been carried out on their private source. This figure is comparable to that of RA survey respondents (82.9%). Of those that were aware (n = 222), 37.4% were aware of some maintenance having been carried out, with the remainder responding that no maintenance had been carried out to their knowledge. Once again, this figure is comparable with that of RA survey respondents (35.1%).32.6% of bored SPS had undergone some form of maintenance with this figure being 49% for hand-dug wells, with a similar pattern evident for RA survey analysis. 52.9% of PrGWS users were aware of previous maintenance of their source. Differences were not found to be significant apropos previous maintenance works being carried out, however a significant difference in awareness was found, with 85% of hand-dug wells and 81.1% of hand-dug wells and bored SPS users aware of whether or not maintenance had been undertaken, with an awareness rate of 37.7% of PrGWS users (χ^2 (3)= 46.876, $p < 0.001$) (see Appendix F6).

8.3.1.9. Important Factors in Well Design and Construction

Figure 8.14 illustrates those factors considered important in well design/construction by SA survey respondents. Water quality was considered an important factor by the majority of respondents (70.4%), with well yield (57.6%) and cost (44.7%) also considered important. This pattern of perceived importance is similar to that shown by RA respondents, however, there is a lower overall importance attributed to water quality and yield among SA respondents, with the equivalent proportions for RA respondents being 95.5% (water quality) and 89.3% (well yield). As potentially important factors, construction cost, well depth, well location and other factors all showed comparable results between RA and SA respondents. This may be due, in part, to a lower sense of "ownership" of the water source and associated problems on behalf of the younger SA respondent dataset. A significant difference was found between the level of importance attributed to water quality and source type (χ^2 (3)= 9.559, $p = 0.023$). While 69% and 67.8% of hand-dug wells (n = 56) and bored SPS (n = 177) users agreed that water quality was an important factor, this figure was 86% for PrGWS users (n = 43).

A total of 55.7% of respondents stated that water quality was the most important factor, with equivalent figures for yield and location of 23.1% and 9.9%, respectively. No significance was found between most important factor and respondents' source type.

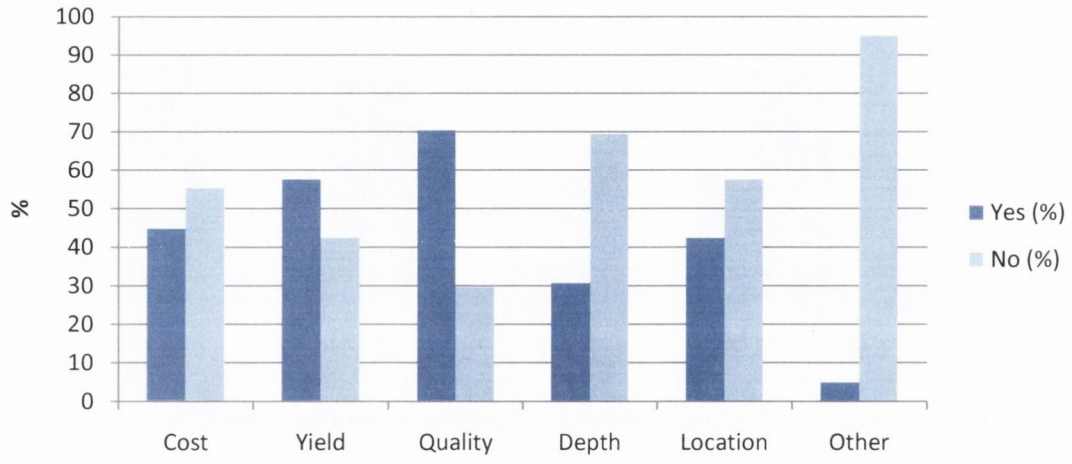


Figure 8.14 Factors considered important by SA respondents (n = 284)

8.3.2 Water Quality Analysis

8.3.2.1. Water Quality Analysis Awareness

Overall, 76.5% of SA respondents (n = 225) were aware whether or not their source had been tested at some point, compared with 93.5% of RA survey respondents. No significant difference was found between respondents' awareness of previous water quality analysis and respondents supply type. Of those that exhibited an awareness of previous analysis,56% (n = 121) indicated that their water had been tested at some point, which is comparable to RA survey respondents(59.8%), with 50% of respondents aware of the water quality analysis results. Awareness of water quality analysis results was much lower among the SA sample than the RA sample. There was a significance found between respondents' knowledge of previous water quality testing results and respondents' source type (χ^2 =15.744, p = 0.003), with 46.2% and 59.5% of hand-dug wells and bored SPS users, respectively, exhibiting an awareness of previous testing results, while this figure stood at 11.8% for PrGWS users. This finding may be expected as users of hand-dug wells and bored SPS will generally bear the cost of water quality testing of their source, however, in the case of PrGWS users, the cost will probably be shared by all users or borne in part by a local authority, and therefore perceived as being of less importance due to a lower financial burden or lower sense of "ownership" of source.

8.3.2.2. Water Testing Importance

Results indicate that there is no significant difference between SA survey respondents' source type and level of importance attributed to regular water quality testing of their source, with all source types following a similar pattern.

Similar respondent proportions assigned low and high importance to regular testing within the RA and SA survey groups. A much lower proportion of SA respondents however, assigned a medium level of importance to regular analysis (14.6%) when compared with RA respondents (50.6%). This is probably due to the relatively high proportion of undecided respondents within this survey group (27.8%), as shown in *Figure 8.15*.

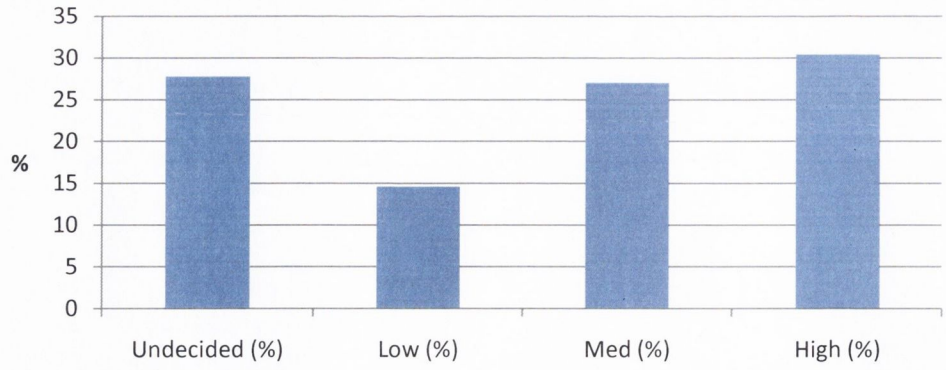


Figure 8.15 Perceived importance of regular water quality analysis (n = 295)

33% of SA respondents (n = 97) indicated that they would be prepared to pay for regular water testing, a marked departure from 61.6% of RA respondents. No significant difference was found between respondents attitudes towards payment for regular water quality testing and source type with 39% of hand-dug wells users agreeing they would pay for regular testing while the equivalent for bored SPS and PrGWS users was 30.4% and 37.58%, respectively.

8.3.3 Groundwater Contamination

8.3.3.1. Contaminant Recognition

Respondents' knowledge of a number of groundwater contaminants, both microbial and chemical, is presented in *Figure 8.16*. Contaminant awareness of SA respondents was generally comparable with those of RA respondents, with similar patterns evident. There was, however, a lower awareness of *E. coli*/faecal coliforms and *Cryptosporidium* among SA respondents. It is unclear as to why this was the case. Significance was found with regard to source type and associated groundwater contaminant awareness. However, further analysis was carried out as proportions did not seem to support this finding i.e. positive awareness of *Giardia* was 13.5% amongst hand-dug wells users, 5.6% for bored SPS users and 8.9% amongst PrGWS users. Investigation of the data found there was a disproportionate level of knowledge among those respondents associated with "other groundwater sources", however these only numbered 7 respondents and were found to significantly reduce the power of statistical tests. These respondents were subsequently excluded from analysis and hereafter no statistically significant difference was found between source types and groundwater contaminant awareness.

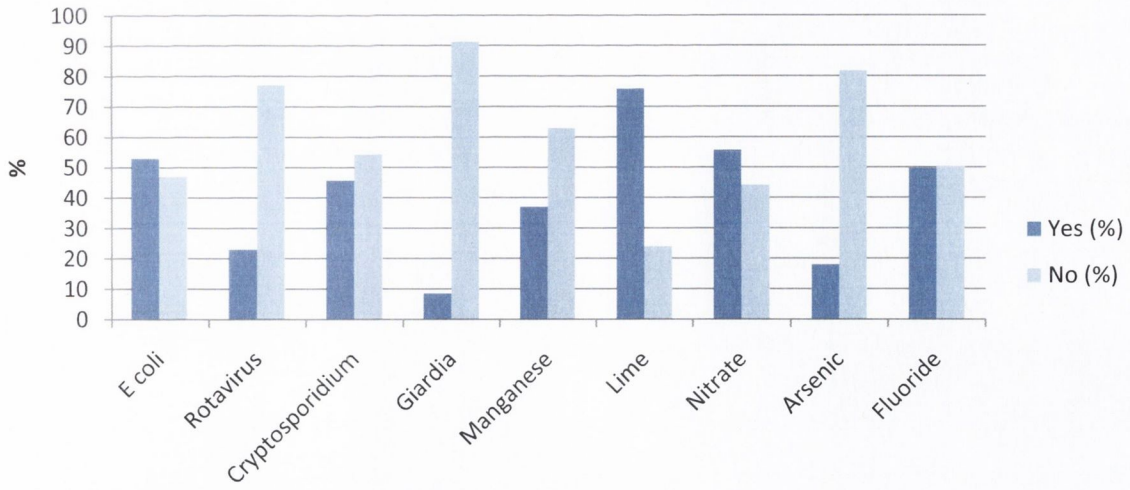


Figure 8.16 SA Groundwater Contaminant Awareness (n = 291)

Similarly to RA survey analysis, there was a low relative awareness of rotavirus (23.1%) and *Giardia* (8.6%) as potential groundwater contaminants, with higher apparent awareness of *E. coli* (52.9%) and *Cryptosporidium* (45.3%).

No significant mean difference was found between the total number of contaminants recognised by respondents and their source type. However, a significant mean difference was found in relation to gender and contaminant recognition ($t = 1.984$, $p = 0.048$), with female respondents exhibiting a median awareness of five out of nine and male respondents a median awareness of four out of nine. A Significant difference was also suggested in relation to contaminant recognition and knowledge of well age($t= -2.508$, $p = 0.013$), with those respondents aware of their source age typically exhibiting a higher overall contaminant awareness. Additionally, those respondents with a higher overall contaminant recognition were found to have a greater level of appreciation of water quality as a factor in well design/construction ($t= -3.152$, $p = 0.002$) and displayed a greater likelihood of selecting "water quality" as the most important factor in well design/construction ($t = 2.344$, $p = 0.02$).

8.3.3.2. Potential Health Effects

An investigation of respondents' awareness of potential health effects stemming from consumption of contaminated groundwater found that only26.7% ($n = 78$) of total SA respondents were aware of at least one potential illness or symptom. This is a marked departure from RA survey results, where 69.8% of respondents were found to demonstrate an awareness of at least one waterborne symptom or illness. It is unclear as to why the awareness of potential health effects was so low among these respondents, as it would be expected to be higher among a younger, educated demographic. This lack of awareness may be due in part to the "lack of ownership" of private sources attributed to this dataset. There was found to be a statistically significant difference between source type and health hazard awareness ($\chi^2 = 11.889$, $p = 0.008$),with a considerably higher level of health hazard awareness exhibited by PrGWS (45.4%) users than either hand-dug wells (26.6%) or bored SPS (20.7%).

As shown in *Table 8.10*, respondents' awareness of the potential health effects of contaminated groundwater consumption had a significant relationship with recognition of all investigated contaminants. A similar pattern was evidenced for RA surveys, however, SA survey respondents had generally stronger relationships, all significant at a 95% level i.e. respondents exhibiting an awareness of *Cryptosporidium* spp. were more likely to exhibit an awareness of potential health effects attributable to contaminated groundwater sources.

Table 8.10 Results of Chi-square analysis of health hazard awareness and contaminant awareness

	χ^2	df	p
E. coli	30.584	1	<0.001
Rotavirus	16.63	1	<0.001
***Cryptosporidium* spp.**	35.79	1	<0.001
***Giardia* spp.**	16.869	1	<0.001
Manganese	7.868	1	0.004
Lime/Hardness	18.312	1	<0.001
Nitrate	33.429	1	<0.001
Arsenic	11.95	1	0.001
Fluoride	20.921	1	<0.001

Awareness of previous water quality analysis was also found to have a relationship with health effect awareness (χ^2 (1)= 4.825, p = 0.029), however, this relationship, although significant at the 95% level, was not particularly strong and was not found to translate to actual previous water quality analysis (χ^2 (1) = 3.782, p = 0.07) at a 95% level. Subsequently, a Mann-Whitney U-test found no statistical difference between perceived importance of regular analysis on the part of the respondent and health effect awareness.

8.3.3.3. Activities potentially affecting water quality

35.7% of total respondents (n = 106) were aware of a potential hazard source within the surrounding area of their source. No significance was found between awareness of a potential groundwater hazard and respondent source type, nor were any significant relationships suggested in relation to residence or contractor awareness. There was however a significant relationship indicated in relation to the respondents' source age awareness and potential hazard source awareness (χ^2 (1)= 7.645, p = 0.004), with those respondents exhibiting an awareness of their source significantly more likely to have an awareness of potentially polluting activities.

Subsequent questioning investigated the presence of a number of potentially contaminating activities within 100 m (generalised inner protection zone) of their source. Findings are summarized in *Figure 8.17* which, when considered together with the aforementioned figure of 35.7% hazard source awareness suggest that in excess of 11.1%, 10.4% and 34.1% of respondents did not consider adjacent farmyards, slurry pits and grazing animals respectively, as potential sources of groundwater contaminants. As previously discussed, this presents a concern due to the relative frequency of these activities in areas with high private groundwater reliance.

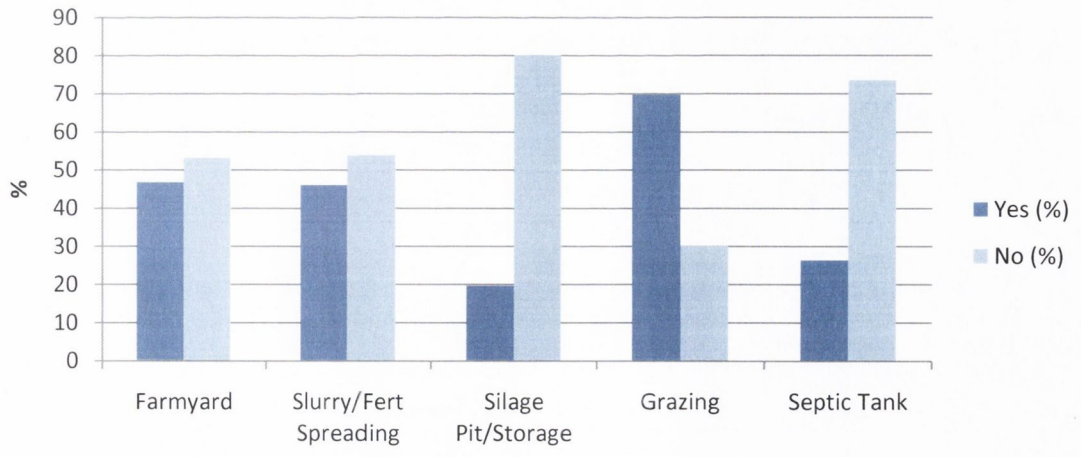


Figure 8.17 Activities within Inner Protection Zone (n = 295)

Bored SPS users exhibited a greater lack of hazard source awareness, with at least 14.7%, 18% and 39.3% not identifying adjacent farmyards, slurry and/or fertiliser spreading and grazing animals, respectively, as potential contaminant sources.

8.3.4 Human Health

8.3.4.1. Household Composition

Table 8.11 summarises household composition, indicating the percentage of households having at least one occupant in each subset.

Table 8.11 Typical household composition

	Infants (<1) (%)	Young children (1-5) (%)	Children (6-18) (%)	Adults (18-65) (%)	Elderly (>65) (%)
Hand-dug wells	0	20	45	100	15
Bored SPS	5.5	17	46.7	100	23.6
PrGWS	11.1	17.8	51.1	100	17.7

A decreased disparity between hand-dug wells and bored SPS household composition was found through SA surveys than suggested by the RA survey. Significant lack of independence was found with reference to the number of infants per household when compared with source type (χ^2 (3)= 13.034, p = 0.011), with PrGWS households having a much higher apparent instance of infant residents. The difference between elderly residence and source type, as previously shown to be significant in the RA survey study, was not found.

Household totals were then analysed. A series of independent samples t-tests found no significant mean difference when comparing source type and total household number. The overall mean household number was 5.48 (St. Dev. 2.16, n = 112), which is significantly higher than both RA survey and national averages, with mean household composition of 5.8, 5.5 and 4.6 relating to hand-dug wells, bored SPS and PrGWS, respectively. Possible reasons for this are unclear; however the low

response rate (37.3%) to this question and hence small sample size may be an explanatory factor. Minimum and maximum household sizes of 2 and 15, respectively, were recorded.

8.3.4.2. Consumption Source

The majority of respondents (91.6%) indicated the primary household consumption point as being a household tap (n = 271).Of the remainder (n = 25), 84% used bottled water. There was no significant difference found with regard to drinking water source and source type. Individual respondents were asked if they themselves consumed from a household tap (treated or untreated), with 88.5% stating that they did (n = 262).No significant difference was found with regard to consumption source and source type. Both household consumption source and individual respondent consumption source were comparable with RA survey findings. Household consumption source was not found to have any apparent association with gender or residency of respondent during source construction (although this was significant at 90% confidence level) as found in the RA survey group.

8.3.4.3. Consumption Patterns

An investigation of daily consumption frequency and volume patterns found a median consumption frequency of 2-3 times per day (n = 271), with no significant difference found between consumption frequency and gender or source type and an associated median range of 1-2 L day^{-1} (n = 271).No significant difference was found between daily consumption volume and gender or source type, however, an expected relationship was found between consumption frequency and consumption volume ($r = 0.667$, $p < 0.001$). These data and further analysis relating to consumption patterns will be presented in detail in *Chapter 10*. All consumption data relate to unboiled water from private groundwater sources not associated with preparation of food or hot beverages.

8.3.4.4. Water-related illness

Only 1.38% (4/289) of respondents reported previous (previous 12 months) instance of themselves or another person in the household having been diagnosed with a water-related illness by a medical practitioner. These were found in one hand-dug wells household, one PrGWS household and two bored SPS households. No statistical significance was found. The equivalent figure for RA surveys was 0.8%.

14.5% of total SA respondents reported themselves or another person in the household having shown potential symptoms of gastroenteritis/AGI in the past 12 months. These figures were 10.5%, 12.3% and 27.3% for hand-dug wells, bored SPS and PrGWS, respectively, with a chi-square test of independence suggesting a significant difference (χ^2 (2)= 7.236, $p = 0.027$) i.e. a significantly higher proportion of potential waterborne illness among PrGWS users. The figure of 27.3% for PrGWS stands out, particularly owing to an apparent 64.5% level of treatment (compared with 19.3% and 25% equivalent for hand-dug wells and bored SPS). However, this may be the case as when contamination of a PrGWS does occur, more widespread infection is possible due to typically higher number of users.

As previously noted, unlike positively diagnosed cases of water-related illness, these responses pertain to potential cases of general gastrointestinal illness and it is therefore probable that a proportion of these were foodborne or related to recreational water as opposed to being a direct result of water consumption. Actual diagnosis figures appear to be somewhat more realistic.

8.3.4.5. Source Satisfaction

Respondents were asked to rate their source satisfaction from 1 (very dissatisfied) to 5 (very satisfied), with the findings presented in *Figure 8.18*. No significant difference was found between hand-dug wells and bored SPS users with regard to overall source satisfaction, however, a significant difference was found between hand-dug wells and PrGWS user satisfaction, with hand-dug wells users typically exhibiting a higher level of source satisfaction ($U = 1{,}065$, $p = 0.036$).

As previously discussed, knowledge of health hazards and overall source satisfaction displayed an association ($U = 6{,}596$, $p < 0.01$), a trend which was also noted in RA survey analysis. This suggests a negative correlation between these two parameters i.e. an existing knowledge of waterborne illness results in a lower level of source satisfaction. Non-parametric tests found no discernable relationship between specific contaminant recognition and overall contaminant recognition and source satisfaction.

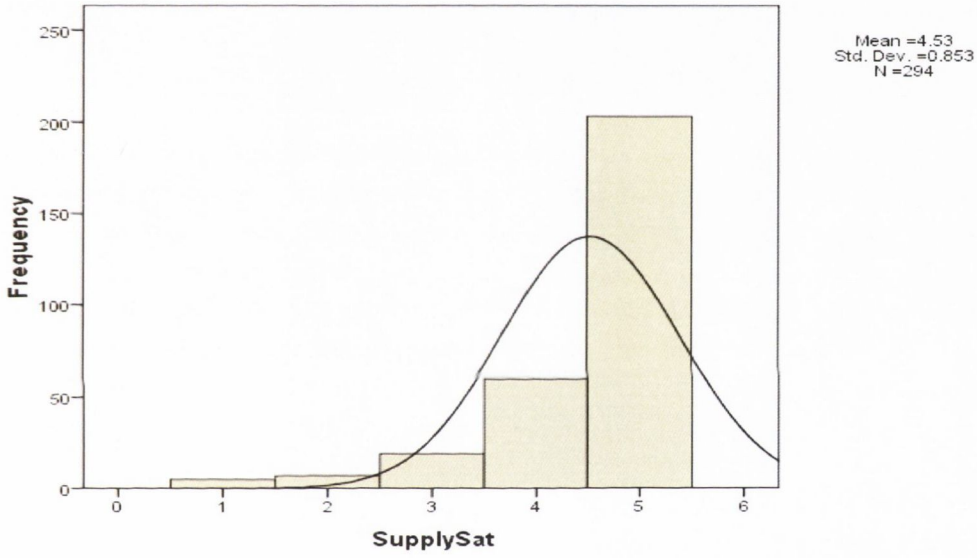


Figure 8.18 Source Satisfaction with overlain Gaussian Curve (n = 294)

Again, as previously observed among RA respondents (*Section 8.2.4.5*), a significant difference was found between household drinking source and source satisfaction ($U = 1{,}814$, $p < 0.001$), and respondent drinking source and source satisfaction ($U = 3274$, $p = 0.003$), with those respective households and respondents not using their well as the primary source of consumption exhibiting lower overall source satisfaction in both cases. Subsequently, correlations were indicated between source satisfaction and consumption frequency ($r = 0.188$, $p = 0.002$), consumption volume ($r = 0.134$, $p = 0.028$) and the instance of potential waterborne infection ($r = -0.169$, $p = 0.004$), indicating a level of sensitivity among respondents of health, consumption patterns and water source. A similar pattern was found in the RA group.

Household size and source satisfaction also exhibited significant difference ($F(4) = 3.988$, $p = 0.005$), suggesting a higher level of source satisfaction associated with larger household sizes, with this apparent correlation was found to be significantly linear ($r = 0.251$, $p = 0.008$). As was the case in the

RA survey group, the source satisfaction distribution is skewed (Skew = -2.144) and therefore, due to an overall high level of satisfaction, inference based on this distribution is difficult.

8.4 Awareness Scoring

As previously mentioned (*Section 8.1*), one of the primary research objectives of the awareness survey study was to define the level of awareness among private groundwater users in Ireland of a number of pertinent issues regarding private groundwater sources, groundwater contamination and the potential human health effects of contamination. The results have been used in two ways:

1. Assess the present situation with respect to the susceptibility of the existing private groundwater sources by including overall user/owner awareness as a parameter in the development of a groundwater source susceptibility model (*Chapter 9*)

2. Gauge the understanding of private well owners and users in Ireland of the relationships between contamination risk, water source and health. These data would provide an essential foundation for subsequent development of guidelines for the owners of new and existing private wells, so as to reduce the adverse health impacts in the future.

An awareness scoring methodology was therefore developed based upon completed surveys, with the scoring model designed to rate respondents perceived groundwater source and groundwater contamination awareness. A review of the literature found that although awareness scoring has been successfully used in a variety of fields ranging from neuropsychology to social sciences (Connolly *et al.,* 1992; Verhey *et al.,* 1993; Clare, 2004; Glazer *et al.,* 2005; Ogden *et al.*, 2006), it has been rarely used to quantify awareness of groundwater contamination or the subsequent health effects. One study (Kien & Ross, 2009), used awareness scoring in this way, however this was carried out in Northern Vietnam in relation to arsenic levels in groundwater.

The awareness questionnaire was reviewed and a total of 14 questions were selected as being suitable for the quantification of awareness on the respondents' part. Questions which focused on respondents' views or opinions, or were used to collect data on household composition or consumption, were not seen as being relevant to "groundwater awareness" and were subsequently excluded from the scoring model. A simple scoring methodology was then devised based on responses, as summarised in *Table 8.12*.

Table 8.12 Awareness Scoring Model

Question	Score
Q1.	
Knowledge of current household source type	
• Hand-dug wells/Bored SPS/PrGWS/PuGWS/Other	1
• Don't know	0
Q3/4 (Amalgamated)	
Knowledge of current source age	
• Number of years	1
• Don't know	0

Q5.
Knowledge of well contractor/digger

- Yes 1
- No 0

Q6.
Knowledge of construction involvement

- Yes/No 1
- Don't know 0

Q7.
Knowledge of well records

- Yes/No 1
- Don't know 0

Q8.
Knowledge of source construction details

- Awareness of 4-6 details 2
- Awareness of 1-3 details 1
- Awareness of 0 details 0

Q9.
Knowledge of source treatment

- Yes/No 1
- Don't know 0

Q11.
Knowledge of previous source maintenance

- Yes/No 1
- Don't know 0

Q14.
Knowledge of previous water quality analysis

- Yes/No 1
- Don't know 0

Q18.
Knowledge of groundwater contaminants

- Awareness of 5-9 contaminants 2
- Awareness of 1-4 contaminants 1
- Awareness of 0 contaminants 0

Q19.
Knowledge of potential health problems from groundwater contamination

- Yes 1
- No 0

Q20/21 (Amalgamated)
Knowledge of potential hazard sources in source vicinity

- Yes 1
- No 0

If respondent answered "No" to Q20, responses to Q21 are investigated

Knowledge of specific activities within inner protection zone

- Specific activities present -1
- Specific activities absent 0

Note: A negative score (-1) was attributed if respondent stated there was no potential hazard source in well vicinity, but further questioning found one or more potentially polluting activities within 100 m of their source.

The above scoring methodology resulted in two separate awareness scores:

1. A groundwater source awareness score, quantifying the overall level of knowledge attributed to a respondent about their own specific groundwater source. This score was calculated using responses from Q1-14 as illustrated in *Table 8.12*, resulting in a scoring range of 11, with minimum and maximum scores of 0 and 10, respectively.

2. A groundwater contamination score, quantifying the respondents' level of awareness concerning potential groundwater contaminants, potential health effects of groundwater contamination and potential contaminant sources in general and in the context of their own source. The score was determined using responses from Q18-21 as presented in *Table 8.24*, resulting in a scoring range of 6, with minimum and maximum scores of -1 and 4. In order to avoid statistical complications, all calculated contamination awareness scores were increased by 1, in order that negative scores were avoided, resulting in adjusted minimum and maximum scores of 0 and 5.

8.5 RA Source and Contamination Awareness

Outline statistics concerning calculated awareness scores are presented in *Figures 8.19 & 8.20*.

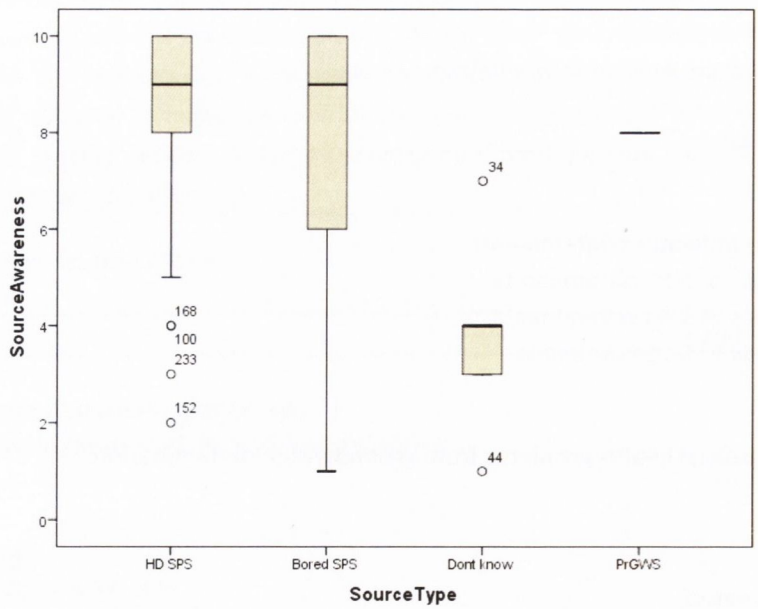


Figure 8.19 Boxplot of RA Source Awareness Scores (n = 245)

The median source awareness score for RA respondents (n = 245) was 9 and consequently somewhat higher than expected. As shown in the *Figure 8.22*, median scores for hand-dug wells and bored SPS were comparable, with their means being 8.5 (95% CI 7.97 – 9.03) and 7.85 (95% CI 7.51 – 8.19) respectively. Therefore hand-dug wells users exhibited a slightly higher overall awareness. Bored SPS users exhibited a wider range of apparent awareness. The mean difference was found to be significant (t (113) = 2.063, p = 0.041).

The mean source awareness score for respondents who were not aware of their source type (n = 7) was 3.8 (95% CI 1.11 – 6.49) and a median awareness score of 4. This is unsurprising as a private groundwater user unaware of their source type would not be expected to exhibit a high overall level of groundwater source and contamination awareness. Median source awareness scores for male and female respondents were 10 and 7 respectively, with male respondents therefore displaying a higher awareness overall than female respondents. This difference was found to be significant (t (1) = -4.718, p < 0.001).

The median contamination awareness score for RA respondents (n = 245) was 3 and therefore proportionately lower than source awareness. As presented in *Figure 8.23*, median scores for hand-dug wells and bored SPS were 3.5 and 3, respectively, with their means being 3.33 (95% CI 2.95 – 3.72) and 3.45 (95% CI 3.25 – 3.65) respectively. Therefore, although individual hand-dug wells users exhibited a slightly higher contamination awareness, bored SPS users displayed a narrower inter-quartile range, with fewer low scores i.e. a lower standard deviation. The mean difference between hand-dug wells and bored SPS users was not found to be significant. The mean awareness score for respondents who were not aware of their source type (N = 7) was 4.2 (95% CI 2.84 – 5) and a median awareness score of 5.

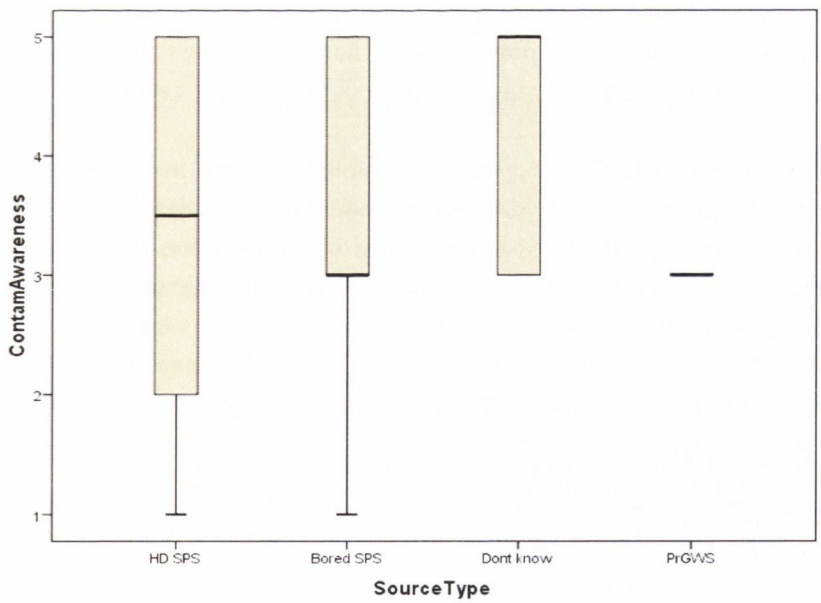


Figure 8.20 Boxplot of RA Contamination Awareness Scores (n = 245)

Median awareness scores for male and female respondents were 3 and 4, respectively, with male respondents therefore displaying lower contamination awareness than female respondents, which

was contrary to results of source awareness. Again however, this difference was not found to be significant.

Exploratory analysis (histograms, Q-Q plots; see Appendix F7) indicated that awareness scoring was not normally distributed among the sample population. Non-parametric correlations (Spearmans r) were therefore used to investigate possible relationships between variables of interest i.e. awareness score and variables not included in awareness scoring (to avoid multi-colinearity).

8.5.1 Source Awareness

Table 8.13 presents all statistically significant relationships found between RA respondents' source awareness score and non-related factors.

Table 8.13 Statistically significant outputs from analysis of source awareness and other factors (RA)

Factor	Test statistic	df	P
Residency during construction	t = -7.95	243	<0.001
Well Records	t = -2.288	204	0.023
Yield as important construction factor	t = -2.531	240	0.012
Depth as important construction factor	t = -2.818	240	0.005
Previous instance of potential related illness	t = 2.037	241	0.043

As shown above, the strongest relationship between source awareness and a non-related factor (i.e. a factor not included in scoring model), was between source awareness and residency during construction, with those in residence during construction (n = 171) displaying a mean source awareness score of 8.62, as opposed to those not in residence (n = 74) having a mean source awareness of 6.32. Another relationship of interest was found between those respondents' reporting a previous potential case of potential waterborne illness and source awareness, with those reporting previous illness (n = 24) typically less aware of their source details.

Perhaps of more interest than the suggested relationships were the expected relationships not borne out by data analysis, with no statistical relationship found between source awareness and groundwater contamination awareness or awareness of potential health effects of contamination. Additionally, no relationship was found to exist between respondents' source satisfaction and source awareness (see Appendix F8).

8.5.2 Groundwater Contamination Awareness

Only one non-related factor displayed a statistically significant difference in the context of contamination awareness, with those respondents exhibiting higher contamination awareness generally displaying a lower source satisfaction (r = -0.201, p = 0.002). Although those respondents from households not using their domestic well as the primary consumption source showed a higher mean contamination awareness (3.90), compared to those that did (3.39), this difference was not found to be significant. However, those respondents (as opposed to households) that did not drink from their household tap (largely preferring to consume bottled water) exhibited no mean contamination awareness difference. These results would seem to suggest that respondents' source

awareness and contamination awareness were distinctly independent, with higher contamination awareness, although seen to negatively affect overall source satisfaction, not altered by knowledge of their source, the household composition or previous instances of illness. Consequently it does not seem to affect overall consumption trends or respondent attitudes or behaviour.

8.6 SA Survey Awareness

Outline statistics concerning calculated source and contamination awareness scores from SA respondents are presented in *Figures 8.21 & 8.22*.

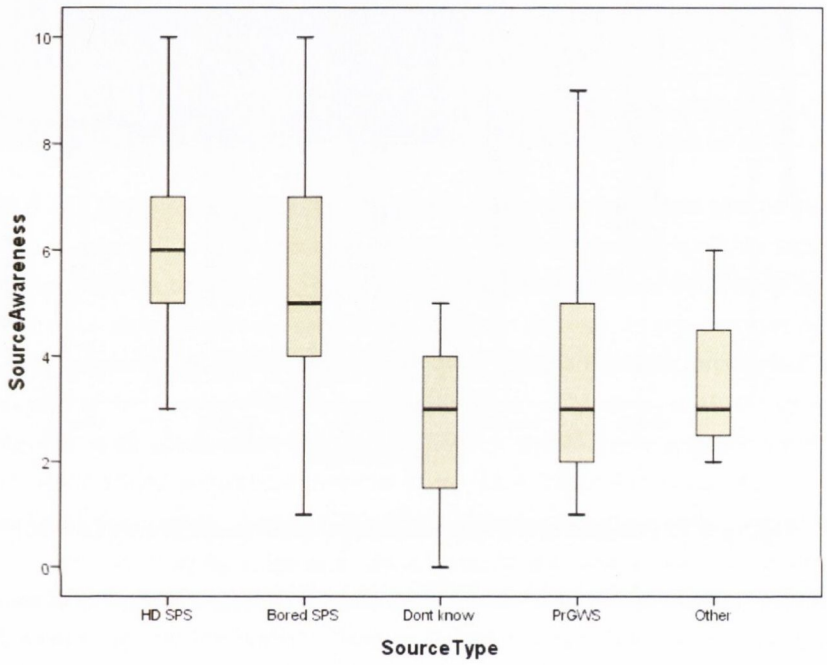


Figure 8.21 Boxplot of SA Source Awareness Scores (n = 300)

The median source awareness score for SA respondents (n = 300) was 5 and therefore a marked decrease when compared with an RA source awareness of 9. This may be due to the younger demographic comprising and also, as previously mentioned, the probable lack of ownership (literally and figuratively) attributed to the SA demographic in relation to their private source. Hand-dug wells users were seen to have higher overall source awareness than bored SPS users, with awareness medians of 6 and 5 respectively, although this difference was not significant. PrGWS users had a median source awareness of 3, which was expected due to a lower level of overall involvement associated with these consumers. A one-way ANOVA test found this result was significant ($F(4) = 12.354$, $P < 0.001$), with a post-hoc Bonferroni multiple comparison test suggesting the overall difference could be attributed to significant mean differences between both hand-dug wells and PrGWS (mean difference = -2.289, $p < 0.001$) and bored SPS and PrGWS (mean difference = -1.855, $p < 0.001$). As shown in *Figure 8.23*, those respondents both unaware of their source type and those categorizing their source type as "other", displayed similar overall source awareness to PrGWS users.

Median source awareness scores for male and female respondents' were 5 and 4 respectively, with male respondents therefore displaying a greater source awareness than female respondents, a significant difference ($t(13) = -2.473$, $p = 0.028$), as previously found for RA respondents'.

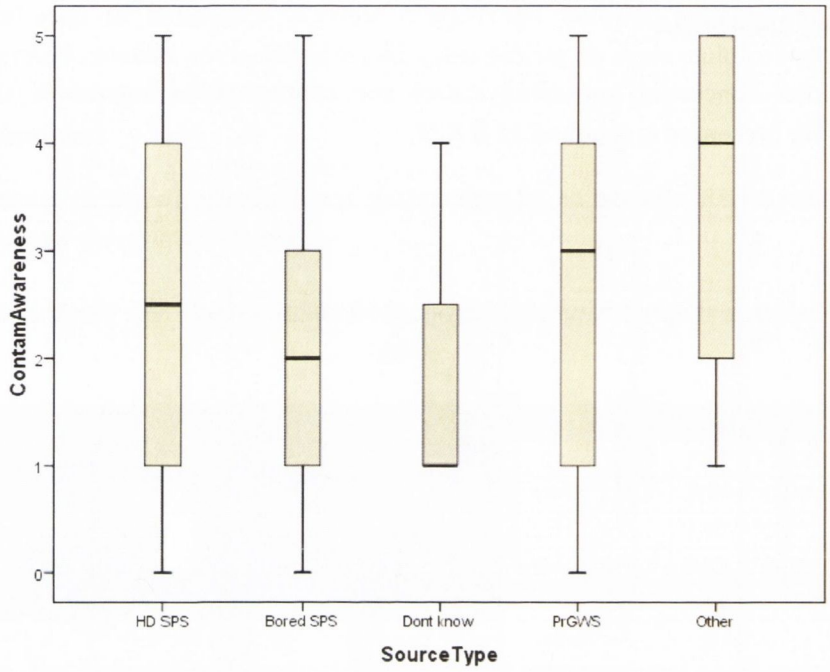


Figure 8.22 Boxplot of SA Contamination Awareness Scores (n = 300)

The median contamination awareness score for SA respondents (n = 365) was 2. As shown in *Figure 8.22*, hand-dug wells users exhibited a higher overall contamination awareness than Bored SPS users, with median contamination awareness scores of 2.5 and 2 respectively, with bored SPS users displaying a narrower overall inter-quartile range. An independent samples t-test found this mean difference was not statistically significant. These findings are comparable with those from the RA dataset. The median contamination awareness score for surveyed PrGWS users (n = 45) was 3. One-way ANOVA tests with associated multiple-comparison tests found that there was no mean difference between contamination awareness and source type. It is unclear why those respondents associated with "other" source types had a higher median contamination awareness score.

Median contamination awareness scores for SA male (n = 287) and female (n = 13) were 2 and 3 respectively, with an independent samples t-test not finding a significant difference.

Exploratory analysis (histograms, Q-Q plots; see Appendix F7) indicated that SA awareness scoring was normally distributed among the sample population. However, as not all potentially correlated factors were normally distributed, non-parametric methods were used to investigate possible relationships between allocated levels of awareness and unconnected variables of interest i.e. variables not included in awareness score calculation.

8.6.1 Source Awareness

Table 8.14 presents all statistically significant relationships found between SA respondents' source awareness score and non-related factors.

Table 8.14 Statistically significant outputs from analysis of source awareness and other factors (SA)

Factor	Test statistic	df	P
Residency during construction	t = -2.814	296	0.005
Well Records	t = -4.268	119	<0.001
Yield as important construction factor	t = -3.193	281	0.002
Contaminant recognition	r = 0.185		0.002
Average daily consumption volume	r = 0.141		0.02
Contamination Awareness	r = 0.125		0.03

As shown (*Table 8.16*), the most significant relationship found was between source awareness score and respondents access to their well records. Although this relationship is highly significant, it is to some extent unimportant as only 19.8% (n = 121) of those respondents exhibiting an awareness of well records went onto state that they had access to these records. In the context of the entire SA dataset, these only account for 8% of respondents. Similarly to results presented in *Table 8.15*, residency was found to be a major factor in overall source awareness, with those respondents in residence during the well construction period exhibiting significantly greater source awareness. Other significant associations included a positive linear association between source awareness and potential contaminant awareness, as well as a similar relationship concerning consumption volume, with those respondents displaying increased source awareness also citing increased average daily consumption volumes. A weak positive linear relationship was evidenced between source and contamination awareness.

8.6.2 Contamination Awareness

Table 8.17 presents all statistically significant relationships found between SA respondents' groundwater contamination awareness score and non-related factors. Those respondents indicating an awareness of their source age, source construction involvement and source components also tended to exhibit a higher overall groundwater contamination awareness, also indicated by the weakly positive linear relationship between source awareness and contamination awareness. Those respondents stating that they did not consume water from their groundwater source, instead choosing to consume from "other source", of which the primary source was bottled water, exhibited significantly lower contamination awareness than those respondents consuming directly from their source. It is unclear as to why this is the case, as the opposite would be expected. Two potential reasons are hypothesized;

1. Drinking water source and contamination awareness are not related, leading to a lower level of interest regarding contamination issues on behalf of those consumers not drinking from their private source, and,

2. Those consumers whose main source of drinking water is bottled water may be located in urban areas for a significant proportion of the time i.e. work, school, etc, and therefore have a lower level of interest in contamination issues

Additionally, a significant mean difference was found between contamination awareness and previous cases of potentially waterborne illness, with those respondents reporting a previous illness, either contracted by themselves or within their household, typically having higher contamination awareness. This is unsurprising, as where a potentially waterborne infection/illness occurs, it is probable that associated consumers awareness level of waterborne contamination will increase.

Table 8.15 Statistically significant outputs from analysis of contamination awareness and other factors (SA)

Factor	Test statistic	df	P
Awareness of well age	t = -2.555	297	0.011
Involvement awareness	t = -3.323	297	0.001
Well component awareness	r = 0.168		0.004
Cost as important construction factor	t = -2.109	282	0.036
Yield as important construction factor	t = -3.447	281	0.001
Quality as important construction factor	t = -2.664	282	0.008
Location as important construction factor	t = -2.067	281	0.04
Respondents consumption source	t = -2.16	294	0.032
Previous instance of potential related illness	t = -3.456	287	0.001
Source Awareness	r = 0.125		0.03

8.7 Awareness and Susceptibility

In order that the potential effects of groundwater source and groundwater contamination awareness could be quantified, a susceptibility paradigm was developed. This model, based upon the Source-Pathway-Receptor framework for risk-based decision making, comprises two distinct hazard receptors and consequently two separate measures of susceptibility, namely source susceptibility (primary susceptibility) and consumer susceptibility (secondary susceptibility), as shown in *Figure 8.23*. The Source-Pathway-Receptor model traditionally encompasses one potential contaminant receptor, usually an aquifer or well; however, consumer susceptibility in this case is being used to represent an extra level of potential consumer protection through a number of consumer controls, namely water treatment, source maintenance/rehabilitation and periodic water quality analysis. Therefore, for the purposes of this research, source susceptibility will be defined as "the susceptibility of a private domestic well to contamination by microbial and/or chemical contamination due to a number of potential causative factors including local hydrogeology, well location, design and construction". Consumer susceptibility is defined as "the susceptibility of a well user to consumption of a microbial and/or chemical contaminant, if present in a private well" and is based upon the assumption that where one or more of the aforementioned controls are in place, in the case of a private well being contaminated, the consumer is less likely to consume potentially harmful substances.

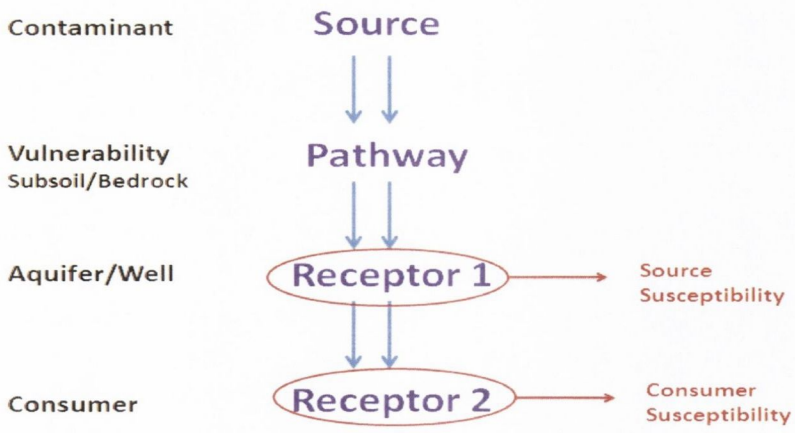


Figure 8.23 Susceptibility Model

In order that respondent awareness and its potential effect on susceptibility be quantified, measures against which awareness scores could be analysed were necessary. In the case of primary susceptibility, this measure is the presence or absence of microbial contamination in a well based on "one-off" sampling. For secondary susceptibility, this measure is the presence or absence of the consumer controls, namely previous well maintenance, utilisation of a water treatment system and previous water quality analysis. Additionally, the number of consumer controls in place was used to give an overall measure of secondary susceptibility.

8.7.1 Primary Susceptibility

In total, 137 awareness surveys were carried out, with which there was an associated "one-off" groundwater sample collected and analysed. These data are now investigated in order to consider the existence of an association between consumer awareness and well susceptibility. Within the sample dataset being used (N = 137), 44 samples tested positive for the presence of *E. coli*, resulting in a contamination rate of 32.1%. As discussed in *Chapter 7*, the overall contamination rate found among all "one-off" groundwater samples was 29.4% and therefore this sub-sample is closely representative.

E. coli presence/absence

Independent samples t-tests were carried out in order to examine if a mean difference awareness score was evident between respondents whose sources tested positive for the presence of *E. coli* and those whose sources tested negative. As shown in *Table 8.16*, in both cases i.e. source awareness and contamination awareness, respondents whose sources tested positive for *E. coli* exhibited slightly higher mean awareness scores. However, the results of the statistical analysis show that neither of these mean differences were considered significant at a 95% level.

Table 8.16 Source/Contamination Awareness V E. coli Presence/Absence

		N	Mean	Std. Dev.	Std. Mean Error
SourceAwareness	Absent	93	7.74	2.489	0.258
	Present	44	8.09	1.927	0.194
ContamAwareness	Absent	93	3.48	1.396	0.145
	Present	44	3.70	1.286	0.194

Further analysis was then undertaken to investigate the existence of relationships between respondent awareness and level of *E. coli* contamination (cfu/100ml), where present. Non-parametric Spearman correlations were used for this purpose. As shown in *Table 8.17*, no significant association between respondent awareness and contamination concentration was found among the amalgamated source type dataset (i.e. hand-dug wells + bored SPS). When source types were separated and statistical tests rerun, no statistically significant relationship was found to exist between hand-dug wells users with regard to level of awareness (both source awareness and contamination awareness) and the presence and/or level of microbial contamination found in their well (*Table 8.18*). The same outcome was found among Bored SPS users (*Table 8.19*).

Table 8.17 Non-Parametric correlations between contamination magnitude (where present) and source and contamination awareness (RA Respondents)

	E. coli (cfu/100ml)	Source Awareness Score	Contamination Awareness Score
Correlation Coefficient	1.000	0.008	0.083
Sig. (2-tailed)	.	0.928	0.332
N	137	137	137

Table 8.18 Hand-dug wells Awareness Vs E. coli Presence

	E.coli Pres/abs	N	Mean	Std. Deviation	Std. Error Mean
SourceAwareness	Absent	14	8.57	2.065	.552
	Present	13	8.38	1.805	.500
ContamAwareness	Absent	14	3.36	1.646	.440
	Present	13	3.77	1.235	.343

Table 8.19 Bored SPS Awareness Vs E. coli Presence

	E. coli pres/abs	N	Mean	Std. Deviation	Std. Error Mean
SourceAwareness	**Absent**	75	7.84	2.348	.271
	Present	30	8.00	2.017	.368
ContamAwareness	**Absent**	75	3.48	1.369	.158
	Present	30	3.63	1.326	.242

8.7.2 Secondary Susceptibility

8.7.2.1. Treatment

In the RA dataset, a significant mean difference was found between respondents with treatment processes in place at their source and both respondents source awareness ($t(241) = 2.659$, $p = 0.008$) and contamination awareness ($t(241) = -2.986$, $p = 0.003$). Those respondents with treatment systems in place ($n = 79$) were found to have significantly lower source awareness, however, conversely these respondents were found to display higher groundwater contamination awareness, which was expected. In the SA dataset, no significant differences were found when comparing the utilisation of treatment systems and either source awareness or contamination awareness. Although the same trends were in evidence, as found in the RA dataset, neither was found to be statistically significant.

Among hand-dug wells users ($n = 117$), a significant mean difference was found between source awareness and the use of treatment systems ($t(115) = 2.555$, $p = 0.012$) with those respondents with treatment systems in place displaying a significantly lower source awareness. No relationship was found with regard to contamination awareness in this dataset. Bored SPS users ($n = 342$) exhibited a significant mean difference between the use of treatment and groundwater contamination awareness ($t(340) = -3.076$, $p = 0.002$), with those respondents using treatment systems having significantly higher contamination awareness, as expected.

Within the total dataset i.e. RA + SA surveys, approximately 84.6% ($n = 99$) and 66.1% ($n = 226$) of hand-dug wells and bored SPS users, respectively, stated that no treatment system was currently in place at their source. The difference between source type and treatment implementation was found to be statistically significant ($\chi^2(1) = 14.219$, $p < 0.001$). There are currently approximately 145,000 households in Ireland supplied by private groundwater sources (excluding PrGWS), therefore, it may be inferred from these data that approximately 90,000 households (or 260,000 consumers) are currently served by entirely untreated private groundwater supplies, with this figure rising to almost 122,000 households when water-softening systems are accounted for.

8.7.2.2. Maintenance

In the RA dataset, those respondents indicating that previous maintenance had been carried out at their source were found to exhibit significantly higher source awareness ($t(201) = -1.984$, $p = 0.049$), with these respondents having an average source awareness of 9.02, as opposed to 8.51 among those respondents who stated that previous maintenance had never been carried out. No significant relationship was found between source maintenance and respondents groundwater contamination awareness. SA respondents ($n = 222$) did not display any statistically significant mean differences concerning awareness scoring and previous maintenance having been carried out, with the same being true among hand-dug wells and bored SPS respondents.

Approximately 52.5% ($n = 53$) and 72.5% ($n = 219$) of hand-dug wells and bored SPS users within the total dataset indicated that no maintenance or rehabilitation work had previously been carried out at their source, with this being found to be statistically significant ($\chi^2(1) = 13.856$, $p < 0.001$). As above, if it is assumed that there are currently approximately 145,000 households in Ireland supplied by private groundwater sources (excluding PrGWS), it may be approximated that almost 110,000 households (or 320,000 consumers) are currently supplied by an unmaintained private groundwater source.

8.7.2.3. Water Quality Analysis

Both source awareness and contamination awareness were found to significantly affect the likelihood of water quality analysis having been carried out at the respondents' source. In both cases, a higher level of awareness was found to increase the probability of water quality analysis having been carried out. In the case of groundwater contamination awareness, the mean difference was found to be highly significant ($t(212) = -3.938$, $p < 0.001$), with source awareness found to be less significant, although still significant at a 95% level ($t(212) = -1.971$, $p = 0.05$). These findings were echoed among the broader bored SPS dataset, with both rising source awareness ($t(290) = -2.642$, $p = 0.009$) and rising contamination awareness ($t(290) = -3.881$, $p < 0.001$) increasing the probability of previous water quality analysis having been carried out.

Similar patterns were evident among SA respondents, however, the mean difference between previous water quality analysis and source awareness was not significant at a 95% level. In the case of contamination awareness, findings again suggested a higher probability of previous analysis having been carried out where there was higher awareness ($t(216) = -2.308$, $p = 0.22$). Again these findings were mirrored among hand-dug wells respondents, with previous water analysis and source awareness not having a significant relationship, and increasing contamination awareness suggesting a higher probability of analysis having been carried out ($t(105) = -2.391$, $p = 0.019$).

In total, it was found that 48.8% ($N = 52$) of hand-dug wells users stated that their source had never been tested, with a figure of 40.1% ($N = 117$) evident among bored SPS users. The difference between source types was not found to be significant. As before, it was assumed that a maximum of 10% of private groundwater sources in Ireland are hand-dug wells, with resulting extrapolations suggesting that up to 245,000 people are supplied by a private groundwater source which has not been previously tested.

8.7.2.4. Consumer Controls

Analysis was carried out to investigate if consumer awareness was related to levels of implemented consumer control (*Tables 8.20 – 8.22*). In the RA dataset, a one-way ANOVA test found no significant difference between respondents' source awareness and the level of consumer control in place at their source. However, highly significant variation between groups was found in relation to respondents contamination awareness and the associated level of consumer control ($F(3) = 8.212$, $p < 0.001$). Consequently, a post-hoc Bonferroni multiple-comparisons test was undertaken to investigate the pattern of this significance. *Table 8.20* details the findings of this statistical test, with statistically significant results presented in red. The evidence suggests that increased groundwater contamination awareness among respondents may result in higher levels of consumer controls in place, with those respondents declaring to have 2 or 3 consumer controls in place typically exhibiting higher contamination awareness.

A one-way ANOVA test found both source awareness and contamination awareness to be significantly associated with level of consumer controls in place (Table 8.21). Again, a post-hoc Bonferroni multiple-comparisons test was undertaken and similar findings resulted, with those respondents displaying increased awareness, both of their source and of groundwater contamination, exhibiting an increased level of self reported consumer control at their source.

When source types were separated and analysed, similar findings arose, with significant differences evident between groups in relation to contamination awareness and level of consumer control in among both hand-dug wells users ($F(3) = 3.665$, $p = 0.015$) and bored SPS users ($F(3) = 7.47$, $p < 0.001$). In both cases, multiple-comparisons tests suggested increased contamination awareness exhibited by respondents resulted in greater levels of consumer control. This finding was particularly significant in the case of bored SPS users (*Table 8.22).* Although no significant difference was found in relation to source awareness and level of consumer control among hand-dug well users, a significant difference was found among borehole users ($F(3) = 6.868$, $p < 0.001$), with multiple-comparisons tests strongly suggestive of increasing consumer control in concurrence with increased awareness.

Table 8.20 Multiple Comparisons Test Results RA Respondents

Consumer Control Number		Mean Difference	Std. Error	Sig.	95% Confidence Interval	
					Lower Bound	Upper Bound
0	1	-.462	.252	.411	-1.13	.21
	2	-1.094	.265	.000	-1.80	-.39
	3	-1.485	.400	.002	-2.55	-.42
1	0	.462	.252	.411	-.21	1.13
	2	-.633	.222	.029	-1.22	-.04
	3	-1.024	.373	.040	-2.02	-.03
2	0	1.094	.265	.000	.39	1.80
	1	.633	.222	.029	.04	1.22
	3	-.391	.382	1.000	-1.41	.63
3	0	1.485	.400	.002	.42	2.55
	1	1.024	.373	.040	.03	2.02
	2	.391	.382	1.000	-.63	1.41

Table 8.20 Multiple Comparisons Test Results RA Respondents

Consumer Control Number		Mean Difference	Std. Error	Sig.	95% Confidence Interval	
					Lower Bound	Upper Bound
0	1	-.462	.252	.411	-1.13	.21
	2	-1.094	.265	.000	-1.80	-.39
	3	-1.485	.400	.002	-2.55	-.42
1	0	.462	.252	.411	-.21	1.13
	2	-.633	.222	.029	-1.22	-.04
	3	-1.024	.373	.040	-2.02	-.03
2	0	1.094	.265	.000	.39	1.80
	1	.633	.222	.029	.04	1.22
	3	-.391	.382	1.000	-1.41	.63
3	0	1.485	.400	.002	.42	2.55
	1	1.024	.373	.040	.03	2.02
	2	.391	.382	1.000	-.63	1.41

Table 8.21 SA Awareness/Consumer Control ANOVA

		Sum of Squares	df	Mean Square	F	Sig.
SourceAwareness	Between Groups	57.034	3	19.011	4.987	.002
	Within Groups	1075.067	282	3.812		
	Total	1132.101	285			
ContamAware	Between Groups	23.633	3	7.878	3.625	.014
	Within Groups	612.818	282	2.173		
	Total	636.451	285			

Table 8.22 Multiple Comparisons Test Results Bored SPS Users

Consumer Control Number		Mean Difference	Std. Error	Sig.	95% Confidence Interval	
					Lower Bound	Upper Bound
0	1	-.582*	.207	.032	-1.13	-.03
	2	-1.065*	.235	.000	-1.69	-.44
	3	-.941*	.353	.049	-1.88	.00
1	0	.582*	.207	.032	.03	1.13
	2	-.483	.218	.162	-1.06	.09
	3	-.359	.342	1.000	-1.27	.55
2	0	1.065*	.235	.000	.44	1.69
	1	.483	.218	.162	-.09	1.06
	3	.125	.359	1.000	-.83	1.08
3	0	.941*	.353	.049	.00	1.88
	1	.359	.342	1.000	-.55	1.27
	2	-.125	.359	1.000	-1.08	.83

8.8. The effect of consumer controls on water quality and site condition

In order to investigate the effect of previous well maintenance, previous water quality analysis and the use of water treatment on the current water quality, a series of statistical tests were carried out. It was found that no statistical dependence was suggested between *E. coli* presence or levels of *E. coli* (where present) and previous maintenance work having been carried out (n = 137). This was also found to be the case regarding bacterial water quality and the presence of water treatment systems. However, although no statistical dependence was found at a 95% level of significance, between the presence of *E. coli* and previous water quality analysis, results implied some relationship at a 90% level (χ^2 (1)= 2.25, p = 0.092), with 26.6% of previously tested wells having tested positive for *E. coli*, while this figure was 38.7% for previously untested wells.

In order to investigate the importance of consumer awareness as a potential indicator of on-site conditions i.e. on-site ground conditions, wellhead radius (<10m) ground conditions, presence of on-site ponding and the presence of a secure boundary (i.e. all parameters over which the well user may exercise some level of control), a number of statistical tests were undertaken. A series of Mann-Whitney U and ANOVA tests found that there no significant relationships were found to exist between consumer awareness and general on-site conditions, wellhead radius ground conditions, the presence of on-site ponding or the presence of a secure boundary.

8.9. Summary

In all, 545 awareness questionnaires were completed by private groundwater users using two survey modalities, namely researcher administration and self (respondent) administration. As expected, hand-dug wells were typically found to be older than bored wells, with RA respondents citing average ages of 39 years and 19 years for hand-dug and bored wells, respectively. It is probable therefore, that a higher level of maintenance is necessary in relation to hand-dug sources in Ireland, although it is noted that these source types are in the minority.

Although approximately 70% and 85% of RA and SA respondents, respectively, acknowledged the involvement of a water diviner at the well design stage, however, only 16% and 1.2% of RA respondents acknowledged the involvement of local authorities and a professional hydrogeologist, respectively. Similar proportions were noted among SA respondents. This finding presents a source of concern, as the primary purpose for employment of a water diviner at the well design stage is the location of a suitable yield, as opposed to sustainable groundwater quality, with many well drilling contractors privately employing water diviners.

A significantly low level of awareness was noted with regard to waterproof sealing of private groundwater sources, among both RA and SA respondents. Users of bored SPS in particular, displayed a very low awareness of appropriate sealing of their source. For example, 81.5% of RA respondents (n = 179) had no awareness of sealing or grouting of their well.

Although the majority of both RA and SA respondents were aware of whether or not their private source was treated, approximately 68% of RA respondents acknowledged that no treatment system of any kind was currently in use at their source. A similar percentage (69.8%) was found among SA respondents. If these figures are transposed nationally, this would imply that there are currently

almost 90,000 households (or 260,000 consumers) in Ireland supplied by entirely untreated private groundwater supplies (excl. PrGWS). Further analysis found that among RA respondents, 58% of respondents used domestic ion-exchange water softeners as their primary treatment process, therefore, just over 18% of respondents had an appropriate water treatment system in place. A significantly lower level of treatment was in evidence with regards to hand-dug wells, which due to their typical design and construction, are almost certainly more susceptible to pathogenic ingress.

Likewise, the majority of respondents showed an awareness of previous maintenance works having been carried out at their source, however, approximately 35% of RA respondents were aware of maintenance having actually been carried out, with a very similar proportion among SA respondents. A higher level of maintenance was apparent among hand-dug well users, although this proportion was less than 50% (46%). Overall, this equates to almost 110,000 households (or 320,000 consumers) currently being supplied by an unmaintained private groundwater source in Ireland.

Moreover, a majority of respondents were also aware of whether or not previous water quality analysis had been carried out at their source, with approximately 60% of RA respondents acknowledging previous sampling and analysis. A comparable proportion was found among SA respondents. No significant difference was found to exist with regard to previous analysis and source type among RA respondents, suggesting that there are currently up to 245,000 consumers (excl PrGWS) supplied by a private groundwater source which has not been previously tested.

Although there was a relatively high recognition of groundwater contaminants such as *E. coli*, *Cryptosporidium* spp., nitrate and fluoride, the very low levels of awareness associated with rotavirus and *Giardia,* evidenced among both RA and SA respondents, were of concern, as both of these microbial contaminants are causative agents of waterborne gastroenteric illness. The relatively high level of awareness associated with *Cryptosporidium* spp. was probably due to a highly publicised recent outbreak in the west of Ireland. No statistically significant relationships were found to exist between groundwater contaminant recognition and other surveyed parameters.

Respondents' awareness of potential health problems caused by groundwater contamination (illnesses and/or symptoms) showed that just under 70% of RA respondents exhibited an awareness of at least one illness or symptom caused by consumption of contaminated groundwater. This figure was only 27% among SA respondents. There was the expected difference found between respondents who had initiated previous water quality analysis and apparent health hazard awareness. Furthermore, analysis found that all potential groundwater contaminants, with the exception of calcium carbonate (lime/hardness) and *Giardia*, had statistically significant relationships, with total contaminant number awareness showing a strong statistical association with health hazard awareness. Respondents with increased awareness of potential groundwater contaminants were found to be more likely aware of potential health hazards arising from contaminated groundwater consumption.

Approximately 56% and 36% of RA and SA respondents, respectively, were aware of at least one potential contaminant hazard in the vicinity of their well, however, further data collection suggests that approximately 14%, 31% and 24% of RA respondents were unaware of the potential threat of adjacent slurry/fertiliser spreading, grazing animals and septic tanks, respectively, with figures of 11%, 10% and 34%, respectively, found amongst SA respondents. These findings may present a

concern, as the majority of private groundwater sources are located in rural areas and these three contaminant sources in particular are ubiquitous throughout rural Ireland.

An awareness scoring methodology based upon completed questionnaires was developed and employed, with the scoring model designed to rate respondents perceived groundwater source and groundwater contamination awareness. Among RA respondents, a median source awareness score of 9 (90%) was found, which was higher than expected. Hand-dug well users exhibited a slightly higher source awareness score than bored well users, which was expected due to a higher level of residency (*Table 8.13*) and design/construction involvement. This pattern of higher source awareness amongst hand-dug well users was also observed in the SA dataset, however the median source awareness score for SA respondents was 5 (50%) and therefore markedly lower than RA respondents. This is probably due to the younger demographic comprising the SA dataset and also, the probable lack of ownership (literally and figuratively) attributed to the SA demographic in relation to their private source. Residency was also found to be an explanatory factor with regard to source awareness among the SA dataset, as were contaminant recognition and daily consumption volume (*Table 8.14*).

The median contamination awareness score for RA respondents was 3 (60%) and therefore proportionately lower than source awareness, with a slightly higher level of awareness again attributed to hand-dug well users. Higher contamination awareness was apparent among female respondents than male respondents and no statistically significant correlation was found to exist between individual respondent source and contamination awareness scores. The median contamination awareness score for SA was 2 (40%) and therefore lower than RA respondents. As the RA dataset was primarily comprised of young (20-30 years) respondents in 3rd level educational institutions, it was expected that this dataset would have a higher contamination awareness, which was not borne out. A statistically significant correlation was found to exist between individual respondent source and contamination awareness scores amongst SA respondents. Respondents' consumption source and previous instance of a potentially waterborne illness were also found to have significant correlations with SA contamination awareness (*Table 8.15*).

In order that the potential effects of groundwater source and groundwater contamination awareness could be quantified, a susceptibility paradigm was developed. No significant relationship was found to exist between source/contamination awareness and *E. coli* presence or magnitude (where present) in the respondents well, and therefore no correlation was found between respondent awareness and source susceptibility i.e. the probability of a well becoming contaminated.

Those respondents with treatment systems in place were found to have significantly lower source awareness, particularly among hand-dug well users, conversely these respondents were found to display higher groundwater contamination awareness, particularly among bored well users, which was expected. Furthermore, those respondents indicating that previous maintenance had been carried out at their source were found to exhibit significantly higher source awareness, with no significant relationship found between source maintenance and respondents' groundwater contamination awareness.

Both source awareness and contamination awareness were found to significantly affect the likelihood of water quality analysis having been carried out at the respondents' source. In both

cases, a higher level of awareness was found to increase the probability of water quality analysis having been carried out. In the case of groundwater contamination awareness, the mean difference was found to be highly significant, with source awareness found to be less significant, although still significant at a 95% level. Therefore it is concluded that the level of awareness attributed to a well user is related to the level of consumer control applied at the respondents' groundwater source, particularly the respondents' contamination awareness (*Tables 8.20-8.22*).

No significant correlation was found to exist between the level of consumer control i.e. treatment in place, previous maintenance and/or previous analysis and either groundwater bacterial quality or site assessment condition.

Chapter 9: Private Source Susceptibility Modelling

9.1 Introduction

The following chapter integrates all data collected as part of the source susceptibility fieldwork programme, in addition to data collected as part of the consumer awareness survey. As the focus of this research is the quantitative analysis of the human health risk factors associated with private groundwater sources in Ireland, all explanatory and predictive models have been based upon *E. coli* data. The results of bivariate statistical analysis with respect to these data and potential explanatory variables are presented, and used to develop a number of multivariate source susceptibility predictive models with regard to microbial contamination and potential risk factors including source location with respect to geological/hydrogeological setting and potential hazard sources, source design, construction and maintenance. Resulting models are subsequently validated using a validation dataset (Castledaly, Co Westmeath) and sensitivity analysis is carried out in order to estimate the relative importance of significant model coefficients (risk factors).

9.2 Bivariate Analysis (Risk Factor Assessment)

Potential relationships between individual risk factors and private source contamination were analysed in order to form working hypotheses regarding spatial and temporal contamination patterns and their likely causative factors. Furthermore, the results of this analysis were used to inform multivariate modelling and correct for potential multi-colinearity among potential risk factors.

Due to the very low instance of protozoan contamination found, it was decided that these data should not be used as model inputs and therefore bivariate analysis was not carried out. In relation to *E. coli* contamination, due to a number of factors including sample collection variation, microbial ecology variation and bacterial source variation, the use of absolute plate count data (cfu/100 ml) was not used as a model input. Instead, the presence or absence of bacterial contamination and plate count ranges (i.e. 1-10 cfu/100 ml, 11-100 cfu/100 ml, >100 cfu/100 ml) were used, with bivariate analysis therefore carried out on these parameters i.e. the data were categorised as to whether they met or exceeded each target. This approach has been previously used in a number of studies (Goss *et al*, 1998; Howard *et al*, 2003; Giannoulis *et al.*, 2005 Twarakavi & Kaluarachchi, 2006).

Due to the differing variable types included for analysis, a number of bivariate techniques were used as appropriate. In the case of *E. coli* presence/absence (dichotomous variable), chi-square tests of independence, Mann-Whitney U tests and independent samples t-tests were used as appropriate when analysing nominal, ordinal and continuous risk factor datasets, respectively. Analysis of ordinal datasets (i.e. plate count ranges) was undertaken using chi square tests of independence, and one-way analysis of variance (ANOVA) as appropriate.

The following section presents the results of individual risk factor analyses for study areas 1-4, vulnerability category, boreholes and hand-dug wells, three physicochemical clusters (see Chapter 7; *Section 7.7*) and the total dataset. Interpretations of significance are also provided, with significance

(p <0.05 used by convention (Agresti, 1996). Due to the large number of potential risk factors involved in analysis, only associations with p < 0.1 are presented. Potential input parameters included all data collected as part of the site assessment (see Appendix D3).

The effect of the "one-off" sampling regime duration on seasonality was investigated i.e. disproportionate numbers of samples collected during particular months or seasons. The results are presented in Appendix G1. It was found that although particular months were not representative (i.e. February – 31 "one-off' samples, June – 3 "one-off" samples), when seasonal sample collection was investigated, no such lack of representivity resulted, with 26%, 24%, 28% and 21.5% of sample collection occurring during spring, summer, autumn and winter, respectively.

9.2.1 Study Area 1 (Belmont, Co Offaly; High vulnerability)

As outlined in Chapter 6 (*Section 6.2.3*), SA1 is a *High* vulnerability area, with a varied superficial geology, with Carboniferous Limestone tills covering approximately 30% of the area, cutover peats approximately 10-15% and the remainder being composed of glaciofluvial limestone sands and gravels, presenting as a large esker. The bedrock geology is also varied and chiefly composed of Devonian Kiltorcan-type Sandstones (DKS) and Dinantian Sandstones, Shales and Limestones (DESSL), surrounded by areas of Dinantian Pure Unbedded Limestones (DPUL) and Dinantian Lower Impure Limestones (DLIL). The aquifer is typically Ll or Lm. Significant relationships between individual risk factors and *E. coli* presence in SA1 are presented in *Tables 9.1 & 9.2.*

Table 9.1 Results of Chi-square analysis between individual risk factors and E. coli presence from SA1 (p <0.1)

Variable	X^2	Sig	df
Source Type	3.964	0.046	1
Landuse	5.109	0.047	1
Wellhead Finish	10.877	0.002	2
Liner Clearance	10.519	0.002	1

Table 9.2 Results of t-test analysis between individual risk factors and E. coli presence from SA1 (p <0.1)

Variable	t	Sig	df	95% CI
Source Age	-1.83	0.08	48	-32.79 – 1.96
Septic Tank Number (<100m)	2.24	0.03	48	0.05 – 0.949
Distance from Point Agricultural Source	1.685	0.098	48	-38.71 – 439.31
Mean Temperature	-1.71	0.093	48	-4.22 – 0.33

48-hour precipitation	-3.178	0.003	48	-11.23 - -2.52
120-hour precipitation	-2.973	0.005	48	-15.59 - -3.01
Monthly precipitation	-2.965	0.005	48	-107.35 - -20.59
Sample watertable	2.842	0.007	43	0.792 – 4.66

As shown above, a number of potential risk factors had significant bivariate associations with the presence or absence of *E. coli* in private sources in SA1. Both septic tanks and point agricultural sources (farmyards, animal housing, feed storage lots) were related to bacterial presence, with non-point sources such as animal grazing and landspreading not identified as being major causative factors, although wells located in agricultural landuse areas were more likely to exhibit bacterial presence (Odds Ratio (OR) 5.806, 95% CI 1.12-29.89). Source design and construction details also had a role in bacterial contamination, with source type (OR 0.278, 95% CI 0.076 – 1.0), wellhead finish and liner clearance (OR 8.333, 95% CI 2.13 – 32.6) all significantly associated with bacterial presence. As expected, there was a higher probability of bacterial contamination in hand-dug wells, with these sources approximately 3.6 times more likely to exhibit *E. coli* presence. This has likely contributed to sample watertable significance as a risk factor. In terms of wellhead finish, those wells finished above ground and in underground chambers were found to have *E. coli* present in 50% and 23% of cases respectively; however, those wellheads finished at ground level were contaminated in 75% of cases. Furthermore, where there was liner flange clearance (>0 mm), either above ground level or the base of the chamber, these wells were over 8 times less likely to have *E. coli* present. Due to multi-colinearity associated with source type, a number of design and construction parameters have been excluded from the SA1-4 analysis. These are however included and quantified in both cluster type and generalised dataset analyses (*Sections 9.2.7 – 9.2.10*).

Throughout Chapter 9, precipitation inputs are defined as the depth (mm) of total rainfall measured at the closest rainfall station during the 24-hour, 48-hour, 120-hour and 30-day period prior to the day of sampling. Mean air temperature is defined at the mean temperature on the day of sampling. Both precipitation rates and mean air temperature were found to be significantly associated with bacterial presence. In the case of mean air temperature, although this may affect bacterial growth rates where present, it is likely this was not a causative factor but instead reflects a seasonal pattern. As for precipitation, both short-term (i.e. 24-h, 48-h, 120-h) and medium-term (i.e. 30-d) rainfall rates were significantly associated with bacterial presence, which may suggest that both ingress due to precipitation runoff/overland flow and rapid recharge are bacterial drivers in SA1.

Perhaps of most interest in SA1 was the lack of significant association between bacterial presence and geological/hydrogeological factors such as bedrock type, subsoil type, subsoil thickness and groundwater vulnerability. This was likely due to the relatively uniform geology/hydrogeology within this study area.

9.2.2 Study Area 2 (Crossbridge, Co Wicklow; Extreme/High vulnerability)

SA2 is comprised of two smaller study areas, namely Crossbridge and Kilquiggin. Subsoils in both areas are dominated by granite tills (>80%), interspersed with pockets of outcrop and subcrop, granite sands and gravels and small veins of undifferentiated alluvium. The study area bedrock geology is characterised as granites and other igneous intrusive rocks (GII) throughout, with aquifers classified as poor with bedrock generally unproductive except in local zones (Pl) and locally important, with bedrock being moderately productive only in local zones (Ll), with Pl aquifers dominant overall. The entire area is categorised as being *High* or *Extreme* vulnerability. A more in-depth description is provided in Chapter 6 (*Section 6.2.3*). Significant relationships between individual risk factors and *E. coli* presence in SA2 are presented in *Tables 9.3 & 9.4*

Table 9.3 Results of Chi-square analysis between individual risk factors and E. coli presence from SA2 (p <0.1)

Variable	X^2	Sig	df
Subsoil Thickness	6.843	0.077	3
Vulnerability	4.902	0.048	1
Surface Spring <100m	5.63	0.048	1
Septic Tank Gradient	16.824	<0.001	2
Point Agricultural Source Gradient	5.193	0.075	2
Well Ponding	3.407	0.1	1

As shown in *Table 9.3*, the hydrogeological setting was found to be a significant risk factor with reference to *E. coli* presence in SA2, with those wells situated in *Extreme* vulnerability areas 5.8 times (OR 5.833, 95% CI 1.084 – 31.37) more likely to have *E. coli* present than those located in *High* vulnerability areas. Furthermore, subsoil thickness was also found to be a significant bivariate risk factor. Due to inherent colinearity between groundwater vulnerability and subsoil thickness, and uniformity of subsoil type throughout SA2, subsoil thickness is a likely causative factor, while vulnerability may be a useful susceptibility indicator.

Table 9.4 Results of t-test analysis between individual risk factors and E. coli presence from SA2 (p <0.1)

Variable	t	Sig	df	95% CI
Point Agricultural Source Distance	2.032	0.094	26	-17.82 – 57.86
Monthly precipitation	-2.418	0.019	48	-70.88 - -6.51
Sample watertable	1.976	0.065	38	-1.08 – 3.98

As for SA1, point contamination sources were again observed as being likely causes of private source contamination in SA2, with septic tank setback gradient (i.e. hydraulic gradient from the well to the septic tank) exhibiting a highly significant relationship in particular. Both the setback distance and gradient from point agricultural hazard sources were also significantly associated. Additionally, surface water springs within a 100 m radius of the wellhead were significantly related to bacterial presence, which would suggest ingress from point or non-point hazard sources. Due to the aforementioned relationships regarding point hazard sources, and the relatively low agricultural stocking rates in this area, point sources are the probable cause of contamination. Sources located within 100 m of a surface spring were over 7 times more likely to exhibit bacterial presence (OR 7.312, 95% CI 1.189 – 44.97).

Generally, well design and construction were not found to be significant potential risk factors in SA2. However, as shown (*Table 9.3*), visible ponding in the well chamber or within a 10 m of the wellhead was associated with elevated rates of bacterial ingress (OR 4.62, 95% CI 0.82 – 26.02).

Conversely to findings from SA1, no association was found to exist between short-term precipitation (24-hour, 48-hour, and 120-hour) and bacterial ingress, however monthly precipitation was observed as being a potential driving factor. This would suggest that where *E. coli* contamination occurred in SA2, direct ingress at the wellhead might not be the cause, and that instead, contamination may have occurred through the subsurface. This would also seem to be suggested by the apparent relationship between sample watertable and *E. coli* presence. Where bacterial presence was noted, a mean watertable of 2.04 m was recorded, while this figure was 3.49 m for uncontaminated sources. As there was a typically high watertable in this study area and source type was not found to be a significant risk factor, this is unlikely due to colinearity.

9.2.3 Study Area 3 (Paulstown, Co Kilkenny; High vulnerability)

Study Area 3 is dominated by Namurian shale and sandstone tills with large pockets of interspersed bedrock outcrop and subcrop in the highlands. In the lower lying areas, Carboniferous limestone tills dominate. The bedrock geology throughout the study area is extremely diverse, as is the aquifer importance; however, the majority of SA3 is composed of poor and locally important aquifers. *Extreme* vulnerability is predominant in the northern half of SA3, with large areas of *High* vulnerability to the south (see *Section 6.2.3*). Potentially significant associations regarding *E. coli* presence in SA3 are outlined in *Tables 9.5 & 9.6*.

Table 9.5 Results of Chi-square analysis between individual risk factors and E. coli presence from SA3 (p <0.1)

Variable	X^2	Sig	df
Bedrock Type	21.351	0.003	7
Subsoil Type	9.419	0.009	2
SW Drains	4.704	0.043	1
Septic Tank Gradient	9.651	0.008	2

Table 9.6 Results of t-test analysis between individual risk factors and E. coli presence from SA3 (p <0.1)

Variable	t	Sig	df	95% CI
Septic Tank Number (<100m)	-3.009	0.004	48	-1.32 - -0.26

As shown in *Table 9.5*, both bedrock type and subsoil type were noted as having some explanatory value with regard to bacterial contamination in SA3. In regard to bedrock type, although this area had diverse bedrock distribution with eight recorded bedrock types, sources located in areas underlain by impure or dolomitised limestone were found to have a higher bacterial contamination rate than those underlain by other rock types in SA3, including Westphalian sandstones and shales, Namurian sandstones and shales, and purer Dinantian limestone. Further investigation found that contamination was significantly associated with these bedrock types (χ^2 = 19.718, p<0.001; OR 50.0, 95% CI 4.62 – 540). This trend was also evident with regard to subsoil type, whereby 40% (n = 4) of sources located in limestone tills were bacterially contaminated, compared to only 12.5% (n = 5) of those sources situated in the predominantly sandstone/shale tills.

Although statistical analysis suggested that the proximity of a surface water drainage ditch was significantly associated with contamination, further interpretation found that a majority of uncontaminated sources were situated within 100 m of a drainage source, and therefore this is unlikely to be a risk factor.

The most apparent trend with regard to hazard sources in SA3 was the significant association of septic tank location and number with contaminated sources, with those sources with >1 septic tank within 100 m (n = 16; 32%) and septic tanks located up gradient of the wellhead (n = 4; 8%) more likely to exhibit bacterial presence. Conversely, there was no suggestion of agricultural hazard sources being involved in source contamination in this area. This is also reflected in the lack of association between mean temperature and bacterial presence, which would suggest a lack of seasonality with regard to agricultural cycles and groundwater contamination in this area. Additionally, it was found that short-term and medium-term precipitation were not significant contamination drivers in SA3.

No significant association was found to exist between bacterial presence in SA3 and any measured source design or construction criteria. This may further suggest that direct and rapid ingress of contaminants at the wellhead are not typical in this area, and gradual recharge from poorly located or designed septic tank systems are more likely.

9.2.4 Study Area 4 (Kilclone, Co Meath; Low vulnerability)

Carboniferous limestone tills are the dominant subsoil in SA4, with Dinantian Upper Impure Limestones (DUIL) dominating the bedrock geology, although a large area of undifferentiated, Namurian bedrock (NU), underlying approximately 15% of the study area, is situated in the northwest. The majority of the area (>80%) is characterised as being *Low* vulnerability. Likewise, the majority of the area (>90%) is classified as a locally important aquifer, which is productive only in local zones (Ll), with the area underlain by Namurian shales and sandstones to the northwest

characterised as being a poor aquifer, which is productive only in local zones (Pl) (see *Section 6.2.3*). Significant relationships between individual risk factors and *E. coli* presence in SA4 are presented in *Table 9.7*. No significant mean differences were found with regard to *E. coli* presence and any continuous parameters.

As shown (*Table 9.7*), there was a significant association (although not at a 95% level) found between supply type and bacterial presence. Further analysis found that shallow hand-dug wells were found to be over three times more likely to be bacterially contaminated than deeper boreholes (OR 0.3, 95% CI 0.067 – 1.169). No geological/hydrogeological parameters were found to have any association with bacterial presence.

Those sources located within 100 m of a rural tertiary or link road were found to have a higher rate of *E. coli* presence (55%) compared with those located near a busier secondary road. This may be due to higher rates of agricultural traffic on small rural roads and associated runoff after rainfall; however, this was not borne out with regard to rainfall response, as no significant relationship was found to exist between bacterial presence and precipitation.

A notable trend occurred in SA4 with regard to ground condition and site ponding, with poor ground condition near the wellhead and visible onsite ponding both positively associated with bacterial presence. As this area is predominantly categorised as being *Low* vulnerability, and therefore overlain by uniform low permeability subsoils, this likely illustrates both the instance of onsite ponding and the lack of significant association between short and medium term precipitation and *E. coli* presence.

Statistical analysis suggests that septic tank systems are the predominant source of microbial contamination in this area, with no agricultural hazards exhibiting significant associations. This agrees with previous work undertaken in this area, with regard to it being considered an "overdeveloped" rural catchment i.e. high septic tank density (McCole, 1992).

Table 9.7 Results of Chi-square analysis between individual risk factors and E. coli presence from SA4 (p <0.1)

Variable	X^2	Sig	df
Supply Type	3.183	0.074	1
Road Type	3.182	0.096	2
Site Ponding	5.633	0.033	1
Septic Tank Gradient	7.388	0.025	2
Wellhead Finish	14.308	0.001	2
Liner Clearance	14.23	<0.001	1
Wellhead Radius Ground Condition	9.79	0.044	4

The significant wellhead design and construction parameters were, as in SA1, wellhead finish and liner clearance. Those sources with the wellhead finished at ground level had evidence of bacterial contamination in almost 92% of cases, whereas wellheads finished in underground chambers and above ground level (i.e. positive liner clearance above ground) were contaminated in 28% and 33% of cases, respectively. Liner clearance (either above chamber floor or ground level depending on wellhead finish) was also significantly associated with *E. coli* presence, with those wells with a lack of clearance over 25 times more likely to have bacteria present (OR 25.66, 95% CI 2.97 – 221.6).

Overall, analysis suggests that poorly located sources in relation to septic tank systems, in association with low subsoil permeability are a major contaminant source in SA4. Additionally, low permeability subsoils leading to surface ponding and consequently surface runoff into poorly finished wellheads may be a contributing factor, and may be a hazard pathway in *Low* vulnerability areas nationally.

9.2.5 Vulnerability category

Significant relationships between individual risk factors and *E. coli* presence in *High* and *Extreme* vulnerability areas are presented in *Tables 9.8 & 9.9*.

Table 9.8 Results of Chi-square analysis between individual risk factors and E. coli presence from High/Extreme vulnerability dataset (p <0.1)

Variable	X^2	*Sig*	*df*
Source Type	5.384	0.031	1
Bedrock Type	126.4	< 0.001	28
Subsoil Type	63.608	< 0.001	10
Aquifer Importance	12.135	0.016	4
Secure Boundary	4.339	0.042	1
Septic Tank Gradient	8.758	0.013	2
Wellhead Finish	8.81	0.012	2
Liner Clearance	8.798	0.005	1
Chamber Ingress	5.755	0.056	1
Chamber Material	8.113	0.088	4

Overall, as shown in *Table 9.8*, there was a significant association found between bacterial presence and source type in *High/Extreme* vulnerability areas, with hand-dug wells found to be approximately 2.6 times more likely to have *E. coli* present (OR 0.375, 95% CI 0.161 – 0.865). Bedrock type is also significantly associated with bacterial contamination, with limestone and sandstone areas having higher levels of *E. coli* incidence than granite and Namurian bedrocks. This is also reflected with regard to subsoil type, with granite and Namurian tills exhibiting lower levels of *E. coli* presence. As before, it is unclear as to whether bedrock or subsoil type is the primary contamination driver, and

to some degree this will depend upon the nature of the contamination and its source. Notably, no significant difference was recorded with regard to bacterial presence and sources located in *High* vulnerability and sources located in *Extreme* vulnerability.

The significance of a secure boundary would suggest that diffuse agricultural activities i.e. livestock grazing, are an important source of bacterial contamination in *High* and *Extreme* vulnerability areas, however, as neither grazing animal distance nor setback gradient were flagged as being significantly associated with *E. coli* presence, this remains inconclusive.

As for private sources in *Low* vulnerability areas, septic tank setback gradient with respect to the wellhead is significantly associated with bacterial presence in *High* and *Extreme* vulnerability areas. Furthermore, wellhead finish and liner clearance are also suggested as potential risk factors and again suggest direct ingress at the wellhead, as do the significance related to well chamber ingress and chamber construction material. It was found that over 20% (n = 18) of well chambers constructed from 100mm concrete blocks were linked with *E. coli* presence, whereas no well chambers finished with lime/mortar plaster were contaminated.

Table 9.9 Results of t-test analysis between individual risk factors and E. coli presence from High/Extreme vulnerability dataset (p <0.1)

Variable	t	Sig	df	95% CI
48-hour Precipitation	-3.082	0.002	158	-6.41 - -1.4
120-hour Precipitation	-1.748	0.082	158	-11.37 – 0.69
Monthly Precipitation	-3.623	<0.001	158	-64.05 - -18.85
Average Temperature	-1.673	0.096	158	-3.16 – 0.26

As shown in *Table 9.9*, high levels of significance existed between both 48-hour precipitation and monthly precipitation and *E. coli* presence. This may suggest that both direct ingress at the wellhead and relatively rapid recharge are driving bacterial contamination of private groundwater resources, with mean average temperature significance perhaps reflecting a seasonal contamination pattern (i.e. agricultural cycles).

It is important to note that bivariate analysis found that no significant difference existed with regard to *High/Extreme* vulnerability and either *E. coli* presence or, where present, *E. coli* magnitude e.g. *High, Extreme* and *Extreme (sur)* vulnerability wells had *E. coli* contamination rates of 23%, 26% and 23.8%, respectively. Therefore, while mechanisms of contaminant transmission may differ among source types located in different vulnerability categories; this does not result in a differing contamination rate or magnitude.

9.2.6 Boreholes

All boreholes sampled and assessed (n = 169) have been similarly analysed in order that further multivariate models may be developed and potential input parameters and their associated importance quantified. Additionally, as previously mentioned many of the parameters pertaining to

source design and construction contained inherent colinearity and were therefore excluded from previous bivariate analysis (SA1 - 4). These parameters have been included in both the source type bivariate investigations. *Tables 9.10 & 9.11* outline significant associations regarding *E. coli* presence and potential risk factors.

Table 9.10 Results of Chi-square analysis between individual risk factors and E. coli presence from boreholes (p <0.1)

Variable	X^2	Sig	df
Bedrock Type	33.881	0.001	13
Subsoil Type	16.002	0.007	5
Aquifer Importance	8.939	0.063	4
Site Ponding	3.496	0.079	1
Septic Tank Gradient	9.239	0.01	2
Wellhead Finish	5.354	0.069	2
Liner Clearance	3.815	0.063	1
Construction Material	6.325	0.097	3
Liner Material	9.545	0.008	2

As shown in *Table 9.10*, a number of geological and hydrogeological characteristics were found to be significantly associated with *E. coli* presence in boreholes and are therefore considered potential risk factors. Bedrock type in particular was a highly significant indicator of bacterial contamination. Dinantian limestones, and particularly impure limestones were typically associated with higher contamination rates, with approximately 35% (n = 24) of bored sources located in Dinantian limestones exhibiting evidence of bacterial ingress, compared with 24% of all assessed bored sources. This was also reflected with regard to subsoil type, with just under 39% (n = 29) of those sources located in limestone tills having shown evidence of bacterial presence, while this figure was 24% for the entire bored source dataset. It is unclear as to whether bedrock type or subsoil type are the dominant potential risk factor, as subsoils are typically highly correlated with the bedrock parent material. Additionally, the hazard source will depend on the hydrogeological setting i.e. localised hazard sources may be linked with highly permeable subsoils, whereas more widespread aquifer contamination would be expected to be associated with bedrock type, particularly in the case of bedded limestones, where horizontal flow of contaminants may be relatively rapid along bedding planes. The emergence of aquifer importance (Ll –Locally important, moderately productive only in local zones) as a potentially significant risk factor, in concurrence with the lack of significance associated with subsoil thickness would seem to suggest localised aquifer contamination.

Table 9.11 Results of t-test analysis between individual risk factors and E. coli presence from boreholes (p <0.1)

Variable	t	Sig	df	95% CI
Liner clearance (mm)	2.212	0.028	164	3.8 – 67.1
48-hour precipitation	-1.783	0.076	167	-5.15 – 0.26

The significance of onsite ponding (OR 2.12, 95% CI 0.95 – 4.7), liner clearance (OR 2.488, 95% CI 0.97 – 4.34) and measured liner clearance suggests that, at least in some cases, direct ingress of surface runoff at the wellhead is leading to bacterial contamination. This is also backed up by the significance of 48-hour precipitation. As this dataset includes SA4, which is predominantly composed of thick (>10 m) low permeability subsoils, rapid recharge of contaminants is unlikely to occur and therefore direct ingress is likely. Typically, boreholes lined with steel and surrounded by an appropriate concrete chamber were found to have a lower bacterial incidence rate than other construction types.

The only direct relationship between a potential hazard source and bacterial presence was found with regard to the septic tank setback distance. Where septic tank systems within a 100 m radius of the wellhead were situated down gradient of the wellhead, a bacterial incidence rate of just less than 18% was recorded (n = 21). However, where septic tank systems were located "on-gradient" or up gradient from the wellhead, incidence rates of 35% (n =9) and 44% (n = 11) were noted.

9.2.7 Hand-dug wells (Cluster 3)

As outlined in Chapter 7 (*Section 7.7*), the hand-dug well dataset also represents Cluster 3, resulting from Bayesian Criterion (BIC) 2-step cluster analysis. This cluster has been identified as being susceptible to bacterial contamination, with both temporal and spatial analysis confirming this. *Tables 9.12 & 9.13* outline significant associations regarding *E. coli* presence and potential risk factors.

Table 9.12 Results of Chi-square analysis between individual risk factors and E. coli presence from hand-dug wells (p <0.1)

Variable	X^2	Sig	df
Bedrock Type	12.095	0.034	5
Aquifer Importance	8.001	0.046	3
Site Gradient	5.047	0.08	2
Road Gradient	7.153	0.028	2
Pump Type	8.257	0.016	2
Wellhead Finish	6.011	0.05	2
Liner Clearance	5.177	0.035	1

Table 9.13 Results of t-test analysis between individual risk factors and E. coli presence from hand-dug wells (p <0.1)

Variable	t	Sig	df	95% CI
Houses supplied	2.285	0.032	40	0.051 – 1.29
Road Distance	2.01	0.056	38	-2.05 – 82.45
Septic Tank Setback Distance	2.406	0.025	40	20.19 – 124.19
Septic Tank Number (<100m)	-1.694	0.098	40	-1.01 – 0.09

As for boreholes (*Section 9.2.6*), both bedrock type and aquifer importance were identified as being potential risk factors, with limestones and sandstones once again displaying a higher bacterial incidence rate than granites/Namurians i.e. 80% of hand-dug sources in sandstone areas were contaminated, compared with 28.5% in granite areas. A higher level of *E. coli* presence was observed in hand-dug wells located in locally important aquifers (65%) than poorly productive aquifers (26%). No hand-dug wells situated on a regionally important aquifer were assessed as part of the fieldwork programme. As it was not expected that geological/hydrogeological factors would have a significant influence on contamination in hand-dug wells due to the nature of their design and construction, a number of further statistical analyses were undertaken. It was found that hand-dug sources in limestone and sandstone areas (i.e. typically high levels of contamination) had mean septic tank setback distances of 55 m and 60 m, respectively, while hand-dug sources in granite and Namurian areas (i.e. typically low levels of contamination), had mean septic tank setback distances of 195 m and 219 m, respectively. A one-way ANOVA found this was a significant difference ($F(5) = 9.127$, $p < 0.001$). Furthermore, similar analysis found the same pattern with regard to aquifer importance, with sources located in Lm and Ll aquifers having a mean septic tank setback distance of 60 m and 64 m, respectively, while sources located in Pl aquifers having a septic tank setback distance of 139 m ($F(3) = 3.745$, $p = 0.019$).

Both road setback gradient and distance were significantly associated with bacterial presence. Hand-dug wells with evidence of bacterial contamination had a mean road setback distance of 32.8 m, compared with a mean setback distance of 74 m in uncontaminated wells. This may be due to either road runoff or well damage due to traffic vibrations; however, as wells located down gradient or "on-gradient" from roads had a higher contamination incidence, this may suggest road runoff as being the cause e.g. agricultural vehicles, etc.

Again, wellhead finish and liner clearance were identified as potential risk factors, which would suggest direct ingress of contaminants at the wellhead. Septic tank systems were isolated as being the potential hazard source of greatest significance, with septic tank setback distance in particular identified as being significantly associated with contamination. Contaminated wells had a mean setback distance of 62 m, whereas uncontaminated wells had a recorded mean setback of 134 m.

Additionally, hand-dug sources supplying one household were observed as being more likely to have *E. coli* present than those supplying greater than one (OR 0.133, 95% CI 0.025 – 0.719). This may be

due to higher levels of consumer control (secondary susceptibility) associated with sources supplying multiple households.

9.2.8 Cluster 1 – Boreholes in limestone/sandstone areas

As outlined in Chapter 7 (*Section 7.7*), a series of BIC 2-step cluster analyses undertaken with regard to groundwater source type and groundwater physico-chemistry resulted in the qualification of three distinct source clusters, namely hand-dug wells, boreholes in limestone areas and boreholes in granite/Namurian areas. Additionally, clusters were found to be significantly associated with E.coli presence. As the bivariate analyses pertaining to Cluster 3 (hand-dug wells) have been undertaken and presented earlier, the results of similar analyses with regard to Clusters 1 and 2 are now presented. *Tables 9.14 & 9.15* outline significant associations regarding *E. coli* presence and potential risk factors in Cluster 1.

As shown, the main hazard source risk factors with significant associations were point contamination sources, namely septic tanks and point agricultural sources, with both septic tank setback distance and gradient significantly associated with bacterial contamination.

A number of source design and construction parameters were also significantly associated, particularly the wellhead finish, liner clearance, well construction material and chamber presence. This would suggest that direct ingress at the wellhead is a major contributing factor in the case of bacterial contamination of boreholes in limestone areas, and as presented, the relatively significant response to 48-hour precipitation and 120-hour precipitation would also suggest direct ingress of surface water run-off or rapid recharge from point contamination sources.

Table 9.14 Results of Chi-square analysis between individual risk factors and E. coli presence from Cluster 1 (p <0.1)

Variable	X^2	Sig	df
Bedrock Type	16.089	0.065	9
Site Ponding	3.681	0.069	1
Point Agricultural Source <100m	8.111	0.006	1
Wellhead Finish	8.996	0.011	2
Liner Clearance	6.883	0.015	1
Construction Material	7.14	0.068	3
Liner Material	6.651	0.036	2
Chamber Presence	3.681	0.069	1

Table 9.15 Results of t-test analysis between individual risk factors and E. coli presence from Cluster 1 (p <0.1)

Variable	t	Sig	df	95% CI
Surface Water Distance	-1.778	0.096	30	-137.5 - -0.51
Septic Tank Number <100m	1.91	0.059	85	-0.017 – 0.844
Septic Tank Setback Distance	2.17	0.033	85	0.7 – 16.11
Liner flange clearance (mm)	1.715	0.09	84	-5.85 – 79.05
Sample WT	1.923	0.059	61	-0.1 – 5.63
48-hour Precipitation	-1.75	0.084	85	-7.4 – 0.47
120-hour Precipitation	-2.042	0.047	85	-9.83 - -0.07

9.2.9 Cluster 2 – Boreholes in granite/Namurian areas

Tables 9.16 & 9.17 outline significant associations regarding *E. coli* presence and potential risk factors among boreholes in "granite/Namurian" areas. Similar to results from Cluster 1, the main potential hazard sources of significance with regard to bacterial contamination in Cluster 2 were point sources, including septic tank systems and point agricultural sources. However, the hydrogeological setting within Cluster 2 was found to be more significant than Cluster 1, with both subsoil type and thickness found to be significant (with vulnerability also significant). A small number of sources within Cluster 2 (n = 5) were underlain by limestone tills (although not associated with limestone bedrock), with 80% of these exhibiting bacterial contamination. This figure was less than 12% (n = 9) among those sources not underlain by limestone tills. Furthermore, boreholes located in areas with <3 m subsoils were bacterially contaminated in 20% of cases, while this was approximately 9% where subsoils were >3 m. This would suggest rapid recharge of microbial contamination from point sources. However, a number of specific design parameters were also significant including visible ingress, well chamber construction material and liner clearance, implying direct wellhead ingress at some sources.

Table 9.16 Results of Chi-square analysis between individual risk factors and E. coli presence from Cluster 2 (p <0.1)

Variable	X^2	Sig	df
Bedrock Type	20.435	0.009	8
Subsoil Type	12.511	0.006	3
Subsoil Thickness	10.084	0.073	5
Vulnerability	8.412	0.078	4
Septic Tank Gradient	12.267	0.002	2

Point Agricultural Source Gradient	5.486	0.064	2
Well Chamber Ingress	4.711	0.095	1
Chamber Material	9.777	0.044	4

Table 9.17 Results of t-test analysis between individual risk factors and E. coli presence from Cluster 2 (p <0.1)

Variable	t	Sig	df	95% CI
Point Agricultural Source Distance	2.281	0.037	44	1.01 – 28.71
Source Age	-1.771	0.084	42	-23.9 – 1.56
Liner Clearance (mm)	1.874	0.073	76	-3.49 – 73.32

As previously outlined (*Chapter 7, Section 7.7*), it has been shown that higher contamination rates were evidenced in Cluster 1 (boreholes in limestone/sandstone areas) than Cluster 2 (boreholes in granite/Namurian areas). Investigation of bivariate analysis of contamination within Clusters 1 and Cluster 2 suggest that similar hazard sources are significant with regard to *E. coli* contamination, particularly septic tank and point agricultural source proximity. Likewise, a number of design and construction characteristics are similarly significant, although this is expected as both clusters pertain to boreholes. The primary sources of dissimilarity are the significance of well chamber characteristics and lack of significance regarding precipitation within Cluster 2. This may suggest that boreholes in limestone/sandstone areas are more sensitive to rainfall events, with 120-hour and 30-day precipitation being more significant contamination drivers in limestone/sandstone areas. Furthermore, poor wellhead design among wells in Cluster 2 is a more significant cause of contaminant ingress, particularly the presence of a well chamber and, where present, chamber construction material.

9.2.10 Total Dataset

Significant risk factors identified within the entire "one-off" fieldwork dataset are presented in *Tables 9.18 & 9.19*. As expected and previously borne out by cluster analysis (Chapter 7), well type was a significant risk parameter with respect to microbial contamination, with shallow hand-dug wells approximately 2.8 times more likely to have *E. coli* present than boreholes (OR 0.352, 95% CI 0.175 – 0.71). As previously mentioned, due to issues regarding colinearity, a number of design and construction details are therefore excluded from analysis within the total dataset.

The primary hazard source of concern was the septic tank setback distance, with uncontaminated wells having a mean setback distance of 109m, whereas those wells with *E. coli* present displayed a mean setback distance of 63m. Additionally, those sources with a septic tank located <100m and up-gradient from the wellhead were 2.8 times more likely to exhibit bacterial presence than those sources not located within 100 m of a septic tank and up-gradient of the septic tank (OR 2.886, 95% CI 1.39 – 5.99).Overall, 48.6%, 41.3% and 22.8% of contaminated wells had a septic tank within 100 m located up gradient, "on gradient" and down gradient, respectively, with this adjudged to be a

significant outcome (χ^2 (2) = 11.232, p = 0.004). This implies that septic tanks located up or on gradient with private wells are definite contamination risk factors. Furthermore, the significance associated with on-site ponding (OR 1.873, 95% CI 0.96 – 3.64) and site gradient may imply poorly located and/or designed septic tank systems, in low permeability subsoils.

Table 9.18 Results of Chi-square analysis between individual risk factors and E. coli presence from total dataset (p <0.1)

Variable	X^2	Sig	df
Bedrock Type	38.158	<0.001	14
Subsoil Type	21.539	0.001	5
Subsoil Thickness	12.756	0.026	5
Aquifer Importance	12.202	0.016	4
Vulnerability	11.332	0.023	4
Site Gradient	8.984	0.011	2
Road Setback Gradient	5.499	0.064	2
Onsite Ponding	3.476	0.076	1
Septic Tank Gradient	10.41	0.005	2
Source Type	8.931	0.004	1

As shown in *Table 9.19*, the precipitation volume (mm) in the 48-hour period prior to sampling may suggest direct wellhead ingress of contaminated runoff, which would seem to be supported by the aforementioned significance relating to on-site ponding and site gradient, however, as specific design and construction parameters have been excluded from this dataset, it is not possible to further illustrate this potential relationship. Nevertheless, short-term rainfall is almost certainly a significant factor in microbial contamination. Lastly, the significance of mean air temperature at sampling (9.4°C at uncontaminated wells, 10.9°C at contaminated wells), may imply a seasonal pattern with respect to bacterial contamination of private wells in Ireland.

Table 9.19 Results of t-test analysis between individual risk factors and E. coli presence from total dataset (p <0.1)

Variable	t	Sig	df	95% CI
Septic Tank Setback Distance	1.926	0.056	209	-1.12 – 92.94
48-hour Precipitation	-2.252	0.025	209	-5.07 - -0.33
120-hour Precipitation	-1.648	0.096	209	-8.59 – 0.7
Mean Temperature	-1.93	0.055	209	-2.94 – 0.03

9.3 Multivariate Statistical Analysis

Bivariate analysis of potential risk factors (explanatory variables) with regard to their relationship to a response variable i.e. *E. coli* presence or *E. coli* magnitude, although useful, does not adequately model and therefore explain the environmental system comprising groundwater/source susceptibility. By examining two or more sets of potential explanatory factors and their associations with the response variable, a better understanding of the system may become apparent. However, an insignificant correlation between a specific explanatory variable and the response variable does not necessarily imply that it will not be a useful (and significant) predictor in a multivariate model (due to combined effects of two or more explanatory variables), with the inverse also being true. As the system under investigation contains multiple environmental hazards, pathways and receptors, a multivariate approach to analysis and modelling was taken in order that multiple explanatory variables and their respective effects upon model outputs was quantified and understood. There are a multitude of multivariate techniques potentially appropriate for the development of a groundwater susceptibility model, as outlined below (Moore & McCabe, 2006; Tabachnick & Fidell, 2007):

- Multivariate analysis of variance (MANOVA) - an extension of ANOVA where more than one explanatory variable is analysed simultaneously.
- Multivariate regression – Description of simultaneous variable responses and their effects on a response variable.
- Logistic Regression Analysis (LRA) – Prediction of group membership (binomial or multinomial) where explanatory variables are continuous, discrete or a combination of both.
- Principal Components Analysis (PCA) – Summary of a variable set using developed ordered orthogonal variables.
- Factor Analysis (FA) - Summary of a variable set using developed synthetic variables.
- Redundancy Analysis (RA) – Multivariate analogue of regression analysis, deriving a minimum set of synthetic variables from a set of explanatory variables in order to investigate maximum variance in a response variable.
- Cluster Analysis (CA) – Classification of data into clusters based on a number of processes i.e. 2-step analysis, k-means analysis.

As set out by Tabachnick & Fidell (2007), there are two research objectives which may be satisfied with the use of multivariate statistical analysis, namely data classification/structure and prediction of group membership, with the research objective typically influencing the appropriate multivariate modelling technique to be used.

Data classification/structure techniques include factor analysis, principal components analysis, Cluster analysis and structural equation modelling and are used to identify similar groups within datasets. These techniques have been successfully employed in a number of previous studies with regard to water contamination (Hagedorn *et al*, 1999; Jenerette *et al*, 2002; Reghunath *et al.*, 2002; Lambrakis *et al*, 2004; Panda *et al*, 2006; Shrestha & Kazama, 2007; Omo-Irabor *et al.*, 2008; Koh *et al*, 2010). Predictive membership techniques are used to predict membership within a set of explanatory variables to pre-identified response groups, with the primary difference between group

membership prediction and data classification being the pre and post identification of response groups respectively. Predictive membership techniques include discriminant function analysis, hierarchical (sequential) discriminant function analysis, frequency analysis (Logit) and logistic regression.

In the context of this study, quantitative prediction of group membership was the preferred statistical approach, as homogenous groupings were already assigned i.e. *E. coli* present or absent, high, moderate or low *E. coli* levels. Additionally, prediction of group membership would allow for the quantification of explanatory risk factor contributions to the overall system. As the primary response variable (*E. coli*) included a high number of "non-detects" i.e. <1 cfu/100ml, a discrete threshold value was specified to define the response categories, therefore, MANOVA and multivariate regression analysis were not considered appropriate, as these techniques are typically used where the response variable is continuous in nature. In order to correctly choose the appropriate technique, an adapted decision tree based on Tabachnick & Fidell (2007) was utilised (*Figure 9.1*).

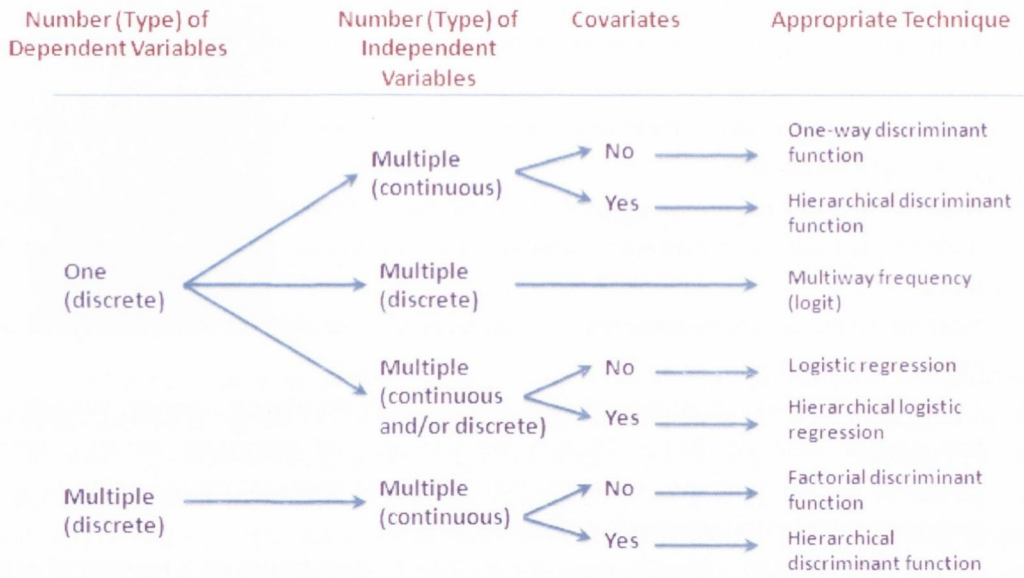


Figure 9.1 Decision tree relating to appropriate statistical technique for susceptibility modelling (adapted from Tabachnick & Fidell, 2007)

As shown (*Figure 9.1*), where a categorical response variable is used, discriminant function analysis is typically employed if all independent variables are continuous and normally distributed (Shrestha & Kazama, 2007); logit analysis may be used if all independent variables are categorical and logistic regression (LR) is often chosen where predictor variables (i.e. potential risk factors) are a mixture of continuous and categorical (Wuensch, 2009). LR makes no assumptions regarding the independent variable distributions (Hailpern & Visintainer, 2003). Furthermore, Nolan (2001) states that logistic regression is well suited to analysis where a high level of "non-detects" characterise the response variable, as was the case in this study. Therefore LR was chosen as the appropriate technique for analysis and subsequent modelling of private source susceptibility for this study. Due to the existence of covariates in the analysis (i.e. secondary variables that can affect the relationships

between the dependent variable and other independent variables of interest e.g. precipitation), hierarchical LR was used.

9.4 Logistic Regression (LR)

LR is typically used to model discrete response variables, which may be binary (i.e. Yes/No) or ordinal (i.e. low/moderate/high) in nature, with the elementary underlying principle being the determination of the probability of an event by examining the correlation between explanatory and response variables (Helsel & Hirsch, 1992). LR has been extensively applied in epidemiological research since the 1960s and is becoming more commonplace in environmental research (Helsel & Hirsch, 1992; Nolan, 2001; Masetti *et al.*, 2008), including assessment of multiple explanatory variables and their respective roles in the contamination of aquifers (Eckhardt & Stackelberg, 1995; Nolan, 2001; Howard *et al*, 2003; Gardner & Vogel, 2005).

In the context of this study, binary hierarchical LR has been employed to predict a dichotomous response variable, namely the occurrence of *E. coli* in private wells in Ireland (*E. coli* absent – 0; *E. coli* present – 1). This approach was taken as it is considered that the prediction of specific *E. coli* concentrations (cfu/100ml) is not realistic due to unquantifiable inherent variations within the system i.e. magnitude of *E. coli* in faeces of particular animal or human sources, high resolution temperature variation (e.g. hourly) affecting microbial die-off, specific on-site geological/hydrogeological settings (e.g. preferential flowpaths). A model which can be used to predict the probability of bacterial occurrence has therefore been favoured.

LR models predict the logit, i.e. the natural log of the odds ratio of a given occurrence (Wuensch, 2009). The odds ratio is based upon the probability of exceeding a predefined threshold (Nolan, 2001), as shown in *Equation 9.1*, which in the case of this study is 1 cfu/100ml:

$$odds\ ratio = \frac{p}{1-p} \qquad \textit{(Eqn. 9.1)}$$

where
p = probability of exceeding the threshold value (i.e. ≥1 cfu/100ml)

The log of the odds ratio (logit), transforms a variable constrained between 0 and 1 into a continuous variable that is a linear function of the explanatory variables. As described by Helsel & Hirsch (1992), the resulting LR equation (*Equation 9.2*) is a linear function, given by:

$$\ln(Odds) = \ln\left(\frac{p}{1-p}\right) = a + bx \qquad \textit{(Eqn. 9.2)}$$

where
a = constant (scalar intercept parameter)
bx = vector of slope coefficients and explanatory variables

Transforming the predicted values of the response variable back into probability units gives the following: (Helsel & Hirsch, 1992; Hosmer & Lemeshow, 2000; Nolan, 2001) (*Equation 9.3*):

$$P = \frac{e^{(a+bx)}}{(1+e^{(a+bx)})} \qquad \textit{(Eqn. 9.3)}$$

where

P = probability that *E. coli* is present (≥1 cfu/100ml)

x = vector of n explanatory variables

a = constant

b = vector of slope coefficient values, so that $bx = b_1X_1 + b_2X_2 + \ldots\ldots\ldots b_nX_n$

Using an iterative process, the SAS/SPSS/PSAW statistical analysis software package calculates values for a and b that maximise the ln likelihood function L, as shown in *Equation 9.4*:

$$L = \sum_{i-1}^{m}[(y_i \times lnP_i) + (1 - y_i) \times \ln(1 - P_i) \qquad \textit{(Eqn. 9.4)}$$

where

m = number of observations in the dataset

y_i = outcome variable set to 1 when *E. coli* ≥1 cfu/100ml; otherwise a value of 0 is assigned (Tesoriero & Voss, 1997)

The probability of the observed results given the parameter estimates is known as the likelihood. The resulting model is subsequently evaluated using the likelihood ratio test statistic (G), which tests the statistical significance of coefficients in the overall LR model (Hosmer & Lemeshow, 2000; Nolan, 2001), by comparing the observed values of the response variable to predicted values obtained from the model, with and without the variable being tested (*Equation 9.5*):

$$G = -2(L_a - L_{model}) \qquad \textit{(Eqn. 9.5)}$$

where

L_a = log-likelihood of intercept only model

L_{model} = log-likelihood of model with one or more explanatory variables

Conversely, the Hosmer-Lemeshow goodness-of-fit (HL) statistic compares observed values to those predicted by the resulting model (Hosmer & Lemeshow, 2000). Observed and predicted probabilities are compared for data grouped from high to low, with ten groups (deciles) commonly used for comparison (Nolan, 2001) i.e. the first group comprising n/10 observations with the smallest predicted probabilities, and the last group comprising n/10 observations with the largest predicted probabilities. The HL statistic is chi-squared distributed, and as the null hypothesis is that the model fits the data, higher p-values indicate better overall fit (Hosmer & Lemeshow, 2000).

The G (-2 log likelihood) statistic has been used in this study to develop optimal models, with the HL goodness-of-fit statistic used to test how well the resulting models fit the data.

Additionally, several pseudo R-square measures have been suggested for the measurement of model quality for LR models. One such measure is known as the Nagelkerke R-square, which is calculated as follows (*Equation 9.6*):

$$f(x) = \frac{1-\exp\left\{-2\frac{(LL_m - LL_0)}{N}\right\}}{1-\exp\left\{\frac{2LL_0}{N}\right\}} \qquad \textit{(Eqn. 9.6)}$$

where

LL_m= log-likelihood of the model

LL_0 = log-likelihood of the intercept (the model without any explanatory variables included)

N = sample size

The Nagelkerke R-square statistic will be used to determine the level of overall system variation explained by the finished LR models.

As previously mentioned, LR modelling has been previously employed to predict aquifer vulnerability and susceptibility, although it has not been used to predict bacterial contamination of private domestic groundwater sources. Howard *et al* (2003) developed LR models in order to investigate the causes of exceedances of water quality targets for thermotolerant coliforms and faecal Streptococci of shallow groundwater in Kampala, Uganda. Likewise, Shanks *et al* (2006) used LR for the analysis of faecal source marker data and their respective explanatory variables in Tillamook Bay, Oregon. Tesoriero & Voss (1997) have also successfully used LR modelling to predict the probability of elevated nitrate concentrations in wells in the Puget Sound Basin, Washington.

9.4.1 Variable colinearity/multi-colinearity

Colinearity is the existence of a linear relationship between two explanatory variables in a multivariate model, with multi-colinearity referring to the existence of a linear relationship between more than two explanatory variables (O'Brien, 2007). Where colinearity or multi-colinearity exists within a regression model, coefficient estimates (B) may change arbitrarily in response to small changes in the model or data (Lipovestky & Conklin, 2001).

It was considered favourable that all potentially significant explanatory variables be included in the development of LR models in order that the maximum level of variance within each system is explained unless they exhibited symptoms of colinearity within the variable set. Colinearity was evaluated within each LR model using variance inflation factors (VIFs), as previously applied by Kayhanian *et al* (2007).The tolerance statistic is interpreted as the proportion of variance in coefficients for individual covariates not accounted for by other independent variables in the model. Variables with a low tolerance statistic (<0.7) contribute little additional information to the model. The VIF statistic is the reciprocal of tolerance, and increasing VIF factors indicate increasing colinearity and unstable estimates of regression coefficients for independent variables with high VIFs.

The main sources of colinearity within the LR model input dataset were associated with well type and associated design and construction characteristics i.e. source depth, liner diameter, construction materials, etc. It was therefore considered that separate well type LR models should be developed which would allow for the inclusion of all collinear parameters. Collinear parameters relating to well type were therefore excluded from all other LR models.

9.5 Binary LR Modelling of bacterial presence/absence

In order to develop multiple regression models to assess source susceptibility to bacterial contamination, the association between each potential risk factor (explanatory variable) and event occurrence (*E. coli* ≥1 cfu/100ml) was evaluated, as set out earlier in this chapter, with the importance of potential model predictors being evaluated based on *t*-values, χ^2 statistics, ANOVA and *p*-values. The presence of a single faecal coliform is a breach of the Drinking Water Regulations (S.I. No. 278 of 2007) and as such is regarded as a serious concern prompting investigation (EPA, 2010a; EPA, 2010c).

Parameter Entry

All potential risk factors of importance were entered into the LG model hierarchically based upon perceived importance from bivariate evaluation. A stepwise method of entry (Forward: Conditional) was used for the majority of predictor variables, with variable entry taking place at p = 0.05 and removal at p = 0.1 (Tesoriero & Voss, 1997) i.e. if an exploratory predictor variable has an individual significance within the model p < 0.05 after 20 iterations it remains in the model, if significance is 0.05-0.1 it is flagged and can be manually taken out, if significance > 0.1, it is automatically removed from the model. If bivariate analysis resulted in a high level of association between a predictor variable and the outcome variable, this predictor was "forced" into the model, due to expected significance. If forced entry revealed the predictor variable was in fact insignificant within the model, stepwise entry was then assigned and the model re-run. A maximum of 20 estimation iterations were run for each model step, with estimation terminated where parameter estimates changed by less than 0.001 between two estimation iterations. After model development, significant predictor parameters in the model were assigned by the researcher to associated identifiable blocks (or hierarchies) e.g. if septic tank setback distance, number of septic tanks <100 m radius, and septic tank setback radius were all significant within a developed model, they were assigned to a "septic tank" hierarchy. The model was consequently re-run with these hierarchies, so that the relative importance (i.e. proportion of system variability, overall significance of the hierarchy) could be quantified. A classification cut-off of 0.5 was used as this was the most conservative approach i.e. classification cut-off < 0.5 = *E. coli* absent; *E. coli*> 0.5 = *E. coli* present.

As discussed in Chapter 8 (*Section 8.9*), due to the lack of measured association between source susceptibility and source or contaminant awareness among respondents, awareness scoring has not been included in multivariate model development. All LR models were developed using SPSS/SAS 16.0 statistical software. An SPSS susceptibility modelling codebook, as developed and used by the researcher is included in Appendix G2.

9.5.1 Study Area 1 (Belmont, Co Offaly)

The final binary LR model for SA1 was developed using four model blocks (hierarchies), namely agriculture (Block 1), well design/construction (Block 2), short (120-h) and medium term (30-d) precipitation (Block 3) and road location (Block 4). The final regression coefficients including coefficient significance, model and hierarchy significance and model R^2 are shown in *Table 9.20*.

Table 9.20 LR model of source susceptibility within SA1

	B	B Sig.	Model Sig.	H/L Sig.	R^2
Block 1:					
Point Agricultural Source <100m	-3.49	0.018			
Grazing Land Up-Gradient	-5.007	0.02	0.219	0.656	0.079
Block 2:					
Liner Clearance	-5.076	0.008			
Wellhead Cover	-3.159	0.046	0.002	0.059	0.378
Block 3:					
120-hour Precipitation	0.272	0.007			
30-day Precipitation	0.024	0.012	<0.001	0.761	0.718
Block 4:					
Road Setback Down-Gradient	3.687	0.041			
Constant	-1.015	0.63	<0.001	0.94	0.795

From *Equation 9.3*:

$$P = \frac{e^{(a+bx)}}{(1 + e^{(a+bx)})}$$

a = -1.015

b_1 = -3.49; X_1 = 0 if point agricultural source >100 m, 1 if point agricultural source <100 m

b_2 = -5.007; X_2 = 0 if grazing land not up-gradient, 1 if grazing land up-gradient

b_3 = -5.076, X_3 = 0 if no liner clearance, 1 if liner clearance

b_4 = -3.159, X_4 = 0 if wellhead uncovered, 1 if wellhead covered

b_5 = 0.272, X_5 = Precipitation (mm) over 120-hour duration prior to sampling

b_6 = 0.024, X_6 = Precipitation (mm) over 30-day duration prior to sampling

b_7 = 3.687, X_7 = 0 if roadway not down gradient, 1 if roadway down gradient

Therefore;

$$P = \frac{[\exp(a + b_1X_1 + b_2X_2 + \ldots\ldots b_7X_7)]}{1 + \exp(a + b_1X_1 + b_2X_2 + \ldots\ldots b_7X_7)}$$

SA1 BM20 was chosen for this worked example, due to the high level of *E. coli* contamination present (161 cfu/100ml). A worked example for a source with no contamination present is included in Appendix G3. The regression equation for SA1 BM20 is:

$$P = \frac{\exp(3.7558)}{1 + \exp(3.7558)} = \frac{42.768}{43.768} = 0.977$$

As previously outlined, a classification cut-off of 0.5 was used and therefore the resulting SA1 regression model correctly predicts that SA1 BM20 has *E. coli* present i.e. 97.7% probability of contamination.

A Nagelkerke R^2 of 0.795 was attributed to the overall model, which would suggest that the model correctly explains 79.5% of variability. Furthermore, the Hosmer-Lemeshow statistic for the model

was χ^2 (7) = 2.325, p = 0.94. As previously mentioned, the HL statistic is chi-squared distributed, and as the null hypothesis is that the model fits the data, higher p-values indicate better overall fit. Therefore, this implies that the model fits the data very closely.

The-2 log likelihood statistic for the overall model was χ^2 (7) = 44.733, p <0.001, therefore, the model is statistically significant (i.e. high likelihood). Additionally, all four model blocks resulted in significant -2 log likelihood statistics (p<0.05).
The resulting SA1 LR model indicates that bacterial contamination of private groundwater sources in SA1 is primarily associated with (point and diffuse) agricultural sources, with the significance of both liner clearance and wellhead cover pointing to direct ingress at the wellhead as the likely contamination mechanism. Monthly precipitation rates and 120-hour precipitation rates would seem to be the main transport mechanisms, which may be due to gradual runoff or relatively rapid recharge. The insignificance of parameters relating to geological and/or hydrogeological setting within the SA1 model was expected, due to relative geological uniformity.

The classification table (*Table 9.21*), shows the overall prediction accuracy obtained using the developed SA1 LR model and subsequent regression equation. As shown, the overall prediction accuracy of the SA1 LR model was 90% (i.e. amalgamated *E. coli* presence and absence prediction rates), with both *E. coli* absence and *E. coli* presence having high prediction accuracies of 93.1% and 85.7% respectively. Further examination of incorrect predictions found that false positives (i.e. prediction of *E. coli* absence where *E. coli* were actually present) occurred where low levels (<10 cfu/100ml) of *E. coli* were present, with false negatives typically occurring during dry periods i.e. the model was highly sensitive to precipitation, as high intensity short duration rainfall was adjudged to be a primary contamination driver, therefore, periods of little or no short-term rainfall resulted in lower levels of predicted *E. coli* presence, which was not always the case.

Table 9.21 LR model classification table (SA1 Model)

	Observed	Predicted	Percent Correct
E. coli Absent	29	27	93.1
E. coli Present	21	18	85.7
			90

9.5.2 Study Area 2

The final binary LR model for SA2 was developed using three model blocks, namely: septic tank systems (Block 1), agriculture (Block 2), and surface water vicinity (Block 3). The final regression coefficients including variable standard error and significance are shown in *Table 9.22*.

Table 9.22 LR model of source susceptibility within SA2

	B	B Sig.	Model Sig.	H/L Sig.	R^2
Block 1:					
Septic Tank "On-Grad"/Up-Grad	-5.079	0.002	<0.001	1	0.391
Block 2:					
Agricultural Landuse	3.858	0.024	<0.001	0.984	0.532
Block 3:					
Surface Spring <100m	-3.745	0.022			
Constant	1.073	0.436	<0.001	0.773	0.681

The overall predictive accuracy obtained using the SA2 LR model was 96%, as shown in *Table 9.23*. This is considered an extremely high level of predictive accuracy, however, due to the low level of *E. coli* presence encountered in SA2 and the relatively uniform geological setting, this level of accuracy may be somewhat inflated. Where the one false negative was predicted (Kn5), a low level of *E. coli* contamination was recorded (6 cfu/100ml), which was similarly found in the SA1 LR model.

The overall LR source susceptibility model for SA2 was simple, with only three parameters necessary to explain approximately 68% of variability and correctly predict contamination with 96% accuracy. The model indicates that the susceptibility of sources within this study area was almost exclusively associated with hazard location, with agricultural landuse and septic tank location denoted as being the primary hazard sources(i.e. as shown in *Table 9.23*, over 53% of variance in the system and 78% of variance explained by the resulting model was attributed to these two risk factors). As for SA1, the geological/hydrogeological setting associated with sources in SA2 was regarded as being insignificant in the model, which was expected due to a high degree of geological uniformity. The insignificance of well type and source design/construction parameters may indicate subsurface contamination in concurrence with a typically high watertable.

Table 9.23 LR model classification table (SA2 Model)

	Observed	Predicted	Percent Correct
E. coli Absent	43	42	97.7
E. coli Present	7	6	85.7
			96

The overall model had a Nagelkerke R^2 of 0.681, and therefore accurately describes 68.1% of variability within the system. The Hosmer-Lemeshow statistic for the model was χ^2 (3) = 1.118, p = 0.773, which would suggest that the model fits the input data well. The -2 log likelihood statistic for the model was χ^2 (7) = 23.751, p <0.001, with all three model hierarchies obtaining significance.

9.5.3 Study Area 3

As shown in *Table 9.24*, the final binary LR model for SA3 comprised four model blocks, which were; subsoil type (Block 1), septic tank systems (Block 2), well design and construction (Block 3) and agriculture (Block 4).

Table 9.24 LR model of source susceptibility within SA3

	B	B Sig.	Model Sig.	H/L Sig.	R^2
Block 1:					
Limestone Tills	-4.567	0.021	0.06	1	0.112
Block 2:					
Septic Tank Up-grad	-5.946	0.008	0.011	0.487	0.27
Block 3:					
Well Chamber Ponding	4.401	0.029			
Wellhead Cover	-4.324	0.048	0.007	0.867	0.4
Block 4:					
Grazing Land Distance	0.17	0.03			
Secure Boundary Presence	-4.802	0.024			
Constant	5.252	0.014	0.001	0.536	0.626

As shown earlier in this chapter (*Tables 9.5 & 9.6*), septic tank systems were significantly associated with *E. coli* presence in SA3 and therefore, forced entry was used to include both septic tank setback distance and septic tank number <100m in the model, in order to improve the overall model accuracy and in particular the *E. coli* presence prediction. When both parameters were included, *E. coli* prediction accuracy was improved by approximately 33% (88.9%), with the Nagelkerke R^2 increasing from 0.626 to 0.861 i.e. a further explanation of approximately 23% of system variability. However, the correlation coefficients for both parameters were statistically insignificant, likely due to risk factor interactions, and were therefore omitted from the final model. Similarly, the inclusion of 24-hour precipitation increased the overall model accuracy to 94%, however, this parameter was also adjudged to be insignificant within the final model.

The LR model indicates that diffuse agricultural sources (animal grazing) and septic tank systems were the primary contaminant sources within SA3, comprising 22.6% (0.626 – 0.4) and 15.8% (0.27 – 0.112) of system variability, respectively. This was expected due to the rural nature of the study area. Additionally, those sources underlain by limestone tills were adjudged to be more likely to have bacterial contamination present. A previous study by Daly (1994a) found that thin layers of limestone tills in this area (Nore Valley, Co Killkenny) had recharge coefficients of 60-90%. All sources in limestone areas of SA3 were underlain by <3 m of subsoils. The model reflects this, implying that sources in areas of limestone tills were sensitive to 120-hr precipitation, with rapid recharge therefore the likely contamination transport mechanism. In areas not underlain by limestone, direct recharge by gradual runoff was the more typical transport route.

The final classification table for the SA3 LR model is presented below (*Table 9.25*). As shown, the overall model accuracy was 86%, however, a relatively low prediction accuracy with regard to *E. coli* presence was noted (55.6%), with a high prediction accuracy for *E. coli* absence accounting for the

high model prediction rate. As for SA1 and SA2, where false negatives occurred, they were typically associated with low level contamination (<10 cfu/100ml).

Table 9.25 LR model classification table (SA3 Model)

	Observed	Predicted	Percent Correct
E. coli Absent	41	38	92.7
E. coli Present	9	5	55.8
			86

As mentioned, the Nagelkerke R^2 for the final model was 0.626, and therefore 37% of variability within the system remains unexplained. The data did however offer a significant model fit as evidenced by the HL statistic (χ^2 (8) = 7.01, p = 0.563).The -2 log likelihood statistic for the model was χ^2 (6) = 24.082, p = 0.001, with all four model hierarchies obtaining significance.

9.5.4 Study Area 4 (Low Vulnerability)

The final *Low* vulnerability (SA4) LR model was developed from a total dataset of 52 private sources located in Kilclone, Co Meath. Four significant model hierarchies were included in the model, including; septic tank location (Block 1), short (48-h, 120-h) and medium term (30-d) precipitation (Block 2), wellhead design/construction (Block 3) and Landuse (Block 4). These are presented in *Tables 9.26 & 9.27.*

The primary hazard source was found to be septic tank systems, with ST number, ST setback distance and ST setback gradient all significant within the LR model. Notably, three precipitation rates were adjudged to be significant, which implies that a number of transport mechanisms may have explanatory value in *Low* vulnerability. The significance of liner clearance and liner cap presence, in concurrence with 48-hour precipitation and 120-hour precipitation would suggest that some level of direct wellhead ingress due to surface runoff was occurring. This was expected, as low permeability soils in SA4 lead to high levels of visible surface ponding over the duration of the fieldwork programme. Additionally, 30-day precipitation would suggest recharge and subsequent subsurface ingress, or alternatively (and less likely), gradual surface runoff.

Table 9.26 LR model of source susceptibility within SA4

	B	B Sig.	Model Sig.	H/L Sig.	R^2
Block 1:					
Septic Tank Setback Distance	-0.117	0.039			
Septic Tank Number <100m	-1.659	0.08			
Septic Tank Up-Grad	-15.174	0.04	0.035	0.349	0.204
Block 2:					
48-hour precipitation	-0.926	0.04			
120-hour precipitation	0.711	0.043			
30-day precipitation	-0.069	0.05	0.03	0.461	0.315
Block 3:					
Liner Clearance	-18.536	0.022			
Liner Cap Presence	-7.865	0.04	<0.001	0.817	0.759
Block 4:					
Agricultural Landuse	5.776	0.041			
Constant	48.812	0.027	<0.001	0.94	0.839

In the absence of septic tank system number (<100 m) as an input parameter, overall prediction dropped to 84.6% (R^2 0.794, HL Sig 0.538), therefore as it was <0.1, it was included in the final LR model. As shown, high overall model significance was achieved, with the model providing for approximately 84% of variance.

Table 9.27 LR model classification table (SA4 Model)

	Observed	Predicted	Percent Correct
E. coli Absent	29	27	93.1
E. coli Present	23	20	87
			90.4

9.5.5 High/Extreme Vulnerability Areas

In all, the *High/Extreme* vulnerability dataset included 160 private sources. The final LR model comprised four significant hierarchies, namely: septic tank location (Block 1), wellhead design/construction (Block 2), subsoil parameters (Block 3), and 30-d precipitation (Block 4). The final model outcomes are presented in *Tables 9.28 & 9.29*.

It was found that thin layers of high permeability subsoils (sand and gravels, limestone tills) were associated with higher levels of bacterial presence, with the primary bacterial source adjudged to be septic tank systems. The significance of liner clearance, wellhead cover and wellhead radius ground condition suggest that direct ingress at the wellhead was a significant bacterial transport mechanism. This was not reflected by significant precipitation rates, however, with short term precipitation insignificant within the final model (high p-value). Instead, 30-day precipitation rates were significantly associated with bacterial presence, which would suggest more gradual transportation mechanisms. It is important to reiterate that no significant association was found

between vulnerability category and source type and therefore, differing transport mechanisms, if present within the *High/Extreme* dataset were due to geological/hydrogeological setting, and not well type.

Table 9.28 LR model of source susceptibility within High/Extreme vulnerability sources

	B	B Sig.	Model Sig.	H/L Sig.	R^2
Block 1:					
Septic Tank Setback Distance	-0.013	0.014			
Septic Tank Down Grad	1.156	0.03	0.004	0.442	0.102
Block 2:					
Liner Clearance	-1.68	0.003			
Wellhead Cover	-1.544	0.011			
Wellhead Rad Ground Cond (10m)	-1.62	0.009	<0.001	0.121	0.249
Block 3:					
Limestone Till	-1.657	0.002			
Subsoil Thickness	1.876	0.001			
Sands & Gravels	-2.27	0.05	<0.001	0.285	0.402
Block 4:					
Monthly Precipitation	0.01	0.004			
Constant	2.639	0.069	<0.001	0.142	0.462

Table 9.29 LR model classification table (High/Extreme vulnerability Model)

	Observed	Predicted	Percent Correct
E. coli Absent	121	116	95.9
E. coli Present	38	21	55.3
			86.2

The overall *High/Extreme* vulnerability LR model, although statistically significant, provided explanation for approximately 46% of total variance within the system, signifying that additional data is needed to further improve the model.

9.5.6 Boreholes

Using the results from bivariate analysis, as set out earlier in this chapter, an appropriately accurate hierarchical LR model could not be developed for boreholes assessed as part of this study, with a maximum level of model accuracy of <70% (Nagelkerke $R^2 = 0.326$) i.e. it was not possible to initialize to the required level of model significance. This is probably due to high levels of inherent variability with regard to geological and hydrogeological setting, and bored well design and construction details. It was therefore considered useful that two distinct models be developed for Cluster 1 (boreholes situated in limestone areas) and Cluster 2 (boreholes situated in granite/Namurian areas), as a proportion of variability would be inherently absent from these datasets, therefore allowing for the development of susceptibility models.

9.5.7 Boreholes in Limestone Areas (Cluster 1)

The Cluster 1 LR model was developed using a total dataset of 87 private boreholes. The final model included four significant input hierarchies, including septic tank location (Block 1), liner clearance (Block 2), agricultural point source vicinity (Block 3), and adjacent roadway setback gradient (Block 4). The final model is presented below (*Tables 9.30 & 9.31*).

Table 9.30 LR model of source susceptibility within Cluster 1

	B	B Sig.	Model Sig.	H/L Sig.	R^2
Block 1:					
Septic Tank Setback	-0.075	0.002	0.025	0.39	0.078
Block 2:					
Liner Clearance	-3.095	<0.001	0.002	0.115	0.184
Block 3:					
Point Agricultural Sources <100m	-4.276	0.001	<0.001	0.714	0.356
Block 4:					
Road Gradient (UG/OG)	-2.288	0.009			
Constant	9.194	<0.001	<0.001	0.104	0.463

Table 9.31 LR model classification table (Cluster 1 Model)

	Observed	Predicted	Percent Correct
E. coli Absent	58	54	93.1
E. coli Present	29	19	65.5
			83.9

Additionally, 120-hr precipitation was found to improve the overall significance of the final Cluster 1 LR model (increased explanatory power of approximately 4%); however, it was not a significant parameter within the model. The HL statistic for the final model, as shown, was not significant ($p = 0.104$), and therefore, the final model is considered to be a good fit. A low Nagelkerke R^2 (0.463) indicates that the model explains approximately 46% of variability within the system, and could therefore be improved upon if more data were available.

Overall, the model suggests that the primary hazard sources associated with boreholes in limestone areas are point sources, including septic tank systems and point agricultural sources. The significance of liner clearance, in concurrence with improved overall model accuracy associated with 120-hour precipitation implies that direct wellhead ingress due to gradual surface runoff is a likely contaminant transport mechanism. The insignificance of geological/hydrogeological parameters within the final model was expected, due to the inherent geological uniformity within Cluster 1.

9.5.8 Boreholes in Granite/Namurian Areas (Cluster 2)

Cluster 2 comprised 80 private bored sources located in predominantly "granite/Namurian" areas. The final model was developed using four significant hierarchies, namely: septic tank location (Block 1), subsoil thickness (Block 2), agricultural land setback gradient (Block 3), and wellhead design/construction (Block 4), as presented in *Tables 9.32 & 9.33*.

The inclusion of 24-hour precipitation increased the models overall predictive accuracy from 92.5% to 93.8%, however, explained variance was only increased by 1.7% to 0.614 and as coefficient significance within the model was >0.1, it was omitted. Similarly, grazing land setback distance and the presence of agricultural point sources within 100 m of the wellhead increased prediction accuracy but are not significant in the model and decreased overall goodness-of-fit. Overall, the -2 log-likelihood and H/L statistics indicate that the model is significant and provides a good fit for the input variables. The overall model accuracy is considered high at 92.5% (*Table 9.33*).

Table 9.32 LR model of source susceptibility within Cluster 2

	B	B Sig.	Model Sig.	H/L Sig.	R^2
Block 1:					
Septic Tank Up Grad	-5.227	0.01			
Septic Tank On Grad	-7.674	0.002	0.009	1	0.193
Block 2:					
Subsoil Thickness	5.755	0.02	0.002	0.64	0.299
Block 3:					
Grazing Land On Grad	-3.168	0.01	<0.001	0.874	0.392
Block 4:					
Liner Clearance (mm)	-0.021	0.029			
Above Ground Wellhead Finish	2.89	0.09			
Wellhead Rad Ground Cond (10m)	2.273	0.09			
Constant	3.397	0.133	<0.001	0.709	0.597

Table 9.33 LR model classification table (Cluster 2 Model)

	Observed	Predicted	Percent Correct
E. coli Absent	68	65	95.6
E. coli Present	12	9	75
			92.5

The final LR model indicates that bacterial contamination among boreholes in granite/Namurian areas is associated with both point (septic tank systems) and non-point (grazing animals) sources, with both rapid recharge and direct wellhead ingress the likely mechanisms of contaminant transport. The significance of subsoil thickness, with higher levels of bacterial presence in those sources associated with <3 m overlying subsoils would suggest relatively rapid recharge through thin (high recharge coefficient) subsoil layers. The significance of wellhead finish, liner clearance and

wellhead ground condition would intimate direct ingress of contamination. The significance of short term precipitation (24-hour) also reflects a relatively rapid transport mechanism.

9.5.9 Hand-dug Wells (Cluster 3)

The total hand-dug well dataset was comprised of 42 private sources in total and therefore, due to the very small overall sample size, general inferences should be treated with caution. The final HD LR model comprises four significant hierarchies: septic tank location (Block 1), liner clearance (Block 2), agricultural point source vicinity (Block 3), and households supplied (Block 4). The final outputs and model are presented below (*Tables 9.34 & 9.35*).

When 24-hour precipitation was added to the LR model, the Nagelkerke R^2 increased to0.851 and overall prediction accuracy was increased to 92.9%, with 95% and 90% of *E. coli* absence and *E. coli* presence correctly predicted, respectively. This would suggest that short-term precipitation may be used to explain an additional 8% of the variability in this system however, as this coefficient was not statistically significant within the overall model, it was omitted. Similarly, both grazing land setback distance and road setback distance explained additional system variance; however neither was significant within the model. No geological/hydrogeological factors were deemed to have significant explanatory value with regards bacterial contamination of hand-dug wells. Although geological factors were borne out by bivariate analysis as being potential predictor variables (*Section 9.2.6*), it is considered that as the majority of hand-dug sources are shallow and typically have minimal interaction with the aquifer, it is not surprising that hazard sources are considered significant predictors, while geological factors are not.

Table 9.34 LR model of source susceptibility within hand-dug wells (Cluster 3)

	B	B Sig.	Model Sig.	H/L Sig.	R^2
Block 1:					
Septic Tank Setback	-0.032	0.015			
Septic Tank Up Grad	-5.82	0.021			
Septic Tank Down Grad	-3.63	0.044	0.008	0.862	0.327
Block 2:					
Liner Clearance	-4.478	0.016	0.001	0.545	0.497
Block 3:					
Point Agricultural Sources <100m	4.438	0.019	<0.001	0.811	0.603
Block 4:					
Number of houses supplied	-3.981	0.022			
Constant	13.469	0.008	<0.001	0.543	0.773

Table 9.35 LR model classification table (hand-dug well Model)

	Observed	Predicted	Percent Correct
E. coli Absent	22	20	90.9
E. coli Present	20	17	85
			88.1

The final LR model for hand-dug sources was noted as being somewhat similar to the final LR model for Cluster 1 sources, with point contamination sources including both septic tank systems and agricultural point sources, found to be the primary hazard sources. The significance of liner clearance within the model suggests that direct wellhead ingress is a predominant bacterial contamination mechanism, with the association, albeit insignificant within the final model, of 24-hour precipitation also pointing to rapid direct ingress. An independent samples t-test found that HD sources associated with >1 household supplied (n = 12; 28.5%) were more likely to have bacterial contamination absent, which may potentially be attributed to higher levels of maintenance associated with increased source shareholder numbers.

9.5.10 Total Generalised Model

The final model which was developed included all sources assessed as part of the overall fieldwork programme (n = 211). As expected, due to the higher overall sample size, the resulting model included a greater number of explanatory variables and had a relatively low overall Nagelkerke R^2 (0.446), due to high levels of variability/uncertainty, potential colinearity and missing data. The final model was constructed using five significant input hierarchies: septic tank system location (Block 1), bedrock and quaternary geology (Block 2), wellhead design/construction (Block 3), roadway setback gradient (Block 4), and 120-h precipitation (Block 5). The final model is shown in *Table 9.36*.

As shown, the primary hazard source indicated by the final LR model is septic tank location, with approximately 11% of variance explained by this model hierarchy. Furthermore, approximately 10% of variation (i.e. 0.211 – 0.107 from Table 9.36) was explained by the amalgamated bedrock material and subsoil thickness hierarchy, with sources located in Dinantian (typically impure and/or unbedded) limestone areas typically more likely to have *E. coli* present, as were those sources located in areas with <3m subsoils overlaying the parent material. The most important model hierarchy was wellhead design/construction, with the significance of liner clearance, wellhead cover and wellhead ingress all suggesting that direct wellhead ingress is typically an important contamination transportation mechanism in private sources in Ireland. Additionally, the significance of wellhead radius ground condition also reflects this conclusion, while also raising questions regarding wellhead radius maintenance. In all, the wellhead design/construction hierarchy was deemed responsible for approximately 20% of total variance within the system (i.e. 0.41 – 0.211 from Table 9.36) and 47% of the explained variance. The significance of 120-hour precipitation also suggests that a substantial proportion of private source contamination is due to localised contamination by way of direct wellhead ingress, or rapid recharge in areas with thin high permeability subsoils, as opposed to general "aquifer contamination".

Table 9.36 LR model of source susceptibility within total dataset

	B	B Sig.	Model Sig.	H/L Sig.	R^2
Block 1:					
Septic Tank Setback Distance (m)	-0.022	0.007			
Septic Tank Up Grad	17.548	0.02			
Septic Tank On Grad	18.223	0.018			
Septic Tank Down Grad	19.032	0.013	0.003	0.141	0.107
Block 2:					
Dinantian Limestone Bedrock	-1.527	0.001			
Subsoil Thickness (>/< 3 m)	1.035	0.02	<0.001	0.95	0.211
Block 3:					
Liner Clearance	-2.173	<0.001			
Wellhead Cover	-1.525	0.002			
Wellhead Rad Ground Cond (10m)	-1.196	0.028			
Wellhead Ingress	0.798	0.05	<0.001	0.369	0.41
Block 4:					
Road On Grad	-0.838	0.07	<0.001	0.623	0.42
Block 5:					
120-hour precipitation	0.024	0.05			
Constant	-32.666	0.027	<0.001	0.637	0.446

It was found that liner cap absence presence and household supply number increased the overall model accuracy to 84.7% by way of an improvement in bacterial presence prediction (62.3%), but as the significance of this parameter was >0.1 within the final model (p = 0.107), it was omitted from the generalised LR model. As shown in *Table 9.37*, the final generalised LR model had an overall accuracy of approximately 84%, with very high predictive accuracy with regard to bacterial absence in particular.

Table 9.37 LR model classification table (Total Generalised Model)

	Observed	Predicted	Percent Correct
E. coli Absent	149	140	93.1
E. coli Present	62	37	59.7
			83.9

9.6 Generalised Model Sensitivity Analysis

As set out by Lenhart *et al.* (2002) and Kousari *et al.* (2010), model input parameters will always contain an inherent level of variability and/or uncertainty, due to a number of factors, including budgetary constraints, spatial and/or temporal variability, instrumentation, etc. Therefore, knowledge of sensitive input parameters is beneficial for model development, application and

validation and will lead to decreased overall uncertainty and increased user confidence in the models predictive abilities. Therefore, sensitivity analysis of the primary input variables in the outlined generalised LR model have been undertaken in order to provide insights regarding parameter trends and resulting outcomes. As outlined by Hensher *et al.* (2005), the central principle underlying sensitivity analysis is the variation of one model input parameter while holding all others fixed, and analyzing the effect of this variation on the overall model prediction output.

Sensitivity analysis has been carried out using both "what-if" analysis and threshold value analysis, as previously used in a number of studies (Chaudhuri & Narasayya, 1998; Guillou & Hall, 2001; Golfarelli *et al.*, 2006). "What-if" analysis was used to vary input parameters in the LR model equation, while threshold value analysis was used to examine the point at which contamination probability exceeded 0.5. All analysis was carried out in Excel 2007, using generic simulation models, with SA1 BM11 (monitored borehole) and BM20 ("one-off" hand-dug well) used as source examples. The majority of sensitivity analyses have been undertaken with respect to continuous predictor parameters i.e. 120-hour precipitation, septic tank setback distance. Due to their continuous nature, they can be increased indefinitely until 100% or 50% probability is reached, allowing for the development of predictive model curves. This is not possible with dichotomous/nominal/ordinal predictor variables.

9.6.1 Septic tank system setback (distance and gradient)

The following section presents results of sensitivity analysis within the generalised model with respect to septic tank setback distance relative to BM11 (BM11 was chosen as it was a borehole of typical construction, and had 20-month monitoring data, during which *E. coli* contamination had been recorded). All other parameters were held constant, with septic tank setback distance increased in 10 m increments from 0 m setback (It should be noted that the septic tank system was located up-gradient of BM11). As shown in *Figure 9.2*, at the overall mean 120-hour precipitation rate of 17 mm, the 0.5 probability threshold was reached at approximately 21 m septic tank setback distance, with probability decreasing approximately linearly. Therefore, the model predicts that typically as septic tank setback distance increases, contamination susceptibility decreases, which is expected. Further analysis was carried out at a higher mean 120-hr precipitation rate of 100 mm (*Figure 9.3*). At the 100 mm 120-hour precipitation rate, the model predicted that the 0.5 probability threshold was reached at approximately 105 m, with contamination probability decreasing approximately linearly with increasing septic tank setback distance. Therefore, the model predicts a higher necessary setback distance during periods of increasingly intense precipitation, as expected.

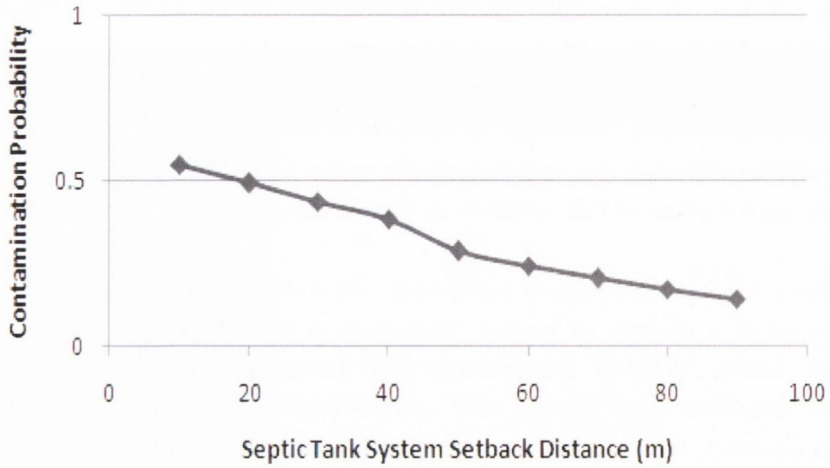


Figure 9.2 Contamination Probability vs. Septic Tank System Setback Distance at mean 120-hr precipitation (17 mm) (BM11)

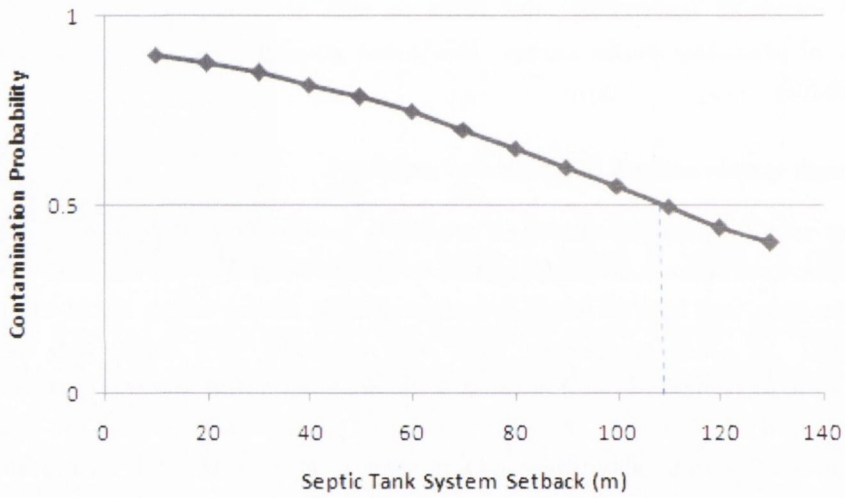


Figure 9.3 Contamination Probability vs. Septic Tank System Setback Distance at 100 mm 120-hr precipitation (BM11)

Furthermore, septic tank setback was additionally affected by the setback gradient with respect to the wellhead location. At a 100 mm precipitation rate, it was found that, as expected, where a septic tank was located up-gradient of a wellhead, it presented a greater contamination probability than septic tanks situated level with and down gradient of the well respectively. As shown in *Figure 9.4*, threshold value analysis found that the 0.5 probability threshold was exceeded by up-gradient wells at an approximate setback distance of 110 m. These figures were 80 m and 40 m for (hypothetical) septic tank systems level with and down gradient of BM11, respectively. Therefore, in this case, the model showed that septic tanks had an approximate "zone of potential influence" extending up to 40 m down-gradient and 110 m up-gradient of BM11, while septic tanks level with this well would seem to have a "radius of potential influence" of approximately 80 m. This finding was expected, and illustrates that the model works logically with respect to septic tank influence on private groundwater source contamination.

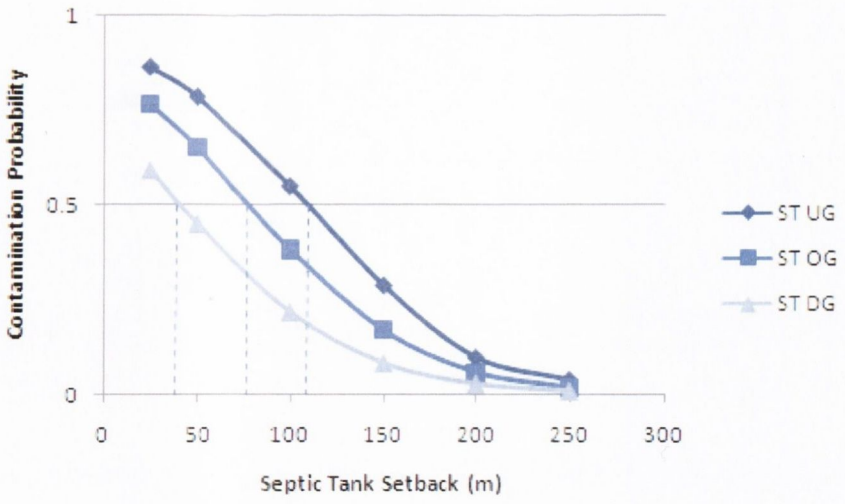


Figure 9.4 Contamination Probability Vs Septic Tank System Setback Distance at 100mm 120-hr precipitation and various septic tank setback gradients (BM11)

The figures presented in Figure 9.4 would seem to lend support to the recent *Code of Practice for Wastewater Treatment Systems for Single Houses* (EPA, 2009d), which provide guideline figures pertaining to minimum septic tank (and percolation areas) setback distances from private groundwater sources, in the context of setback gradient. Guidance setback distances of 60 m, 25 m and 15 m are stated for septic tanks located up-gradient, alongside (no gradient) and down-gradient of domestic wells, respectively (see Appendix G4). Although the figures of 110 m, 80 m and 40 m outlined in Figure 9.4 are considerably higher, this is due to the use of a 100 mm (i.e. atypically high) 120-hour precipitation rate in the sensitivity analysis process. As shown in Figure 9.2, a septic tank setback distance of 21 m is the point at which contamination probability becomes <0.5 at the more typical 120-hour rainfall rate of 17 mm. It is important to note however, that the model indicates that increased setback distances are necessary with increased precipitation rates.

9.6.2 Bedrock Type

BM11 was subsequently "transposed" into a "Dinantian limestone area" i.e. a private source with identical design, construction and location. As shown in *Figure 9.5*, the generalised LR model predicts higher contamination probability among those wells situated in areas with Dinantian limestone bedrocks (as opposed to other bedrock types including "non-Dinantian" limestone). This finding was expected, as both Dinantian limestone and associated limestone tills are highly permeable. Therefore, precipitation and consequently contaminants transported via precipitation will travel more quickly through these media, with a higher chance of reaching the source while bacteria remain viable. This finding has already been presented in SA4, Cluster 1 and Cluster 2, with limestone typically found to represent higher overall contamination susceptibility.

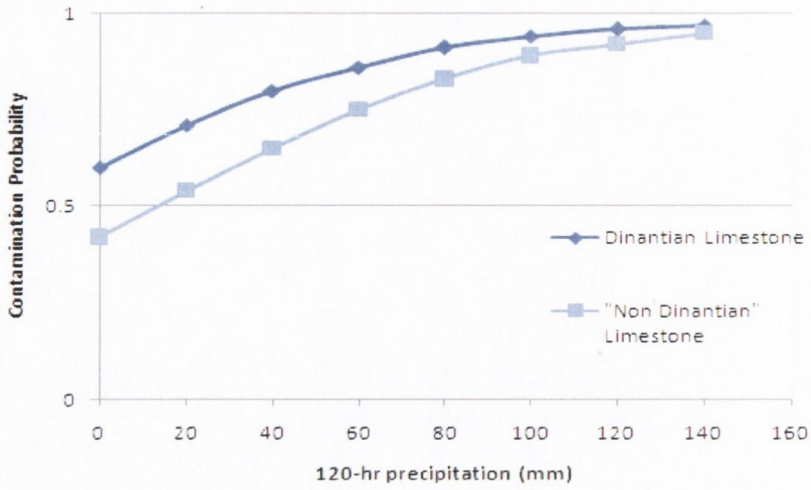


Figure 9.5 Contamination Probability Vs. 120-hr precipitation and Dinantian limestone/"Non-Dinantian" limestone bedrock (BM11)

9.6.3 Subsoil Thickness

As set out by Daly (2000), the pathway for contaminants from hazards to wells is typically by two means: (i) through the soils, subsoils and/or bedrock; or (ii) surface water in the immediate vicinity of the well having direct access to the well. Therefore, it is expected that where natural attenuation/filtration by subsoils is decreased or entirely absent due to thin subsoil layers, a higher level of susceptibility exists with regard to groundwater contamination. For model development, subsoil layers were simplified and dummy coded into subsoil layer 1 (<3 m) and subsoil layer 2 (>3 m). As shown in *Figures 9.6 & 9.7*, when "what-if" analysis was carried out, the LR model indicated that thinner subsoil layers resulted in higher overall contamination probability, over a range of septic tank system setback distances and 120-hr precipitation rates. However, subsoil thickness was sensitive to subsoil type/bedrock type, with Dinantian limestone sources typically exhibiting higher contamination probabilities than sources located in "non-Dinantian" limestone areas.

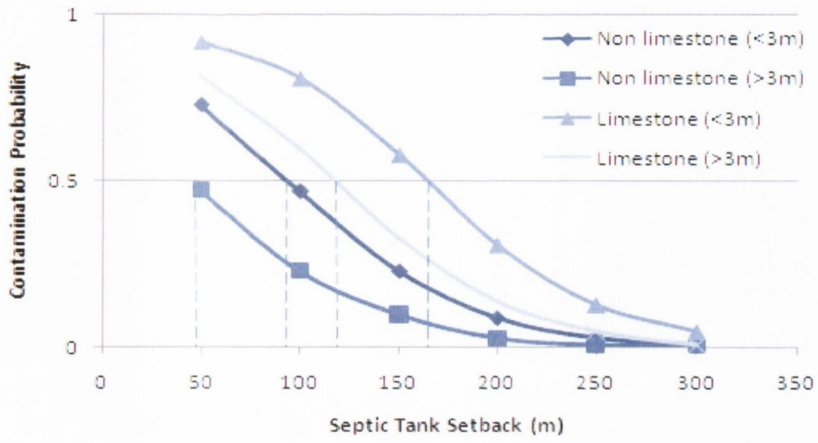


Figure 9.6 Contamination Probability vs. Septic Tank System Setback Distance at 100mm 120-hr precipitation including Dinantian limestone/"Non-Dinantian" limestone bedrock and <3m/>3m subsoils (BM11)

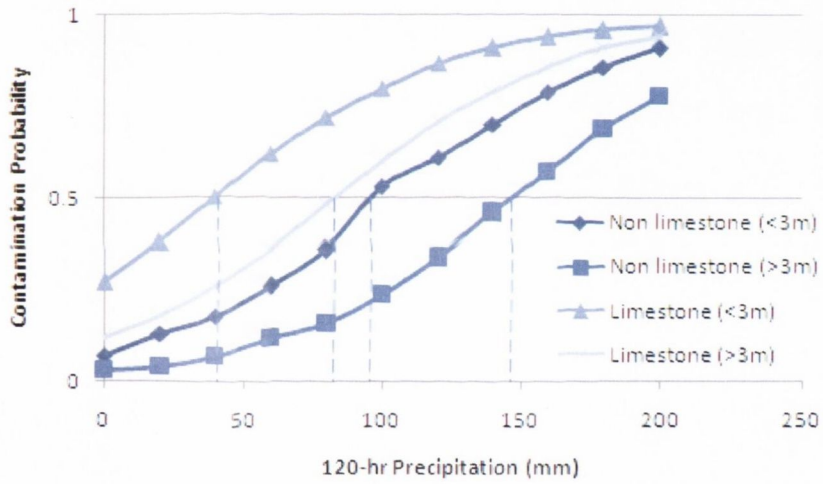


Figure 9.7 Contamination Probability Vs. 120-hr precipitation including Dinantian limestone/"Non-Dinantian" limestone bedrock and <3m/>3m subsoils (BM11)

9.6.4 Wellhead Design and Construction

As shown in *Figure 9.8*, when all other input parameters were held constant apart from 120-hr precipitation rate, the LR model indicated that those sources with a positive liner clearance (i.e. >0 mm protrusion of liner above ground or chamber floor), regardless of wellhead finish, would be less likely to be bacterially contaminated. It was found that even at 0 mm 120-hr precipitation, the 0.5 probability threshold was already exceeded. Similarly, the 0.5 probability threshold was exceeded in the absence of 120-hr precipitation in the case of sources without any wellhead cover in place (*Figure 9.9*).

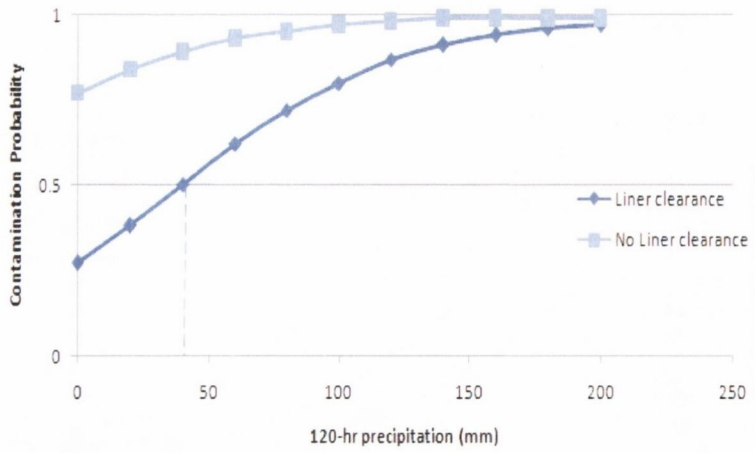


Figure 9.8 Contamination Probability Vs. 120-hr precipitation including well liner clearance presence and absence (BM11)

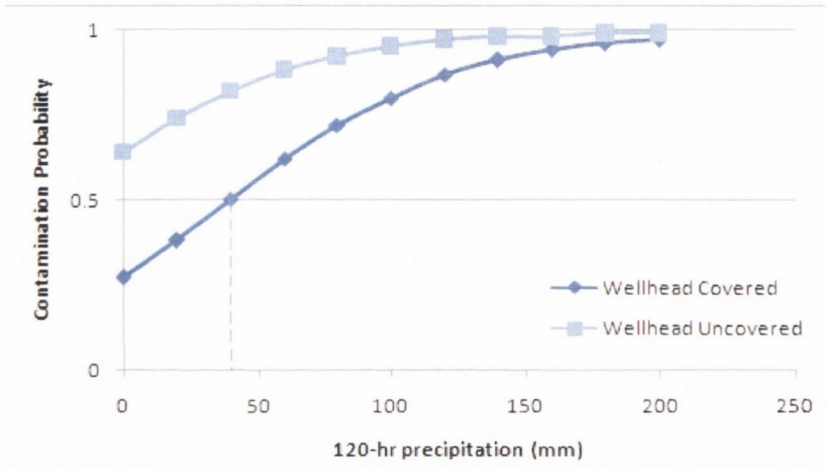


Figure 9.9 Contamination Probability Vs. 120-hr precipitation including wellhead cover presence and absence (BM11)

Although an exceedance of the 0.5 probability threshold was not in evidence in the case of 0 mm 120-hr precipitation in the case of wells adjacent to poor ground condition (i.e. surface ponding, etc), it is clear that these wells had higher overall susceptibility to bacterial contamination. "What-if" analysis found that the 0.5 probability threshold was exceeded at approximately 40 mm 120-hr precipitation where wellhead ground condition was adjudged to be poor, whereas this figure was approximately 85 mm where adjacent ground condition was categorised as good (*Fig. 9.10*).

A similar pattern, albeit more sensitive, emerged with regard to wellhead/chamber ingress. As shown in *Figure 9.11*, where definite surface water/shallow groundwater ingress was evident at the wellhead due to lack of sufficient liner clearance, chamber cracking, lack of waterproof finish, a higher probability of bacterial contamination was noted over a range of 120-hr precipitation rates. The 0.5 probability threshold was exceeded at approximately 8 mm 120-hr precipitation in the case of sources with definite ingress, whereas this figure stood at approximately 42 mm 120-hr precipitation where ingress was not evident.

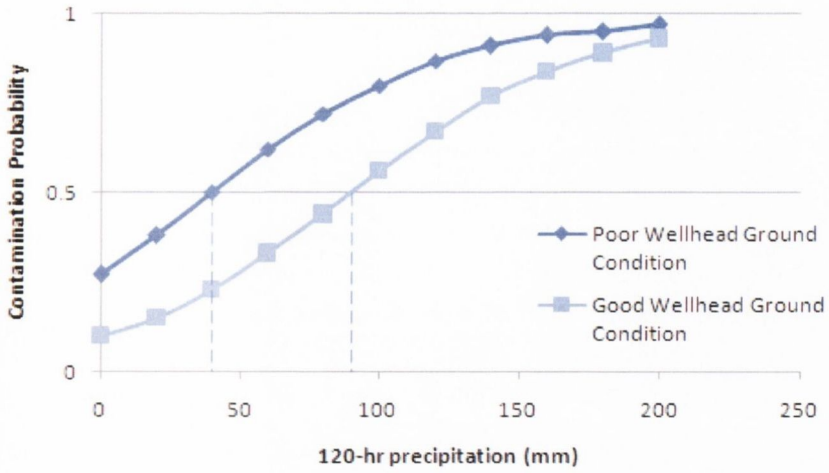


Figure 9.10 Contamination Probability Vs. 120-hr precipitation including wellhead radius ground condition (BM11)

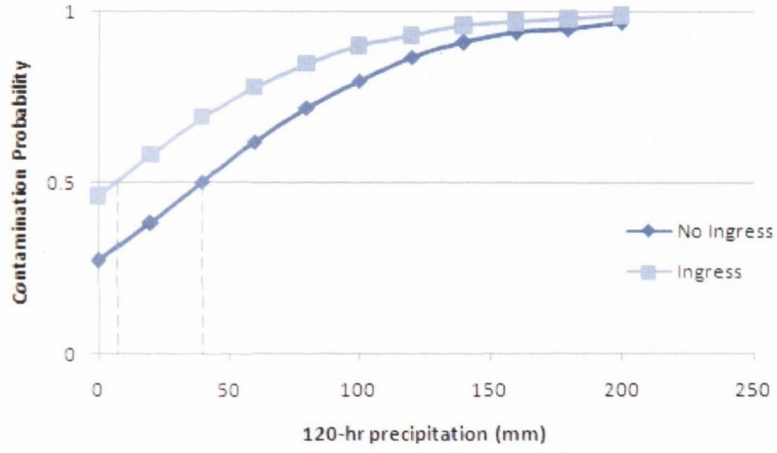


Figure 9.11 Contamination Probability Vs. 120-hr precipitation including wellhead/chamber ingress potential (BM11)

9.6.5 Source susceptibility categorisation

As previously mentioned, all developed models had inherent limitations i.e. none possessed 100% overall predictive accuracy, although all presented models were shown to provide good predictive abilities. Furthermore, as shown by sensitivity analysis (i.e. Figure 9.5; BM11 located in Dinantian limestone), in particular instances, the model predicts contamination probability >0.5 at all times i.e. source is always contaminated, which is not considered realistic, except in the most extreme cases. Therefore, the researcher proposes a generalised susceptibility model based upon contamination probability classification bands, which may be used in the absence of precipitation data i.e. continuous variable, to indicate the overall susceptibility of individual sources. For example, a three-tier equidistant classification scheme, as shown:

Low susceptibility < 0.33
Moderate susceptibility 0.33 – 0.66
High < 0.66

In the absence of 120-hour precipitation data, susceptibility scores for BM11 and BM20 were therefore, 0.32 and 0.57, respectively. Therefore, BM11 may be categorised as a low susceptibility source, with monthly monitoring figures reflecting this, as only 15.7% (n = 3) of samples contaminated with *E. coli*, typically during high rainfall periods. Conversely, BM20 (HD SPS) may be adjudged to be a moderate susceptibility source (with a contamination probability score of 0.57 being close to the theoretical cut-off point of 0.66), as this source is a hand-dug well with no liner clearance, and a septic tank located <20 m from the wellhead.

9.7 Model Verification (Validation Dataset)

Model validation is possibly the most important step in the model building sequence (Picard & Cook, 1984), particularly in the case of LR regression as models obtained using such data-driven methods are prone to increased type 1 errors. Therefore proper validation of such models is crucial (Coffey *et al*, 2004). A number of model validation methods were considered for the verification of the developed LR susceptibility models including data-splitting and cross-validation, however, according to Harrell (2001), both of these methods have inherent drawbacks and the use of a validation set was therefore favoured.

As set out by Picard & Cook (1984), the training dataset i.e. the data used to train or develop a model, are unsuitable for model validation as the LR model fitting process ensures that accuracy of the model based upon the "training set" is as high as possible and is therefore specifically suited to the dataset. Therefore, the use of test set data for model validation would result in exaggerated accuracy estimation, with the aforementioned test set providing a more realistic estimate of model performance (Coffey *et al*, 2004). In order that the developed models could be validated, an additional study area was selected, surveyed and assessed during the period October-December 2010 for use as a validation set (Castledaly, Co Westmeath). This dataset comprises 51 private groundwater sources and is described in Appendix G5.

Model validation was undertaken by substituting relevant source data into the developed LR equations as model coefficients. A classification cut-off of 0.5 was used as before, with a test model output of <0.5 recorded as a negative prediction and an output of >0.5 recorded as a positive prediction. The observed and expected frequencies were then compared in order to provide an estimate of model validity. Validation was undertaken on two developed LR models:

9.7.1. SA1 LR Model

This model was chosen for validation due to its geographical proximity to the test set study area (>20km) and geological/hydrogeological similarity. Furthermore, it was considered advantageous to perform validation one at least one localised (SA) LR model in order to estimate their accuracy outside the context of the test set. As shown (*Table 9.38*), the SA1 LR model correctly predicted approximately 57% of observed outputs within the validation set, comprising a 40.6% negative contamination accuracy and an 84% positive contamination accuracy. As the SA1 LR model provided

an overall fit of 90% within the SA1 test set, this is considered a poor overall fit and therefore it is concluded that the SA1 LR model is suitable for localised usage only. There was no significant mean difference between overall prediction accuracy within the validation set and well type, with correct predictions occurring in 51.5% of boreholes and 66.6% of hand-dug wells.

Table 9.38 LR model validation classification table (SA1 Model)

	Observed	Predicted	Percent Correct
E. coli Absent	32	13	40.6
E. coli Present	19	16	84.2
			56.8

Examination of predictions leads the researcher to conclude that the high reliance and weighting associated with precipitation data in the SA1 LR model leads to an overestimation of contamination probability. As the duration of the sampling programme for the development of the validation set took place over a short period (3 months), during which time there was above average precipitation, this is considered the primary reason for the poor level of validation. Therefore, localised models may be appropriate for use in geologically/hydrogeologically similar areas, however, as the validation set was not collated over a long enough duration, this was not possible to ascertain.

9.7.2. Total Generalised Model

This model was chosen for validation as it was the primary, generalised susceptibility model resulting from this stage of the research. It was considered important to validate this model with "unseen" data in order to gauge its efficiency for use throughout Ireland, and examine where any limitations may exist within the model. As shown in *Table 9.39*, the overall model validation accuracy was 80.4%, which was comparable with the overall training set accuracy of 83.9%. The LR model exhibited a very high level of accuracy with regard to prediction of bacterial presence (100%); however, this may also reflect some level of overestimation of bacterial presence (i.e. the model incorrectly predicted that 31.3% of uncontaminated sources were in fact contaminated). This is again likely due to the short duration of data collection associated with the validation set i.e. validation set collated over a 2-month period, during which higher than average precipitation occurred.

Table 9.39 LR model validation classification table (Total Generalised Model)

	Observed	Predicted	Percent Correct
E. coli Absent	32	22	68.7
E. coli Present	19	19	100
			80.4

As for the SA1 LR model, no significant mean difference was found between overall prediction accuracy within the validation set and well type, with model accuracies of 81.8% and 77.7% among bored and hand-dug sources respectively.

The validation results showed that the localised LR models, although useful within study areas and small, relatively uniform catchments should not be used generally. However, the final generalised model has been shown to provide a high level of accuracy (>80%) among both test and validation datasets and could therefore, be suitable for general use. However, there are inherent limitations within all LR models, highlighted by both false positive and false negative predictions in the models previously presented. It is considered that these models may be further improved i.e. prediction accuracy, variability, by further data collection.

9.8 Modelling Bacterial Contamination Magnitude

It was considered that, in addition to accurately modelling the risk of an adverse outcome i.e. the presence of *E. coli* in private wells, it would also be useful to estimate the likely level of contamination where present. While modelling several dichotomous outcomes was considered i.e. absence of *E. coli* versus low *E. coli* presence, absence of *E. coli* versus moderate *E. coli* presence, the approach does not fully utilise the available information and is less efficient for the case of ordered responses, specific ordinal techniques were favoured. In water resources applications, response variables may often be classified into more than two response categories, which have a natural ordering of severity or certainty (Helsel & Hirsch, 1992; Ananth & Kleinbaum, 1997). For this study, the following ordinal categories were developed: low *E. coli* (1-10 cfu/100ml), moderate *E. coli* (11-100 cfu/100ml) and high *E. coli* (>100 cfu/100ml). These water quality targets i.e. 1 cfu/100ml, 11 cfu/100ml and 101 cfu/100ml were selected and based upon the range of results obtained. Sources with <0 cfu/100ml are not included in this model as it is considered they have been appropriately modelled through LR i.e. *E. coli* absence.

Extensions of logistic regression for binary responses are available for analysis of these occurrences. It was considered that ordinal logistic regression (also known as the Polytomous Universal Model or PLUM procedure in SPSS/SAS/PASW) was an appropriate statistical technique for this analysis, which has been successfully used in a previous groundwater quality study by Twarakavi & Kaluarachchi (2005).

9.8.1 Ordinal Logistic Regression (PLUM)

The SPSS/SAS/PASW ordinal regression procedure (PLUM) is an extension of the general linear model to ordinal categorical data. As described earlier (see *Equation 9.2*), the LR equation is a linear function (Helsel & Hirsch, 1992). The log of the odds or logit (ln(odds)) is the probability of an event occurring i.e. *E. coli* 1-10 cfu/100ml, divided by the probability that this event does not occur. When there are more than two potential events - i.e. low, moderate, and high - the binary LR model can be modified to incorporate the ordinal nature of the dependent variable by defining the probabilities differently. Rather than considering the probability of an event occurring, the probability of that event and all events ordered before it are included i.e. in order to model the probability of *E. coli* 11-100 cfu/100ml, the probability of *E. coli* 11-100 cfu/100ml and *E. coli* 1-10 cfu/100ml are modelled (Helsel & Hirsch, 1992; Twarakavi & Kaluarachchi, 2005). The ordinal LR model is fitted to the observed responses using the maximum likelihood approach (Hosmer & Lemeshow, 2000), with this method typically producing values of the unknown model parameters (i.e. potential risk factors) that best fit the predicted and observed probability values (Twarakavi & Kaluarachchi, 2005).

Therefore the upper threshold i.e. *E. coli*>100 cfu/100ml (Y = 3), is modelled as (Helsel & Hirsch, 1992; O'Connell, 2006) (*Equation 9.7*):

$$\log\left(\frac{Prob(y=3)}{Prob(y=1)+Prob(y=2)}\right) = Prob(X > Y^*) = b_0 + b_1X \qquad \textit{(Eqn. 9.7)}$$

where b_0 is the estimate of intercept and b_1 is the estimate of slope. Likewise, the probability of being above the lower threshold (E coli > 0 cfu/100ml) is modelled using *Equation 9.8*:

$$\log\left(\frac{Prob\ (y=2)+Prob\ (y=3)}{Prob\ (y=1)}\right) = Prob\ (X > Y^* - \alpha) = b_0 + b_1X \qquad \textit{(Eqn. 9.8)}$$

where α is a shift parameter, estimated by the statistical programme (Harrell, 2001). As the responses are ordered (ordinal), the slope b_1 is common to all thresholds, and represents the proportional effect of X on the underlying and unobserved *Y**.

As stated by Twarakavi & Kaluarachchi (2005), an initial step using bivariate statistical analysis should be included prior to model development in order to identify all potentially influencing model variables. This was carried out using chi-square tests of independence and/or one-way ANOVA as appropriate, with results presented in *Tables 9.40 & 9.41.* Bivariate analysis suggests that the *E. coli* magnitude (where present) is primarily affected by the type and proximity of adjacent hazard sources including both point (point agriculture <100 m, septic tank number <100 m) and non-point sources (secure boundary area). Furthermore, while it was found that the presence of E.coli was typically related to 120-hour and 30-day rainfall, where present, the concentration is affected by 48-hour rainfall i.e. shorter, high intensity events. When considered concurrently, these results imply that point contamination sources are responsible for higher levels of *E. coli* ingress, with high intensity short-term rainfall causing rapid ingress of contaminants, before the on-set of bacterial die-off i.e. high bacterial concentrations and short travel times. It should be noted that due to low rates of bacterial presence and consequently low sample numbers, only the total dataset has been modelled using the PLUM procedure.

Table 9.40 Results of Chi-square analysis from PLUM dataset (p <0.1)

Variable	X^2	Sig	df
Bedrock Type	29.336	0.081	20
Dinantian Limestones	5.914	0.052	2
High Vulnerability	5.333	0.069	2
Point Pollution Source <100m	24.138	<0.001	6

Table 9.41 Results of t-test analysis from PLUM dataset (p <0.1)

Variable	F	Sig	df
Septic Tank Number <100m	1.596	0.1	2
Secure Boundary Area	10.375	0.001	2
48-hour Precipitation	2.198	0.1	2

9.8.2 Ordinal LR (PLUM) modelling of level of bacterial contamination

Once the association between each potential risk factor (explanatory variable) and event threshold was evaluated, based on χ^2 statistics, one-way ANOVA and associated *p*-values had been evaluated; risk factors were entered into the PLUM model. Potential risk factors were entered into the model in order of increasing p-values i.e. smallest p-value (potential point source of pollution ($p < 0.001$)). Nominal and ordinal coefficients were entered into the model as model factors, with all continuous variables entered as model covariates. A forced method of factor/covariate entry is used. PLUM models were developed using a maximum of 100 iterations, zero log-likelihood convergence, the *logit* link function and a 95% confidence interval, all of which are default PLUM settings.

As for binary LR, due to the apparent lack of association between source susceptibility and source awareness or contaminant awareness among respondents, awareness scoring has not been included in multivariate model development. All PLUM models were developed using SPSS/SAS 16.0 statistical software.

9.8.3 PLUM Diagnostics

In order to diagnose the overall model fit and uncertainty, a number of PLUM diagnostics are available for use. Both Nagelkerke R^2 and 2 log-likelihood statistics are used similarly to their use for binary LR models (as previously outlined in this chapter). Additionally, a number of diagnostic statistics are used specifically for ordinal LR models, including testing parallel lines (parallelism), deviance goodness-of-fit and Pearson goodness-of-fit.

In fitting an ordinal LR model, there is an inherent assumption that the relationships between the explanatory variables (risk factors) and the logits are the same for all logits, meaning the results of the model are a set of parallel lines (planes), for each category of the dependant variable (Helsel & Hirsch, 1992). In order to test this assumption, the model coefficients are allowed to vary, performing estimations of the coefficients and testing for equality (Harrell, 2001; O'Connell, 2006). Therefore, the 2 log-likelihood of the constrained model (i.e. the model that assumes parallelism) is tested against models with differing coefficient planes. The null hypothesis states the presence of parallelism. Therefore, if the null hypothesis is not rejected (high p-value), the parallel is considered adequate. Rejection of the null hypothesis is often attributed to the use of an inappropriate link function (O'Connell, 2006; Norušis, 2010)

Both the Deviance and Pearson goodness-of-fit statistics use the observed and expected frequencies to estimate the overall fit of the final model, as shown in *Equations 9.9* and *9.10*:

The deviance measure is

$$D = 2\sum\sum O_{ij} \ln\left(\frac{O_{ij}}{E_{ij}}\right) \qquad \textit{(Eqn. 9.9)}$$

The Pearson goodness-of-fit statistic is

$$\chi^2 = \sum\sum (O_{ij} - E_{ij})^2 / E_{ij} \qquad \textit{(Eqn. 9.10)}$$

As stated by Norušis (2010), a well fitting model will have similar observed and expected counts; therefore the value of both the deviance measure and Pearson statistic will be small, with high associated significance levels. Therefore, the null hypothesis that the overall model fits is rejected if the observed significance level of the goodness-of-fit statistics is small.

9.8.4 Final Ordinal Model

The final PLUM OLR model is presented in *Table 9.42* below.

Table 9.42 PLUM (OLR) model of contamination magnitude

	B (Est)	B Sig.
Threshold:		
E. coli Group 1 (<10 cfu/100ml)	-30.36	<0.001
E. coli Group 2 (11-100 cfu/100ml)	-28.969	<0.001
Factors:		
On-Site Surface Ponding = 0	-1.659	0.016
Septic Tank Up Grad = 0	-15.719	<0.001
Septic Tank On Grad = 0	-14.787	<0.001
Septic Tank Down Grad = 0	-15.367	<0.001
Potential Point Source <100m = 0	-3.983	<0.001
Potential Agricultural Point Source <100m = 0	3.978	<0.001
Covariates:		
24-hour Precipitation	-0.157	0.04
48-hour Precipitation	0.084	0.05

In order to calculate probabilities, a cumulative probability approach was adopted. The probability of affiliation with a specific group is then ascertained by subtraction, with the highest group probability denoting affiliation. A worked example is shown below (CB11, SA2):

$$Prob(event) = \frac{1}{1+e^{a+bx}} \qquad \textit{(Eqn. 9.11)}$$

where
α = -30.36 (group 1), -28.969 (group 2)
b_1 = -1.659
b_2 = -15.719
b_3= -14.787
b_4= -15.367

b_5= -3.983
b_6= 3.978
b_7= -0.157
b_8 = 0.084

Therefore;

$$Prob\ (group\ 1) = \frac{1}{1 + e^{3.682}} = \frac{1}{40.725} = 0.024$$

$$Prob\ (group\ 2) = \frac{1}{1 + e^{2.742}} = \frac{1}{16.517} = 0.06$$

$$Prob\ (group\ 1\ or\ 2\ or\ 3) = 1$$

Therefore;

$$Prob\ (group\ 3) = 1 - (0.024 + 0.06) = 0.916$$

The final PLUM model predicts membership of group 3 (*E. coli*>100 cfu/100ml), with 91.6% probability. CB11 (SA2) was found to have 256 cfu/100ml and therefore, in this case the model provides an accurate prediction.

As shown in *Table 9.43*, the overall model is statistically significant, with the final providing a better overall fit and prediction than the intercept only model. A test of parallelism resulted in an insignificant p-value (*Table 9.44*), therefore as previously mentioned, the null hypothesis is rejected, and the logit link function is considered appropriate for use (Norušis, 2010).

Table 9.43 -2 Log Likelihood of final PLUM model

Model	-2 Log Likelihood	X^2	df	Sig.
Intercept only	112.2			
Final	79.3	29.9	8	<0.001

Table 9.44 Results of test of parallelism on PLUM model

Model	-2 Log Likelihood	X^2	df	Sig.
Null Hypothesis	80.989			
General	71.302	9.687	9	0.376

As previously mentioned, a model which provides a good fit for the input data will have similar expected and observed counts, with the deviance and Pearson statistics being the primary measures for goodness-of-fit for PLUM models. As shown in *Table 9.45*, both of these statistics had high associated significance levels (0.993 and 0.705 respectively), and therefore the null hypothesis that the overall model provides a good fit is accepted. The Nagelkerke R^2 for the final PLUM model was 0.489, meaning that approximately 51% of variability/uncertainty remained unexplained.

Table 9.45 Deviance and Pearson goodness-of-fit statistics for final PLUM model

	X^2	df	Sig.
Deviance	73.392	106	0.993
Pearson	97.708	106	0.705

As shown in *Table 9.46*, ordinal prediction accuracy resulting from the developed PLUM model (based on the contaminated dataset) was extremely high in regard to low level bacterial contamination (97.4%) and relatively good at high level contamination prediction (61.5%). However, there was a complete lack of accurate prediction with regard to *E. coli* levels in the moderate range (11-100 cfu/100ml). This is likely due to a high level of uncertainty associated with source design/construction parameters and transient hazard sources, which may be improved through further data collection and analysis.

Table 9.46 PLUM model classification table

Observed	Predicted			
	E. coli Low	*E. coli* Moderate	*E. coli* High	Percent Correct
E. coli Low	37	0	1	97.4
E. coli Moderate	8	0	3	0
E. coli High	5	0	8	61.5
Overall Percentage				72.5

In order to characterise the significant model inputs and their respective influence with regard to bacterial contamination levels, both 2-step Cluster analysis and principal component analysis were carried out, using contamination group as the primary classification group. However, neither method resulted in the development of meaningful Clusters or components. Characterisation of contamination level has therefore been carried out using the PLUM model factors/covariates and their perceived importance within each contamination group (*Table 9.47*).

Table 9.47 PLUM model factors/covariates and their perceived importance

Risk Factor	>10 cfu/100ml	11-100 cfu/100ml	>100 cfu/100ml
On-site surface ponding (%)	26.4	54.5	30.7
Septic Tank Up Grad (%)	26.3	27.3	38.4
Septic tank on grad (%)	21	9.1	23.1
Septic tank down grad (%)	50	63.6	38.4
Potential Point Source <100m (%)	47.3	45.4	76.9
Agricultural Point Source <100m (%)	44.7	36.3	23.1
24-hour precipitation (mean)	3.85	2.32	3.26
48-hour precipitation (mean)	7.67	9.3	13.6

As shown above, typically those sources with low *E. coli* concentrations were characterised by a relatively high percentage of agricultural point sources within a 100m radius and low mean 24-hr and 48-hr precipitation rates. Conversely, high *E. coli* concentrations were associated with a high percentage of non-agricultural point hazard sources and high mean 48-hr precipitation rates, which would suggest that high contamination rates may be attributable to high short-term precipitation rates in concurrence with adjacent point hazard sources. Additionally, this is reflected in high bacterial concentrations being associated with the highest relative percentage of septic tank systems situated up-gradient from the wellhead. The moderate contamination group are characterised by high rates of on-site surface water ponding.

9.9 Summary

The focus of Chapter 9 was the integration of all data collected as part of the source susceptibility fieldwork programme (Chapter 7), in addition to data collected as part of the consumer awareness survey (Chapter 8). Bivariate analysis was carried out with respect to *E. coli* presence and all potential predictor variables, in order to inform subsequent multivariate modelling. Multivariate modelling was used to develop statistical models, capable of explaining previous bacterial contamination and consequently predicting future contamination events. The developed multivariate models were verified for accuracy using an independent model validation (training) dataset. Finally, similar modelling techniques were used to model contamination magnitude, where present.

Bivariate Analysis

Generally, it was found that point sources of contamination (i.e. septic tank systems, farmyards, animal housing, etc) were significantly associated with *E. coli* presence. This agrees with a previous study by Francy *et al.* (2000), who found that the presence of an adjacent septic tank was the primary risk factor with respect to *E. coli* occurrence in groundwater samples in the USA. Diffuse sources (i.e. grazing animals, landspreading), were only found to be significant within one dataset, namely sources located in *High/Extreme* vulnerability areas.

Source design and construction characteristics were found to be potential risk factors within a number of datasets, with wellhead finish, the presence of a wellhead cover and well liner clearance being the most significant among datasets. Typically, wells with little or no liner clearance were more likely to have *E. coli* present, as were wells finished at ground level, as opposed to those finished above or below ground, suggesting that contamination ingress may be occurring directly at the wellhead. A number of previous studies have reported similar findings (Simpson, 2004; Swistock & Sharpe, 2005; Gonzales, 2008).

Short and medium term precipitation (i.e. 48-hour, 120-hour, and 30-day) was associated with *E. coli* presence within most datasets, with higher rainfall rates typically occurring in concurrence with higher levels of *E. coli* occurrence. This suggests that precipitation is a primary contamination driver with respect to private groundwater sources in Ireland, with increased precipitation, particularly in areas underlain by moderate and high permeability subsoils (*High/Extreme* vulnerability) leading to both increased shallow groundwater infiltration and gradual recharge.

A number of risk factors were significant within both the *High/Extreme* vulnerability and *Low* vulnerability datasets, including septic tank system location and several well design characteristics. However, diffuse contamination sources including animal grazing and a number of geological/hydrogeological characteristics were also significant potential contamination predictors within the *High/Extreme* vulnerability dataset. This is suggestive of animal faeces being more rapidly recharged to groundwater in areas with thin overlying layers of high permeability subsoils, due to decreasing levels of natural protection afforded by soil layers, increasing the levels of viable bacterial ingress. Similar findings have been reported in previous studies (Spalding *et al.*, 1993; Goss *et al.*, 1998; Taylor *et al.*, 2004; Liu *et al*, 2005). Due to the relative homogeneity of the *Low* vulnerability dataset (i.e. all from SA4), the absence of geological/hydrogeological parameters as significant predictors was expected.

The most significant predictors with respect to bacterial contamination in hand-dug wells were the location of septic systems, and on-site characteristics including site gradient, and road setback distance and gradient. Antecedent precipitation was not a significant predictor, indicating that contamination in hand-dug wells is largely "non-temporal". Although septic tank system proximity was also adjudged to be significant with respect to boreholes, borehole design characteristics were adjudged to be more significant; suggesting that contamination of private boreholes is more source specific and based upon poor design and/or construction. The significance of 48-hour precipitation points to rapid direct ingress at poorly designed or constructed wellheads as a primary contamination mechanism.

The primary sources of bivariate dissimilarity between Clusters 1 and 2 were the significance of well chamber characteristics and lack of significance regarding precipitation within Cluster 2. This suggests that boreholes in limestone/sandstone areas are more sensitive to rainfall events. Furthermore, poor wellhead design among wells in Cluster 2 was a more significant cause of contaminant ingress.

Multivariate Analysis (E. coli Presence)

LR models largely reflected bivariate analyses, with point sources of contamination, particularly septic tank systems, significant within all developed models. This agrees with previous studies by Beller *et al.* (1997) and Borchardt *et al.* (2010), both of which reported that old and/or poorly maintained septic tank systems are potential causes of groundwater contamination, and consequently, large waterborne disease outbreaks. The only LR model which did not include septic tank location as a significant predictor variable was the SA1 model, which implicated both point and diffuse agriculture as primary hazard sources. Likewise, source design and construction characteristics were significant within all models, except SA2, with liner clearance and wellhead cover being the design/construction parameters of most importance. Antecedent precipitation was also identified as a significant predictor within a number of models, with 120-hour and 30-day rainfall the most significant rainfall periods, indicating that both rapid ingress/infiltration at or adjacent to the wellhead and gradual recharge to the aquifer are both likely modes of contaminant transport. However, it was found that within the total generalised model, only 120-hour precipitation was a significant predictor. This may suggest that relatively rapid shallow groundwater infiltration is a significant mode of transport (i.e. 24-hour and 48-hour precipitation may be associated with direct ingress of surface runoff, while 30-day precipitation may be associated with

gradual recharge). Significantly, the developed SA4 model (*Low* vulnerability), was the only model which included 48-hour precipitation as a predictor variable. This is suggestive of rapid contamination by way of direct ingress at the wellhead (surface runoff) as a contamination mechanism in *Low* vulnerability areas, particularly at hand-dug wells or poorly designed boreholes.

Geological and hydrogeological characteristics were found to be significant within a number of developed models, particularly within larger models, presumably due to decreased homogeneity i.e. the effects of geological and/or hydrogeological parameters may remain "hidden" within geologically homogenous areas. Within the final model, sources located in areas underlain by Dinantian limestones or overlain by <3 m subsoils were adjudged to have higher levels of inherent susceptibility.

The significance of on-site conditions and wellhead radius conditions as risk factors in a number of models (*High/Extreme* vulnerability; Cluster 2; Generalised model), suggests that "maintenance issues" are associated with source susceptibility and therefore, by extension, consumer control and consumer awareness is involved in source susceptibility. Future guidance should therefore, include on-site and wellhead vicinity maintenance as parameters of importance (see Chapter 11).

Typically, smaller datasets (e.g. study areas $n \approx 50$) resulted in models with higher prediction accuracies due to increased homogeneity, leading to decreased variability and/or uncertainty. For example, all individual study area models had overall prediction accuracies of approximately 90%, while the total generalised model had a prediction accuracy of 84%. It was not possible to measure a number of potentially significant predictor variables as part of this study (e.g. the presence of a waterproof seal, depth of well liner, etc.) and these potentially significant variables could therefore not be included in multivariate modelling. Future work should focus on these parameters, as their addition may be used to further refine and improve models through reduction of uncertainty/variability.

Previous work by Gaut (2005) found significant bivariate correlations between microbiological water quality in 49 private wells in Norwegian crystalline bedrock aquifers and (i) wellhead completion, (ii) type and thickness of superficial deposits, and (iii) landuse and contamination sources. Both the localised and total generalised model produced in this chapter would seem to suggest similar associations existing in private wells in Ireland.

Sensitivity Analysis

The effect of septic tank setback distance from the wellhead was found to be approximately linear, with contamination probability decreasing with increasing setback distance, as expected (Figure 9.2). At higher precipitation rates, higher contamination probabilities were typically evidenced, as expected, with the overall pattern again approximately linear (Figure 9.3). Septic tanks located up-gradient of wellheads typically had a greater potential "zone of influence" than those septic tanks located "on-gradient" and down-gradient (Figure 9.4). These findings would seem to lend support to the recent *Code of Practice for Wastewater Treatment Systems for Single Houses* (EPA, 2009d), while also highlighting the point that atypically high precipitation rates will increase necessary setback ranges, and therefore an appropriate "factor of safety" may be recommended. Furthermore, sources located in limestone areas and/or in areas with <3 m of overlying subsoils were more susceptible to

bacterial ingress, as were sources with uncovered wellheads, sources with no liner clearance, and sources with categorically poor wellhead radius ground conditions.

A source susceptibility categorisation method is proposed (Section 9.6.5), based upon the final generalised LR model, whereby the 120-hour precipitation parameter is excluded from the model, resulting in classification of the overall source design, construction and setting with respect to geology, hydrogeology and potential contaminant sources.

Model Verification

Two developed models were used for model validation, namely the SA1 LR model (due to its geographical proximity ([>20km] and geological/hydrogeological similarity) to the training set (Castledaly, Co. Westmeath), and the final generalised model (the model of principal importance).

The SA1 LR model correctly predicted approximately 56.8% of observed outputs within the validation set (*Table 9.38*). As the developed SA1 LR model provided an overall fit of 90% within the SA1 test set, this was considered a poor level of validation, with the high reliance and weighting associated with precipitation data in the SA1 LR model leading to an overestimation of contamination probability. Due to a short data acquisition period associated with the training set and atypical precipitation patterns during this period, it is concluded that localised models may be appropriate for use in hydrogeologically similar areas, however, longer data acquisition periods are recommended.

The total generalised model provided a high level of verification, with a validation accuracy of 80.4%, comparable with the overall training set accuracy of 83.9% (*Table 9.39*). It was concluded that the final generalised model was shown to provide a high level of prediction accuracy (>80%) among both test and validation datasets and could therefore, be suitable for general use.

PLUM Modelling (E. coli magnitude)

The proximity of point sources of contamination, including septic tank systems and agricultural sources, were found to be significant predictors of *E. coli* magnitude, where present. Increased levels of point sources of contamination were associated with higher bacterial concentrations. Furthermore, the presence of visible on-site surface ponding was also a significant predictor of increased bacterial concentrations, suggesting direct ingress at the wellhead as a significant mode of contaminant transport. Short-term precipitation (24-hour, 48-hour) was also significant within the final PLUM model. Typically, medium-term precipitation (120-hour; 30-day) was found to be a risk factor with regard to bacterial presence, however, contamination magnitude was more sensitive to short term precipitation; which again indicates direct ingress of surface runoff from point sources (agricultural, domestic and commercial) is a primary mechanism of contamination.

PLUM modelling was useful for discerning between high and low levels of bacterial contamination, however, the model could not effectively predict moderate bacterial contamination levels.

Chapter 10: A Quantitative Human Health Risk Assessment of Microbial Contamination of Private Groundwater Supplies in Ireland

10.1 Introduction

As set out in *Chapter 4*, both chemical and microbial contaminants of private groundwater supplies may pose potential human health problems, however, the main private groundwater contaminants of concern in terms of human health in Ireland are microbial pathogens, including:

- Bacteria, such as verocytotoxigenic *E. coli*
- Protozoa, such as *G. lamblia* and *Cryptosporidium* spp.
- Viruses, including rotavirus

The following chapter uses both existing data and data collected by the researcher to develop a quantitative Monte Carlo simulation model to evaluate the annual risk of infection from microbial contamination of private groundwater sources in Ireland. Three existing deterministic health risk assessment models have been employed to estimate the magnitude of risk to health from contamination of private supplies. The deterministic models and their outputs are subsequently employed for the development of a new stochastic model. The model considers a number of input variables including private groundwater contamination occurrence, magnitude and seasonality, private groundwater consumption trends, decontamination (water treatment), and vulnerable sub-populations. Sensitivity analysis is employed to identify and quantify the model input variables of most importance in the context of model outputs i.e. increased/decreased human health effects. The model developed in this study may be useful for local authorities, government agencies and other stakeholders to evaluate the likely risk of infection given some basic input data on source water quality, treatment processes employed and consumer trends. Additionally, it is notable that private water supplies in Ireland are subject to significantly less surveillance than public supplies, therefore, the model and subsequent outputs may be used to compare the predicted levels of human health burden among the general and private groundwater populations in Ireland.

10.2 Objectives and Scope of Quantitative Risk Assessment (QRA)

Recognising the public health burden posed by pathogens such as *E. coli* and *Cryptosporidium* spp., and the potential role of private groundwater supplies in their transmission, this study aims to answer a number of risk assessment questions:

- What is the likely magnitude of human health risk attributable to consumption of pathogenic microorganisms via private groundwater sources in Ireland?
- Are these risks reflected in national reporting data and if not, what are the likely levels of under-reporting?
- Is there a difference between bored and hand dug wells with regard to potential human health risks associated with waterborne contamination?
- What is the daily probability of illness/infection in different risk groups (vulnerable subpopulations) and what is the annual probability of infection?

- What impact do consumer practices (i.e. consumption volume, use of treatment systems, etc) have on the predicted human health risk?
- Is there a large risk characterisation difference between deterministic and stochastic QRA methods?

10.3 Hazard Identification

As previously outlined (Chapter 5), hazard identification is the first step in the "4-Step Risk Assessment Approach", which has been adopted for this study. Although hazard identification, including health risk identification, pathogen identification and transmission route identification has been previously outlined in Chapter 4, this is now expanded upon with respect to the two specific pathogens being assessed.

10.3.1 E. coli

A number of waterborne outbreaks of *E. coli* associated with private groundwater supplies have been identified in Ireland during the period 1999-2009. In 2003, the North Eastern Health Board (NEHB) reported an outbreak of VTEC O157 which was positively linked to a rural sports club. Four human cases were confirmed, three of which required hospitalisation. Drinking water at the location was supplied from an untreated private well and microbial analysis was positive for the outbreak strain, with epidemiological evidence also positively linking human infection with water consumption from the water source. Two additional family outbreaks were also reported by the NERB during 2003, with both being linked to waterborne transmission routes. Microbial evidence positively linked water from a private well in one of these outbreaks. For the second, water from a private well was found to be positive for *E. coli,* but a definite epidemiological link was not established (HPSC, 2004a). Furthermore, a third family outbreak was reported by the Western Health Board (WHB) during 2003 as being transmitted either by water or by animal contact. Both water from a small group water scheme employed by the family, and samples taken from sheep on the family farm, tested positive for VTEC O157 that were indistinguishable from those isolated from the human cases (Coffey *et al.,* 2007). Overall, during 2003, there was documented evidence from six family outbreaks served by private water supplies (comprising 10 confirmed cases) (HPSC, 2004a; Coffey *et al.,* 2007). This is the first instance of an Irish health board and the Health Protection Surveillance Centre (HPSC) specifically detailing the potential human health effects posed by contaminated private wells.

During 2006, as in previous years, evidence again suggests that untreated drinking water played a significant role in *E. coli* transmission in Ireland (HPSC 2008a). There were two incidents where examination of water from the private wells of affected households confirmed the presence of *E. coli* O157 indistinguishable from the associated human isolates. This report reiterated the point that drinking water from untreated private water supplies at this time remained an important risk factor for VTEC infection in Ireland.

In 2008, an elevated concentration of Verotoxigenic *E. coli* (VTEC) notifications were reported in HSE-South, with over 50% of the regions VTEC notifications occurring during an 8-week period from early June. As shown (*Figure 10.1)*, during that period, thirteen cases of VTEC (8 confirmed/5 probable) were notified, with all except two cases associated with a drinking water supply from a

private domestic well, with the same report stating that provisional data suggest that drinking water from houses with private wells could be a highly significant factor in the increased number of reported VTEC cases (HPSC, 2008b).

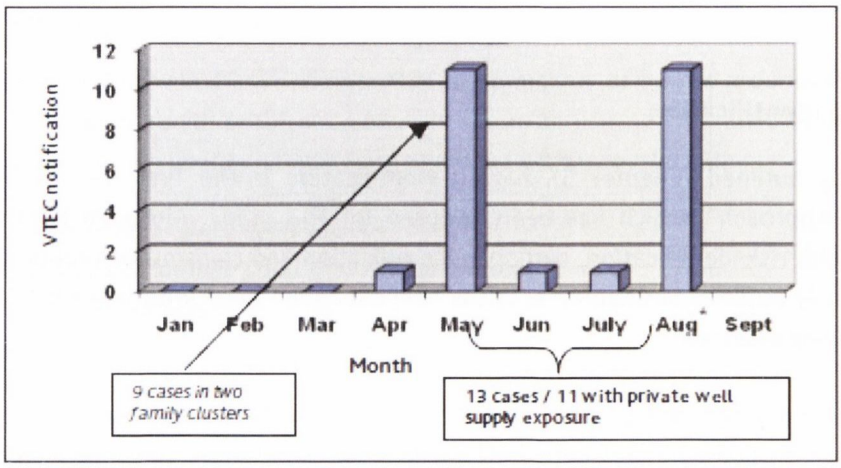


Figure 10.1 VTEC Notifications HSE-South 2008 (HPSC, 2008b)
**Until 20/08/08*

Overall Incidence

As shown *(Figure 10.2)*, the overall incidence of reported VTEC cases in Ireland has shown an upward trend (R^2 = 0.816) over the 10-year period 1999-2009. Although, as already outlined, not all VTEC cases were attributed to waterborne transmission routes, as shown in *Table 10.1*, HPSC (2008a) data attribute approximately 17% of VTEC outbreaks with known transmission routes to waterborne sources, with 12.5% of confirmed VTEC cases and 17.4% of VTEC illnesses. Therefore, it is likely that increased overall incidence rates are also reflected in increased waterborne incidence rates, at least some of which may be attributed to untreated private groundwater sources.

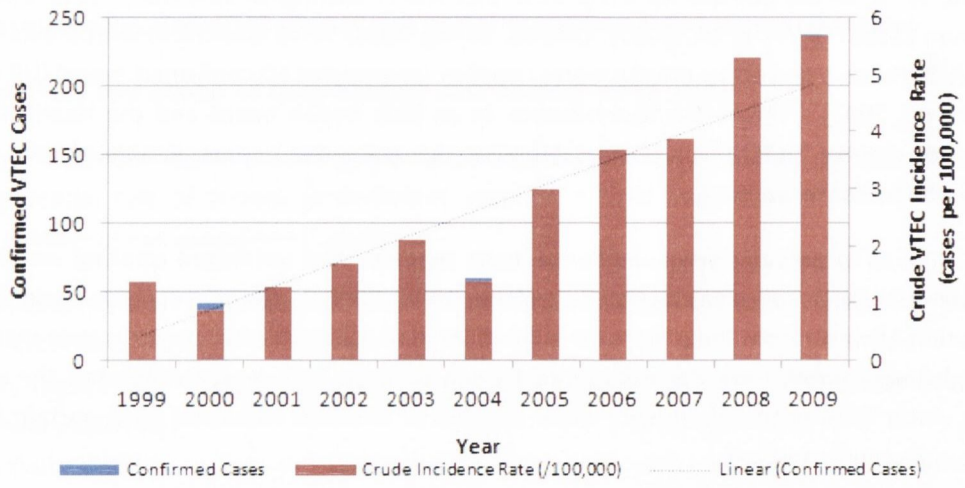


Figure 10.2 Confirmed VTEC cases and crude incidence rate in Ireland 1999-2009

(NB: Crude Incidence Rate (CIR) refers to the number of notified/diagnosed waterborne illnesses per 100,000 population) (HPSC, 2010b)

Table 10.1 VTEC outbreaks by suspected mode of transmission, 2006

Suspected mode of transmission*	No. of outbreaks	No. of confirmed cases	No. reported illnesses
Animal contact	1	3	3
P-P	5	21	8
Waterborne	1	2	2
P-P & Foodborne	3	9	7
Foodborne	1	2	1
P-P & Waterborne	1	3	2
Unknown/Unspecified	18	50	43
Total	30	90	66

*(HPSC, 2008a); *P-P denotes suspected person-to-person transmission*

Seasonality

Typically, in Ireland the largest concentration of VTEC cases are reported during the 3rd quarter (July-September), as shown in *Figure 10.3* (HPSC, 2005). This third quarter peak was ubiquitously reported in all HSE areas in 2006. A similar pattern was reported for 2008, with 43.4% (98/226) of cases reported during the period July-September in all HSE areas apart from the primarily rural HSE-East and the HSE-Midlands, in which a large VTEC outbreak occurred during the first quarter (HPSC, 2009b). As outlined in Chapter 7, similar peaks relating to *E. coli* presence and magnitude were found during seasonal groundwater monitoring at both SA1 (*Figure 7.44*) and SA2 (*Figure 7.51*) during the 3rd quarter and therefore it is likely that at least a proportion of the reported annual increase in VTEC cases during this period is due to increased *E. coli* levels in private groundwater supplies.

This seasonal trend is typically observed in rural areas, and therefore may be attributable to agricultural cycles. Furthermore, it is likely that increased precipitation rates lead to increased contamination rates, due to increased levels of contaminant transport, where present. As shown in *Figure 10.4*, VTEC incidence rates are often lowest in the primarily urban HSE-East area.

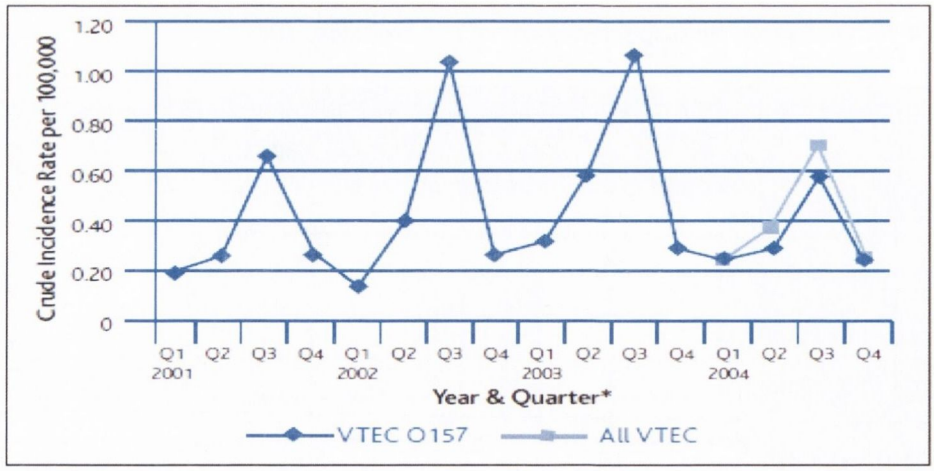


Figure 10.3 Confirmed VTEC cases by quarter of onset of symptoms; Ireland, 2001-2004

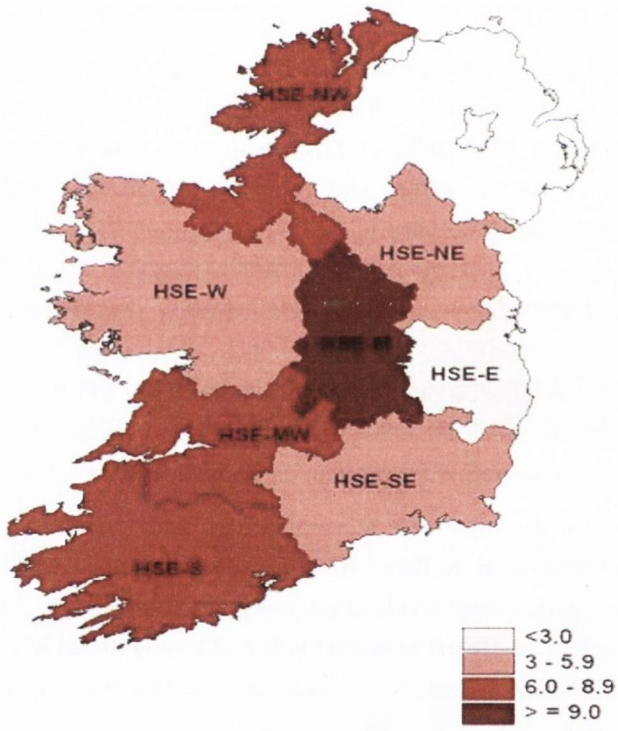


Figure 10.4 Map of Ireland showing VTEC crude incidence rates (cases/100,000) by HSE-area, Ireland 2008 (HPSC, 2009b)

Age Distribution

As shown (*Table 10.2; Figure 10.5*) there is typically a significantly elevated age-specific incidence rate (ASIR) among children <5 years, decreasing among children 5-14 years and again among adults >15 years (HPSC, 2009b). This has been the case in Ireland during the period 1999-2009, for example; during 1999 there were 51 documented cases of *E. coli* O157:H7 in Ireland, with 28 of

these cases in children <10 years (HPSC, 2010b). Furthermore, as shown in Figure 10.5, the ASIR is also elevated in those consumers >64 years.

Table 10.2 Age distribution of VTEC cases and age-specific incidence rate, 2006

Age group	Total Cases (Confirmed and probable)	Age-specific incidence rate (/100,000)
<5 years	59	19.5
5-14 years	36	6.4
>15 years	63	1.9

(HPSC, 2008a)

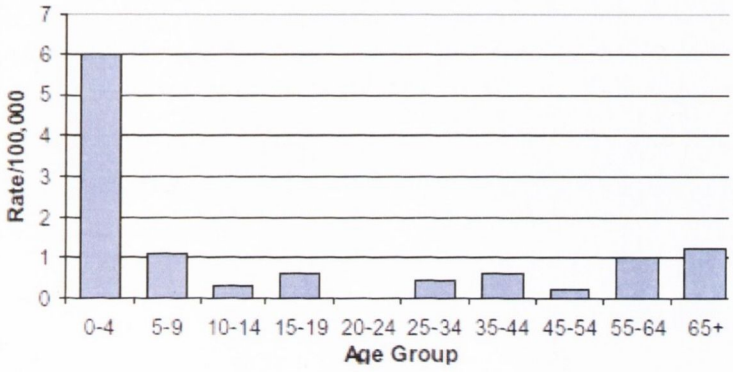


Figure 10.5 Age specific incidence rate of VTEC O157 in Irish residents, 2000 **(HPSC, 2001)**

10.3.2 Cryptosporidium spp.

As previously outlined (Chapter 4, *Section 4.4.1*), human cryptosporidiosis only became a notifiable disease in Ireland on January 1, 2004 (Garvey & McKeown, 2004), and hence, the true burden of human cryptosporidial disease in Ireland prior to 2004 remains undetermined. Overall, three well documented *Cryptosporidium* outbreaks have occurred in Ireland during the past decade, all of which were associated with waterborne transmission routes. These were Lough Owel (April 2002; Midland HSE), Carlow town (Spring 2005; Midland HSE) and Galway city (Spring 2007; Western HSE). These outbreaks have been discussed in Chapter 4 (Section 4.4.1).

Overall Incidence

As shown *(Figure 10.6)*, the overall incidence of reported cryptosporidiosis cases in Ireland has remained relatively constant over the 6-year period 2004-2009.The importance of water as a potential transmission route is reflected in a recommendation by the EPA that risk assessments be carried out by each sanitary authority to determine the vulnerability of public water supplies to *Cryptosporidium* (HPSC, 2008c).The primary *Cryptosporidium* species, where known, is *typically C. parvum (Table 10.3).* As shown in *Table 10.4*, during 2008, three *Cryptosporidium* outbreaks were adjudged to have waterborne transmission routes, attributing approximately 24% (7/29) of outbreak illnesses during that year (HPSC, 2009c).

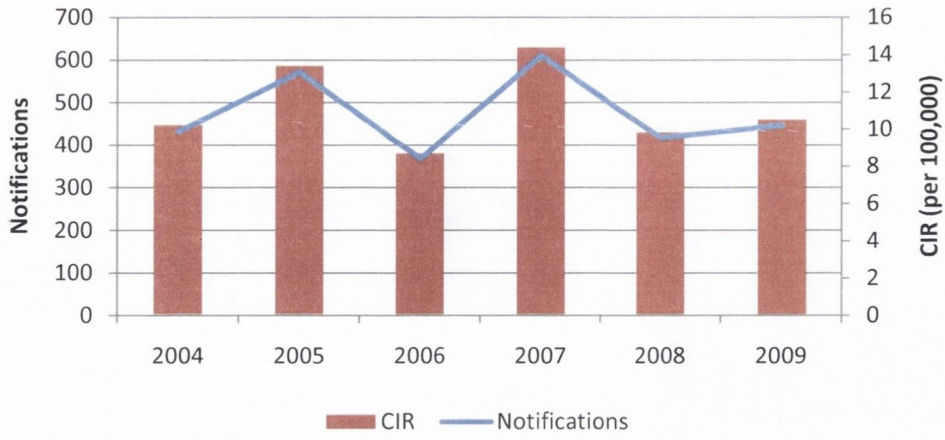


Figure 10.6 Confirmed cryptosporidiosis cases and crude incidence rate in Ireland 2004-2009 (HPSC, 2010a)

Table 10.3 Species distribution of Cryptosporidium cases, Ireland 2009

Species	No. of cases	% of cases
C. parvum	68	15.3
C. hominis	7	1.6
C. sp:nontypeable	1	0.2
Not specified	369	82.9
Total	609	100

(HPSC, 2010a)

Table 10.4 Cryptosporidiosis outbreaks Ireland 2008

Month	HSE Area	Transmission route	Type	Location	No. illnesses
Jan	SE	P-P	General	Crèche/Household	3
Jan	W	WB	Family	Private house	3
Feb	NE	P-P	General	Crèche	5
Mar	MW	Unspecified	General	Community	3
Mar	S	P-P/WB	Family	Private house	2
Apr	SE	P-P/Animal	General	Community	7
Jul	SE	WB/Animal	Family	Private house	2
Sep	NW	P-P	Family	Private house	4

(HPSC, 2009c); P-P denotes person-to-person; WB denotes waterborne

Seasonality

As previously outlined (Chapter 4; Section 4.4.1), there is typically a spring peak in human cryptosporidial infections, coincident with the lambing and calving season. This strong spring peak would, as shown in *Figure 10.7*, is also consistent with the hypothesis that *C. parvum* is the predominant species in Ireland and that agriculture is the primary hazard source. Notably, the number of cases in 2007 peaked slightly earlier than in previous years with 62% occurring between March and May. However, this was largely due to the outbreak in the HSE-W (Galway city). When the HSE-W cases are excluded, the seasonal pattern is similar to previous years, with a peak in April-June (HPSC, 2009c).

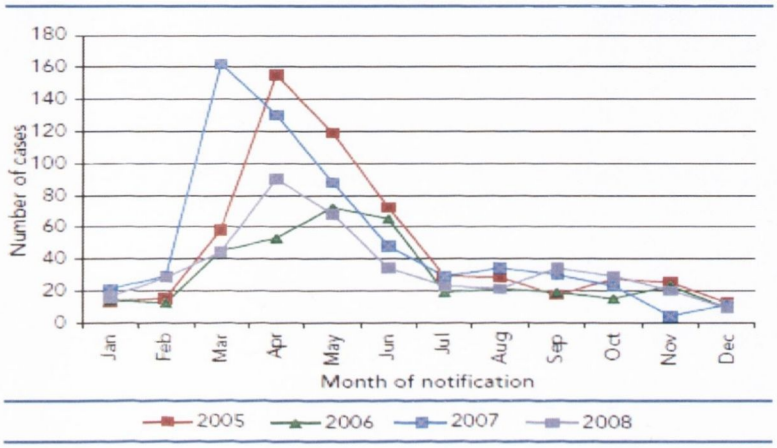


Figure 10.7 Seasonal distribution of cryptosporidiosis cases 2005-2008 (HPSC, 2009c)

As noted by Corbett-Feeney (1987), and HPSC (2009c), there is a large variation in incidence between the different health board areas. Two studies conducted concerning cryptosporidiosis in Irish children, found significantly higher incidence among children from rural backgrounds (Corbett-Feeney, 1987; Carson, 1989). Furthermore, as shown in *Table 10.5*, similar to VTEC infection, there is a particularly low rate of cryptosporidiosis hospitalisations among the ERHA population, which is consistent with this observation (HPSC, 2009c). Similar findings have been reported by the HPSC throughout the notification period 2004-2010. It is important to consider however that the striking variation in regional incidence in Ireland may also reflect different diagnostic and hospitalisation practices between regions.

Table 10.5 Number of notified cases and CIRs cryptosporidiosis by HSE area 2007

HSE Area	No. of cases	CIR (cases/100,000)
Eastern Region (ER)	22	1.4
Midlands (M)	34	12.8
Mid Western (MW)	57	16.0
North Eastern (NE)	24	5.6
North Western (NW)	25	10.6
South Eastern (SE)	79	16.9
Southern (S)	60	9.9
Western (W)	308	75.8

(HPSC, 2009c)

Age Distribution

There is typically a significantly elevated age-specific incidence rate among children <5 years, as shown in *Figure 10.8* (HPSC, 2006). Again, this pattern has been repeatedly reported by the HPSC throughout the notification period 2004-2010. Furthermore, although there is little existing data in Ireland regarding age-specific incidence rates of cryptosporidiosis infection, Haas *et al.* (1999) report that a number of previous studies have reported mortality rates of approximately 0.1% in elderly and infant populations.

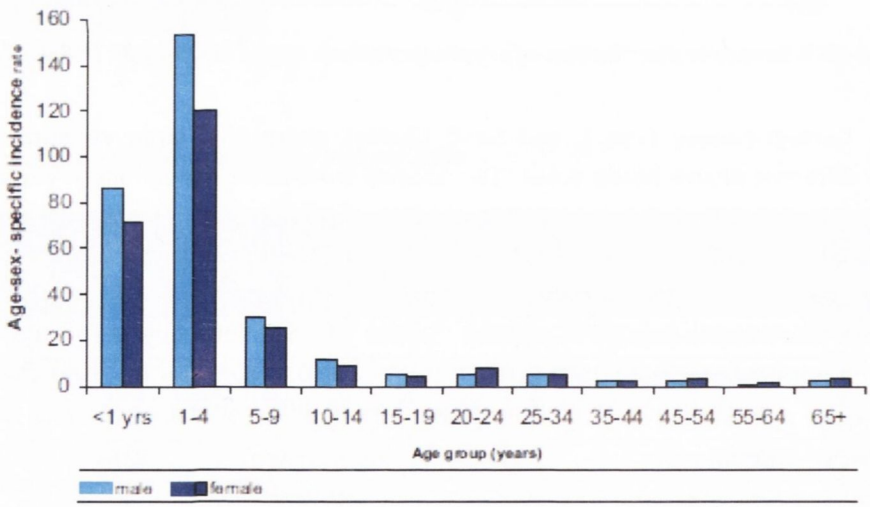


Figure 10.8 Age-sex-specific incidence rates for cryptosporidiosis in Ireland, 2005 (HPSC, 2006)

10.4 Exposure Assessment

As outlined in *Chapter 5*, the waterborne exposure assessment is a qualitative estimation of the extent of a contaminant in a single or daily serving from a private Irish groundwater source and its determination is based upon the route of exposure, the exposure duration and exposure magnitude. In the context of this study, the quantitative microbial risk assessment is based upon the number of

microorganisms at the time of consumption and therefore the accuracy of the exposure assessment is highly dependent on the quality of the data applied in generating the assessment. Ideally, data relating to microbial prevalence, magnitude, and virulence, in addition to data pertaining to consumption rates and frequency, seasonality and treatment system efficacy will be derived from actual process measurement. Where possible, this approach has been adopted throughout this study. However, due to factors including time restraints and lack of resources, not all necessary input data have been derived and therefore, these data are inferred from existing sources and as such are explicitly stated. In conducting an exposure assessment, it is necessary to include data relating to particular processes within the "private groundwater consumption chain". To do this, a consumption chain exposure assessment model has been developed and is presented in Figure 10.9. The data and their use within the exposure assessment are now outlined.

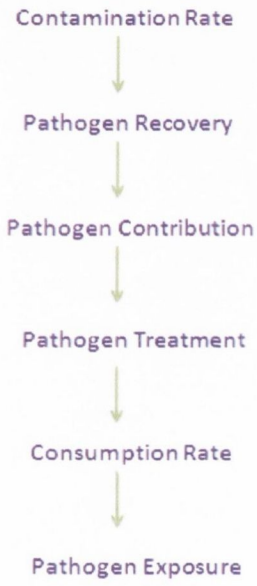


Figure 10.9 Flow Diagram of Exposure Assessment Model

10.4.1 Exposure Assessment Data

As previously outlined (Chapter 5, Section 5.6), the primary difference between deterministic and stochastic risk assessments is the manner in which assessment inputs are expressed and introduced into the risk analysis model. As both deterministic and stochastic methods are being employed for this study, exposure assessment input variables will reflect this difference. The deterministic inputs are expressed in terms of data means (or medians where appropriate), with stochastic inputs expressed as probability distributions developed from fieldwork data. These are now presented.

Contaminant Loading

Where bacterial contamination had occurred, it was considered important that the rate of bacterial loading be characterised, in order that these data could be included in both deterministic and stochastic risk assessment models. In the case of deterministic characterisation, due to the high proportion of "uncountable" bacterial culture results, resulting in high numbers of outliers, it was decided that the use of median rather than mean bacterial loads was appropriate. The overall median bacterial load (where found) was 4 cfu/100ml, however, as for bacterial occurrence, it was

also considered appropriate that source types be modelled individually, as both source types exhibited differing bacterial trends throughout the study. Median bacterial loading rates for bored and hand dug sources were found to be 3 cfu/100ml and 5 cfu/100ml, respectively.

In order to account for uncertainty and variability, stochastic distribution fitting was applied to characterise the bacterial loading rates for stochastic QRA. MS Excel and @Risk 5.5 were employed for this purpose. A fixed lower limit bound of 1 cfu/100ml and open upper limit bound (i.e. extending to infinity) were chosen to test the bacterial loading rates among bored and hand dug wells against 17 existing distributions, with parameter estimation adopted as the fitting method. The distribution with the smallest chi-square likelihood value was consequently adjudged to provide the most appropriate distribution fit to existing loading data.

The Geometrical distribution (μ= 53.9, δ = 103) was found to provide the most significant fit (χ^2 = 75.64) to bacterial loading rates among assessed bored wells (Figure 10.10), while a Geometrical distribution (μ= 56.15, δ = 85.8) also provided the most adequate fit for bacterial loading among hand dug wells (Figure 10.11) (χ^2 = 20.11). As previously outlined, due to low sample volumes, it was not possible to accurately enumerate the level of protozoan contamination, where present. Therefore, the rate of protozoan contamination is based upon a previous study. This study (The National *Cryptosporidium* Survey Group, 1992)has been chosen due to its proximity and similarity (private wells in the UK, with 5.8% of wells testing positive), and found typical groundwater *Cryptosporidium* concentrations, where positive, of 0.004-0.922 oocysts/L (mean 0.23 oocysts/L). As no contamination loading distribution is available in existing literature, a uniform contamination loading distribution has been assumed (Figure 10.12).

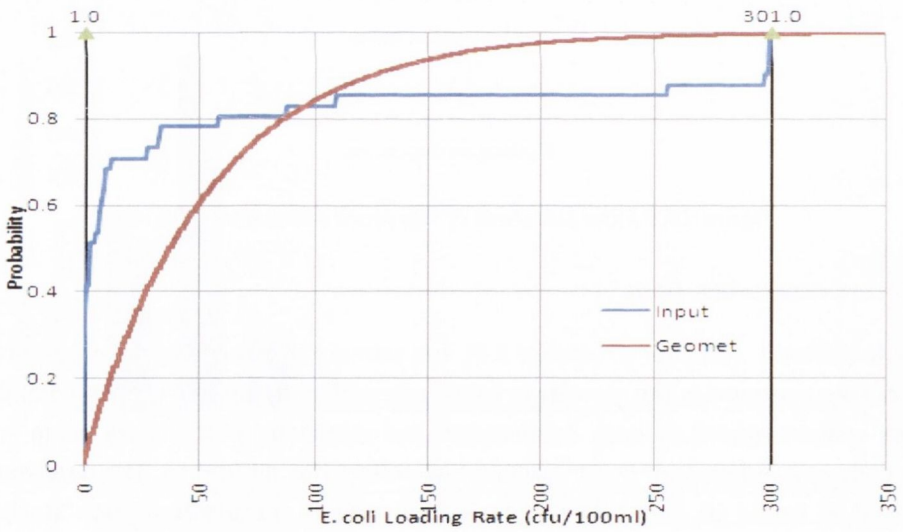


Figure 10.10 Fit comparison for E. coli loading rate among contaminated bored wells (Geometrical 0.018214)

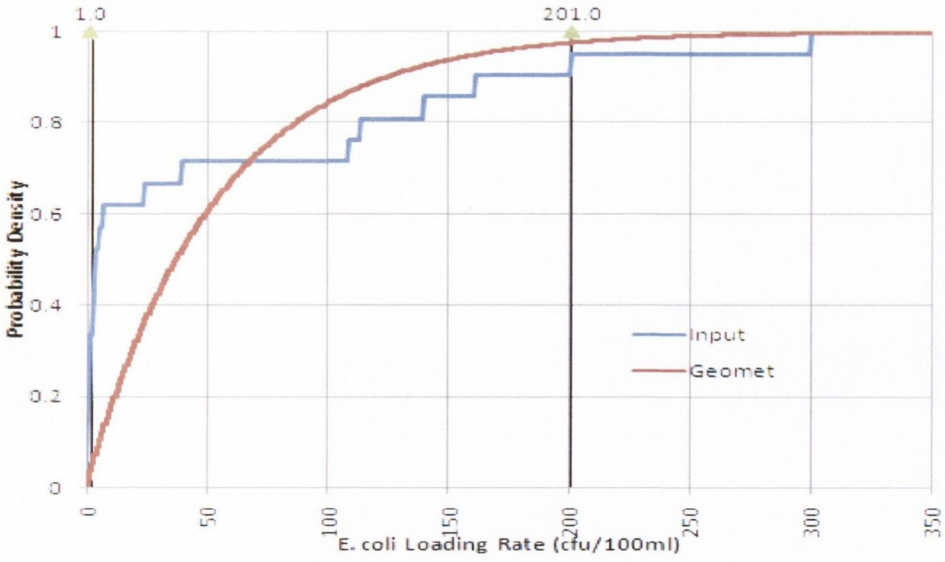


Figure 10.11 Fit comparison for E. coli loading rate among contaminated hand-dug wells (Geometrical 0.018309)

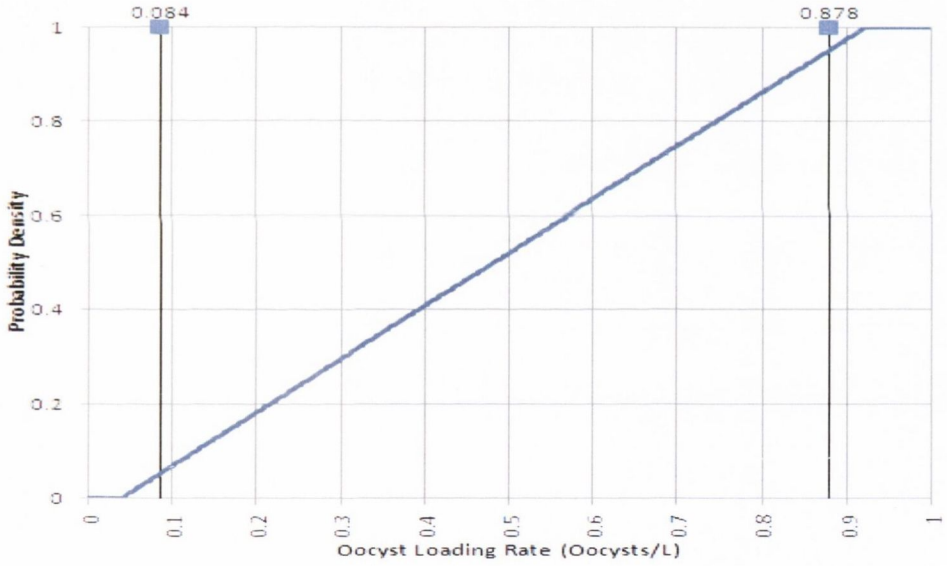


Figure 10.12 Uniform distribution representing oocyst loading rate (0.02-0.92 oocysts L^{-1})

Pathogen Recovery

The pathogen recovery rate is the proportion of pathogen that can be recovered through laboratory techniques. In the case of pathogenic *E. coli*, it will be assumed that recovery rates were 100% (Dufour *et al.,* 1981). Aldom & Chagla (1995) reported *Cryptosporidium* spp. recovery rates of approximately 70% when using flow cytometry. QA analysis of positive *Cryptosporidium* controls by the researcher found very similar recovery rates (75-85% recovery). Therefore, a 0.7 recovery rate will be used in reference to *Cryptosporidium* spp.

Pathogen Contribution

The pathogen contribution, or pathogenicity, of a microbial organism may be defined as the ability of the organism to produce an infectious (symptomatic or asymptomatic) illness in a host. For the purposes of this QRA, deterministic pathogen contribution rates will be adopted. Haas *et al.* (1999), report that approximately 8% (0.08) of *E. coli* are pathogenic, with this figure also adopted by WHO (2004) and Howard & Pedley (2006). Therefore, 0.08 will be utilised here as the deterministic figure for *E. coli* pathogenic contribution.

As reported by Chen *et al.* (2003b), ingestion of as few as 2-10 oocysts are sufficient for human infection. Therefore, due to this high level of infectivity, in order to avoid under-estimation of the potential disease burden associated with *Cryptosporidium*, it will be assumed that 100% of oocysts are potentially pathogenic.

Water Treatment

As described in Chapter 3 (Section 3.12), a number of treatment processes exist for the amelioration of pathogenic organisms and other contaminants. Although treatment rates at private groundwater sources are thought to be low, they are employed in some cases and where present, there will be a subsequent reduction in both contaminant occurrence and magnitude. This reduction will be reflected in overall exposure and therefore must be included in any exposure assessment of drinking water (Cummins *et al.*, 2010). As illustrated in *Figure 10.13*, conventional treatment processes comprise three treatment steps, including primary treatment which is primarily physical in nature; secondary treatment using chemical methods (i.e. coagulation/flocculation agents) and tertiary treatment focusing on biological inactivation.

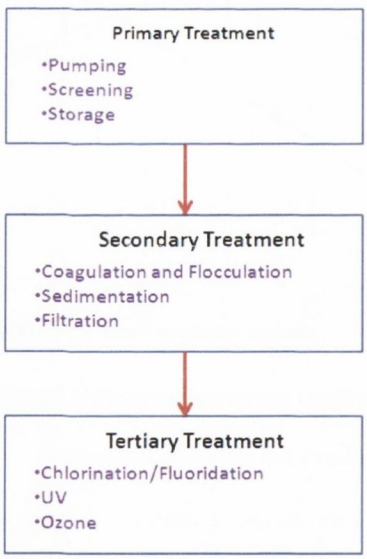


Figure 10.13 Flow Diagram of Water Treatment Processes

As part of the awareness survey (Chapter 8), the use and, where applicable, the type of treatment system in place were investigated. Results of this investigation are presented in *Figures 10.14 &*

10.15. As shown, approximately 68% of respondents stated that no treatment process was currently in use at their source. Of the remaining 32%, the majority of respondents (58%) employed domestic ion-exchange water softeners as their primary treatment process. As previously illustrated (*Section 8.2.1.7*), previous studies (Fleming, 1987; Parsons, 2000) have found that domestic water softeners have no treatment efficacy with regard to pathogenic organisms and in some cases ion exchange resins contained in these systems may even support microbial regeneration and growth. Therefore, domestic water softening systems will not be regarded as part of this exposure assessment.

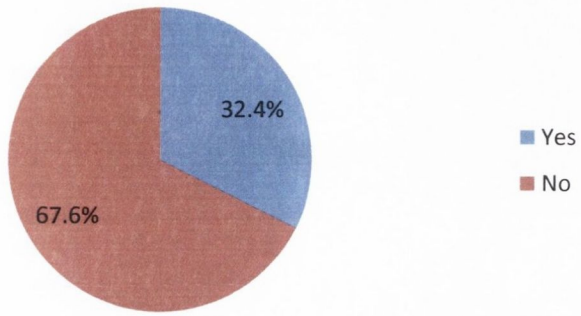


Figure 10.14 Use of Water Treatment (n = 243)

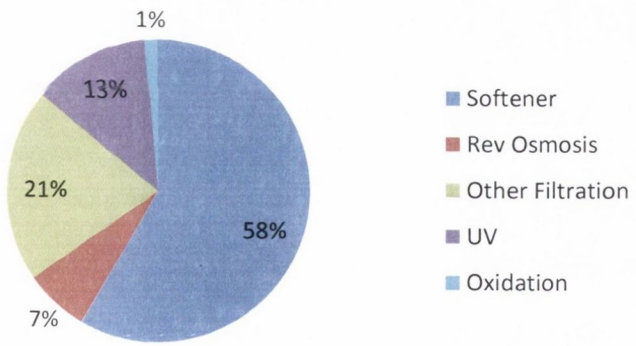


Figure 10.15 Treatment Process Frequency (n = 58)

All the remaining treatment processes, including reverse osmosis (RO), filtration, UV irradiation and chemical oxidation have the potential to successfully reduce pathogen numbers in drinking water (Teunis *et al.*, 1997), although chemical oxidation is very rarely employed as a "single-house system". Therefore, reverse osmosis (RO), filtration, UV irradiation have been included in the exposure assessment.

Al-Bastaki (2004) reported 100% microbial reduction by both UV and RO systems. Craik *et al.* (2000), Craik *et al.* (2001) and Linden *et al.* (2002) have shown UV irradiation inactivation rates of 2-4 $\log_{10}$(99% - 99.99%) for both *Cryptosporidium* and *Giardia*. For the purposes of this assessment, both

UV and RO systems will be assumed to inactivate 1 $\log_{10}$ (90%) of microbial pathogens (i.e. conservative approach).

Also previously outlined in Chapter 3 (Section 3.1.2), there are a number of filtration processes currently in use ranging from media filtration ($3x10^{0}$ – $3x10^{2}$µm pore size) to nanofiltration ($1x10^{-4}$ – $3x10^{-3}$µm pore size) (Kiely, 1997; Masters & Ela, 2008). All these systems have been shown to reduce microorganisms in solution including algae, bacteria, protozoa and viruses, however, different systems reduce to different degrees (Taylor & Wiesner, 1999; Van der Bruggen *et al.*, 2003). The collection of data for the exposure did not include the delineation of filtration systems and therefore a conservative approach will be taken, whereby filtration systems will be assumed to be media filtration systems ($3x10^{0}$ – $3x10^{2}$µm pore size), thereby providing an over-estimate of exposure, which is considered favourable to under-estimation. Jacangelo *et al.* (1995) report that MF membranes, where intact, are capable of removing 99.999% of protozoan pathogens, thereby almost completely eliminating exposure (Teunis *et al.*, 1997). It will be assumed that a 1 $\log_{10}$ rate of pathogen elimination results from intact filtration systems (i.e. conservative approach).

Therefore, as part of the exposure assessment, a 13.6% effective treatment rate has been adopted (i.e. 42% of 32.4%; overall level of treatment excluding water softening) in the case of *E. coli* and *Cryptosporidium* spp., with a 1-$\log_{10}$ inactivation rate.

Consumption Rate

A number of deterministic human health risk assessments, including Gerba *et al.* (1996a), Chick *et al.* (2001), Webb *et al.* (2003) and Ashbolt (2004) assume a daily water consumption rate of 2 L day^{-1}. However, other studies have shown that this assumption is inaccurate and the average person, particularly in temperate climates, typically consumes less than this (Ryan *et al.*, 2000; Robertson *et al.*, 2000; Jones *et al.*, 2006; Lloyd Hough *et al.*, 2010). Additionally, Mons *et al.* (2007) advocate the use of country-specific water consumption data for all QRA, due to climactic and dietary variation. Additionally, this approach allows for a more accurate estimation in the context of an entire population rather than a "representative consumer". Several similar studies have been carried out, with a summary of results presented in the Appendix (H1).

As for domestic consumption source, several questions were included in the awareness survey (Chapter 8, Section 8.2.4.3) in order that data pertaining to private groundwater usage in Ireland could be collected. Questions regarding daily consumption of un-boiled, "raw" groundwater i.e. straight from household tap, were included in this survey, in order that probability density functions for daily water consumption in Ireland could be included in the calculation of exposure. Only the volume of cold tap water without heat treatment (i.e. boiling for hot beverages), was investigated; therefore, only those respondents whose primary drinking water source was un-boiled water from a groundwater source were included in consumption distribution development. These data were collected from 545 respondents and are presented in *Table 10.6* and *Figure 10.16.*

Table 10.6 Summary statistics relating to daily un-boiled water consumption (ml/day)

Consumption Vol	N	Range	Min	Max	Mean	Med	Std. Dev.	Var
	483	2125	125	2250	926	750	610	377

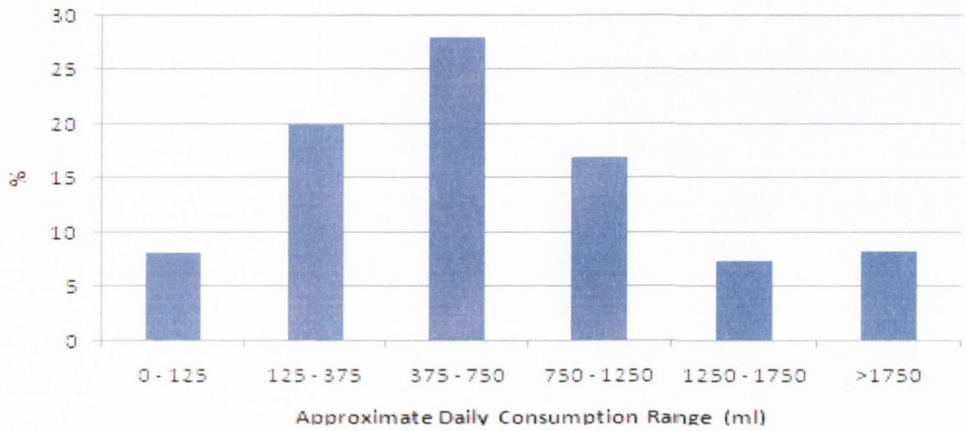


Figure 10.16 Daily un-boiled water consumption

As previously discussed (Chapter 5, Section 5.9.2), in the case of deterministic QRAs, a deterministic point estimate for input variables is typically employed, which in the case of daily groundwater consumption for this study will be the calculated mean daily consumption i.e. 0.926 L/day. This figure is similar to those from a number of studies (Ryan *et al.*, 2000; Robertson *et al.*, 2000; Jones *et al.*, 2006; Lloyd Hough *et al.*, 2010).

In the case of stochastic QRAs, however, input variables are typically in the form of statistical distributions, accounting for both uncertainty and variability. Previous studies have proposed that daily water consumption approximately complies with a lognormal distribution (Roseberry & Burmaster, 1992; Lloyd-Hough *et al.*, 2010). Mons *et al.* (2007) have proposed both the Poisson and exponential distributions as providing a better fit for daily water consumption. However, exploratory analysis of the data collected as part of the current research (See Appendix H2) found that the lognormal distribution provided a slightly better fit than both the Poisson and exponential distributions. As a greater number of studies lend support to use of the lognormal distribution, the lognormal distribution ($\mu = 0.926$, $\sigma = 0.61$) will be applied in carrying out the exposure assessment.

An independent samples t-test found no significant association between gender and consumption volume. Similar statistical analysis found no association between source type and consumption patterns. Data was also collected and analysed with regard to consumption patterns among vulnerable sub-populations, with a particular emphasis on infants and young children (Gerba *et al.*, 1996b). However, although consumption rates among these sub-populations is less than among adults, dose-response curves relating to *E. coli* and *Cryptosporidium* were not available and therefore, these have not been adopted.

10.5 Hazard Characterisation

As for the exposure assessment, a hazard characterisation model has been developed, beginning with a daily exposure level (from the exposure assessment process), and ending with an estimated annual probability of waterborne illness. This is presented below (*Figure 10.17*).

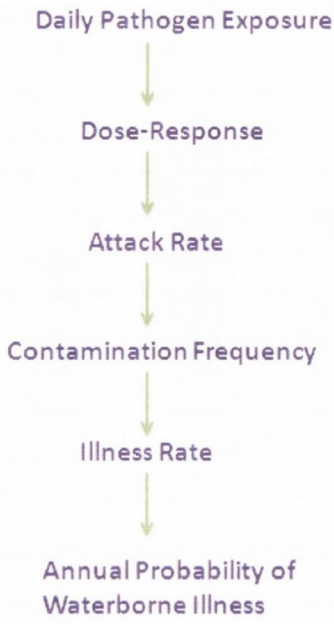


Figure 10.17 Flow Diagram of Hazard Characterisation Model

Dose Response

As outlined in Chapter 5 (Section 5.5), dose-response models describe observed infectivity as a function of dose (Messner *et al.*, 2001). Also outlined in Chapter 5 (Section 5.5.1), the primary difference between the various QRA methodologies tends to be the specific dose-response model employed. As the three deterministic models used in this study, in addition to the developed stochastic model, utilise different dose-response inputs, their application is discussed at the hazard characterisation step for each model.

Attack Rate

The attack rate is the proportion of those people infected by a specific waterborne pathogen that go on to show symptomatic signs of illness (morbidity) i.e. diarrhoea, nausea, vomiting, etc. As the collation of these data were not within the remit of the current study, and no previous studies have been undertaken in Ireland with regard to waterborne pathogen attack rates, previously published studies from developed countries were reviewed.

Olsen *et al.* (2000) reported an illness/attack rate of 0.23-0.5 among residents of Alpine, Wyoming during an *E. coli* 0157:H7 outbreak, which was associated with an unchlorinated water supply. Overall, 157 cases of illness were identified, with two "follow-up" cohort studies finding that illness was significantly associated with consumption of untreated water from the contaminated supply. An

attack rate of 0.5 is therefore applied in reference to *E. coli* contamination of all groundwater sources. Dalle *et al.* (2003) reported on an outbreak of waterborne cryptosporidiosis in South Burgundy, France, during which a *Cryptosporidium* illness/attack rate of 0.508 was found. Similar figures have been reported by a number of studies including Teunis *et al.* (2002) (0.52-0.54), Ortega *et al.* (2006) (0.46) and Hajdu *et al.* (2008) (0.67). An attack rate of 0.508 has been adopted throughout this chapter.

Illness:Infection Ratio

The risk of developing acute symptomatic illness once infected by *E. coli*, as stated by Howard & Pedley (2006), is difficult to estimate, due to limited consolidated data. At present, the proportion is set at 25% (0.25) based on the morbidity ratio for Shigella. This figure has been previously employed by Haas *et al.* (1999) and Howard & Pedley (2006), and is applied here. An illness:infection ratio of 100% is employed in the case of *Cryptosporidium* spp. (WHO, 2004; Howard & Pedley, 2006)

Seasonal Contaminant Occurrence

From the 21-month temporal sampling programme, it was found that bored wells had some level of bacterial presence on 28.2% of temporal sampling occasions (n = 11), while hand dug wells had bacteria present 53.8% of the time (n = 4). This equates to a mean of 102 contaminated days per annum among bored wells (range 0-228 days) and 196 contaminated days among hand dug wells (range 136-283 days). The aforementioned mean values will be used for the deterministic quantification of adverse human health effects associated with contaminated private groundwater sources. The stochastic quantification necessitated the development of statistical distributions relating the temporality of bacterial contamination to probability; this was undertaken with Excel and @Risk 5.5.

Temporal sampling results from both bored and hand dug wells were fit with the most appropriate distribution, using a chi-square statistic to adjudge goodness-of-fit. Temporal contamination among bored wells was found to fit a geometrical distribution (μ = 109; δ = 74.6) (Figure 10.18);while contamination seasonality among hand dug sources was Poisson distributed (μ = 195; δ = 57.1) (Figure 10.19). It is considered that the temporal sampling results associated with hand-dug wells provided a relatively poor fit, due to the low number of monitored hand-dug wells, however, the Poisson distribution provided the best available fit for the available data. Both temporal contaminant occurrence distributions were truncated (Bored – 0%, 95%; HD – 0%, 89%), in order that minimum and maximum values of 0 and 365 days were randomly selected during simulation modelling.

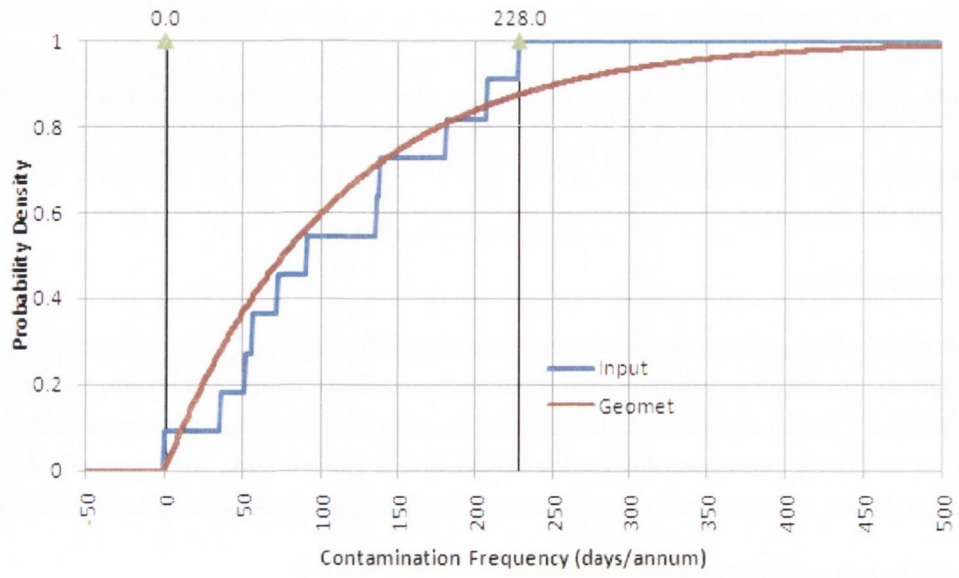


Figure 10.18 Fit comparison for contamination frequency among contaminated bored wells (Geometrical 0.009061)

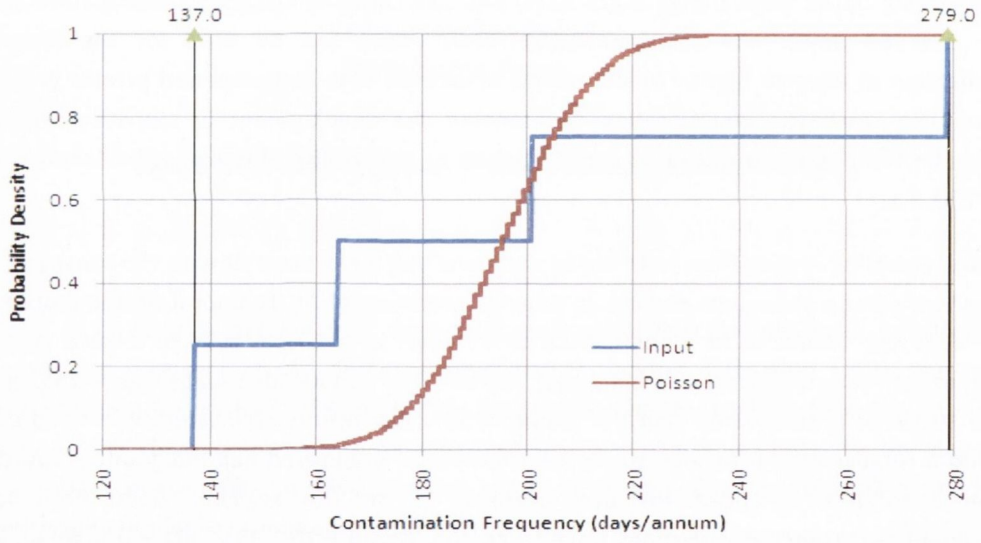


Figure 10.19 Fit comparison for contamination frequency among contaminated hand-dug wells (Poisson 195.25)

10.6 Risk Characterisation

A risk characterisation model developed for this study is presented in *Figure 10.20*. All risk characterisation input variables were deterministic in nature.

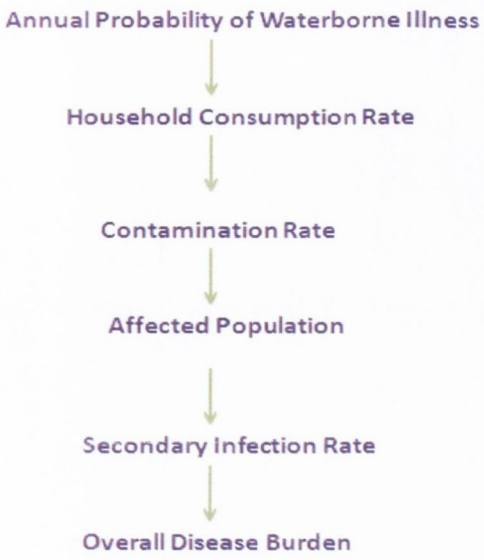


Figure 10.20 Flow Diagram of Risk Characterisation Model

Household (Non) Consumption

It was considered that not all consumers associated with rural households supplied by a private groundwater source would actively consume from these sources. In order to accurately estimate the proportion of "active" groundwater consumers, questions relating to same were included in the awareness survey. Overall, it was found that 89.3% of respondents (n = 483) consumed "raw water" directly from a household tap at least once per day, with the remaining 10.7% (N = 58) primarily utilising bottled water or water from other sources for consumption. As the overall sample size (n = 541) was considered appropriately large for determination at a 95% level of confidence, this finding (89.3%/10.7%) is adopted throughout the deterministic and stochastic QRA process. A chi-square test of independence found no significant association between gender and consumption source.

Spatial Contaminant Occurrence

In light of data obtained from the fieldwork programme, suggesting significantly higher levels of bacterial occurrence (independent of bacterial concentration) among hand-dug wells than bored wells (Chapter 7, Section 7.6.7), it was considered appropriate that source types remain separate throughout both the deterministic and stochastic risk assessment processes, as these well types were likely associated with differing levels of human health risk. Overall, 29.4% of "one-off" sampled wells (62/211) showed evidence of bacterial contamination (faecal coliforms) upon sample analysis, comprising 24.8% of bored sources (42/127) and 47.6% of hand dug sources (20/42). As the presence of a single faecal coliform in a drinking water supply is a breach of the Drinking Water Regulations (S.I. No. 278 of 2007) and constitutes a human health threat, these proportions were

employed for risk assessment. These findings are comparable with previous monitoring data in Ireland, with EPA (2010c) reporting that 34.8% of groundwater monitoring sites had faecal coliforms present during the period 2007-2009.

In order to allocate a likely dose-response value to the consumption of bacterially contaminated groundwater, median contamination rates were employed, as it was considered that the use of mean values would result in an overestimation of contamination magnitude due to a small number of outliers (>300 cfu/100ml). Therefore, from Chapter 7 (Section 7.6.7), median contamination rates of 3 and 5 cfu/100ml were calculated and used for bored SPS and hand dug SPS respectively.

Furthermore, as also outlined in Section 7.6.7, flow cytometric analysis resulted in 4.93% of groundwater samples (4/81) found to be positive for *Cryptosporidium* spp. Due to the low percentage, this figure is adopted for both bored and hand dug wells. These *Cryptosporidium* spp. contamination rates were generally consistent with earlier groundwater quality findings, for example, Hancock *et al.* (1998) reported *Cryptosporidium* spp. or *Giardia* detections in 5% of vertical wells in the US, with this result reported as being consistent with earlier findings from Hibler (1988) and Rose *et al.* (1991). Moreover, Smith *et al.* (1995) found a 7% rate of *Cryptosporidium* spp. presence in wells in Scotland, with Wallis *et al.* (1996) finding a 4.5% rate of *Cryptosporidium* presence in drinking water samples in Canada.

As sample volumes used for flow cytometry were small (500 ml) and oocyst numbers (where found) were very low, it was not possible to accurately enumerate the level of protozoan contamination, where present. Therefore, the rate of protozoan contamination is based upon a previous study. This study (The National *Cryptosporidium* Survey Group, 1992, cited in Lisle & Rose, 1995), found typical groundwater protozoan levels of 0.004-0.922 oocysts/L, in 138 samples from public and private supply wells in the UK.

Private Groundwater Reliant Population

From CSO census data (2007), the number of persons whose main household supply is a private domestic groundwater source is estimated as:

SPS – 145,341 (households supplied by private domestic source)

This figure was multiplied by 2.95 which is the average rural household size in Ireland, as all households supplied by a private groundwater source are assumed to be located in relatively rural areas. Therefore:

SPS – 428,756 (persons supplied by private domestic source)

The total number of persons supplied by private groundwater sources in Ireland (excluding PrGWS) is therefore approximately 428,756. As previously mentioned, those populations served by bored SPS and hand dug SPS are modelled separately, as hand-dug sources are typically more susceptible to contamination. As no data is available regarding their relative proportions, expert opinion has been sought (Misstear, pers. comm.; Ball, pers. comm; Hunter Williams, pers. comm.), leading to an assumption that no more than 10% of the Irish private groundwater population is currently supplied by hand dug wells, with this percentage likely to decrease over coming years. Therefore, for the purpose of risk assessment, it is assumed that 10% of the SPS population (14,500 households) are

supplied by a hand dug source (42,775 consumers), with the remaining 90% (130,800 households) supplied by a bored source (385,860 consumers).

However, as discussed in Chapter 8 (Sections 8.2.4.2 & 8.3.4.2), 10.7% of awareness questionnaire respondents indicated that they did not consume water from a private domestic groundwater source on a daily basis, preferring bottled water or other sources. Therefore, it is assumed that 10.7% of persons supplied by a private domestic groundwater source in Ireland do not directly consume from this source and are therefore not at risk of waterborne infection from this source. Accordingly, the "groundwater population associated with direct daily consumption" will comprise 89.3% of the total private groundwater population.

For this risk assessment exercise, it was necessary to take account of vulnerable sub-populations (Gerba *et al.,* 1996b). These include children (<5years), the elderly (>65years) and the immuno-compromised (transplant patients, AIDS/HIV patients, pregnant women). Due to a lack of data regarding the number and distribution of the immuno-compromised sub-population, it will be assumed that this sub-population is primarily located in urban areas supplied by treated public supplies and/or are not drinking untreated private groundwater. Therefore, the vulnerable sub-population mainly comprises children (<5years) and the elderly (>65years). Assuming these sub-populations are uniformly distributed throughout the country and independently distributed in regard to household water source, this population comprises 20.25% of the population, distributed as follows (CSO, 2007):

Infants (<1year) – 1.44%

Children (1-5years) – 8.5%

Elderly (>65years) – 10.31%

The vulnerable sub-population therefore comprises approximately 114,500 of the total private groundwater population (PrGWS + SPS) and 86,800 in the population supplied by private domestic groundwater sources (SPS).

Secondary Infection Rate

A secondary infection rate of 25% will be assumed for *Cryptosporidium* spp. (Macler & Merkle, 2000), while a 15% secondary infection rate will be applied for pathogenic *E. coli* (Snedeker *et al.*, 2009).

10.7 Human Health Risk Assessment

The following section will apply three existing deterministic models, which have been previously applied for quantitative microbial human health risk assessment from waterborne sources. These models were chosen based on a number of factors including data dependence, data availability, simplicity, and accuracy. Relevant Irish data, where available, are substituted into these models, resulting in deterministic morbidity and mortality estimates for the Irish population supplied by private groundwater sources. A stochastic risk assessment model has been developed by the researcher, based on considered strengths and weaknesses of the deterministic models and their outputs.

The deterministic models adopted were:

- Teunis *et al.*, 1997
- Macler & Merkle, 2000
- Howard & Pedley, 2006

The following data were applied for both deterministic and stochastic risk assessments.

Pathogen Mortality Rates

A pathogen mortality rate is the proportion of those persons showing symptoms of a waterborne pathogenic illness resulting in a fatality. As for illness/attack rates, analysis of clinical data were not considered under the remit of the current study, therefore, previously published data from developed countries were applied.

Published figures are often cited for the vulnerable population, which, as previously stated, accounts for approximately 20.25% of the current Irish population. As outlined by Rangal *et al.* (2005), based on *E. coli* O157:H7, a mortality rate of 0.5% is expected within these vulnerable populations. Additionally, previous studies have reported that approximately 2.1% of *E. coli* illnesses may contract haemolytic uremic syndrome (HUS), 3-5% of whom may result in a fatality (Rangel *et al.*, 2005; Boyce *et al.*, 1995).

In the case of *Cryptosporidium* spp., a number of previous studies have reported mortality rates of approximately 0.1% in elderly and infant populations, approximately 0.016% in "non-vulnerable" populations, and as much as 50% among immuno-suppressed patients (Haas *et al.*, 1999).

10.8 Deterministic Model 1: Teunis *et al.*, 1997

The Teunis *et al.* (1997) model was previously employed to assess the significance of protozoan parasites, namely *Cryptosporidium* and *Giardia*, to public health from drinking water sources. It applied an exponential dose-response model, calculating for probability of infection per consumer for a specified period, characterised as follows:

$$Dose = C \times {}^{1}/_{Rr} \times I \times 10^{-DR} \times V \qquad \textit{Eqn. 10.1}$$

and

$$P_i = 1 - e^{-r \times Dose} \qquad \textit{Eqn. 10.2}$$

where,

C = Contaminant concentration

Rr = Recovery rate

I = Pathogen infectivity

DR = Removal or inactivation efficiency of the treatment process, expressed as its decimal reduction factor (i.e. DR = 1 equates to a 90% reduction or inactivation efficiency)

V= Daily consumption rate

P_i = Probability of infection

r = Dose-response

E. coli

The deterministic QRA using the Teunis *et al. (1997)* model was carried out in four distinct modules, namely; Module 1 – Untreated, Bored SPS; Module 2 – Untreated, HD SPS; Module 3 – Treated, Bored SPS; Module 4 – Treated, HD SPS. Calculations pertaining to Module 1 are presented here, with calculations associated with Modules 2-4 presented in Appendix H3. All deterministic model 1 input data are summarised in Appendix H4.

From Eqn. 10.1:

$$Dose = 3\frac{cfu}{100ml} \times \frac{1}{1} \times 0.5 \times 10^0 \times 9.26$$

Therefore, a mean daily dose of 13.9 cfu/day *E. coli* is associated with consumers supplied by contaminated untreated private boreholes. Using a linear approximation of $1x10^{-3}$ per organism (O157:H7) from Howard & Pedley, (2006; based on *Shigella*, From Rose & Gerba, 1991; Haas *et al.*, 1999);

From Eqn. 10.2

$$P_i(day^{-1}) = 1 - e^{-1x10^{-3}x\ 13.9} = 1.38 \times 10^{-2}$$

Therefore, a daily probability of infection of 1.38×10^{-2} is associated with consumers supplied by contaminated untreated private boreholes.

Frequency of contamination

From Chapter 9 (*Tables 7.33, 7.37 & 7.40*), it was found that bored sources had a mean bacterial presence frequency of 30% (3.6/12 months), while this figure was 41% (4.9/12 months) in the case of hand dug sources. Therefore, if it is assumed that each contamination event has a 5-day duration, once per month (Artz & Killham, 2002), the annual probability of infection ($P_{i(annual)}$) associated with Model 1 sources were calculated as follows:

$$(\frac{3.6}{12} \times 5)1.38 \times 10^{-2} = 0.0207$$

From Chapter 8 (Section 8.2.1); it was found that approximately 11.6% of surveyed HD SPS respondents indicated that a treatment system was in place, compared with 41.3% of Bored SPS respondents. However, as 58% of these systems were softener systems (Section 10.4.1), which are not considered to have any treatment efficacy with regard to pathogenic organisms (Parsons, 2000), the adjusted pathogen treatment figures are therefore 4.9% and 17.3% for hand dug and bored sources respectively. The calculated numbers of expected yearly infections attributable to *E.coli* contamination of private untreated boreholes (Module 1) in Ireland are calculated as:

Affected Population (Pp) x Contamination Rate x E. coli pathogenicity

x Annual Prob Infection x Secondary Infection Rate *Eqn. 10.3*

Therefore, from Eqn. 10.3:

$$[130{,}800 \times 2.95 \times 0.827 \times 0.893] \times 0.248 \times 0.08 \times 0.0207 \times 1.15 = 135\ infections/year$$

(CIR; 34.9/100,000 private groundwater consumers)

A summary of the outputs of deterministic model 1 (*E. coli*) are presented in *Table 10.7.*

Table 10.7 Predicted Illnesses and mortalities per annum attributable to E. coli from Private Groundwater Sources in Ireland using deterministic model 1 (*including Haemolytic Uremic Syndrome - HUS)

Module	Source Type	Illnesses	Illness CIR	HUS Cases	Deaths*
1	Bored; Untreated	135	35	2.8	
2	Hand-dug; Untreated	72	168	3.5	
3	Bored; Treated	2.6	3.8	0	
4	Hand-dug; Treated	0.4	17.6	0	
Total		210	49	4.4	0.6-0.7

***Cryptosporidium* spp.**

The Teunis *et al.* (1997) approach was also employed to approximate the levels of morbidity and mortality attributable to *Cryptosporidium* in private groundwater sources in Ireland. Due to lower levels of available data, this was undertaken by way of two QRA modules; namely; Module 1 – Untreated private groundwater supplies, and; Module 2 – Treated private groundwater supplies. For the purposes of this exercise, it was assumed that, where *Cryptosporidium* were present (4.93% of private groundwater sources; Chapter 7, Section 7.6.7), the oocyst concentration was 0.922 oocyst/L (Rose, 1997). Furthermore, similar decontamination decimal reduction rates as those applied to quantify *E. coli* infection have been adopted for assessment of *Cryptosporidium* related illness. Calculations associated with module 2 are presented in Appendix H5, while model inputs pertaining to both modules are presented in Appendix H6.

From Eqn. 10.1:

$$Dose = 0.922 \times {}^{1}/_{0.7} \times 1 \times 10^{0} \times 0.926$$

Dose = 1.22 oocysts/person/day (where contamination has occurred)

From WHO (2004), a low-dose approximation of 4.0 x10^{-3}/oocyst was assumed i.e. *Cryptosporidium* dose-response is approximately linear at low doses, due to exponential dose-response curve. Therefore, the daily probability of illness, where *Cryptosporidium* are present, was calculated from Eqn. 2, as follows:

$$P_i(day^{-1}) = 1 - e^{-4x10^{-3}\,x\,1.22} = 4.87\,x\,10^{-3}$$

Assuming a 30-day period of contamination (Taylor *et al.*, 2004), once per year (Hancock et al., 1998; 2000):

$$P_{i(annual)} = 1.21 \times 10^{-3} \times 30 = 0.146$$

The calculated numbers of expected yearly infections attributable to *Cryptosporidium* spp. contamination of private untreated wells (Module 1) in Ireland are calculated as (From Eqn. 10.3):

$$[145{,}300 \times 2.95 \times 0.853 \times 0.893] \times 0.049 \times 0.146 \times 1.25 = 2{,}919\ infections/year$$

(CIR; 196.6/100,000)

A summary of the outputs of deterministic model 1 (*Cryptosporidium* spp.) are presented in *Table 10.8.*

Table 10.8. Predicted Illnesses and mortalities per annum attributable to Cryptosporidium spp. from Private Groundwater Sources in Ireland using deterministic model 1

Module	Source Type	Illnesses	Illness CIR	Deaths
1	Untreated	2919	798	
2	Treated	49	78	
Total		2968	692	0.6

10.9 Deterministic Model 2: Macler & Merkle, 2000

As outlined in *Section 5.15*, the Macler & Merkle deterministic model was developed to estimate the likely ranges of disease attributable to waterborne rotavirus from groundwater-based community water systems and all public supplies using groundwater in the US.

A simple model is used, as follows:

$$P_{gw} \times R_{contam} \times C_{path} \times R_{inf} \times R_{recov} \times r \times R_{secinf} = WI \qquad \textit{Eqn. 10.4}$$

where,

P_{gw}= Groundwater population

R_{contam}= Contamination rate

C_{path}= Pathogen contribution

R_{inf}= Infection rate

R_{recov} = Recovery rate

r = Dose response

R_{secinf} = Secondary infection rate

WI = Number of waterborne infections

E. coli

The deterministic QRA using the Macler & Merkle model (2000) was carried out in two distinct modules, namely; Module 1 – Bored SPS and Module 2 – HD SPS. Calculations associated with module 2 are presented in Appendix H7, while model inputs pertaining to both modules are presented in Appendix H8.

From Eqn. 10.4:

$$[130{,}800 \times 2.95 \times 0.893] \times 0.248 \times 0.5 \times 1 \times [0.017] \times 1.15 = 835\ infections/year$$

Vulnerable population (<5 &>65 years): 0.2025 x 835 = 169 illnesses in vulnerable population

$$169 \times 0.005 = 0.84\ deaths/annum$$

"Non-vulnerable" population: 0.7975 x 835 = 666 illnesses in non-vulnerable population

$$666 \times 0.0016 = 1.06\ deaths/annum$$

A summary of the outputs of deterministic model 2 (*E. coli*) are presented in *Table 10.9.*

Table 10.9 Predicted Illnesses and mortalities per annum attributable to E. coli from Private Groundwater Sources in Ireland using deterministic model 2 (*including HUS)

Module	Source Type	Illnesses	Illness CIR	HUS Cases	Deaths*
1	*Bored*	*835*	*216*	*17-18 (17.5)*	
2	*Hand-dug*	*293*	*682*	*6-7 (6.1)*	
Total		*1,128*	*263*	*23-25*	*3.3-3.7*

***Cryptosporidium* spp.**

As for *E. coli*, the deterministic QRA using the Macler & Merkle model (2000) was carried out in two distinct modules, namely; Module 1 – Bored SPS and Module 2 – HD SPS. Calculations associated with module 2 are contained in Appendix H9, with model inputs pertaining to both modules presented in Appendix H10.

From Eqn. 10.4:

$$[130{,}800 \times 2.95 \times 0.893] \times [0.0493] \times 0.508 \times \frac{1}{0.7} \times [0.028] \times 1.25 = 431\ infections/year$$

Vulnerable population (<5 &>65 years): 0.2025 x 431 = 87 illnesses in vulnerable population

$$87 \times 0.001 = 0.087\ deaths/annum$$

A summary of the outputs of deterministic model 1 (*Cryptosporidium spp.*) are presented in *Table 10.10.*

Table 10.10. Predicted Illnesses and mortalities per annum attributable to Cryptosporidium spp. from Private Groundwater Sources in Ireland using deterministic model 2

Module	Source Type	Illnesses	Illness CIR	Deaths
1	Bored	431	111	
2	Hand-dug	48	111	
Total		479	111	0.1

10.10 Deterministic Model 3: Howard & Pedley, 2006

This model was employed by Howard & Pedley (2006) to assess the disease burdens associated with a number of pathogens, namely *Cryptosporidium parvum*, *E. coli* O157:H7 and rotavirus, from the piped water supply in Kampala, Uganda. It is based on a methodology set out in *Quantifying Public Health Risk in the WHO Guidelines for Drinking-water quality* (WHO, 2004), with the assessment undertaken by Howard & Pedley (2006) primarily using faecal indicator data as opposed to direct pathogen data. The methodology is based upon a number of calculated terms, as set out below:

$$CD = CR \times (1 - PT) \qquad \textit{Eqn. 10.5}$$

where,

CD = Drinking water quality

CR = Average raw water quality

PT = Treatment effect

$$E = CD \times V \qquad \textit{Eqn. 10.6}$$

where,

E = Exposure by drinking water, organisms per litre

V = Consumption of drinking water

$$P_{inf,d} = E \times r \qquad \textit{Eqn. 10.7}$$

where,

$P_{inf,d}$ = Risk of infection per day

r = Dose-response

$$P_{inf,y} = P_{inf,d} \times 365 \qquad \textit{Eqn. 10.8}$$

where,

$P_{inf,y}$ = Risk of infection per year

$$P_{ill} = P_{inf,y} \times P_{ill/inf} \qquad \textit{Eqn. 10.9}$$

where,

P_{ill} = Risk of diarrhoeal disease

$P_{ill/inf}$ = Risk of diarrhoeal disease given infection

E. coli

As deterministic model 3 includes a treatment efficacy parameter, deterministic calculations were carried out on four distinct modules, similar to the approach taken for deterministic model 1. Calculations associated with modules 2-4 are contained in Appendix H11, while model inputs pertaining to all four modules are presented in Appendix H12.

Step 1: CD (from Eqn. 10.5)

$$CD_{Bored,untreated} = 30\frac{cfu}{L} \times (1 - 0) = 30cfu/L$$

Step 2: E (from Eqn. 10.6)

$$E_{Bored,untreated} = 30\frac{cfu}{L} \times 0.926 = 27.8\, cfu/day$$

Step 3:$P_{inf,d}$ (from Eqn. 10.7)

(using linear approximation of 1x10^{-3}per organism (O157:H7) from Howard & Pedley, 2006 based on *Shigella*, From Rose & Gerba, 1991; Haas *et al.*, 1999)

$$P_{inf,d;Boreduntreated} = 27.8\frac{cfu}{day} \times 0.08 \times 1x10^{-3} = 0.0022/day$$

Step 4: $P_{inf,y}$ (from Eqn. 10.8)

(using 30% and 41% seasonal contamination rates and 5-day assumption as adopted for deterministic model 1)

$$P_{inf,y;Boreduntreated} = 0.0022 \times \left[\frac{3.6}{12} \times 5\right] = 0.0033/year$$

Step 5:P_{ill} (from Eqn. 10.9)

(Where $P_{ill/inf} = 0.25$ (*From* Haas *et al.*, 1999; Morbidity data from *Shigella*)

$$P_{ill,Boreduntreated} = 0.0033 \times 0.25 = 8.25x10^{-4}/year$$

Risk Characterisation (from Eqn. 10.3)

$$[130{,}800 \times 2.95 \times 0.827 \times 0.893] \times 0.248 \times 0.000825 \times 1.15 = 67\ infections/year$$

(CIR = 21/100,000)

Vulnerable population (<5 &>65 years): 20.25% - 14 illnesses in vulnerable population

$$14 \times 0.005 = 0.06\ deaths/annum$$

"Non-vulnerable" population: 79.75% - 53 illnesses in non-vulnerable population

$$648 \times 0.0016 = 0.085\ deaths/annum$$

HUS Rate (2.1/100 illnesses; Rangal *et al.*, 2005) – 1.4 HUS cases /annum, leading to 3-5% mortality rate (Siegler, 1997) – 0.04-0.07/annum

A summary of the outputs of deterministic model 3 (*E. coli*) are presented in *Table 10.11.*

Table 10.11. Predicted Illnesses and mortalities per annum attributable to E. coli from Private Groundwater Sources in Ireland using deterministic model 3 (*including HUS)

Module	Source Type	Illnesses	Illness CIR	HUS Cases	Deaths*
1	Bored; Untreated	67	21	14	
2	Hand-dug; Untreated	37.4	91.9	0.8	
3	Bored; Treated	1.3	2	0	
4	Hand-dug; Treated	0.2	9.5	0	
Total		106	24.7	15	0.95

***Cryptosporidium* spp.**

Calculations associated with modules 2-4 are contained in Appendix H13, while model inputs pertaining to all four modules are presented in Appendix H14.

Step 1: CD (from Eqn 10.5)

$$CD_{Bored,untreated} = 0.922 \frac{oocyst}{L} \times (1 - 0) = 0.922\ oocyst/L$$

Step 2: E (from Eqn. 10.6)

$$E_{Bored,untreated} = 0.922 \frac{oocyst}{L} \times 0.926 = 0.85\ oocyst/day$$

Step 3:$P_{inf,d}$ (from Eqn. 10.7)

(using dose-response point estimate of 4×10^{-3} per organism from WHO (2004))

$$P_{inf,d;Boreduntreated} = 0.85 \frac{oocyst}{day} \times [4 \times 10^{-3}] = 0.0034$$

Step 4: $P_{inf,y}$ (from Eqn. 10.8)

(Assuming a 30-day period of contamination (Taylor *et al.*, 2004), once per year (Hancock et al., 1998; 2000))

$$P_{inf,y;Boreduntreated} = 0.0034 \times 30 = 0.102$$

Step 5:P_{ill} (from Eqn. 10.9)

Where $P_{ill/inf} = 0.30$ (From Havelaar & Melse, 2003)

$$P_{ill,Boreduntreated} = 0.102 \times 0.3 = 0.03$$

Risk Characterisation (from Eqn. 10.3)

Bored; Untreated

$$[130{,}800 \times 2.95 \times 0.827 \times 0.893] \times 0.049 \times 0.03 \times 1.25 = 523\ infections/year$$

(Crude Incidence Rate; 163.9)

Vulnerable population (<5 &>65 years): 20.25% - 106 illnesses in vulnerable population

$$106 \times 0.001 = 0.1\ mortalities/annum$$

A summary of the outputs of deterministic model 3 (*E. coli*) are presented in *Table 10.12.*

Table 10.12. Predicted Illnesses and mortalities per annum attributable to Cryptosporidium spp. from Private Groundwater Sources in Ireland using deterministic model 3

Module	Source Type	Illnesses	Illness CIR	Deaths
1	Bored; Untreated	523	163.9	
2	Hand-dug; Untreated	66	162.2	
3	Bored; Treated	10-11 (10.5)	2.6	
4	Hand-dug; Treated	0-1 (0.3)	0.7	
Total		600	139.9	0.12

A summary of the outputs from all three deterministic models is presented in *Tables 10.13 & 10.14.*

*Table 10.13.Predicted Illnesses and mortalities per annum attributable to E. coli from Private Groundwater Sources in Ireland (*including HUS)*

Model	Source Type	Illnesses	Illness CIR	HUS Cases	Deaths*
1	Bored	138	35.7	2.9	
	HD	72	168.3	1.5	
	Total	210	49	3.4	0.6-0.7
2	Bored	835	216	18	
	HD	293	682	7	
	Total	1128	263	25	3.3-3.7
3	Bored	68	17.6	1.4	
	HD	38	88.5	0.8	
	Total	106	24.7	2.2	0.95

Table 10.14. Predicted Illnesses and mortalities per annum attributable to Cryptosporidium spp. from Private Groundwater Sources in Ireland

Model	Source Type	Illnesses	Illness CIR	Deaths
1	Untreated	2,919	798	
	Treated	49	78	
	Total	2,968	692	0.6
2	Bored	431	111	
	HD	48	111	
	Total	479	111	0.1
3	Bored	534	138	
	HD	66	154	
	Total	600	140	0.12

As shown in *Table 10.13*, deterministic models 1 and 3 resulted in generally comparable results with regard to the estimation of annual disease burden attributable to pathogenic *E. coli* within the private groundwater population of Ireland (CIR 25-50/100,000). However, deterministic model 2 yielded estimates approximately 5 and 10 times those of models 1 and 3, respectively. The estimated yearly burden of illness was in the range 106-1128 (CIR 25-263/100,000), which is 0.44-4.7 times the number of reported VTEC cases in 2009 (approximately 240 reported VTEC infections in Ireland in 2009). To some extent, this may be due to both under- and over-estimation by the models; particularly with respect to models 1 and 3, due to the necessary assumptions with respect to contamination frequency. There is a likely higher CIR among the private groundwater population than within the overall national population due to increased exposure (Garvey & McKeown, 2011), which is considered to be reflected by model 2 estimates. Additionally, there is likely a high degree of under-reporting of VTEC/gastrointestinal illness among the population, particularly among the adult population, as illnesses are typically self-limiting and of relatively short duration.

With regard to cryptosporidial illness within the private groundwater population, as shown in Table 10.14, the three deterministic models results were not generally comparable, with crude incidence rate (CIR) predictions ranging from 111.7/100,000 to 692.4/100,000 i.e. 0.11-0.69%.During the period 2004-2009, the national cryptosporidiosis CIR has ranged from approximately 9 - 14/100,000 (Figure 10.6). Therefore, taking into account the likely high level of underreporting and higher levels of pathogen exposure among private groundwater users, these estimates equate to 7.9 – 49.4 times national figures, respectively. As deterministic models 2 and 3 resulted in comparable predicted outputs, it is likely that these are more representative of the current public health burden posed by *Cryptosporidium* spp. through consumption of contaminated private groundwater sources in Ireland.

10.11 Deterministic Model Assumptions and Limitations

As previously discussed (Chapter 5, Section 5.9.2), a significant limitation of all deterministic models, is that if a deterministic approach is being adopted, it is necessary to model the risk at a discrete contaminant concentration i.e. for deterministic model 1, the researcher had to model health risk from waterborne *E. coli* at 3 and 5 cfu/100ml concentration for bored and hand-dug sources, respectively. However, it is improbable that *E. coli* are present in the majority of contamination events at this concentration. Additionally, the use of a deterministic water consumption rate does not account for the high degree of variability and/or uncertainty associated with this parameter, and therefore does not account for the overall consumption distribution. Consequently, the low-dose approximations applied above may not be relevant for all consumers' i.e. high levels of contamination in concurrence with high levels of consumption. Moreover, it is considered that the necessary assumption made with respect to *E. coli* frequency in models 1 and 3 is a significant source of uncertainty. Therefore, a stochastic model has been employed to quantify likely human health burden associated with both pathogenic *E. coli* and *Cryptosporidium* spp. while accounting for both uncertainty and variability in input parameters, particularly contamination rates and frequency, consumption rates and dose-response rates.

Furthermore, the three deterministic models used all have inherent limitations specific to the models themselves. These are outlined below:

Model 1 (Teunis *et al.*, 1997):

- Secondary infection not considered
- Pathogenic contribution not considered
- Contamination frequency not considered

Model 2 (Macler & Merkle, 2000):

- Treatment effect not considered
- Contamination frequency not considered

Model 3 (Howard & Pedley, 2006):

- Secondary infection not considered
- Constant contamination assumed
- Pathogen recovery rate not considered

In order to address these limitations and assumptions associated with the three deterministic models, the following 3-step stochastic model was developed, based upon an amalgamation of all three deterministic models:

Step 1 Exposure Assessment

$$E = C \times {}^{1}/_{R} \times PI \times [1 - PT] \times V \qquad \textit{Eqn. 10.10}$$

where,

E = Daily Exposure per person

C = Pathogen Concentration (Stochastic Input)

R = Recovery (Deterministic Input)

PI = Pathogen Contribution (Deterministic Input)

PT = Treatment Efficacy (Deterministic Input)

V = Volume (Stochastic Input)

Step 2 Hazard Characterisation

$$P_{inf,year} = E \times DR \times AR \times Inf;ill \times Freq \qquad \textit{Eqn. 10.11}$$

where,

E = Daily exposure per person (Stochastic Input from Step 1)

DR = Dose:Response (Stochastic Input)

AR = Infection Rate (Stochastic/Deterministic Input)

Inf;ill = Infection:Illness Ratio

Freq = Contamination Frequency (Stochastic/Deterministic Input)

Step 3 Risk Characterisation

$$P_{ill,year} \times APop \times [1 - NonCon] \times CR \times SI \qquad \textit{Eqn. 10.12}$$

where,

APop = Affected population (Deterministic Input)

Non Con = Level of non-consumption (Deterministic Input)

CR = Contamination Rate (Deterministic Input)

Pill,year = from above (Stochastic Input from Step 2)

SI = Secondary Infection (Deterministic Input)

10.12 Stochastic Risk Assessment Modelling

All stochastic risk simulations were created and run in Excel with the add-on package @Risk 5.5™ (Pallisade Corporation, New York, USA). All models were based on five concurrent simulations comprising 10,000 iterations/recalculations, using Monte Carlo simulation sampling type and a Mersenne Twister MT19917 pseudo-random seed generator. Due to the "randomness" of Monte Carlo sampling, under- and over-sampling may occur from various parts of variable distributions (Vose, 2000), therefore a larger number of iterations were favoured, to accurately replicate the input distribution shapes.

10.12.1(a) Stochastic Exposure Assessment – E. coli

As shown in Eqn. 10.10, the developed exposure assessment model comprised five input parameters, two of which were expressed stochastically, namely pathogen concentration and consumption volume and were derived from fieldwork data, as were microbial recovery rates. The daily *E. coli* exposure assessment was undertaken using four modules, as follows:

1. Untreated, bored private sources (Module 1)
2. Untreated, hand-dug private sources (Module 2)
3. Treated, bored private sources (Module 3)
4. Treated, hand-dug private sources (Module 4)

The likely daily human exposure distribution pertaining to private bored wells contaminated with *E. coli* is shown in *Figure 10.21*. Overall, it was found that 95% of well users drinking from contaminated wells were consuming 1-139 cfu/day, with a mean daily consumption of 39.95 cfu/day and a standard deviation of 54.17 cfu/day. The maximum rate of ingestion was 1,078 cfu/day, found at very high contamination loadings in concurrence with very high consumption rates.

The modelled daily human exposure distribution associated with private hand-dug wells contaminated with *E. coli* is shown in *Figure 10.22*. As shown, 95% of those drinking from contaminated hand-dug wells were found to ingest 1-145 cfu/day, with a mean daily ingestion of 41.5 cfu/day and a standard deviation of 57.2 cfu/day. A maximum ingestion rate of 1,364 cfu/day was also found.

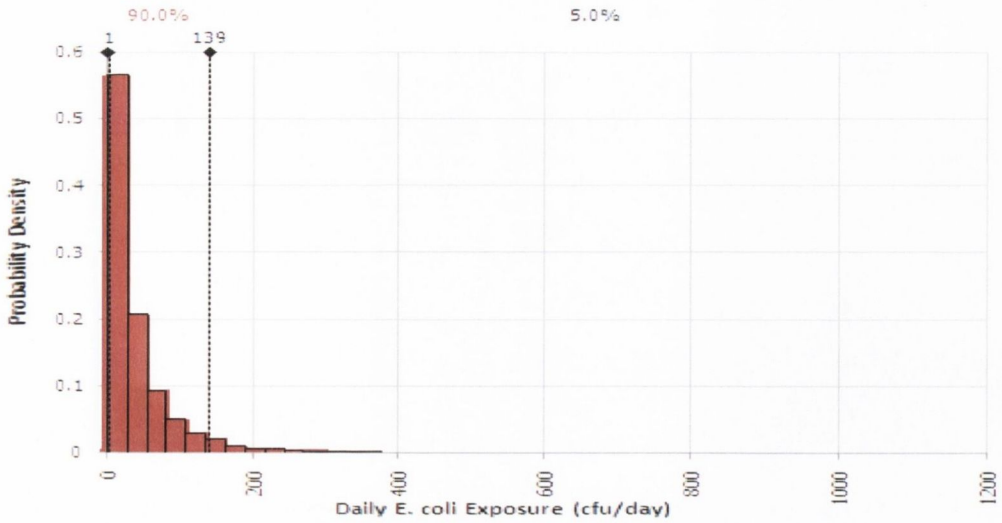


Figure 10.21 Probability density distribution for rate of E. coli exposure per day at contaminated private boreholes in Ireland

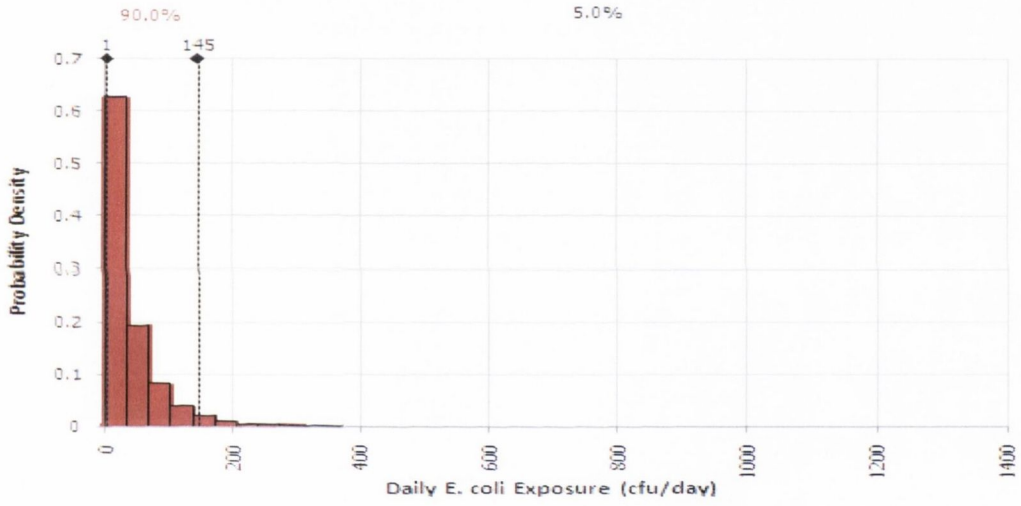


Figure 10.22 Probability density distribution for rate of E. coli exposure per day at contaminated hand-dug wells

Daily *E. coli* exposure with regard to treated wells, including bored and hand-dug sources, were simulated and modelled using the same exposure assessment methodology and simulation approach, with a decimal reduction factor. Therefore, projected daily exposure patterns followed similar distributions, albeit 90% reduced i.e. mean daily *E. coli* exposure predicted for contaminated, but treated bored wells was 3.9 cfu/day.

10.12.1(b) Stochastic Exposure Assessment – Cryptosporidium spp.

The same exposure assessment methodology was adopted for *Cryptosporidium* spp., with the same four modules also utilised. The modelled daily oocyst exposure for untreated bored and hand-dug wells is presented in *Figure 10.23*. As shown, 95% of consumers drinking from these source types may be expected to ingest 0.05-1.7 oocysts/day, with an approximate mean ingestion of 0.61

oocysts/day and an associated standard deviation of 0.57 oocysts/day. As for *E. coli*, the presence of an effective treatment system was assumed to decrease daily exposure by 90%. Therefore, as shown in *Figure 10.24*, the expected mean daily oocyst exposure associated with treated private groundwater supplies was 0.06 oocyst/day, with a 90% range of 0.005-0.173 oocyst/day.

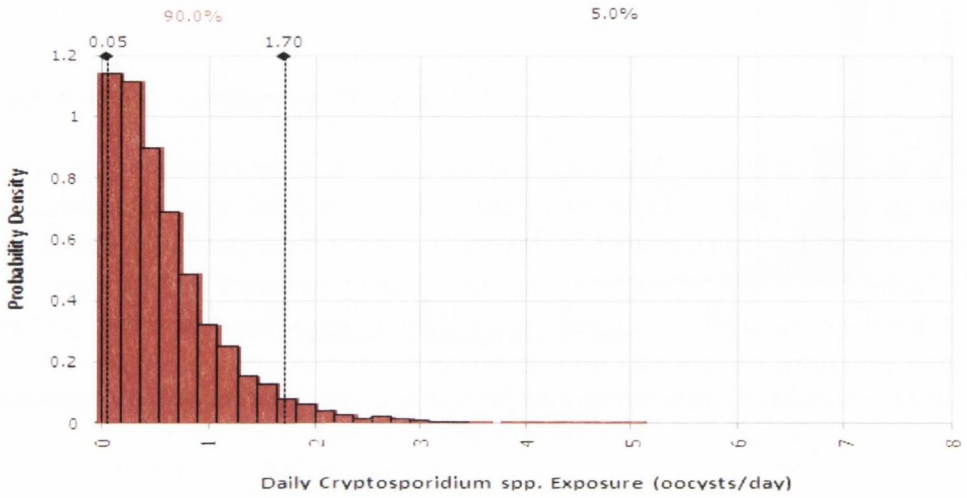


Figure 10.23 Probability density distribution for rate of Cryptosporidium spp. exposure per day at contaminated untreated private groundwater sources

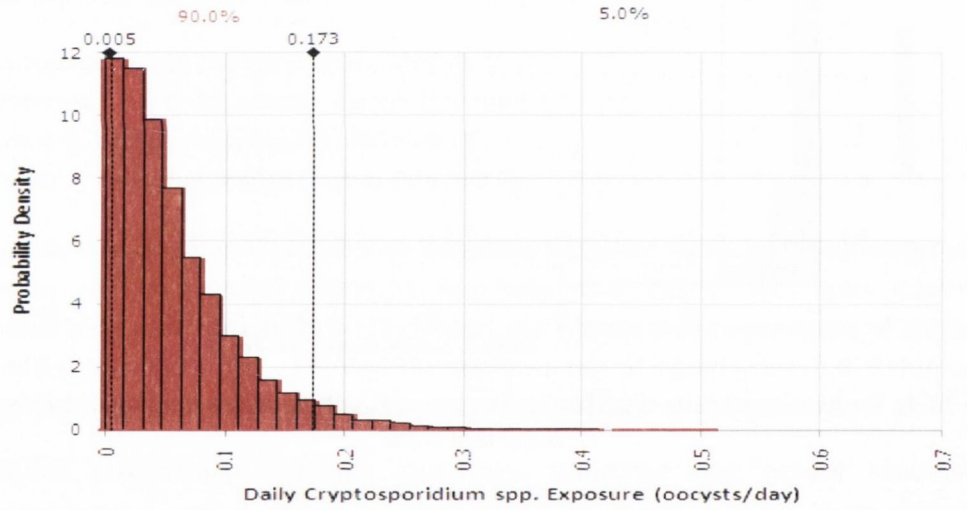


Figure 10.24 Probability density distribution of rate of Cryptosporidium spp. exposure per day at contaminated private groundwater sources with a treatment system in place

The sensitivity of stochastic model inputs to model prediction was modelled by rank order correlation sensitivity analysis. The results are presented in *Figures 10.25 & 10.26*. As shown in *Figure 10.27*, the overall *E. coli* loading rate was the parameter having the most impact on predicted *E. coli* exposure (correlation coefficient 0.73), while consumption rate had a significant albeit less significant impact (correlation coefficient 0.497).

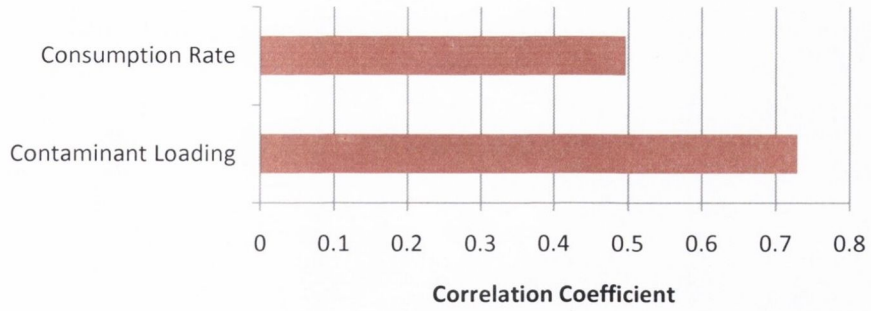


Figure 10.25 Sensitivity analysis between E. coli daily exposure and stochastic model inputs

In regard to *Cryptosporidium* spp. exposure, a similar pattern was evident, with oocyst loading (correlation coefficient 0.746) adjudged to be the parameter of most direct importance on overall daily exposure (*Figure 10.26*), however, consumption rate had a higher correlation with daily exposure than that found in the case of *E. coli* exposure (correlation coefficient 0.603).

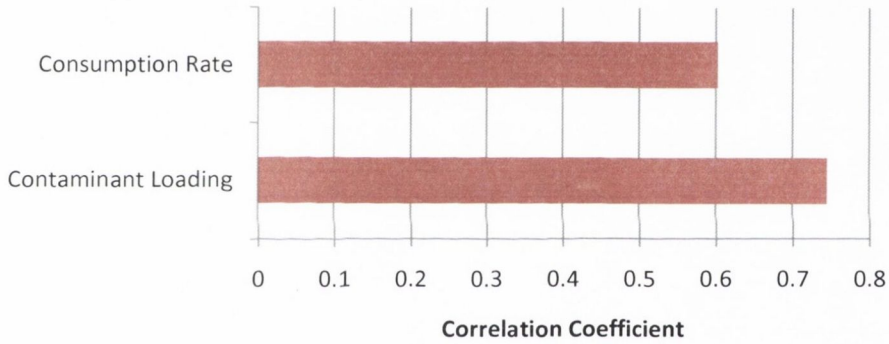


Figure 10.26 Sensitivity analysis between Cryptosporidium spp. daily exposure and stochastic model inputs

10.12.2 Stochastic Hazard Characterisation

As shown in Eqn. 10.13, the developed hazard characterisation model comprises five input parameters. The first input parameter (E) is the previously presented exposure assessment output. The remaining model inputs, two of which are stochastic ($P_{inf:ill}$; *Freq)*, have been previously outlined (Section 10.11). Following on from the exposure assessment, hazard characterisation was undertaken in four modules for both *E. coli* and *Cryptosporidium* spp.

As previously outlined (Section 5.5), a number of dose-response relationships have been proposed for both pathogenic *E. coli* and *Cryptosporidium* spp. In the case of *E. coli*, Powell *et al.* (2000) have proposed using both *Shigella dysenteriae* and EPEC dose-response curves for QRA, while WHO (2004) and Howard & Pedley (2006) have both employed a low-dose deterministic dose-response relationship. With regard to *Cryptosporidium* spp., Barbeau *et al.* (2000) have previously applied an

exponential dose-response association, while WHO (2004) and Howard & Pedley (2006) have again adopted a deterministic low-dose relationship. In the following section, the various dose-response relationships have been utilised and compared for adequacy, leading to probabilistic hazard characterisation estimates.

10.12.2(a) E. coli

Table 10.15 below outlines the mean hazard characterisation (probability of *E. coli* infection per annum) across the four modules using the three previously applied dose-response relationships. These are based on five simulations, comprising 10,000 recalculations each.

Table 10.15 Predicted annual probability of E. coli infection using three dose-response models

	***S. dysenteriae* (Prob)**	**EPEC (Prob)**	**WHO (Det)**
Module 1	0.2826	1.037×10^{-6}	0.0145
Module 2	0.2976	1.075×10^{-6}	0.0151
Module 3	0.0265	1.039×10^{-7}	0.00145
Module 4	0.0298	1.091×10^{-7}	0.00154

As previously outlined by Powell *et al.* (2000) and Duffy *et al.* (2006), EPEC dose-response curves are adopted to represent the lower bound of an *E. coli* O157:H7 dose-response function, based on the assumption that *E. coli* O157:H7 is likely to be more pathogenic than EPEC, while the S. Dysenteriae dose-response relationship may represent the upper bound of the *E. coli* O157:H7 dose-response function, as *S. dysenteriae* is unlikely to be less pathogenic than *E. coli* O157:H7. Therefore, where either one of these dose-response functions is applied in isolation, a large level of under-estimation or over-estimation of the response is probable. As shown in *Table 10.10*, this has occurred, with the use of the EPEC dose-response function resulting in extremely low daily likelihood of infection (i.e. 1 chance in a million), while the use of the *S. dysenteriae* dose-response function results in extremely high response rates (i.e. 28-29% daily probability of infection). In this case, the results from the deterministic low-dose approximation appear to be the most realistic (i.e. 1.4-1.5% daily probability of infection), and may represent the dose-response function with the greatest applicability i.e. low-dose approximation for all pathogenic *E. coli*.

When developing a probability distribution function for *E. coli* loading (where present), it was necessary to use to overall contaminant mean, however, due to a number of outlying *E. coli* loading figures, the resulting means (53.9 cfu/100ml in bored sources; 56.1 cfu/100ml in hand-dug sources) were considered to be inflated, likely leading to inflated risk estimates. Exclusion of these outliers was considered, but rejected as it was felt they were important to include within the analysis, an approach generally recommended within water resources research (Helsel & Hirsch, 2002). Conversely, median loading figures (3 cfu/100ml in bored sources; 5 cfu/100ml in hand-dug sources) were considered unrealistically low. Therefore, a mean/median conversion factor was introduced in order to counteract inflation. These conversion factors were calculated as follows:

Bored sources:

$$Mean\left(53.9\frac{cfu}{100ml}\right) \div Median\left(3\frac{cfu}{100ml}\right) = 17.967$$

Hand-dug sources:

$$Mean\left(56.1\frac{cfu}{100ml}\right) \div Median\left(5\frac{cfu}{100ml}\right) = 11.228$$

Therefore, the final risk characterisation model was amended as follows:

$$P_{inf,year} = \frac{(E \times DR \times AR \times Inf;ill \times Freq)}{Conversion\ Factor} \qquad Eqn.\ 10.13$$

A series of Monte Carlo simulations (5 x 10,000) were subsequently run, with results for untreated bored and hand-dug sources presented in *Figures 10.27 & 10.28*.

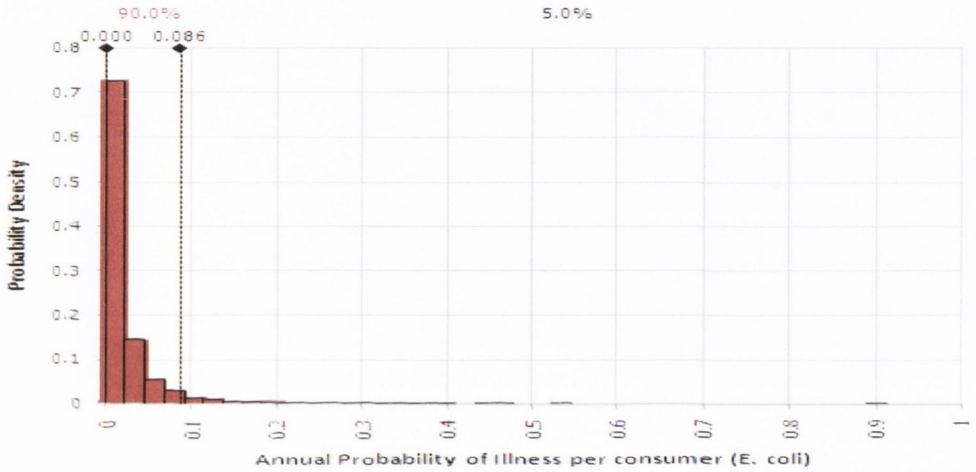


Figure 10.27 Probability density distribution for annual probability of E. coli illness among users of contaminated private boreholes

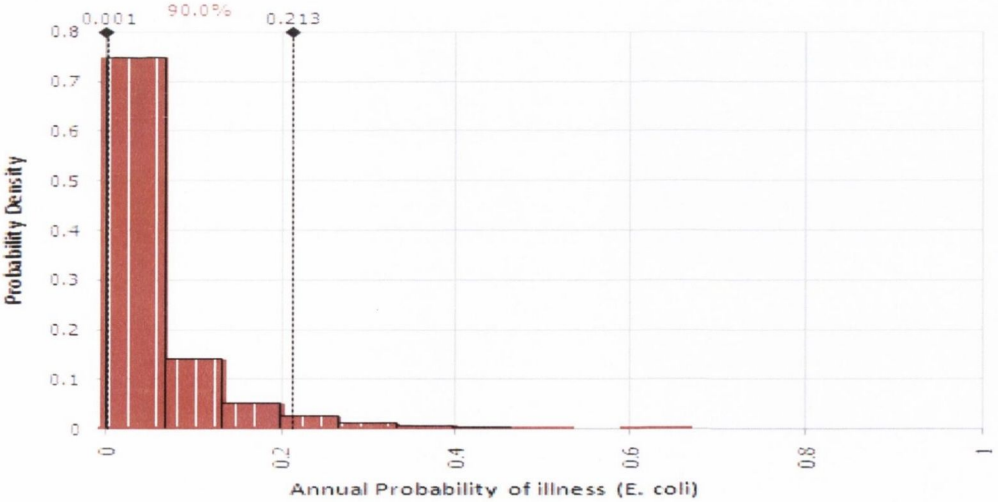


Figure 10.28 Probability density distribution for annual probability of E. coli illness among users of contaminated private hand-dug wells

The mean annual probability of illness associated with untreated bored wells was found to be 0.0226 or 2.26% per annum, with 90% of consumers within the range 0 – 0.086. The mean annual probability of illness associated with untreated hand-dug wells was 0.0581 or 5.81%, with a 90% range of 0.001-0.213. The mean annual probability of infection associated with treated bored and hand-dug well users were $2.2x10^{-3}$ and $5.8x10^{-3}$ respectively.

As shown in *Table 10.15*, the daily probability of infection associated with untreated bored and hand-dug wells were very similar (0.014 and 0.015 respectively), however, as shown in *Figures 10.27 & 10.28*, the predicted annual rate of illness associated with hand-dug wells over twice that of bored well users, mainly due to a higher frequency attributable to hand-dug wells and a higher median *E. coli* loading rate (i.e. low mean:median conversion factor).

10.12.2(b) Cryptosporidium spp.

Table 10.16 below outlines the mean hazard characterisation (Probability of cryptosporidial infection per annum) across the four modules using the two previously applied dose-response relationships. These are based on five simulations, comprising 10,000 recalculations each.

Table 10.16 Predicted annual probability of Cryptosporidium spp. infection using two dose-response models

	***Barbeau et al* (Prob)**	**WHO (Det)**
Module 1	0.00132	0.00123
Module 2	0.00126	0.00122
Module 3	0.000128	0.000124
Module 4	0.000127	0000124

As shown, the exponential dose-response relationship (Barbeau *et al.*, 2000) and low-dose approximation (WHO, 2004) resulted in very similar annual risk estimates per consumer. To an extent this result was expected, as *Cryptosporidium* spp., where detected, were present in relatively low concentrations. In order that all potential uncertainty and variability be accounted for in the final risk characterisation, the deterministic input parameter will therefore be employed.

As shown in *Figure 10.29*, 95% of private groundwater consumers, consuming from untreated supplies (both bored and hand-dug sources) have a mean yearly probability of waterborne cryptosporidial illness from 0.0002-0.0049 (i.e. 0.02%-0.49%). The annual risk of illness is decreased by approximately 90% in cases where a private groundwater source is treated (*Figure 10.30*).

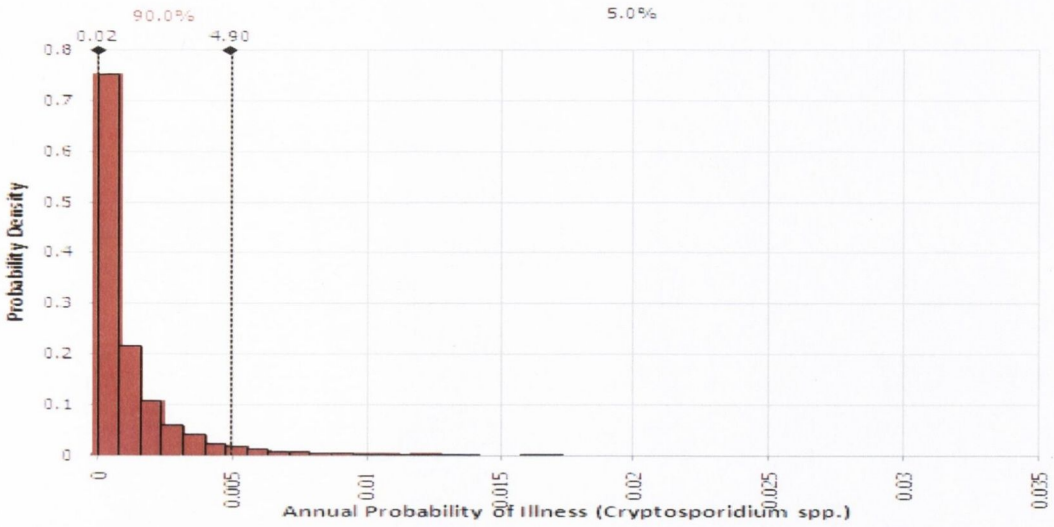


Figure 10.29 Probability density distribution for annual probability of cryptosporidial illness among users of untreated private groundwater sources

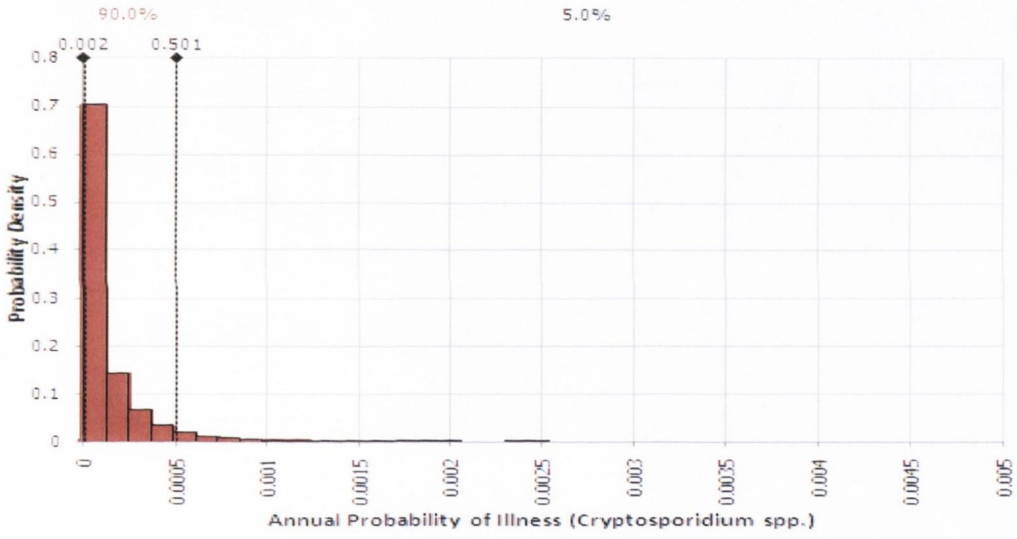


Figure 10.30 Probability density distribution for annual probability of cryptosporidial illness among users of private groundwater sources with treatment systems in place

Rank order correlation sensitivity analysis was subsequently carried out to ascertain which input parameters were had the greatest overall effects on the hazard characterisation outputs.

10.12.3(a) Model Sensitivity Analysis E. coli

Figure 10.31 presents sensitivity analysis performed on untreated bored wells. As shown, the overall *E. coli* loading rate was again the parameter having the most impact on predicted probability of private waterborne *E. coli* illness (correlation coefficient 0.867), with consumption rate and *E. coli* attack rates having correlation coefficients of 0.432 and 0.14 respectively. The correlation coefficients and their respective proportions were found to be almost identical among hand-dug and

treated wells, due to the primary differences between these sources being deterministic model inputs.

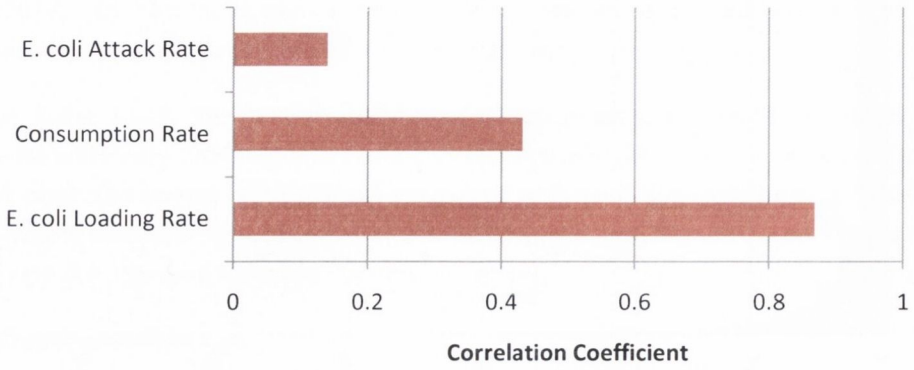


Figure 10.31 Sensitivity analysis between predicted annual probability of E. coli illness and stochastic model inputs

10.12.3(b) Model Sensitivity Analysis Cryptosporidium spp.

As shown in *Figure 10.32*, the inclusion of a deterministic dose-response relationship in the hazard characterisation model has a large overall effect on model output (Correlation Coefficient 0.716). This may represent a model limitation with regard to *E. coli*. Oocyst loading rate and daily consumption rates have correlation cofficients of 0.49 and 0.375 respectively.

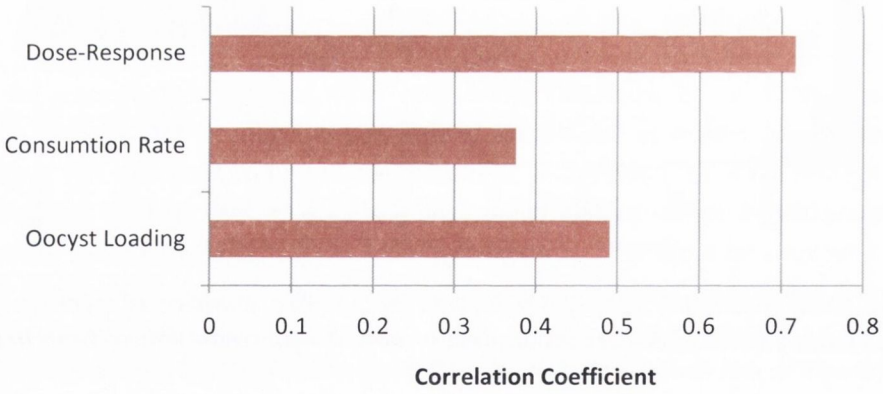


Figure 10.32 Sensitivity analysis between predicted annual probability of cryptosporidial illness and stochastic model inputs

10.12.4 Risk Characterisation

As previously outlined in chapter 5 (Section 5.7), risk characterisation is seen as the final risk assessment step and comprises the integration of the information gained from exposure assessment, dose-response, and hazard characterisation, in order to estimate the magnitude of the

public health problem and to evaluate variability and uncertainty. The risk characterisation model developed and applied here, was as follows (Eqn. 10.12):

$$P_{ill,year} \times APop \times [1 - NonCon] \times CR \times SI$$

Where,

APop = Affected population (Deterministic Input)

Non Con = Level of non-consumption (Deterministic Input)

CR = Contamination Rate (Deterministic Input)

Pill,year = from above (Stochastic Input from Step 2)

SI = Secondary Infection (Deterministic Input)

Module 1 – Untreated bored sources; *E. coli*

The mean calculated number of illnesses attributable to *E. coli* contamination of untreated private bored sources in Ireland was found to be 1,791, with a standard deviation of 3,178 illnesses. As shown in *Figure 10.33*, a range of 0 – 6,900 illnesses encompassed the 95% probability density.

General Mortality Rate (From Macler & Merkle, 2000):

Vulnerable population (<5 &>65 years): 20.25% - 363 illnesses in vulnerable population

$$363 \times 0.005 = 1.8\ deaths/annum$$

"Non-vulnerable" population: 79.75% - 1,428 illnesses in non-vulnerable population

$$1{,}428 \times 0.0016 = 2.2\ deaths/annum$$

HUS Rate (2.1/100 illnesses; Rangal *et al.*, 2005) – 38 HUS cases /annum, leading to 3-5% mortality rate (Siegler, 1997) – 1.9 HUS related deaths/annum

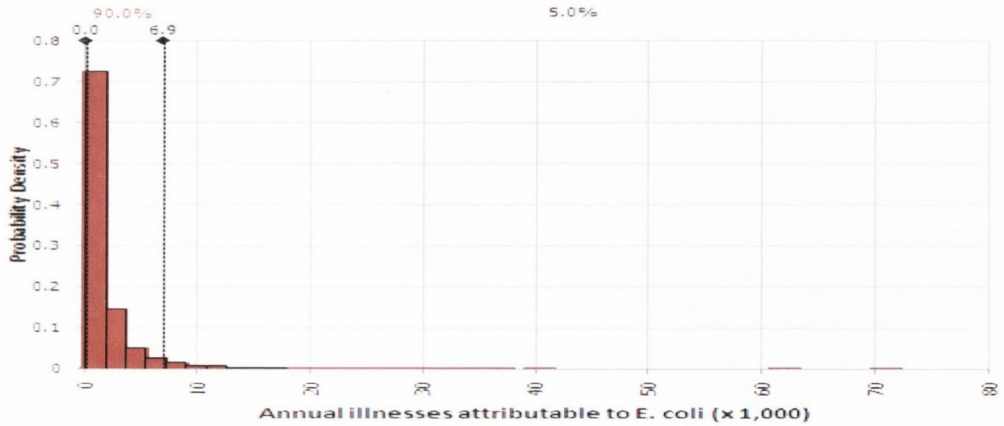


Figure 10.33 Probability density distribution of annual E. coli illnesses among the Irish population served by contaminated untreated private boreholes

Module 2 – Untreated hand-dug sources; *E. coli*

The mean calculated number of illnesses attributable to *E. coli* contamination of untreated private hand-dug sources in Ireland was found to be 1,187, with a standard deviation of 1,929 illnesses. As shown in *Figure 10.34*, a range of 30 – 4,350 illnesses encompassed the 95% probability density.

General Mortality Rate (From Macler & Merkle, 2000):

Vulnerable population (<5 &>65 years): 20.25% - 240 illnesses in vulnerable population

$$240 \times 0.005 = 1.2\ deaths/annum$$

"Non-vulnerable" population: 79.75% - 946 illnesses in non-vulnerable population

$$946 \times 0.0016 = 1.5\ deaths/annum$$

HUS Rate (2.1/100 illnesses; Rangal *et al.*, 2005) – 25 HUS cases /annum, leading to 3-5% mortality rate (Siegler, 1997) – 1.25 HUS related deaths/annum

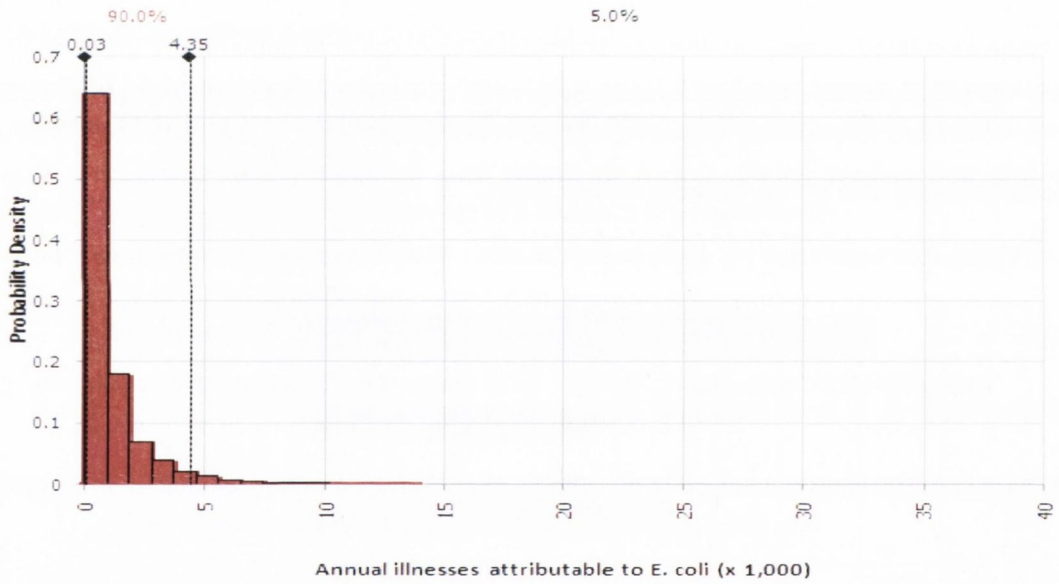


Figure 10.34 Probability density distribution for annual E. coli illnesses among the Irish population served by contaminated untreated private hand-dug wells

Module 3 –Treated bored sources; *E. coli*

The mean calculated number of illnesses attributable to *E. coli* contamination of treated private bored sources in Ireland was found to be 37, with a standard deviation of 66 illnesses. As shown in *Figure 10.35*, a range of 1-138 illnesses encompassed the 95% probability density.

General Mortality Rate (From Macler & Merkle, 2000):

Vulnerable population (<5 &>65 years): 20.25% - 7 illnesses in vulnerable population

$$7 \times 0.005 = 0.035\ deaths/annum$$

"Non-vulnerable" population: 79.75% - 30 illnesses in non-vulnerable population

$$946 \times 0.0016 = 0.048\ deaths/annum$$

HUS Rate (2.1/100 illnesses; Rangal *et al.*, 2005) – 0.77 HUS cases /annum, leading to 3-5% mortality rate (Siegler, 1997) – 0.038 HUS related deaths/annum.

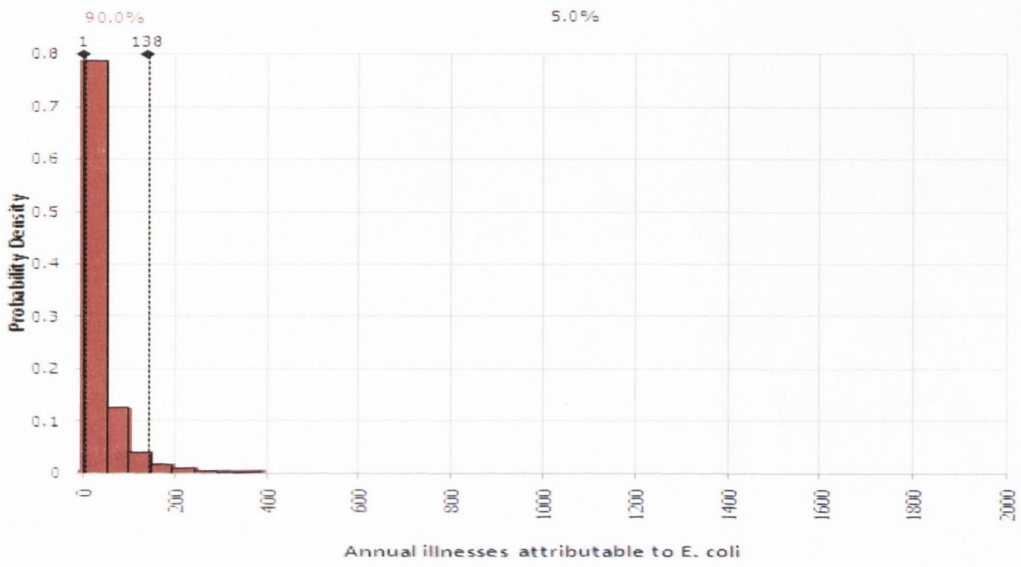


Figure 10.35 Probability density distribution for annual E. coli illnesses among the Irish population served by contaminated private boreholes with treatment systems in place

Module 4 – Treated hand-dug sources; *E. coli*

The mean calculated number of illnesses attributable to *E. coli* contamination of untreated private hand-dug sources in Ireland was found to be 6.04, with a standard deviation of 9.7 illnesses. As shown in *Figure 10.36*, a range of 0-22 illnesses encompassed the 95% probability density.

General Mortality Rate (From Macler & Merkle, 2000):

Vulnerable population (<5 &>65 years): 20.25% - 1.2 illnesses in vulnerable population

$$1.2 \times 0.005 = 0.006\ deaths/annum$$

"Non-vulnerable" population: 79.75% - 4.8 illnesses in non-vulnerable population

$$4.8 \times 0.0016 = 0.007\ deaths/annum$$

HUS Rate (2.1/100 illnesses; Rangal *et al.*, 2005) – 0.12 HUS cases /annum, leading to 3-5% mortality rate (Siegler, 1997) – 0.006 HUS related deaths/annum.

Among all four risk assessment modules, a mean total of 3,021 illnesses per annum among private water consumers are predicted, leading to a predicted annual CIR of 704/100,000. A 95% range of 31 – 11,410 illnesses/annum was also predicted. A full breakdown of predictions with regard to *E. coli* morbidity and mortality are presented in *Table 10.17*. No sensitivity analysis was carried out on risk estimation model, as no additional deterministic inputs were included from the hazard characterisation step.

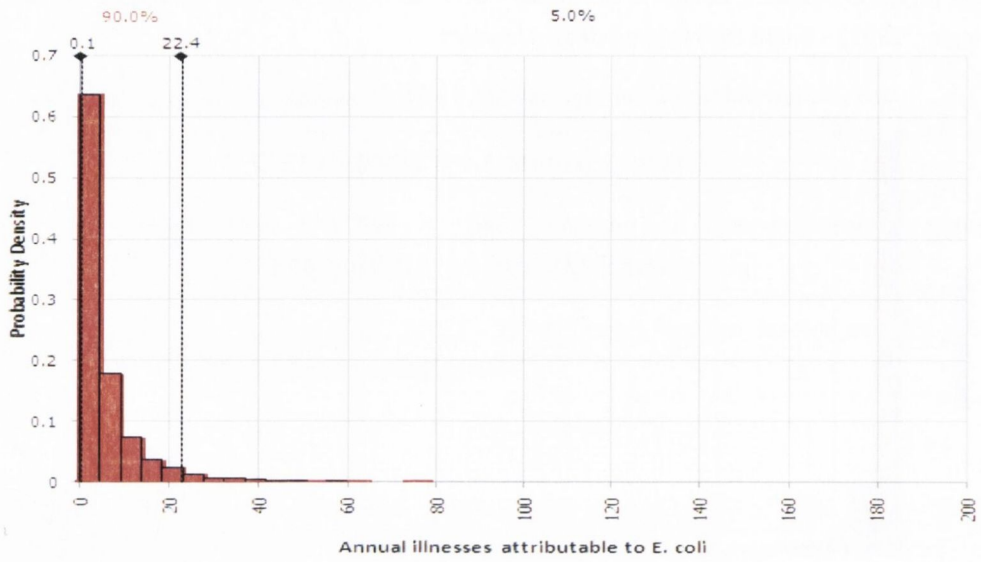


Figure 10.36 Probability density distribution for annual E. coli illnesses among the Irish population served by contaminated private hand-dug wells with treatment systems in place

Table 10.17 Predicted Illnesses and mortalities per annum attributable to E. coli from Private Groundwater Sources in Ireland (*Incl HUS)

	Source Type	Illnesses	Illness CIR	HUS Cases	Deaths*
Module 1	Untreated, bored	1,791	561/100,000	38	5.9
Module 2	Untreated, hand-dug	1,187	2,967/100,000	25	3.9
Module 3	Treated bored	37	55/100,000	1	0
Module 4	Treated, hand-dug	6	287/100,000	0	0
Total		3,021	704/100,000	64	9-10

Module 1 – Untreated bored sources; *Cryptosporidium* spp.

The mean calculated number of illnesses attributable to *Cryptosporidium* spp. contamination of untreated private bored sources in Ireland was found to be 656, with a standard deviation of 608

illnesses. As shown in *Figure 10.37*, a range of 50 – 1,820 illnesses encompassed the 95% probability density.

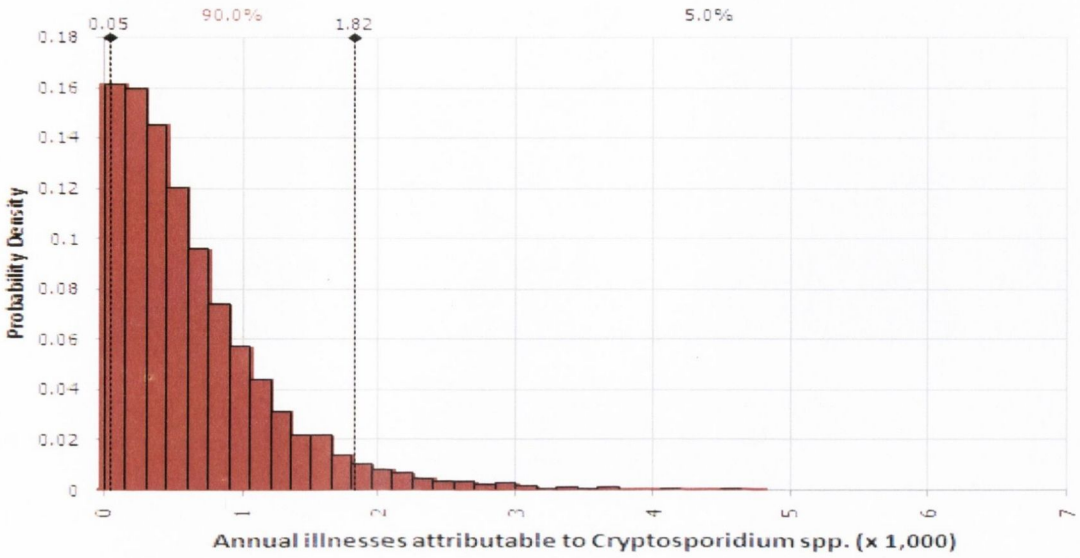


Figure 10.37 Probability density distribution for annual cryptosporidial illnesses among the Irish population served by contaminated untreated private boreholes

From Teunis *et al.* (1997):

Vulnerable population (<5 &>65 years): 0.2025 x 656 = 133 illnesses in vulnerable population

$$133 \times 0.001 = 0.13\ deaths/annum$$

Module 2 – Untreated hand-dug sources; *Cryptosporidium* spp.

The mean calculated number of illnesses attributable to *Cryptosporidium* spp. contamination of untreated private hand-dug sources in Ireland was found to be 99, with a Std. Dev. of 95.2 illnesses. As shown in *Figure 10.38*, a range of 7 – 275 illnesses encompassed the 95% probability density.

From Teunis *et al.* (1997):

Vulnerable population (<5 &>65 years): 0.2025 x 99 = 20 illnesses in vulnerable population

$$133 \times 0.001 = 0.02\ deaths/annum$$

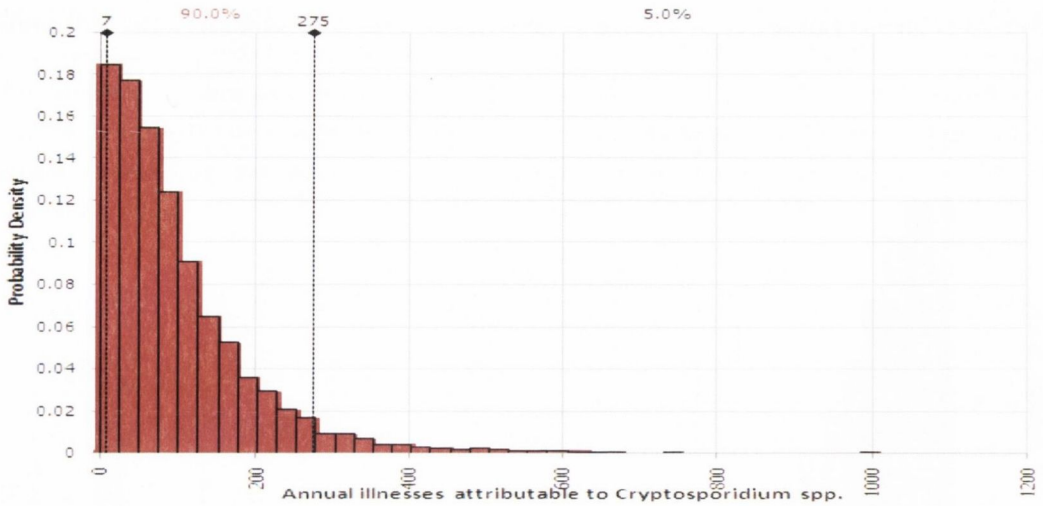


Figure 10.38 Probability density distribution for annual cryptosporidial illnesses among the Irish population served by contaminated untreated private hand-dug wells

Module 3 –Treated bored sources; *Cryptosporidium* spp.

The mean calculated number of illnesses attributable to *Cryptosporidium* spp. contamination of untreated private hand-dug sources in Ireland was found to be 13.4, with a standard deviation of 12.8 illnesses. As shown in *Figure 10.39*, a range of 1 – 37 illnesses encompassed the 95% probability density.

From Teunis *et al.* (1997):

Vulnerable population (<5 &>65 years): 0.2025 x 13.4 = 2.7 illnesses in vulnerable population

$$2.7 \times 0.001 = 0.003\ deaths/annum$$

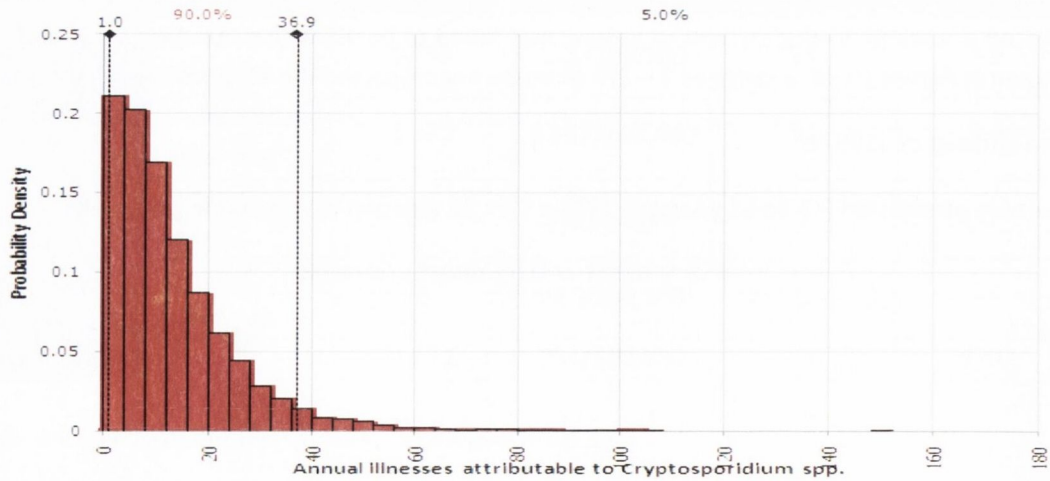


Figure 10.39 Probability density distribution for annual cryptosporidial illnesses among the Irish population served by contaminated private boreholes with treatment systems in place

Module 4 –Treated hand-dug sources; *Cryptosporidium* spp.

The mean calculated number of illnesses attributable to *Cryptosporidium* spp. contamination of treated private hand-dug sources in Ireland was found to be 0.43, with a standard deviation of 0.41 illnesses. As shown in *Figure 10.40*, a range of 0-1 illnesses encompassed the 95% probability density.

From Teunis *et al.* (1997):

Vulnerable population (<5 &>65 years): 0.2025 x 0.43 = 0-1 illnesses in vulnerable population

$$2.7 \times 0.001 = 8.7x10^{-5} deaths/annum$$

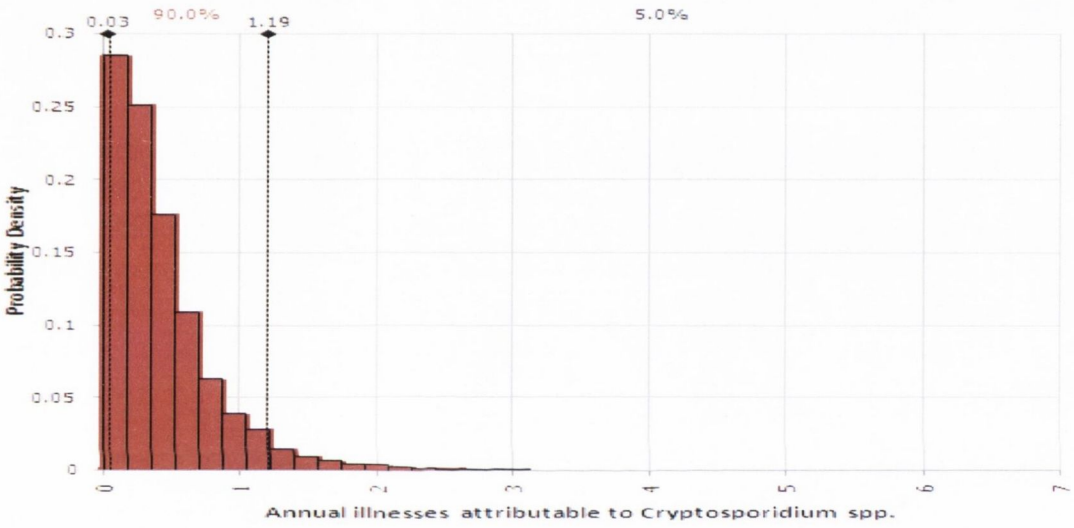


Figure 10.40 Probability density distribution for annual cryptosporidial illnesses among the Irish population served by contaminated private hand-dug wells with treatment systems in place

Among all four risk assessment modules, a mean total of 769 illnesses per annum among private water consumers are predicted, leading to a predicted annual CIR of 179.4/100,000. A 95% range of 58 – 2,133 illnesses/annum was also predicted. A full breakdown of predictions with regard to *E. coli* morbidity and mortality are presented in *Table 10.18*.No sensitivity analysis was carried out on the final risk estimation model, as no additional deterministic inputs were included from the hazard characterisation step.

Table 10.18 Predicted Illnesses and mortalities per annum attributable to Cryptosporidium spp. from Private Groundwater Sources in Ireland

	Source Type	Illnesses	Illness CIR	Deaths
Module 1	Untreated, bored	656	205/100,000	0.13
Module 2	Untreated, hand-dug	99	243/100,000	0.02
Module 3	Treated, bored	13	19.5/100,000	0
Module 4	Treated, hand-dug	0-1	21.5/100,000	0
Total	Total	769	179/100,000	0.15

10.13 Summary

As outlined in the Introduction to this chapter (Section 10.1), both existing data and data collected by the researcher were utilised in the development of a quantitative Monte Carlo simulation (stochastic) model to evaluate the annual risk of infection from microbial contamination (*E. coli* and *Cryptosporidium* spp.) of private groundwater sources in Ireland. Three existing deterministic health risk assessment models were additionally employed to estimate the magnitude of risk to health from contamination of private supplies, following "Good Risk Assessment Principles" as set out by Haas *et al.* (1999).

10.13.1 E. coli

As shown in Table 10.19, all three deterministic models and the developed stochastic risk assessment model predict higher crude incidence rates of waterborne illness attributable to *E. coli* among users of hand-dug wells than borehole users, with predicted hand-dug crude incidence rates ranging from 3.1 (deterministic model 2) to 5.8 (stochastic model) times that of predicted crude incidence rates among bored well users. This is due to a number of factors, including;

- Higher *E. coli* presence rates in hand-dug wells
- Higher magnitude of *E. coli*, where present, among hand-dug wells
- Lower levels of treatment among hand-dug wells

As shown in Figure 10.31, a sensitivity analysis of the stochastic model suggests that the *E. coli* loading rate (i.e. amalgamation of *E. coli* presence and magnitude) is in fact the input with the greatest overall contribution on model outputs and therefore, improved source protection, particularly in the case of existing wells, is necessary in order to minimize the overall health burden attributable to private groundwater sources in Ireland. Although ideally the use of hand-dug wells should be avoided, particularly in *High/Extreme* vulnerability areas, where existing hand-dug wells are in use, improved source protection should be a priority.

Table 10.19 Predicted Illnesses and mortalities per annum attributable to E. coli from Private Groundwater Sources in Ireland (*including HUS)

Model	Source Type	Illnesses	Illness CIR	HUS Cases	Deaths*
Deterministic 1	Bored	138	35.7	2.9	
	HD	72	168.3	1.5	
	Total	210	49	3.4	0.6-0.7
Deterministic 2	Bored	835	216	18	
	HD	293	682	7	
	Total	1128	263	25	3.3-3.7
Deterministic 3	Bored	68	17.6	1.4	
	HD	38	88.5	0.8	
	Total	106	24.7	2.2	0.95
Stochastic	Bored	1828	474	38	
	HD	1193	2789	25	
	Total	3021	705	63	9.8

As shown in Figure 10.2, the number of laboratory-confirmed VTEC cases, and consequently, the overall national (Republic of Ireland) VTEC CIR has consistently risen over the 10-year period 1999-2009, with approximately 240 laboratory-confirmed VTEC cases in 2009. As shown in Table 10.19, both deterministic model 2 and the stochastic model predict significantly larger numbers of illnesses (and consequently CIR) attributable to waterborne *E. coli* in Ireland. The assumptions made with respect to contamination frequency are thought to be the main source of under-estimation attributable to deterministic models 1 and 3. As noted by Frost *et al.* (1996; In: Haas *et al.*, 1999), there is likely to be a substantial difference between confirmed cases and total disease burden, both locally and nationally, as a number of specific steps must occur for case recognition, particularly in the case of "environmentally caused" infectious disease. Where one of these steps is not followed (i.e. acquisition of patient stool sample), it is likely that a case will not be positively confirmed, therefore, current local and national statistics are likely an under-estimate of the national disease burden. Furthermore, national CIRs are often associated with epidemic/outbreak conditions as opposed to the underlying endemic disease rate. For example, Haas *et al.* (1999) report that while an average of approximately 200 outbreak cases per year of water- and foodborne illnesses attributable to *Campylobacter* spp. are reported in the USA, estimates of the disease burden suggest about 2,100,000 cases per year (i.e. approximately 10,000 cases per confirmed case).

All three deterministic models predict lower levels of illness, HUS and death attributable to waterborne *E. coli* than predicted by the developed stochastic model, as outlined in Section 10.11. However, all three deterministic models are considered to contain a number of inherent limitations, several which have been addressed within the stochastic model. Overall outputs by the stochastic model predict that approximately 0.7% (704.8/100,000) of private groundwater users in Ireland will contract a symptomatic illness attributable to waterborne *E. coli* per year (0.47% of borehole users, 2.78% of hand-dug well users). The main sources of variability within the stochastic model are contaminant loading, consumption rate and pathogen attack rate, with contaminant loading and consumption rate being the factors over which the consumer has some level of control. Moreover, although it was, by necessity, a deterministic model input and therefore not included in sensitivity

analysis, the effect of treatment was significant, with the developed stochastic model predicting overall CIRs of 561/100,000 (0.56%) and 55/100,000 (0.05%) among untreated and treated boreholes, respectively.

Overall, a number of risk management recommendations may be made with regard to QRA modelling of the predicted human health burden attributable to *E. coli* contamination of private groundwater sources in Ireland, including:

- the use of boreholes as opposed to hand-dug wells, where possible,
- improved source protection programmes in order to minimise contamination events and magnitude,
- improved siting, design and construction of both hand-dug wells and boreholes (Chapter 11)
- the use of treatment systems

10.13.2 Cryptosporidium spp.

Unlike *E. coli*, as shown in Table 10.20 the three deterministic models and the developed stochastic risk assessment model do not predict significantly different crude incidence rates of waterborne illness attributable to *Cryptosporidium spp.* with regard to well type. Predicted crude incidence rates associated with boreholes ranged from 111.7/100,000 (0.11%) to 173.4/100,000 (0.17%), while crude incidence rates pertaining to hand-dug wells ranged from 111.7/100,000 (0.11%) to 233.7/100,000 (0.23%). As deterministic model 1 was undertaken for treated and untreated sources as opposed to well type, it is not possible to directly compare these predicted CIR ranges; however, this model predicted significantly higher crude incidence rates with regard to cryptosporidial disease burden, probably due to the nature of the dose-response input parameter (Eqn. 10.2). The lack of significant difference between bored and hand-dug wells in the case of cryptosporidial illnesses (and subsequently CIR and mortality) is due to the necessary assumption that *Cryptosporidium* spp. presence and (where present) magnitude were the same between well types. This assumption represents a source of uncertainty within all model predictions, and it is likely that, similar to *E. coli* presence, higher levels of *Cryptosporidium* presence and magnitude occur in hand-dug wells due to design and construction parameters, however, to date this has not been studied.

As shown in Figure 10.6, over the 6-year period 2004-2009, *Cryptosporidium* spp. notifications and crude incidence rates have remained relatively constant, with reported CIRs in the range 9-14/100,000. As shown in Table 10.20, the developed stochastic model predicts an annual CIR of approximately 12-20 times this figure i.e. approximately 12-20 cases per confirmed case. As previously outlined, the developed stochastic model and outputs presented here predict endemic as opposed to epidemic/outbreak incidence rates, and therefore predict an endemic CIR of approximately 0.18% of private groundwater users contracting a symptomatic illness attributable to waterborne *Cryptosporidium* spp. per year in Ireland (i.e. 179.4/100,000). Furthermore, the model predicts 0.15 mortalities per annum or 1 mortality every 6-7 years.

Table 10.20 Predicted Illnesses and mortalities per annum attributable to Cryptosporidium spp. from Private Groundwater Sources in Ireland

Model	Source Type	Illnesses	Illness CIR	Deaths
Deterministic 1	Untreated	2,919	798.4	
	Treated	49	77.7	
	Total	2,968	692.4	0.6
Deterministic 2	Bored	431	111.7	
	HD	48	111.7	
	Total	479	111.7	0.1
Deterministic 3	Bored	534	138.4	
	HD	66	154.3	
	Total	600	139.9	0.12
Stochastic	Bored	669	173.4	
	HD	100	233.7	
	Total	769	179.4	0.15

Sensitivity analyses of the stochastic model inputs (Figure 10.32) show that, unlike *E. coli*, the dose-response rate associated with *Cryptosporidium* spp. is the highest overall contributor to model outputs (Correlation Coefficient 0.716), suggesting that particularly vulnerable sub-populations are likely to contribute significantly to the overall health burden associated with waterborne *Cryptosporidium* spp. in Ireland. This analysis also suggests that although appropriate source protection is a means of decreasing the overall health burden associated with waterborne *Cryptosporidium* spp., it is less effective than imporved source protection with regard to waterborne *E. coli*. As for *E. coli*, although it was, by necessity, a deterministic model input and therefore not included in sensitivity analysis, the effect of treatment was found to be significant, with the developed stochastic model predicting overall CIRs of 205/100,000 (0.2%) and 19.5/100,000 (0.02%) among untreated and treated boreholes, respectively.

Overall, a number of risk management recommendations may be made with regard to QRA modelling of the predicted human health burden attributable to *Cryptosporidium* spp. contamination of private groundwater sources in Ireland, including:

- consumer protection, particularly in the case of vulnerable sub-populations, through periodic water quality testing, use of alternative water sources where previous contamination has occurred, boiling all water, improved diagnostics, systematic stool sampling, etc
- improved source protection programmes in order to minimise contamination events and magnitude,
- improved location, design and construction of wells (see Chapter 11)
- the use of treatment systems

10.13.3 Model Limitations

Although it is considered that the developed stochastic models presented here reflect an improved approach to endemic health burden prediction by including Irish data where possible, a number of limitations still exist, which could be the focus of future work. These include:

- The use of age-specific/population-specific consumption rates and dose-response curves, leading to improved predictions relating to vulnerable sub-populations (i.e. immuno-suppressed, neonates/pregnant women),
- the inclusion of "transient consumers" i.e. holiday makers, rural/farm labourers, customers in rural businesses (public houses, etc) i.e. account for both "travellers syndrome" and gradual immunity (Olsen *et al.,* 2000),
- concurrent contamination by multiple organisms leading to higher attack rates (Kopilovic *et al.,* 2008),
- Both deterministic and stochastic models were heavily reliant upon previously reported data, particularly clinical data (e.g. morbidity, mortality, HUS rate, etc). These assumptions produce significant uncertainty/variability and are therefore sources of error within the models. Although not possible to derive and/or improve under the remit of this work, the use of "up to date" and where possible, country-specific clinical data is preferable.

Chapter 11: Private Groundwater Contamination Prevention – The Role of the Consumer

11.1 Introduction

There are currently estimated to be 100,000 – 200,000 (c 165,000) private domestic wells in Ireland, serving between 450,000 and 500,000 consumer's on a daily basis. Furthermore, private group water schemes, which serve < 50 consumers and are typically from a groundwater source, account for an additional 200,000 – 250,000 consumer's. Therefore, private groundwater supplies in Ireland are the primary water source for up to 750,000 consumers, or approximately 18% of the national population. A further 8% of the population is served by public supplies derived from groundwater, accounting for a total national groundwater reliance of approximately 26% or 1.1 million people. However, in some areas, groundwater reliance is significantly higher, for example, 85% of the population of Co Roscommon rely on groundwater supplies, while this figure is approximately 72% with respect to Co Offaly (Spain, 2000). The following chapter provides practical guidance for users and owners of private domestic groundwater sources in Ireland on the potential health effects of private groundwater contamination, possible sources of these contaminants and how to prevent contamination of their own supplies. The research used to inform these guidelines included the completion of 545 groundwater user awareness surveys (described in Chapter 8), "one-off" sampling and susceptibility surveying of 211 private groundwater sources over five study areas and continuous monthly monitoring of a number of these sources (Chapter 7). The following guideline framework also builds upon previous work by agencies such as the Environmental Protection Agency (EPA), Geological Survey of Ireland (GSI), Institute of Geologists of Ireland (IGI), Health Service Executive (HSE) and the National Rural Water Monitoring Committee (NRWMC). The guidance acknowledges that the majority of well design and construction tasks are usually undertaken by a drilling contractor, particularly in the case of boreholes, therefore, those remedial/preventative actions available to the well owner are the focus of this chapter. It is also considered that, as the primary end-user of the groundwater source, improved awareness on behalf of the well owner will minimise the overall contamination susceptibility associated with the source, and therefore location, construction and design parameters are also outlined.

The following guidance is based upon the "Hazard – Pathway – Receptor" contamination model, whereby, in order for an adverse outcome to occur at a receptor, both a hazard and pathway must exist. In the case of the current study, a "dual-receptor" approach has been developed (Figure 11.1). The primary receptor is considered to be the well or aquifer, therefore "primary susceptibility" may be defined as, the susceptibility of a private groundwater source or aquifer to microbial and/or chemical contamination through inappropriate location, design and/or construction, leading to increased probability of contaminant ingress with the associated adverse effect being pollution of this source by chemical and/or microbial contaminants. The secondary receptor is the private groundwater consumer, therefore, secondary susceptibility is considered as the susceptibility of a well user of consumption of a microbial and/or chemical contaminant, if present in a private well, with the adverse effect being illness due to waterborne infection. An "exclusionary principle" is evoked, whereby the exclusion of the hazard and/or pathway is considered the best way of minimising "receptor susceptibility", thereby protecting the potential receptor.



Figure 11.1 Hazard-Pathway-Receptor Model

The overall objective of the guideline framework, presented below, is to aid in the development of formal guidelines for the owners of both new and existing wells on the location, construction and maintenance of private groundwater supplies in Ireland, so as to reduce potential adverse human health effects in the future. These guidelines will take account of the current level of public awareness of water contamination and health.

11.2 Groundwater Contamination and Potential Human Health Effects

The health risks associated with small-scale drinking water supplies depend on a variety of factors, including: the type of contaminant; the contamination source and loading rate; the type, location and construction of the well; the presence or absence of water treatment (notably disinfection); and the awareness (or lack of awareness) of the health risks amongst the consumers.

There is a wide range of possible illnesses that can result from pathogenic infection (see Chapter 4). The most common illness is generalised acute gastrointestinal illness (AGI), also known as gastroenteritis. Symptoms of AGI include nausea, vomiting, diarrhoea and stomach cramps. Most microbial-related illnesses are relatively minor and of short duration, but some AGI cases can be severe, particularly amongst the elderly, the very young and the immuno-compromised (i.e. "vulnerable sub-populations"). Other potential illnesses which may result as a direct result of consumption of contaminated groundwater are cryptosporidiosis (*Cryptosporidium* spp.), *Giardia*sis (*Giardia* spp.), shigellosis (*Shigella* spp.) and viral gastroenteritis (adenovirus, rotavirus, norovirus), with all of these infections presenting similar symptoms as AGI (i.e. nausea, vomiting, diarrhoea and stomach cramps). The main sources of pathogens in water are human and animal sewage from a variety of potential sources including septic tanks, grazing animals, animal housing, farmyards, landfills and landspreading.

As shown in Figure 11.2, monitoring of groundwater sources (both public and private) during the 14-year period 1995-2009 have found that 27-48% of these sources had *E. coli* present, with a mean incidence of just under 35%. 'One-off' sampling of private wells undertaken as part of the study on which this framework is based found that approximately 29% of wells (62/211) tested positive for *E. coli* during the period 2007-2010 (see Chapter 7), consistent with EPA figures and reinforcing the consideration that pathogenic contamination of private groundwater sources in Ireland is both widespread and frequent.

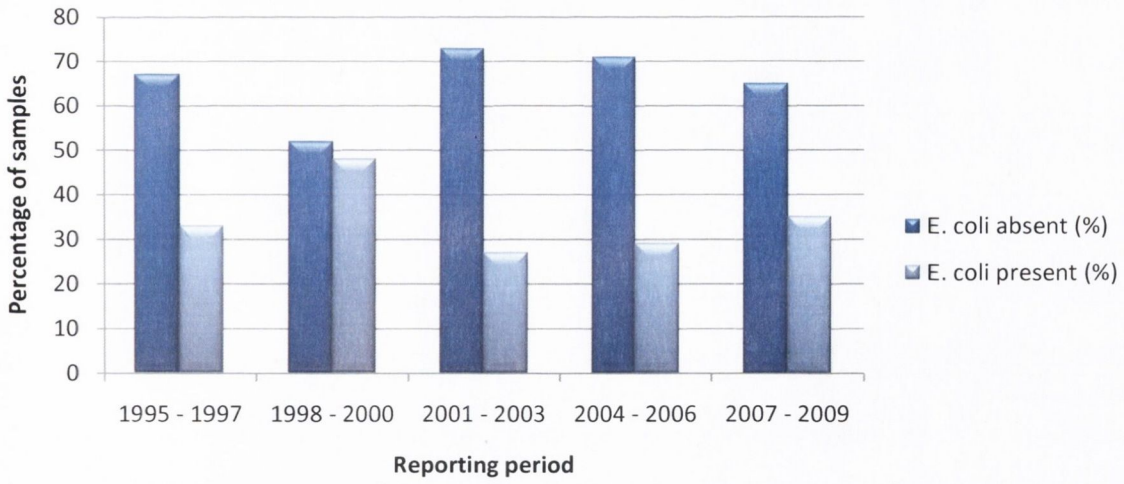


Figure 11.2 Percentage of groundwater samples with E. coli over EPA reporting periods 1995 – 2009 (EPA, 2010c)

As well as microbial contaminants, there are other substances that may occur in small private water supplies that can have significant health consequences. Three inorganic contaminants of health concern that can be found in groundwater supplies are: nitrate, fluoride and arsenic. High nitrate concentrations are most often the result of pollution from agricultural fertilisers, septic tank systems and waste disposal. The main health concern about nitrate relates to the condition known as *infantile methaemoglobinaemia*.

It is strongly recommended that in the case of anyone drinking from a well contracting any of the symptoms outlined above; professional medical advice should be sought. Furthermore, medical professionals should be informed that the patient's primary source of drinking water is a private well, so that the source may be excluded or pinpointed as the likely infection source. This is particularly important with respect to potentially vulnerable populations including infants, young children, the elderly or immune-compromised individuals. Water quality analysis should also be carried out in concurrence with professional medical advice.

11.3 How does contamination occur and from where?

The main sources of pathogens in private groundwater supplies are human and animal faecal matter. As previously outlined by Daly (2004), before a private well has been located, designed and constructed, groundwater will have been flowing by the potential well site, frequently from tens of metres to, in rare instances, several kilometres, depending on a number of local hydrogeological parameters, including bedrock type, subsoil type and thickness and rainfall frequency and magnitude. Upon construction and subsequent pumping of a private well, groundwater is drawn into the well as nearby flow-lines are directed towards the well. If contaminants are present in the adjacent groundwater, particularly up-gradient of the well from which the majority of groundwater will be taken, these will also be drawn into the well; whereupon they may be potentially consumed by the well user. The land surface area from where water is drawn into a well is called the Zone of Contribution (ZOC), with hazard sources located within the ZOC considered to be potential sources of contamination with respect to an individual well (Figure 11.3). Therefore, a knowledge and

understanding, by both well drilling contractors and well owners, of the ZOC, is important at the well location and design phase, so that an effective hazard assessment may be carried out.

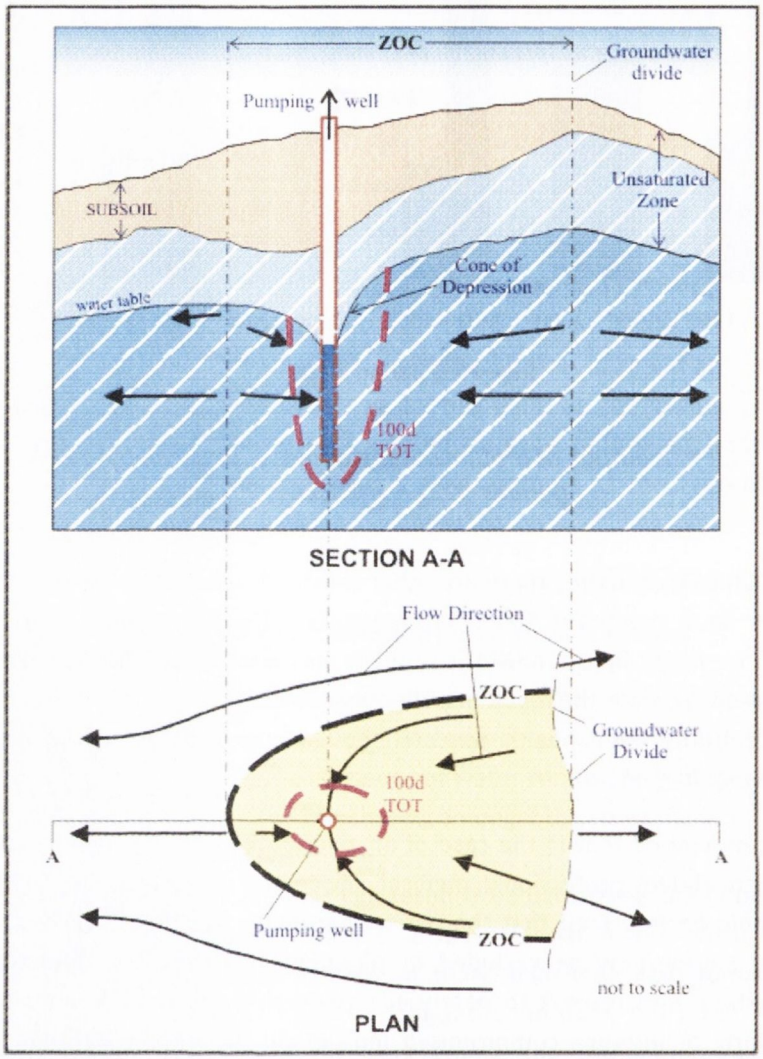


Figure 11.3 Conceptual schematic of the Zone of Contribution (ZOC) (DELG/EPA/GSI, 1999)

Typically the size of a ZOC will depend upon the abstraction rate (i.e. volume of water taken from the well over a specific time period) and recharge (i.e. volume of rainfall which infiltrates through topsoil and subsoil into the saturated zone) over the same period of time. In Ireland, due to relatively high recharge rates (due to the temperate climate), private domestic wells, particularly those not being used to supply agricultural activities, are generally associated with small ZOCs. Daly (2004) states that a typical ZOC for a private domestic well in Ireland may be 1,000 – 2,500 m^2, which may seem large, however this only constitutes a ZOC radius of 18 – 28 m for an approximately circular ZOC (i.e. flat hydraulic gradient and homogenous and isotropic subsoil and bedrock materials).

As shown in Figure 11.3 however, very few sites comprise a completely level hydraulic gradient, which is reflected in the shape of the ZOC, with a larger area of the ZOC occurring up-gradient of the wellhead. Generally, the watertable can be viewed as a subdued reflection of the surface topography, with local surface water flow also providing an insight into the likely groundwater flow

direction. As set out by Daly (2004), the ZOC may be visualized as a 15-30 m wide zone, extending from approximately 150 m up-gradient of the wellhead to 10 m down-gradient, however, this may be significantly larger in permeable bedrocks and karst areas.

When a potentially polluting activity is located within the ZOC around a private well, there are three main pathways which a contaminant may take to gain access to the well. As shown in Figure 11.4, these are: a) surface runoff, b) shallow groundwater infiltration, and, c) aquifer contamination.

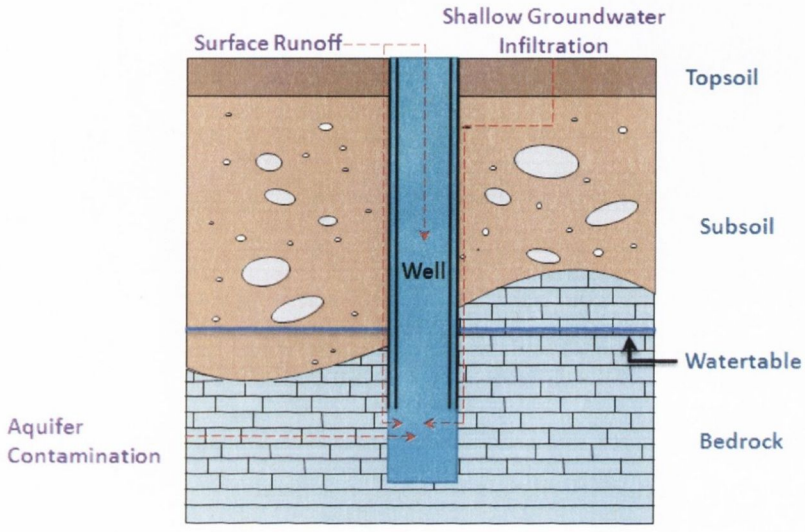


Figure 11.4 Potential contamination pathways

Direct ingress of contaminants by surface runoff may occur during high precipitation periods, particularly in areas dominated by runoff from low permeability soils and subsoils, where surface ponding occurs. In order for surface runoff to directly ingress the well, an access point must exist i.e. uncovered wellhead and/or liner, insufficient or absent liner clearance, ineffective or absent well chamber drainage, etc. Shallow groundwater infiltration may occur in various geological and hydrogeological settings, and typically occurs where the well design/construction itself has created a preferential pathway or conduit for contaminants, which may then enter the well in the producing sections. This may occur due to ineffective or absent well sealing/grouting, poor liner installation or installation of the liner to an insufficient depth. Ingress of surface runoff and shallow groundwater infiltration may both be viewed as localised or "source-specific" contamination, as they are both caused by inappropriate location, design or construction of the well (Figure 11.5). Aquifer (generalised) contamination occurs due to the large scale contamination of a significant proportion or all of an aquifer, whereby a serious contamination event (typically from a point source) may take place many hundreds of metres, or even kilometres, away from the well. Natural groundwater flow may then over time transport contaminants through the aquifer, whereby they may be drawn into the production section of the well upon pumping.



Figure 11.5 Hand-dug well at risk of surface runoff ingress and shallow groundwater infiltration due to lack of wellhead cover and appropriate well liner

11.4 Well Records

As a result of the lack of formal regulations, standards or planning conditions associated with private groundwater sources in Ireland, there is currently no obligation on behalf of well owners or well drilling contractors to provide or log data pertaining to well construction specifications. Furthermore, the awareness survey carried out as part of this research found that less than 22% (80/370) of private well users had records pertaining to the design or construction of their own well. The majority of hand-dug well users (96%) did not have any design or construction records, while this figure was 70% in the case of borehole users. Frequently, hand-dug wells are typified by a low level of design/construction, and as they are typically older, details associated with excavation may not have been recorded. However, the majority of boreholes are constructed by specialised contractors, who would be expected to supply documentation upon completion of a borehole. This documentation may include (Daly, 2004):

- Well location
- Drilled well depth
- Drilled well diameter
- Depth of lining
- Diameter of lining
- Details of sealing/grouting
- Depth to bedrock
- Subsoil type
- Bedrock type
- Water entry levels
- Depth to cavities (if encountered)
- Static watertable depth
- Measured pumping rate

- Results of pumping/step drawdown tests (if carried out)
- Estimated yield
- Results of chemical and/or microbial analyses (if carried out)
- Drilling contractor name and contact details
- Drilling date
- Drilling method

It is useful and therefore recommended for a well owner to request and maintain a record of these parameters, for a number of reasons:

1. If a contamination event does occur, a volume of information is available about the source, which may help both well users and groundwater professionals alike in pinpointing and remediating the problem.
2. Householders purchasing an existing dwelling with an associated private groundwater source will be aware of the well location and associated design/construction parameters in case a problem does arise.
3. If a treatment system is being installed, well records including results of previous chemical and/or microbial analysis, may be used by well users and/or water quality managers in choosing an appropriate treatment system.
4. Well records should be submitted to the Geological Survey of Ireland (GSI) (www.gsi.ie), which may be used to characterise the number, type and susceptibility of wells in a geographical area, and subsequently aid in the location/design/construction of new wells and pinpoint drilling contractors in the area.

Also, it is recommended that the new well owners and new owners of existing wells have the well tested, both chemically and microbially, and maintain the results of analysis with the well records.

11.5 Appropriate Well Location including groundwater vulnerability

In order for a private well to become contaminated, and thus constitute a potential human health burden, the hazard – pathway – receptor link must exist and remain intact. Therefore, when properly located in order to minimise potential hazard sources in the ZOC, the risk of well contamination is significantly minimised i.e. contamination cannot occur in the absence of contaminants. Accordingly, proper well location may be viewed as the principal step in sanitary protection of private groundwater supplies, and is a consideration over which the well owner has significant control. Proper well location depends upon a number of considerations, as previously set out (Daly, 2004; IGI, 2007):

- Contaminant loading (i.e. the volume of contaminants produced by a potential hazard source, for example, point sources of contamination such as septic tanks and farmyards typically release significantly greater contaminant loads than non-point sources including grazing animals);
- Groundwater vulnerability of the site/area (higher volume of contaminants may leach downwards through high permeability subsoils, while a greater level of overland flow is associated with low permeability subsoils);

- Proximity of the potential hazard source with respect to the wellhead (i.e. typically a septic tank located up-gradient of the well within the ZOC will pose a higher threat than a septic tank located down-gradient of the wellhead outside of the ZOC);
- Hazard density (i.e. greater numbers of grazing animals/septic tanks will typically result in higher level of source susceptibility to contamination);
- Risk assessment and management.

The following section provides some guidance on the principal contaminant sources encountered in Ireland and how private groundwater sources may be best located with these in mind. To this end, it is recommended that all well owners undertake a "hazard assessment" of their source. This may be carried out for both new and existing wells. This may be undertaken by visually approximating the ZOC at the (proposed) wellhead, and subsequently determining the existence of any potential hazard sources (see below) within the ZOC. In the case of a planned well, it is advisable that well location is based upon the minimisation of potential hazard sources within the ZOC and a groundwater professional may be consulted in this regard (i.e. hydrogeologist, environmental scientist, local authority engineer). SEPA (2010) recommend that "drinking water supplies be located at least 50 m from any potentially polluting activity". For existing wells, if potential hazard sources are located within the ZOC, it is recommended that the owner undertake a sanitary inspection of his/her well (see Section 11.6.2 – 11.6.6), and consider the level of secondary (consumer) protection currently in place (see Section 11.7).

11.5.1 Septic Tanks

On-site wastewater treatment systems (septic tanks) are numerous throughout Ireland and represent a significant potential source of contamination due to their "point-source" nature. Furthermore, they are the primary source of human-specific pathogens in both surface and groundwater in rural Ireland. As shown (Table 11.1), the recently published *Code of Practice for Wastewater Treatment Systems for Single Houses* (EPA, 2009) recommends a minimum set-back distance of 30-60 m from the septic tank percolation area to the wellhead, depending on subsoil conditions in the area. Furthermore, depending on the setback gradient from the wellhead to the septic tank, this minimum setback recommendation may increase or decrease. The current study reflects this relationship, although as shown in Figure 11.6, greater setback distances are recommended where possible (depending on site availability). It is also worth noting that, during particularly heavy precipitation events, the "zone of influence" associated with a particular source may vary due to increasing surface runoff and subsurface recharge, and therefore it is recommended that all wells be located up-gradient of existing septic tanks and the setback distance maximised, with a minimum of 40 m setback, where possible. Additionally, although typically septic tanks provide a greater threat to wells located in moderate and high permeability subsoil areas, the current study found that septic tanks were a contaminant source of significance in all areas, including *Low* vulnerability areas.

Table 11..1 Recommended Minimum Setback Distance between a Receptor and Septic Tank

T/P[3]-value[1]	Type of soil/subsoil[2]	Depth of soil/subsoil (m above bedrock) (see Notes 1, 2, 3, 6)	Minimum distance (m) from receptor to percolation area or polishing filter[5]				
			Public water supply	Karst feature	Down-gradient domestic well or flow direction is unknown (see Note 5)	Domestic well alongside (no gradient)	Up-gradient domestic well
>30	CLAY; sandy CLAY (e.g. clayey till); SILT/CLAY	1.2 >3.0	60	15	40 30	25	15
10–30	Sandy SILT; silty SAND; silty GRAVEL (e.g. sandy till)	1.2 >8.0	60	15	45 30	25	15
<10	SAND; GRAVEL; silty SAND	2.0[3] 2.0[4] >8.0[4]	60	15	60 40 30	25	15

[1]The T-value (expressed as min/25 mm) is the time taken for the water level to drop a specified distance in a percolation test hole. For shallow subsoils the test hole requirements are different and hence the test results are called P-values. For further advice see Annex C.
[2]BS: 5930 descriptions.
[3]Water table 1.2–2.0 m.
[4]Water table >2.0 m.
[5]The distance from the percolation area or polishing filter means the distance from the periphery of the percolation area or polishing filter and not from the centre.

EPA, 2009

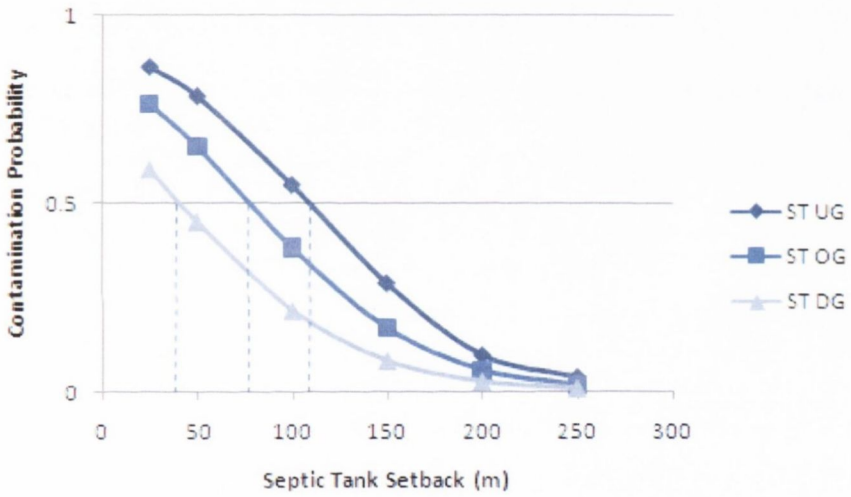


Figure 11.6 Example of Contamination Probability Vs Septic Tank System Setback Distance at 100mm 120-hr precipitation and varying septic tank setback gradients from wellhead

11.5.2 Point Agricultural Sources

To date, groundwater protection responses are available for a number hazards, including landfills, landspreading and septic tanks. However, the current study found that point agricultural sources, including farmyards, animal housing units, animal feed storage and dungsteads were adjudged to be significant impacters on private groundwater quality, particularly in *High* or *Extreme* vulnerability areas and in boreholes in these areas (see Chapter 9). Both setback distance and gradient were adjudged to be significantly associated with the presence of *E. coli* in a number of datasets and settings. It is therefore recommended that new wells be located up gradient of point agricultural sources of potential contamination and a minimum setback distance of 100 m, particularly with respect to hand-dug wells and boreholes located in limestone/sandstone areas (see Sections 9.5.7 and 9.5.9).

11.5.3 Landspreading/Grazing Animals

In the context of the current study, due to the difficulty encountered in separating grazing land and land on which organic wastes may be spread, these potentially contaminating activities have been amalgamated, as it is presumed the majority of grazing land in Ireland is spread with organic materials during the year. For all other farm activities generating slurries, landspreading should be carried out in accordance with the following guidance: (a) Control of Farm Pollution (DAFF, 1992); and (b) Code of Good Agricultural Practice to Protect Waters from Pollution by Nitrates (DoE & DAFF, 1996), incorporating current Teagasc crop fertility recommendations and production techniques. The risk to private groundwater from landspreading and grazing activities is mainly influenced by:

- the chemical and microbiological content of the waste;
- the loading rate, method and timing of application;
- the groundwater vulnerability;
- the proximity of a groundwater source;
- the groundwater resource;
- the type and state of vegetation;
- the weather.

The topsoil and subsoil, depending on their type, permeability and thickness, play a critical role in preventing groundwater contamination and mitigating the impact of many potential pollutants. Therefore, the groundwater vulnerability may be viewed as the most important factor in deciding on the control measures for any area. The present study found that typically, private wells located in *High/Extreme* vulnerability areas were most significantly affected by landspreading/grazing activities, regardless of bedrock type, and that boreholes in particular were particularly sensitive to adjacent landspreading and/or animal grazing. As set out in *Groundwater Protection Schemes* (DoELG/EPA/GSI, 1999), *Groundwater Protection Response Matrix for Landspreading* (Table 11.2), it is recommended that in High/Extreme vulnerability areas, landspreading should not take place within 100 m of the wellhead, and that normal good practice should be exercised in *Low/Moderate* vulnerability areas.

Table 11.2 Response Matrix for Landspreading (DoELG/EPA/GSI, 1999)

VULNERABILITY RATING	SOURCE PROTECTION AREA		RESOURCE PROTECTION Aquifer Category					
			Regionally Important (R)		Locally Important (L)		Poor Aquifers (P)	
	Inner	Outer	Rk	Rf/Rg	Lm/Lg	Ll	Pl	Pu
Extreme (E)	R4	R4	R3[2]	R3[2]	R3[1]	R3[1]	R3[1]	R3[1]
High (H)	R4	R2[1]	R1	R1	R1	R1	R1	R1
Moderate (M)	R3[3]	R2[1]	R1	R1	R1	R1	R1	R1
Low (L)	R3[3]	R2[1]	R1	R1	R1	R1	R1	R1

R1 Acceptable, subject to normal good practice.
R2[1] Acceptable subject to a maximum organic nitrogen load not exceeding 170 kg/hectare/yr.
R3[1] Not generally acceptable, unless a consistent minimum thickness of 1 m of soil and subsoil
R3[2] Not generally acceptable, unless a consistent minimum thickness of 2 m of soil and subsoil
R3[3] Not generally acceptable, unless it is shown that there are no alternative areas available (No spreading will be allowed within a 50 m radius of a groundwater source.)
R4 Not acceptable.

In the case of animal grazing, however, it is not practical to recommend a 100 m "non-grazing" radius, as this would effectively exclude up to 30,000 m^2 of land from production around each well. It is therefore recommended that, where possible, a 10 m secure boundary be constructed and maintained surrounding the wellhead, to exclude grazing animals. (see Figures 11.7 and 11.14).



Figure 11.7 Private well located on grazing land in Co Westmeath with no secure boundary present

11.5.4 Surface Water Bodies

Surface water bodies, including surface springs, surface water drainage ditches, rivers, lakes and streams, represent the static watertable at a particular point, and are typically therefore associated with high adjacent watertables (i.e. shallow overlying subsoil layers of varying permeability's). Furthermore, during particular periods, surface water bodies may contribute to groundwater and therefore present ready access for contaminant ingress. For example, work undertaken during the present study found that private groundwater sources located within 100 m of a surface spring in the predominantly *High/Extreme* vulnerability SA2 (Co Wicklow), were over 7 times (Odds Ratio 7.312) more likely to have *E. coli* present. It is therefore recommended that, where possible, private groundwater sources should be located up-gradient of surface water bodies and a setback distance of 100 m maintained.

11.5.5 Site Ponding

Although on-site ponding is not considered a potential contaminant source in itself, it is typically indicative of poor on-site drainage (low permeability subsoil) and depressed topography, indicative of susceptibility to flooding. If there is insufficient permeability in the surrounding subsoils to accommodate potential contaminant loadings from hazard sources, surface ponding and subsequent runoff of contaminants may occur, particularly in *Low* vulnerability areas, where low permeability subsoils are predominate and where wellheads are poorly/incorrectly completed. For example, bivariate analysis within the total dataset associated with the present study found that where on-site ponding was recorded, wells were approximately 1.8 times more likely to show evidence of bacterial contamination, regardless of well type. It is therefore recommended that GSI vulnerability

maps (www.gsi.ie) be consulted prior to location of a new well in order to gauge the potential for surface runoff during high precipitation periods. Furthermore, Teagasc soil testing may be carried out to investigate the soil type and permeability prior to well location. Additionally, it is recommended that new and existing well sites be assessed during winter months (high precipitation) and a thorough visual assessment of the proposed well area be undertaken to ensure that the well is not located in a poorly drained area i.e. exclusion of a potential contamination pathway. The presence of vegetative indicators of poor drainage should also be assessed.

11.5.6 Roads

The present study found that the proximity of roads was significantly associated with contamination of private groundwater sources in Ireland, particularly in boreholes and areas underlain by limestones or sandstones. A number of potential contamination mechanisms may be created or supported by road proximity, including:

- Direct surface runoff, particularly from agricultural traffic during high precipitation periods,
- Creation of preferential flowpaths within the surrounding aquifer during road construction and/or maintenance works, and,
- Damage to well structure including potential cracking of well lining or grout seals due to vibration caused by adjacent vehicular traffic.

It is therefore recommended that, where possible, wellhead setback from roads be maximized based on site availability and that wells be located up-gradient of the road.

11.6 Design and Construction of New Well

As mentioned at the outset of this chapter, it is acknowledged by the author that, particularly in the case of private borehole design and construction, the majority of control pertaining to design and construction parameters, including well depth, well diameter, drilling method, depth of lining, diameter of lining, is exercised by the drilling contractor and in most cases will depend upon the drilling equipment available, the local hydrogeological characteristics and the experience/expertise of the driller. However, there are a number of "sanitary protection" measures that may be carried out or insisted upon by the well user during and after well installation. Furthermore, it is considered that even with limited control over design and construction parameters, improved awareness on behalf of the well owner of these parameters and their potential effects with regard long-term contamination risks, may lead to improved overall design and construction. Therefore the following section is separated into two distinct sub-sections:

- A list of design and construction parameters controlled by the drilling contractor, including appropriate and inappropriate practice (Table 11.3).
- Potential sanitary protection measures which may be implemented by the well owner to decrease contamination susceptibility

11.6.1 Well Design and Construction

As previously noted by Wright (1995), Ball (2000) and Daly (2004), in Ireland a private borehole (or hand-dug well) is perhaps the most notable feature of a new or existing dwelling that is not currently legislated by any formal regulations, standards or planning conditions. Table 11.3 is intended to provide a checklist for well owners of some of the important well design and construction parameters and, as such, is not an exhaustive or prescriptive guide to well design or construction. For a more comprehensive overview of well construction techniques, it is recommended that the reader consult Guidelines for Drilling Wells for Private Water Supplies (IGI, 2007). Furthermore, thorough reviews of well design and construction are provided by Misstear *et al.* (2006) and Bloetscher *et al.* (2007).

Table 11.3 Recommended practices for well design and construction

	Recommended Practice	Objective
Well Drilling	Drilling fluids should be free from contaminants	Minimise the risk of contaminants being introduced to the well via drilling equipment or drilling fluid
Permanent Casing: Type	British Standard, oil industry (AP) standard or water well standard (Not drainage or sewer pipes!)	Seal off shallow, unstable or contaminated ground; exclude shallow groundwater or surface water from well
Permanent Casing: Material	Steel; Well grade thermoplastic	Seal off shallow, unstable or contaminated ground; exclude shallow groundwater or surface water from well
Permanent Casing: Jointing	Welded; screwed and socketed (Not "push-fit")	Seal off shallow, unstable or contaminated ground; exclude shallow groundwater or surface water from well
Permanent Casing: Diameter	Suitable for installation of dip tube and rising main; Suitable for installation of grouting i.e. 50 mm annulus	Seal off shallow, unstable or contaminated ground; exclude shallow groundwater or surface water from well
Permanent Casing: Depth	In rock, casing should be installed and grouted to a minimum depth of 20 m below ground, or to 10 m into bedrock, whichever is the greater.	Seal off shallow, unstable or contaminated ground; exclude shallow groundwater or surface water from well
Grout Seals	Pressure grouted from base of permanent casing to surface	Seal off shallow, unstable or contaminated ground; exclude shallow groundwater or surface water from well

From Daly, 2004; IGI, 2007; SEPA, 2010

11.6.2 Wellhead Completion

The wellhead is described as the part of the well which exists at the ground surface. Typically, three wellhead completion configurations may be employed, namely above ground, "at-surface" and underground (Figure 11.8). The current study found that where the wellhead was appropriately

completed and located, no difference existed with regard to contamination probability between underground and above ground wellhead completion. However, a significantly higher level of *E. coli* presence was found in wells where "at-surface" completion was employed, regardless of wellhead completion or location. This is probably due to direct surface runoff ingress being a major contamination mechanism at these wells. It is therefore recommended that "at-surface" wellhead completion is not used and, where possible, above ground completion is preferable, due to potential drainage accumulation in below ground wellhead chambers (see Section 11.6.4). In areas where vehicular traffic is likely, underground wellhead completion may be preferable. See Figures 11.11 and 11.12 for recommended wellhead completion configurations.



Borehole wellheads finished; A) below ground level, B) above ground level, and, C) at ground level

Figure 11.8 Borehole Wellhead Completion Configurations

11.6.3 Liner/Casing Clearance

The permanent casing should protrude above the surface of the chamber floor, or where a chamber is not employed, above the concrete slab. This provides adequate clearance in order that surface runoff and/or drainage accumulation cannot gain direct ingress over the top of the casing. A minimum of 100 mm clearance should be maintained, with up to 300 mm clearance preferable (IGI, 2007; SEPA, 2010), where possible (See Figures 11.11 and 11.12).

11.6.4 Wellhead and Liner Cover

In order to prevent accumulation of water or contaminated drainage in the wellhead chamber, particularly in cases where the wellhead has been completed at or below ground level, it is recommended that an appropriate wellhead cover be used. As set out by SEPA (2010), it is recommended that wellhead covers be load-bearing to suit traffic, completely cover the chamber frame and consequently provide a water-proof seal. Similarly, an appropriate liner/casing cap should

be fitted in order to prevent surface runoff (in the case of "at-ground" wellhead completion) or contaminated drainage accumulation (in the case of below ground wellhead completion) gaining direct entry to the well. Daly (2004) recommends a securely fitted, watertight liner cap (see Figure 11.9). This is particularly important in *Low* vulnerability areas, with higher levels of overland flow and accumulated drainage, for example, the presence of an appropriate liner cap (and liner clearance) accounted for approximately 44% of variation in the SA4 (*Low* vulnerability) LR model.



Figure 11.9 Poorly covered hand-dug wellheads (A & B), Poor borehole casing cap (C), and Correctly finished wellhead with steel wellhead cover (D)

11.6.5 Pump Chamber

It is recommended that, regardless of the wellhead configuration, a wellhead chamber should be constructed around the wellhead, to protect the liner and fittings. The wellhead chamber base should be constructed of 150 mm concrete, with <100 mm thick concrete or natural ground being avoided (Daly, 2004; IGI, 2007; SEPA, 2010). Drainage should be provided to minimize accumulation of water on the chamber base. This should be 25 mm ID (Daly, 2004; IGI, 2007), leading to a surface outlet (overground chamber) or effective soakaway (underground chamber). The chamber sides should be constructed with precast concrete sections, engineering brick or waterproof rendered brick/blockwork (bonded to base). It is important that the chamber is waterproof, and therefore "non-waterproof" construction materials are not recommended. Furthermore, waterproofing should be ensured, by regular inspection and maintenance i.e. repairing of cracking, maintenance of drainage, etc.

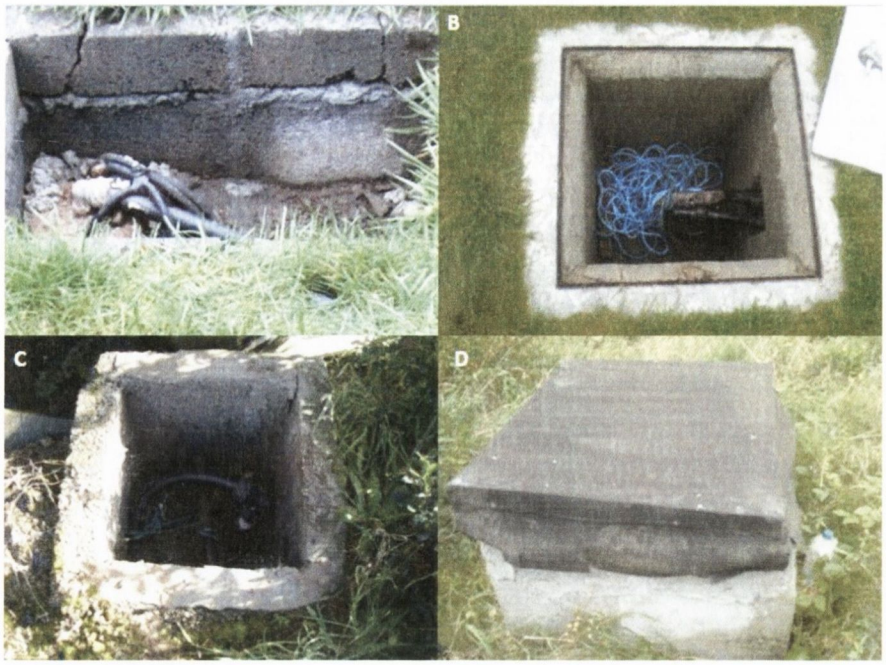


Figure 11.10 A) Well chamber cracking, B) Underground well chamber, C) Underground well chamber with drainage accumulation and "non-waterproof" construction materials, D) Overground well chamber

11.6.6 Concrete Surround

In order to avoid surface runoff and/or shallow groundwater infiltration directly adjacent to the wellhead, it is recommended that a concrete slab of 150 mm thickness be keyed in around the casing (Daly, 2004). This should be installed to a distance at least 0.5 m surrounding the casing, with 1 m recommended (i.e. "direct wellhead vicinity"), and should be properly maintained (see Section 11.7.3)

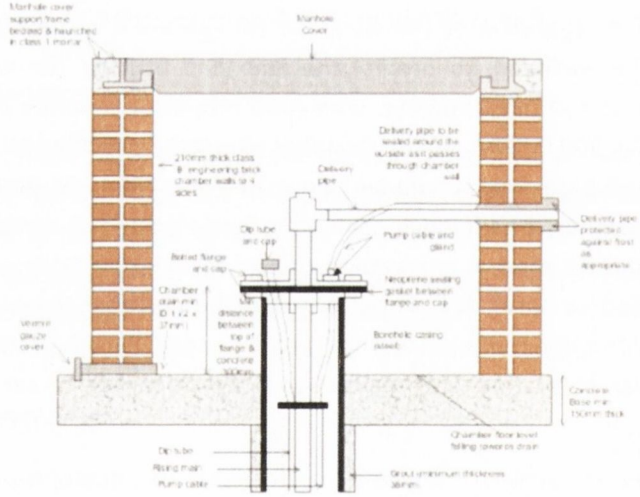


Figure 11.11 Recommended overground wellhead completion (SEPA, 2010)

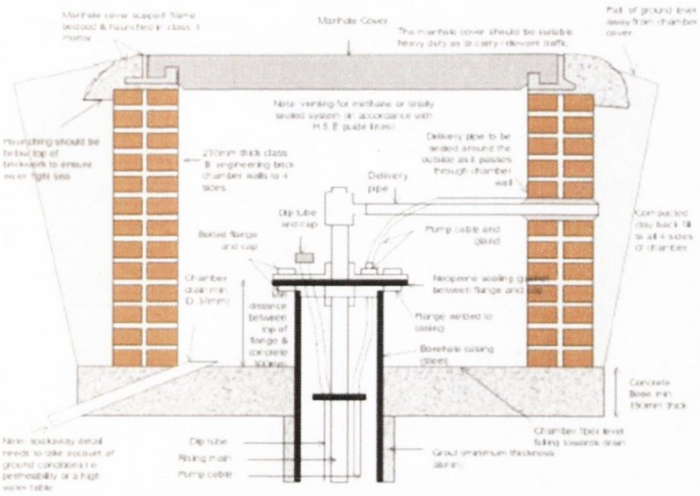


Figure 11.12 Recommended underground wellhead completion (SEPA, 2010)

11.7 Consumer control/protection

As outlined by EPA (2010), "*the sources of contamination of groundwater quality may be unknown, or are beyond the control of the owner of the supply . . . . . and although general improvements in well design, knowledge of source protection and good land-use practice are essential, these may not eliminate the risk of contamination of private sources*". However, there are a number of "secondary (consumer) controls" available to well shareholders that may considerably reduce the likelihood of well users consuming groundwater contaminants, particularly potentially infectious pathogens, namely:

1. Installation of appropriate water treatment system,
2. Regular water quality analysis, and,
3. Regular well maintenance.

All three consumer control measures are effective means of protecting human health in the event of a contamination event. However, any one consumer control used in isolation cannot be viewed as an appropriate "single-barrier" groundwater protection strategy, due to the possibility of failure of any one measure over time. Therefore a "multi-barrier" approach is recommended, whereby all three measures are employed concurrently.

11.7.1 Water Treatment

The awareness study carried out during this study (see Chapter 8) found that approximately 69% of groundwater users in Ireland acknowledged that no water treatment system of any kind was currently in use at their source. Further analysis found that 58% of respondents classified domestic ion-exchange water softeners as their primary treatment process. Previous studies have found that ion-exchange processes (softeners) generally do not affect pathogen or microbial indicator populations in raw water and are therefore ineffective as pathogen treatment systems. Therefore, approximately 13% of respondents had an appropriate water treatment system in place (i.e. 0.42 x (100 – 69). Furthermore, a significantly lower level of treatment was in evidence with regards to

hand-dug wells, which due to their typical design and construction are more susceptible to pathogenic ingress. Therefore, Table 11.4 is provided, to aid users of new and existing private wells on choosing an appropriate water treatment system for their household.

Table 11.4 Treatment Process Selection

Treatment Process	Contaminants Excluded	Domestic Expense	Appropriate pathogen treatment for private wells
Water Softener	Calcium, Magnesium, Bicarbonates, Sulphates	Low	No
Chlorination	Bacteria	Low	No
Media Filtration	Bacteria, Algae, Cysts, Silt, Sand, Clays	Low	Yes
Microfiltration	Bacteria, Algae, Cysts, Clays, Viruses	Moderate/High	Yes
Ultrafiltration	Viruses, Humic Acids, Aqueous Salts, Metal Ions	High	Yes
Nanofiltration	Humic Acids, Aqueous Salts, Metal Ions	High	Yes
Reverse Osmosis	Aqueous Salts, Metal Ions	High	Yes
UV Disinfection	Bacteria, Protozoa, Viruses	Moderate	Yes
Oxidation	Bacteria, Protozoa, Viruses	Moderate/High	Yes

It is important to note that the above table is not prescriptive, and therefore advice should be sought from a water treatment professional prior to choosing an appropriate water treatment system, or combination of systems. Additionally, it is recommended that at least one water sample be collected and analysed from the groundwater source prior to installation of a water treatment system, in order to inform both the well owner and water treatment professional with respect to contaminants which may be present and need to be removed.

11.7.2 Water Quality Analysis

This research has shown that approximately 40% of private wells in Ireland have not been previously analysed for the presence of chemical or microbial contamination. As previously noted by Daly (2004), regular analysis of water quality provides direct information on contamination of wells, although it is important to understand that periodic monitoring only provides an indication of current contamination and is not suggestive of prior or future contamination events i.e. a well which does not have microbial indicators present during summer may become contaminated during winter, or perhaps even shortly after the sample was analysed. It is also important to note that once contaminants reach groundwater bodies, they may remain there for weeks, months and even years, due to very low flow rates.

It is recommended that water quality analyses of private groundwater sources should be carried out:

- Directly after construction of the well,
- Prior to initiation of treatment,
- After initiation of treatment,
- At least once per year, and if possible, seasonally i.e. four times per year,
- Upon any noticeable change of colour, taste, smell or pressure.

It is recommended that well owners contact their local authority (Water or Environment Section) or local Health Service Executive (HSE) in the case of water quality analysis being required. Also, the well owner should ensure that water quality analyses are undertaken in accredited laboratories by qualified personnel.

Upon implementation of water quality analysis, although it is expected that qualified personnel will provide adequate interpretation of results, it is considered favorable that well users should also be able to interpret results, in order that likely sources of contamination (where present) are recognised, and consequently, appropriate remedial actions can be taken. The following table presents an interpretive guide to water quality parameters, associated thresholds and potential sources of contamination (where thresholds are exceeded).

Table 11.5 Water Quality Assessment

Indicator Parameter	Threshold (mg/l)	Assessment
***E. coli* present**	N/A	Organic waste source nearby (excl karst areas), usually septic tank or agriculture
***E. coli* absent**	N/A	Uncontaminated or die-off due to attenuation or time of travel
NO_3>25mg L^{-1}	25	Organic fertiliser or organic waste source nearby
NH_4>0.15mg L^{-1}	0.15	Organic waste source nearby, fertilisers not an issue
K > 5.0mg L^{-1}	4	Organic waste source nearby
K:Na Ratio >0.4	0.4	Agricultural waste rather than septic tank effluent, if <0.4 no conclusion possible
Cl >30 mg L^{-1}	25-50	Organic waste source (excl areas close to coastline <20km)

11.7.3 Well Maintenance

Furthermore, the awareness study has shown that approximately 65% of respondents acknowledge that their private wells have not had previous maintenance works carried out. A higher level of maintenance was apparent among hand-dug well users, although this proportion was less than 50% (46%). For the purposes of this guidance, general maintenance works have been separated into three distinct types:

- Well maintenance
- Wellhead maintenance
- Site Maintenance

a. Well Maintenance

This refers to subsurface maintenance and comprises two main maintenance tasks: a) disinfection, and, b) well pumping/purging.

Disinfection of the well is the last step in well construction and is recommended to ensure a safe groundwater supply (Macler & Merkle, 2000), with Lehr *et al.* (1988) suggesting that this procedure is often neglected by drilling contractors and/or well diggers. Furthermore, disinfection may be carried out at any stage during the lifecycle of the well. The following are recommended times at which disinfection may take place (Adapted from Daly, 2004):

- Directly after construction of the well,
- Prior to initiation of treatment,
- At least once per year (each Autumn is suggested),
- Any noticeable change of colour, taste, smell or pressure,
- Following water quality analysis, whereby total or faecal coliforms were present.

A number of guidelines are available for recommended well disinfection procedures including HSE South East (2000), AWWA (2004), Bloetscher *et al.* (2007) and CNA Environmental Inc (2005), with the majority of disinfection methods involving chlorine in some form, typically hypochlorite (Westerhoff *et al.*, 2005). The well disinfection guidelines outlined below are those recommended by IGI (2007).

Materials

1. 900 g chloride of lime (containing 35% available chlorine), or
2. 450 g "High test Hypochlorite" (50-70% available chlorine), or
3. 9 L 3% Sodium Hypochlorite (3-5% available chlorine)

For materials 1 or 2, dissolve in 23 L water. For material 3, make up to 23 L with water. Mix thoroughly.

Pour half of solution into well, start pump and open all taps until water from each tap has a distinct chlorine small i.e. distribution system is inundated. Stop pump and add the remainder of disinfectant solution. Allow to stand for 12-24 hours. Pump to waste until chlorine smell has cleared.

Well purging or pumping is typically carried out upon completion of well construction and/or disinfection, as a well development technique, however, it may be carried out at any time during the lifetime of the well in order to "flush" the well of contaminants which may be present, in addition to sediments in the groundwater, particularly after heavy rainfall.

b. Wellhead maintenance

The wellhead should be regularly assessed to ensure that contaminants cannot directly enter the well at the surface, and that the wellhead itself is not a source of contamination. A number of maintenance objectives and associated recommendations are presented in Table 11.6. Figure 11.13 shows an example of a poorly maintained wellhead.

Table 11.6 Wellhead maintenance objectives and recommendations

Maintenance Objective	Maintenance Recommendation
Removal of corrosion or deposits from pipes and fittings i.e. rising main, well liner, etc	Dismantle & High pressure jetting Dismantle & brushing/swabbing Dismantle & chemical treatment
Ensure that surface runoff cannot enter well	Install appropriate liner cap/wellhead cover Replace faulty/inappropriate well cap/wellhead cover
Ensure that surface ponding does not occur in wellhead chamber (if present)	Regularly clean wellhead chamber and remove sediment build-up/detritus where evident Ensure that wellhead chamber is suitably drained
Ensure that potential hazard sources are not located immediately adjacent to wellhead	Regularly assess wellhead vicinity and clean when necessary, including removal of domestic animal faeces, bags, fuel, etc (See Figure 11.13)

Adapted from Misstear *et al.*, 2006



Figure 11.13 Poorly maintained wellhead with detritus (waste materials) evident

c. Site Maintenance

Similar to wellhead maintenance, the primary objective underlying general site maintenance is the prevention of well contamination by ensuring that potential hazard sources are not in the immediate well vicinity, and that direct contaminant ingress cannot occur at the surface. Preventative actions may be directly related to appropriate well location (i.e. setback distance and gradient from potential hazard sources), which have been outlined previously. Notably, however, the generalised model of well susceptibility, outlined in Chapter 9, which integrates data from over 200 private wells in Ireland, found that the assessed ground condition within 10 m of the wellhead (i.e. surface ponding/drainage, animal faeces, general state of repair) was a major contributory factor in predicting the presence of microbial contamination in the well (Figure 11.14). For example, the absence of a secure 10 m boundary associated with wells in *High/Extreme* vulnerability areas was

found to be associated with increased bacterial contamination. A number of remedial actions may be taken during the lifecycle of the well, to minimize contamination risks (Table 11.7)

Table 11.7 On-site maintenance objectives and recommendations

Maintenance Objective	Maintenance Recommendation
Ensure that a "secure boundary" is maintained to exclude animals from immediate wellhead vicinity	Construction and regular upkeep of a securely fenced boundary around wellhead
Ensure that on-site surface ponding does not occur	Maintenance of adequate drainage in wellhead vicinity
Ensure that potential hazard sources are not located immediately adjacent to wellhead	Regular upkeep of site to remove detritus including waste materials, etc

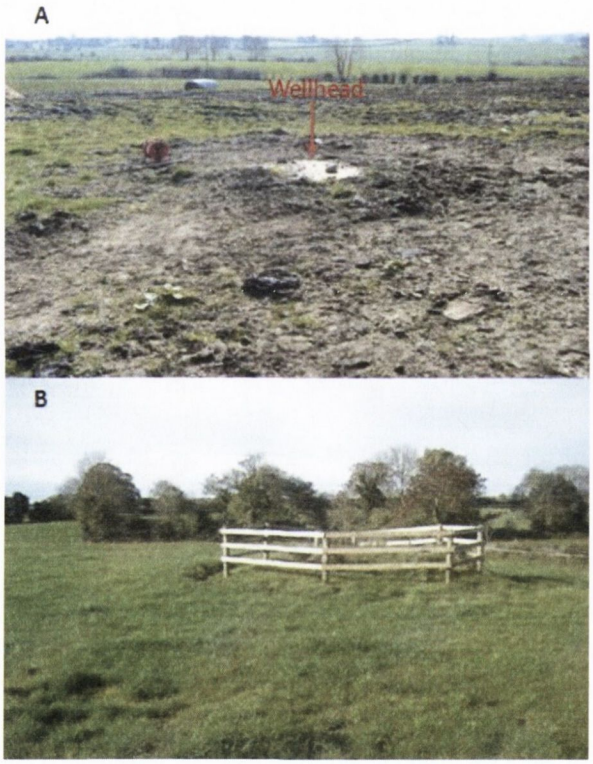


Figure 11.14 A) Poorly maintained on-site conditions with "secure boundary" absent and animal faecal matter in close proximity to wellhead; B) Suitably maintained wellhead area with appropriately maintained "secure boundary"

Chapter 12: Conclusions and Recommendations

12.1 Introduction

The following chapter presents conclusions and recommendations pertaining to the research study presented in this doctoral thesis, which has included comprehensive data acquisition, analysis and modelling. It is considered that a number of novel aspects have been incorporated, including the development of a detailed private well database, consolidating well-specific, site-specific and water quality data, which may be used as a foundation for future similar research. Furthermore, the development and completion of a "groundwater awareness survey" with over 500 respondents has culminated in the identification of a number of knowledge gaps among private well users in Ireland, and as no survey has previously been undertaken, it is envisaged that these results may be used for future development of measures to support relevant environmental policy. The development of a multivariate model using hierarchical logistic regression with step-wise parameter entry which may be used to predict, and therefore prevent, pathogenic contamination of private wells in Ireland has also not previously been completed. This model has been verified using an independent "training" dataset and is considered applicable on a national basis. Lastly, an iterative quantitative microbial risk assessment (QMRA) has been undertaken, again for the first time in Ireland, using novel data and techniques (Monte Carlo simulation modeling; correlation coefficient sensitivity analysis), resulting in original and previously unknown estimates of the current human health burden posed by private well contamination in Ireland. Moreover, a number of recommendations are made, based on model outputs, which may be used to inform risk minimization and prevention measures.

12.2 Conclusions

Research Objective 1

(i) Groundwater User Awareness

While 70% of 545 interviewed respondents (RA + SA) acknowledged the participation of a water-diviner during the well design phase, only 16% and 1.2% of respondents acknowledged the involvement of local authorities and a professional hydrogeologist, respectively. This finding is a source of concern, as the primary purpose for employment of a water diviner at the well design stage is not the location of sustainable groundwater quality, with many well drilling contractors privately employing water-diviners.

Although overall consumer awareness was higher than expected among private groundwater users, a number of potentially significant knowledge gaps were identified. A low level of awareness was noted with regard to waterproof sealing of private groundwater sources, with borehole users especially displaying a very low awareness of appropriate sealing of their source. This is of particular concern in light of earlier findings suggesting that surface ingress is a significant contamination mechanism in private groundwater sources in Ireland. Furthermore, although there was a relatively high level of awareness with regard to *E. coli*, *Cryptosporidium* spp. and nitrates as potential groundwater contaminants, very low levels of awareness were evident with regard to a number of other potential contaminants including *Giardia* spp. and rotavirus. Additionally, less than half of respondents exhibited an awareness of at least one illness or symptom caused by consumption of

contaminated groundwater. Again, this represents cause for concern, as this knowledge gap may contribute to the underreporting of waterborne infections attributable to private groundwater sources in Ireland.

Approximately 14%, 31% and 24% of 245 interviewed respondents (RA) were unaware of the potential threat of adjacent slurry/fertiliser spreading, grazing animals and septic tanks, respectively. As the majority of private groundwater sources are located in rural areas and these three contaminant sources in particular are found throughout rural Ireland, it is therefore important that these knowledge gaps should be addressed in future.

An investigation of the level of "consumer control/susceptibility" (i.e. treatment, maintenance and water quality analysis found that although there was a relatively high awareness of the presence of these consumer controls, approximately 68%, 65% and 40% of 545 respondents acknowledged that treatment, previous maintenance and water quality analysis, respectively, were not carried out at their source.

An awareness scoring methodology found that hand-dug well users exhibited a higher overall level of both source and contamination awareness. No significant relationship was found to exist between source or contamination awareness and *E. coli* presence or magnitude (where present) in respondent sources. Therefore, no quantifiable link was found between respondent awareness and source susceptibility. However, both source and contamination awareness were found to affect the likelihood of "consumer controls" being in place, which in turn would affect the probability of a consumer contracting an infection subsequent to their source becoming contaminated.

Research Objective 2

(ii) Private Groundwater Source Contamination

Approximately 29% of wells (n = 62) tested positive for *E. coli*, of which 68% were boreholes, and 32% were shallow hand-dug wells. Proportionately, boreholes and hand-dug wells exhibited positive "one-off" bacterial results in 24.8% and 47.6% of cases, respectively, suggesting that shallow, hand-dug wells are more susceptible to microbial ingress and therefore pose a greater potential risk to well users than deeper boreholes. A high level of spatial variability was shown with respect to *E. coli* contamination of private wells, with contamination rates ranging from 45% in the *Low* vulnerability study area to 11% in a *High* vulnerability study area, with lower than expected levels of *E. coli* in areas categorised as being predominantly of *High* and *Extreme* groundwater vulnerability. The higher than expected level of bacterial contamination encountered in the *Low* vulnerability study area suggests that source design and/or construction characteristics are primary risk factors with regard *E. coli* contamination. It is therefore hypothesised that contamination in *Low* vulnerability areas may occur largely due to surface ingress at the wellhead, while contamination in *High/Extreme* vulnerability areas may occur due to both subsurface infiltration as well as surface ingress. A mean K:Na ratio of 0.22 was found in wells with no *E. coli* present, while a mean ratio of 0.48 was found where *E. coli* were present, suggesting agricultural contamination sources are a concern in Irish groundwater sources.

A mean measured nitrate concentration of 12.8 mg L^{-1} NO_3 was found among the 211 sampled wells, with only one exceedance of the EU MAC (50 mg L^{-1} NO_3) recorded. Therefore it is concluded that, nitrates from private groundwater sources assessed as part of this study do not currently pose a significant human health burden. Moreover, over the 20-month monitoring period, a slight mean decrease was noted, possibly due to improving agricultural practices and/or environmental legislation, which is reflective of current national trends.

Bayesian Inference Criterion (BIC) two-step clustering resulted in three distinct physicochemical clusters, based on "one-off" spatial groundwater sampling and analysis. These clusters were characterised as boreholes located in predominantly limestone or sandstone areas (Cluster 1), boreholes located in predominantly granite or Namurian areas (Cluster 2) and hand-dug wells (Cluster 3). Further analysis found that there was a statistically significant association between cluster membership and *E. coli* presence, suggesting that these clusters not only appropriately describe the source type and geology, but may also be used to describe source susceptibility. Notably, no association existed between cluster membership and groundwater vulnerability category, reinforcing the hypothesis that source susceptibility (particularly susceptibility to microbial contamination) and groundwater vulnerability category have a greater level of independence than generally presumed i.e. groundwater vulnerability categorisation alone, although useful, only represents a portion of overall source susceptibility, and does not indicate the full contamination risk.

Overall, 93% of continuously monitored wells showed evidence of *E. coli* contamination at least once during the 20-month monitoring period. It is important to note that a proportion of continuously monitored wells were based upon previous contamination being evident. However, only 40% of these showed evidence of *E. coli* contamination upon initial "one-off" sampling. Therefore, it would appear that for microbial analysis, seasonal sampling was important, with a high level of variability and seasonality evident. Typically, *E. coli* presence and magnitude (where present) was increased during mid to late summer and early to mid winter at all monitored sources, with agricultural contamination sources, increased septic tank loadings and increased precipitation rates being likely causes of these peaks. A higher than expected rate of seasonal *E. coli* contamination was again encountered in the *Low* groundwater vulnerability area, with *High/Extreme* vulnerability areas both having lower rates of monthly contamination. Hand-dug wells typically had higher levels of seasonal *E. coli* contamination than boreholes, with mean contamination rates of 53.5% and 30% attributed to hand-dug wells and boreholes, respectively.

Typically, higher rainfall intensity during the 120-hour and 30-day periods prior to sampling was found where *E. coli* were present, particularly among hand-dug wells. Furthermore, a significant correlation between *E. coli* and the level of precipitation in the 120-hour period prior to sampling was found at the *Low* vulnerability study area, suggesting rapid ingress of contaminants to these sources as a direct result of short-term precipitation. Overall, correlations between *E. coli* presence and effective precipitation rates (potential recharge) were less than measured associations between *E. coli* presence and total precipitation rates. This is suggestive of rapid ingress due to poor wellhead design or lack of sanitary seal (i.e. preferential flow at well interface), as opposed to relatively slow contaminant recharge through subsoil layers.

Overall, 4.9% of sampled "one-off" sources had evidence of *Cryptosporidium* spp. present, with none found to have *Giardia* spp. present. Due to the relatively small number of wells sampled and analysed for protozoan contamination (n = 81), it is difficult to make general conclusions regarding the specific sources or mechanisms associated with ingress of protozoa to private wells in Ireland. However, as the 4.9% *Cryptosporidium* spp. incidence rate is generally consistent with a number of previous international studies, it may be positively inferred that private rural wells in Ireland constitute an environmental source of exposure to highly infectious protozoan organisms.

(iii) Private Groundwater Source Susceptibility

Hierarchical logistic regression with stepwise parameter entry was successfully used to develop a number of predictive susceptibility models pertaining to *E. coli* presence (It was not possible to develop a predictive model for *Cryptosporidium* spp. due to the low levels encountered). Localised models, although similar, did exhibit inconsistencies, suggesting spatial variability with regard to private groundwater susceptibility in Ireland. For example, the majority of localised models included septic tank location with respect to the wellhead as a significant predictive parameter. However, this was not the case for the model in Study Area 1 (SA1), which implicated both point agriculture and non-point agriculture as the primary hazard sources. This suggests that although groundwater vulnerability may be used as an indicator of contamination probability, the susceptibility of specific areas and sources should be quantified independently. The generalised model that was developed, based upon all surveyed sources, suggests that the main parameters of concern with respect to *E. coli* presence are septic tank location with respect to the source, soil type and thickness (vulnerability), well design and construction characteristics (particularly liner clearance, wellhead cover, and wellhead ground condition), and 120-hour precipitation rate. Typically, point sources of contamination were found to be more significant model inputs than non-point sources. Furthermore, as ground condition close to the wellhead (<10m) was found to be a significant input parameter, and this may be considered to be a "consumer control", it is concluded that increased overall awareness on the part of well owners/users, particularly with respect to source maintenance, may in fact decrease source susceptibility.

An independent validation set was used to test the generalised model. The total generalised model provided a high level of verification, with a validation accuracy of >80%, comparable with the overall training set accuracy of 84%, and a particularly high level of accuracy with regard to prediction of bacterial presence (100%). Therefore, it is concluded that the developed model has a high level of both explanatory and predictive value with regard to private groundwater contamination susceptibility in Ireland.

Sensitivity analyses carried out on the generalised model found that the overall "zone of influence", particularly up-gradient of the wellhead, increases with increasing 120-hour precipitation rates, particularly with respect to septic tank location, highlighting the point that atypically high precipitation rates will increase necessary setback ranges. It is therefore concluded that future guidelines or legislation should take this finding into account, and provide an appropriate "factor of safety" with respect to wellhead setback from potentially polluting activities.

Polytomous Universal Modelling (PLUM) was employed to model the measured *E. coli* magnitude, where present, with three contamination ranges (Low, Moderate, and High). The proximity of point sources of contamination, including septic tank systems and point agricultural sources, were found to be significant predictors of *E. coli* magnitude. Furthermore, the presence of visible on-site surface ponding was also a significant predictor of increased bacterial concentrations, which is suggestive of direct ingress at the wellhead and high local watertables being significant pathways for contaminants. This lends further support to the hypothesis that private sources located in *Low* vulnerability areas may in fact have higher associated levels of contamination susceptibility during wetter periods, due to increased surface ingress. Moreover, short-term precipitation rates (24-hour, 48-hour) were also significant within the final PLUM model. Typically, precipitation over longer periods (120-hour; 30-day) were found to be risk factors with regard to bacterial presence, however, contamination magnitude was more sensitive to short term precipitation; once again suggesting direct surface ingress as a significant contamination mechanism.

While PLUM modelling was considered useful for distinguishing between high and low levels of bacterial contamination, the model could not effectively predict moderate bacterial contamination levels.

Research Objective 3

(iv) Potential Human Health Implications of Private Groundwater Contamination in Ireland

Three existing deterministic health risk assessment models and a stochastic health risk assessment model developed for this research were employed to estimate the magnitude of risk to health from contamination (*E. coli*; *Cryptosporidium* spp.) of private supplies in Ireland. The developed stochastic model predicts an overall endemic crude incidence rate (CIR) of 0.7% attributable to *E. coli* contamination of private groundwater supplies in Ireland i.e. 0.7% of the private groundwater population predicted to acquire an infection annually. All three deterministic models and the developed stochastic risk assessment model predict higher crude incidence rates of waterborne illness attributable to *E. coli* among users of hand-dug wells than boreholes, with predicted hand-dug CIRs ranging from 3.1 to 5.8 times those of borehole users. Sensitivity analysis of the stochastic model suggests that the *E. coli* loading rate is the model input with the greatest overall contribution of the model outputs and therefore, improved source protection, particularly in the case of existing wells, is necessary in order to minimize the overall health burden attributable to private groundwater sources in Ireland. Particularly in the case of hand-dug wells, which have been shown to exhibit a higher level of inherent contamination susceptibility, improved source protection should be a priority.

The developed stochastic model predicts an overall endemic CIR of 0.18% attributable to *Cryptosporidium* spp. contamination of private groundwater supplies in Ireland, with two of the three deterministic models predicting similar rates (0.11 – 0.23%). Furthermore, the stochastic model predicts 0.15 mortalities per annum or one mortality every 6-7 years. Sensitivity analysis of the stochastic model inputs show that, unlike *E. coli*, the dose-response rate associated with *Cryptosporidium* spp. is the highest overall contributor to model outputs, suggesting that vulnerable sub-populations are likely to contribute significantly to the overall health burden associated with

waterborne *Cryptosporidium* spp. in Ireland and therefore "consumer protection" should be a priority. It is important to note however, that although *Cryptosporidium* spp. presence rates found during the study were comparable with previous international studies, this rate was based on a relatively low number of analysed samples.

Endemic crude incidence rates predicted by the developed model associated with both *E. coli* and *Cryptosporidium* spp. are significantly higher than reported national epidemic rates, however, due to large probable levels of under-reporting and misdiagnosis, current endemic levels associated with these organisms are not available nationally, and therefore a direct comparison is not possible.

Research Objective 4

See Chapter 11

12.3 Recommendations for Further Research

- Further work should focus on the quantification of contamination mechanisms in areas with differing groundwater vulnerability categories and well types. Furthermore, future studies should be initiated to describe differing levels of pathogen concentrations in the developed clusters. It is recommended that microbial source tracking (MST) may be a suitable technique for these investigations. The use of MST techniques would allow for accurate apportionment of pathogen sources (i.e. human or animal), which is important as it generally recognised that pathogens from human sources pose a higher level of human health burden, due to human-specificity.

- The national level of reliance on shallow hand-dug wells in Ireland is currently unknown. In order to accurately quantify the endemic human health burden posed by waterborne pathogens in Ireland, and subsequently provide suitable risk management and communication strategies, the collection of data on the usage of hand dug wells is necessary.

- A national "awareness" programme should be initiated, providing information to private groundwater users in Ireland regarding private well location, design, construction and maintenance. In particular, this programme should focus on the potential health hazards associated with pathogenic contamination of private sources, the potential sources of groundwater pathogens, and the importance of "consumer controls" in reducing secondary susceptibility to consumers.

- Future quantification of private groundwater susceptibility should include a number of parameters including depth of lining, the presence of a sanitary seal and the type of seal used (where applicable). It is considered that the inclusion of these parameters will improve both the performance and utility of the developed models as part of this study.

- As outlined in Chapter 11, the role of road proximity/setback in a number of LR models could not be ascertained as part of this study i.e. it is unclear what mechanism of contamination

may be created or supported by road proximity. It is therefore recommended that future work be carried out in order to ascertain and quantify this mechanism(s), in order to provide further groundwater protection.

- Future work should be carried out to further refine the source susceptibility characterisationmodel proposed in Chapter 9. This may result in a qualitative characterisationmethod (i.e. high, moderate or low depending on probability score) which may be effectively undertaken by groundwater shareholders, and used to provide guidance on a source-by-source basis.

- A similar study should be undertaken for a major unconsolidated aquifer, such as the Curragh sand and gravel aquifer in Co Kildare, to compare with the results from the fractured bedrock aquifers described in this thesis. Furthermore, as *Moderate* vulnerability areas were not examined as part of this research, it is also recommended that similar work be carried out in an area predominantly categorised as *Moderate* vulnerability.

- Future quantitative risk assessments (QRA) should be focused on endemic human health burdens posed by enteric viruses (rotavirus, adenovirus, norovirus) and *Giardia* spp. in private groundwater supplies in Ireland. Furthermore, the developed QRA model presented in Chapter 10 may be further refined through the use of monitoring data and future groundwater quality research data.

- Data relating to age and population-specific dose-response, consumption and exposure should be collated, with a particular emphasis on vulnerable sub-populations. The availability of these data will allow for improved risk characterisationand subsequently, improved risk management and communication strategies.

- Due to a lack of data pertaining to transient populations, acquired immunity and multiple exposures (i.e. concurrent contamination by more than one pathogen), these issues have, to date, not been considered with regard to the human health burden posed by contaminated private groundwater sources in Ireland. Future work should address these knowledge gaps.

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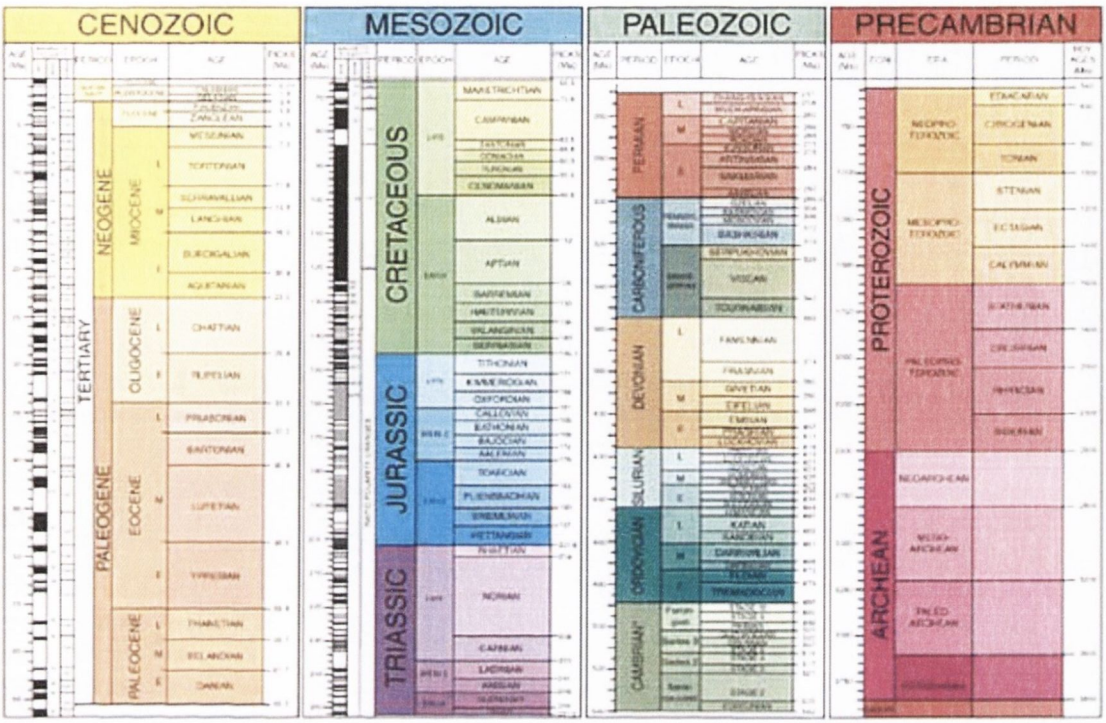
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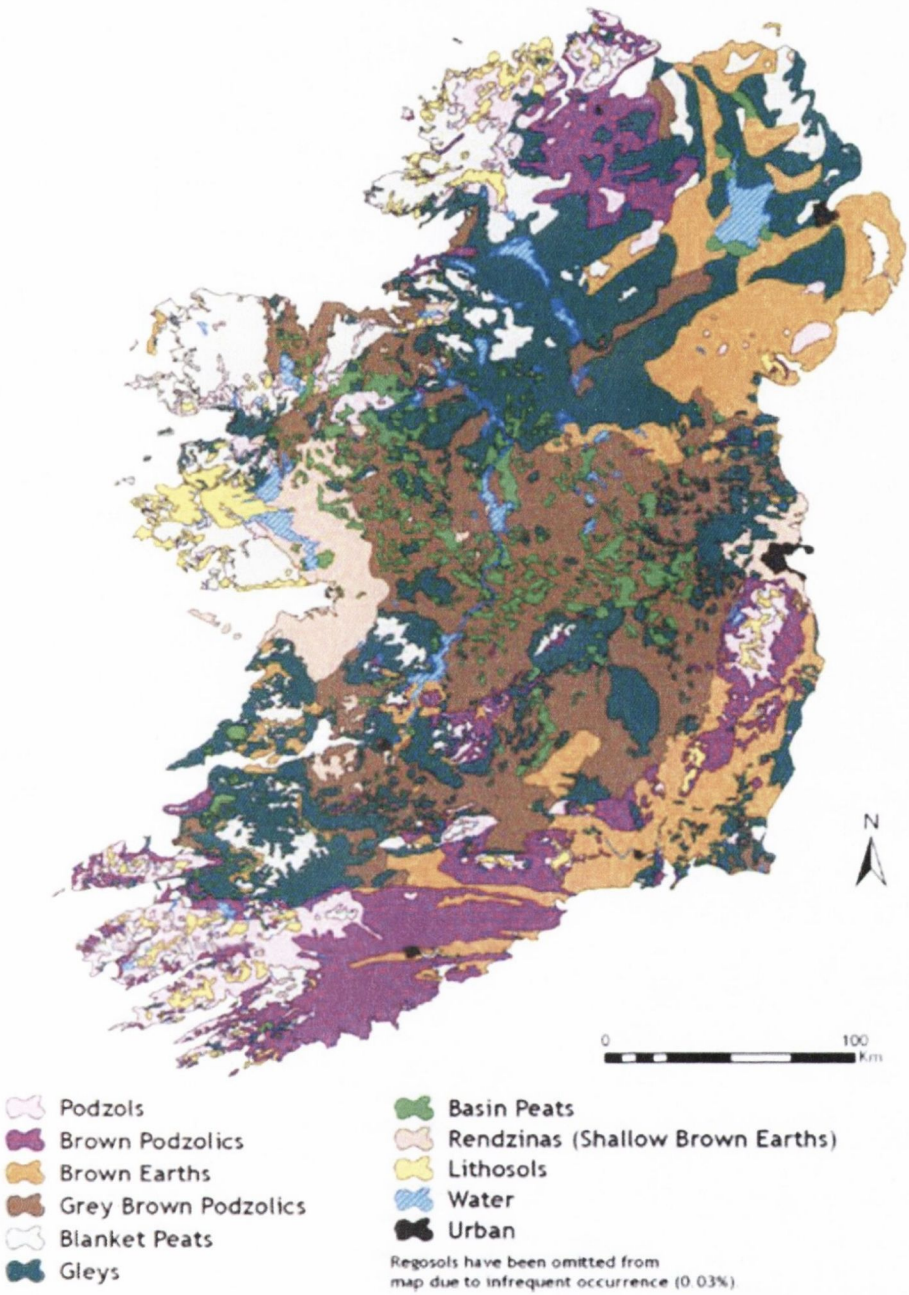
Appendix A1


CENOZOIC
MESOZOIC
PALEOZOIC
PRECAMBRIAN
TERTIARY
NEOGENE
PALEOGENE
MIOCENE
OLIGOCENE
EOCENE
PALEOCENE
CRETACEOUS
JURASSIC
TRIASSIC
PERMIAN
CARBONIFEROUS
DEVONIAN
SILURIAN
ORDOVICIAN
CAMBRIAN
PROTEROZOIC
ARCHEAN

Appendix A2

The soils in Ireland have been classified using a modification of an American soil classification system that has been applied in the United States by the Department of Agriculture since 1938. The ten main soil groups in Ireland are, as shown in *Figure 2.3* (Fay *et al.,* 2002; SSSI, 2004):

- *Podzols*: these poor, acidic soils are formed in hill and mountain areas; they are grey in colour and low in humus and can be found in Counties Wicklow, Donegal and Kerry.
- *Brown Podzolics*: these soils are similar to the Podzols but less depleted; they have a good mix of mineral and organic matter in the surface layer and can be found in Counties Cork, Waterford and Tipperary.
- *Brown Earth*: these mature, well drained soils with a uniform, brown soil horizon show high fertility and can be found in Counties Clare, Wexford and Wicklow.
- *Grey Brown Podzolics*: these soils are formed on calcareous parent material, which counteracts leaching processes; they are all-purpose soils and can be found in Counties Meath, Westmeath and Kildare.
- *Gleys*: these soils developed under waterlogged conditions; they mostly show poor physical conditions and are grey/blue in colour and can be found in Counties Monaghan, Cavan and Leitrim.
- *Blanket Peats*: these soils have accumulated in areas of high rainfall and have poor drainage capacities; they have a characteristic high organic matter content, which can be over 30 percent and are at least 30 cm deep; they can be found in Counties Galway, Mayo and Sligo.
- *Basin Peats*: these soils have formed in hollows, river valleys or lake basins under conditions where the subsoil is not permeable; there are two types: raised bogs and fen peats; they are widespread in the midlands.
- *Redzinas*: these shallow soils are not more than 50 cm deep and can be found together with Lithosols in Counties Dublin and Clare.
- *Lithosols*: these soils are very stony and are situated over solid or crushed bedrock and can be found together with Redzinas in Counties Dublin and Clare.
- *Regosols*: these soils show no horizon development and lack the B-horizon and can be found in County Waterford.



Top-soil map of Ireland (Gardiner & *Radford, 1980)*

Appendix B1

Recommended 8-step procedure for drilling wells into bedrock aquifers in Ireland (IGI, 2007)

1. Drilling into bedrock; a hole is drilled through the soil into the upper bedrock, with this hole being at least 250 mm in diameter.
2. A temporary 250 mm ID steel conductor casing is fitted to ensure non-collapse of the hole. The casing should be removed at a later stage and therefore hard driving of the casing into bedrock is not recommended.
3. With the conductor casing in place the well is deepened into the aquifer at 250 mm diameter to the necessary depth, with fracture flow encountered below 30 m typically yielding sufficient water for a domestic supply in Ireland.
4. Permanent 150 mm ID casing is installed in the 250 mm hole. Where PVC casing is being used, the casing will comprise an end cap to exclude grout and is filled with water to minimise buoyancy.
5. Cement grout is injected through a tremie (grout) pipe down the annulus between borehole side and casing exterior. The annulus is filled from the bottom, expelling water due to its greater density. Where possible, the temporary steel casing should be retrieved at this stage. Grouting will stop when the entire annulus has been filled.
6. The grout is allowed to cure, ideally being given 48 hour curing time. Admixtures may be added to the grout to decrease the necessary curing time; however, these may reduce the overall strength of the seal.
7. The production (open, uncased) section of the borehole is then drilled at a diameter of <150 mm i.e. maximum OD through which drill tools may pass.
8. Well development to ensure a clean supply.

Typical collapse strengths of selected casing materials

Material	Casing wall thickness (mm)	Collapse strength (kPa)
Steel	7.9	5245
PVC	13	790
ABS	13	690
Fibreglass	6	690

Adapted from Misstear *et al.*, 2006

Tensile strengths of selected casings

Material	Specific Gravity	Tensile Strength (MPa)
Low carbon steel	7.85	414
Stainless Steel (Type 304)	8.0	552
PVC	1.4	55
ABS	1.04	31

Fibreglass	1.89	115

Adapted from Misstear *et al.*, 2006

Collapse strength of steel well casing

Nominal Diameter (mm)	Casing wall thickness (mm)	Collapse strength (kPa)
203	6.35	5209
203	7.94	8213
254	6.35	3178
254	7.94	5240
304	6.35	2110
304	7.94	3590

Adapted from Bloetscher *et al.*, 2007

Appendix C1

***Cryptosporidium* species (21)**

Cryptosporidium andersoni
Cryptosporidium bailey
Cryptosporidium bovis
Cryptosporidium canis
Cryptosporidium cichlidis
Cryptosporidium cuniculus
Cryptosporidium felis
Cryptosporidium galli
Cryptosporidium hominis
Cryptosporidium meleagridis
Cryptosporidium muris
Cryptosporidium nasoris
Cryptosporidium parvum
Cryptosporidium pestis
Cryptosporidium reichenbachklinkei
Cryptosporidium saurophilum
Cryptosporidium scophthalmi
Cryptosporidium serpentis
Cryptosporidium suis
Cryptosporidium varanii
Cryptosporidium wrairi

Appendix C2

Global Parameter Natural Background Levels (EPA, 2010; From OCM, 2007)

Parameter	Bedrock Type	10th Percentile	Median	90th Percentile
Nitrate (mg L^{-1})	Global	0.31	3.3	9.2
Orthophosphate (µg L^{-1})	Global	2	6	20
Chloride (mg L^{-1})	Global	12	18	24
Sulphate (mg L^{-1})	Global	2.8	11	37
Iron (µg L^{-1})	Global	18	25	130
Manganese (µg L^{-1})	Global	1	8	32
Zinc (µg L^{-1})	Global	1.5	8	55
Chromium (µg L^{-1})	Global	0.5	2	8
Copper (µg L^{-1})	Global	0.5	2.5	23
Barium (µg L^{-1})	Global	5	38	162
Fluoride (mg L^{-1})	Global	0.05	0.1	0.21
Nickel (µg L^{-1})	Global	0.5	1.7	8

NB Global refers to parameters not significantly influenced by bedrock lithology

Bedrock Influenced Parameter Natural Background Levels (EPA, 2010; From OCM, 2007)

Parameter	Bedrock Type	10th Percentile	Median	90th Percentile
Electrical Conductivity (µS cm^{-1})	Lower Palaeozoic	448	572	800
	Devonian	137	191	228
	Karst	445	544	731
	Mixed	377	595	727
	Granite	40	67	129
	Confined Limestones	668	721	750
	Confined Sandstones	422	449	520
pH	Lower Palaeozoic	6.6	7.4	8.4
	Devonian	5.3	6.3	7.4
	Karst	7	7.3	7.7
	Mixed	6.7	7.3	7.6
	Granite	5.6	6.1	6.7
	Confined	7.3	7.5	7.7

	Limestones			
	Confined Sandstones	7	7.3	7.8
Potassium ($mg\ L^{-1}$)	Lower Palaeozoic	0.5	2	6
	Devonian	0.4	1.1	2
	Karst	0.8	1.6	2.8
	Mixed	0.7	1.6	3.8
	Granite	-	-	-
	Confined Limestones	1.2	1.9	2.5
Sodium ($mg\ L^{-1}$)	Lower Palaeozoic	8	19	37
	Devonian	7.7	9	18
	Karst	7	11	19
	Mixed	9.6	13	26
	Granite	-	-	-
	Confined Sandstones	7	18	37
Calcium ($mg\ L^{-1}$)	Lower Palaeozoic	68	82	96
	Devonian	3.8	17	33
	Karst	54	89	132
	Mixed	39	86	111
	Confined Limestones	45	84	107
	Confined Sandstones	34	47	66
Magnesium	Lower Palaeozoic	9	20	67
	Devonian	2.2	3.3	8
	Karst	4.1	8.4	14

(mg L^{-1})	Mixed	3.4	9.9	24
	Confined Limestones	14	23	35
	Confined Sandstones	9	22	40
Alkalinity (mg L^{-1})	Lower Palaeozoic	220	304	435
	Devonian	13	73	93
	Karst	174	246	342
	Mixed	91	236	308
	Confined Limestones	187	290	398
	Confined Sandstones	170	227	278
Total Hardness(mg L^{-1})	Lower Palaeozoic	219	311	446
	Devonian	21	73	98
	Karst	182	258	364
	Mixed	79	242	364
	Confined Limestones	200	333	382
	Confined Sandstones	180	212	237

Appendix C3

EU Drinking Water Standards

Council Directive 98/83/EC on the quality of water intented for human consumption. Adopted by the Council, on 3 November 1998

Chemical Parameters

Parameter	Symbol/Formula	Parametric Value ($mg\,L^{-1}$)
Acrylamide	C_3H_5NO	0.0001
Antimony	Sb	0.005
Arsenic	As	0.01
Benzene	C_6H_6	0.001
Benzo(a)pyrene	$C_{20}H_{12}$	0.00001
Boron	B	1.00
Bromate	Br	0.01
Cadmium	Cd	0.005
Chromium	Cr	0.05
Copper	Cu	2.00
Cyanide	C=N	0.05
1,2-dichloroethane	$Cl\,CH_2\,CH_2\,Cl$	0.003
Epichlorohydrin	C_3H_5OCl	0.0001
Fluoride	F	1.5
Lead	Pb	0.01
Mercury	Hg	0.001
Nickel	Ni	0.02
Nitrate	NO_3	50
Nitrite	NO_2	0.5
PAHs	$C_2H_3N_1O_5P_{13}$	0.0001
Selenium	Se	0.01
Tetrachloraethene/trichloroethene	C_2Cl_4/C_2HCl_3	0.01
Trihalomethanes (Total)		0.1
Vinyl chloride	C_2H_3Cl	0.0005

Microbiological Parameters

Parameter	Parametric Value
Escherichia coli (*E. coli*)	0 in 250 ml
Enterococci	0 in 250 ml
Pseudomonas aeruginosa	*0 in 250 ml*
Colony count 22°C	100/ml
Colony count 37°C	20/ml

Indicator Parameters

Parameter	Symbol/Formula	Parametric Value
Aluminium	Al	0.2 mg L^{-1}
Ammonia	NH_4	0.5 mg L^{-1}
Chloride	Cl	250 mg L^{-1}
Clostridium perfringens		0/100 ml
Colour		Acceptable to consumer
Conductivity		2500 µS cm^{-1} at 20°C
Hydrogen ion concentration	$[H^+]$/pH	>6.5 and <9.5
Iron	Fe	0.2 mg L^{-1}
Manganese	Mn	0.05 mg L^{-1}
Odour		Acceptable to consumer
Oxidisability		5.0 mg L^{-1} O_2
Sulphate	SO_4	250 mg L^{-1}
Sodium	Na	200 mg L^{-1}
Taste		Acceptable to consumer
Colony count 22°C		No change
Coliform bacteria		0/100 ml
Total Organic Carbon	TOC	No change
Turbidity		Acceptable to consumer
Tritium	H_3	100Bq L^{-1}
Total Indicative Dose		0.10mSv/year

Appendix D1

Site Selection List

The following comprises the initial list of site which underwent matrix selection. This list was developed using a number of sources including Environmental Protection Agency, Geological Survey Ireland, various Local Authorities, and researchers from a number of institutions including National University of Ireland Galway, University College Dublin, Trinity College Dublin, Institute of Technology Sligo.

Co. Carlow
Knockbeg, Co. Carlow

Co. Clare
Ballyvaughan, Co. Clare
Drumcliff, Co. Clare
Ranaghan, Corofin, Co. Clare
Tuamgraney, Co. Clare

Co. Cork
Fermoy, Co. Cork

Co Galway
Clarenbridge, Co. Galway
Craughwell, Co. Galway
Gort, Co. Galway
Kilosolan, Co. Galway
Kinvara, Co. Galway
Menlough, Co. Galway
New Inn, Co. Galway

Co. Kildare
The Curragh, Co. Kildare
Gormanstown, Co. Kildare
Kilrathmurray, Clonard, Co. Kildare
Narraghmore-Liptown, Co. Kildare

Co. Kilkenny
Bennetsbridge, Co. Kilkenny
Carrigeen, Co. Kilkenny
Danesfort, Co. Kilkenny
Freshford, Co. Kilkenny
Glenmore, Co. Kilkenny
Monavadroe, Co. Kilkenny
Paulstown, Co. Kilkenny

Piltown, Co. Kilkenny
Thomastown, Co. Kilkenny
Urlingford, Co. Kilkenny

Co. Laois
Durrow, Co. Laois
Fermoyle, Co. Laois
Killenard, Co. Laois
Kyle & Orchard, Co. Laois
Swan, Co. Laois

Co. Meath
Kilclone, Co. Meath
Kilmessan, Co. Meath
Rushwee, Slane, Co. Meath

Co. Offaly
Agall/Tully, Co. Offaly
Ballyboy, Co. Offaly
Belmont, Co. Offaly
Dunkerrin, Co. Offaly
Moneygall, Co. Offaly
Mountbolus, Co. Offaly
Mountlucas, Co. Offaly
Toberdaly, Co. Offaly
Toberfin Springs – Killeigh & Meilaghan, Co. Offaly
Tully – Rahan, Co. Offaly
Walsh Island, Co. Offaly

Co. Roscommon
Ballinagard, Co. Roscommon
Ballybane/Ballinlough, Co. Roscommon
Castlerea, Co. Roscommon
Killeglan, Co. Roscommon
Mount Talbot/Cloonlaughnan, Co. Roscommon
Rockingham/Boyle-Ardcarn, Co. Roscommon

Co. Sligo
Bricklieve, Co. Sligo
Carrowmore, Co. Sligo
Geevagh, Co. Sligo

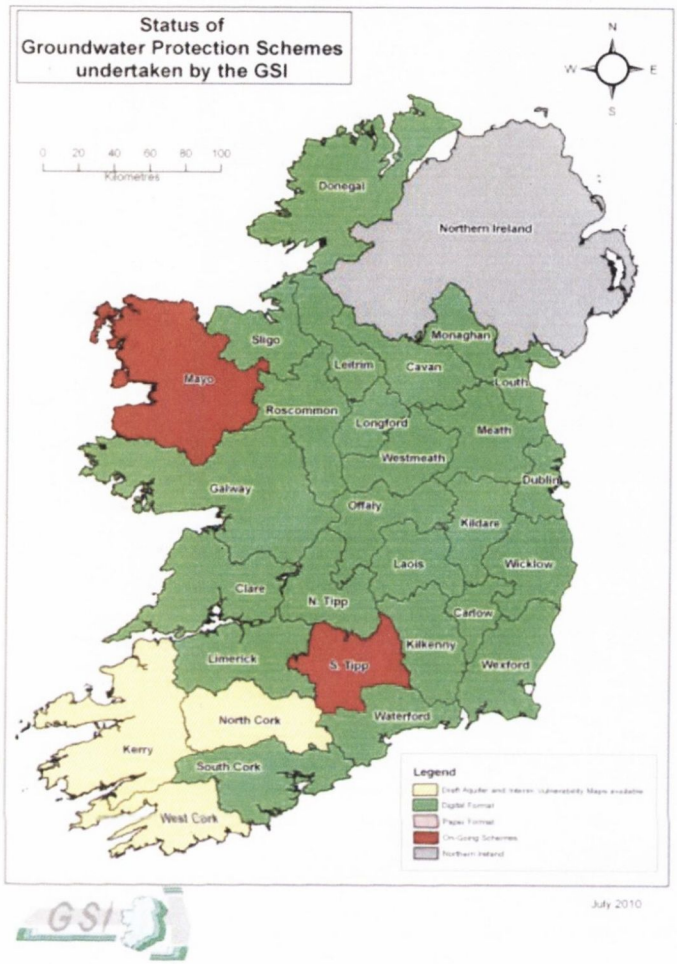
Co. Westmeath
Moate, Co. Westmeath
Rosemount, Co. Westmeath

Castledaly, Co. Westmeath

Co. Wicklow

Baltinglass, Co. Wicklow
Blessington, Co. Wicklow
Crossbridge, Co. Wicklow
Redcross, Co. Wicklow
Roundwood, Co. Wicklow

Appendix D2


Status of
Groundwater Protection Schemes
undertaken by the GSI
N
W
E
S
0 20 40 60 80 100
Kilometres
Donegal
Northern Ireland
Sligo
Monaghan
Leitrim
Cavan
Louth
Mayo
Roscommon
Longford
Meath
Westmeath
Dublin
Galway
Offaly
Kildare
Laois
Wicklow
Clare
N. Tipp
Carlow
Kilkenny
Limerick
S. Tipp
Wexford
North Cork
Waterford
Kerry
South Cork
West Cork
Legend
Digital Format
Paper Format
On-Going Schemes
Northern Ireland
July 2010

Appendix D3





The University of Dublin
Trinity College

Site Assessment

Date: **Assessment Number:**

Step 1: General Details

Well Reference Number			
Site Location (County)			
Townland			
Grid Reference			
Contact Number			
Number of Residents			
Water Supply	PrGWS	Bored SPS	HD SPS

Step 2: Desk Study

Bedrock Lithology		
Soil	**Type**	**Thickness Range**
Aquifer	**Importance Cat**	**Vulnerability**
Outcrop/Subcrop		
GSI GPWS Ref		
Land Slope/ Direction of Flow		

Step 3: Visual Site Assessment

Landscape Position			
Slope	Steep(>1:5)	Shallow(1:5-1:20)	Flat (<1:20)
Surface Features			
Houses (supplied)			
Site Boundries			
Roads			

Land Use			
Surface Water Ponding (On-Site)			
Surface Water Features	**Rivers**	**Lakes**	**Drains**
Karst Features			
Wells	**Working**	**Abandoned**	
Springs			
Vegetation Indicators			
Ground Condition (On-Site)			
Comments			

Step 4: Potential Sources of Pollution

Septic Tanks	**Number (100m)**	**Distance**	**Hydraulic Gradient**
On Site Treatment	**Number (100m)**	**Distance**	**Hydraulic Gradient**
Agriculture	**Arable/Grazing**	**Grazing Type**	**Buildings/Storage**
Landfill Sites (1km)			
Other			
Comments			

Step 5: Wellhead Assessment

Wellhead Details/Protection		
Secure Boundry	Present:	Radius (if present):
Above/below gr level		
Pumphouse	Present:	Material:
Flowmeter		
Well Cover	Present:	Material:
Well Cap	Present:	Material:
Surface ponding or ingress		
Waterproof Finish/Seals		
Evidence of cracking		
Ground condition within 10m radius of well		
Ground Condition within approximate ZOC (see		

maps)		
Well Details		
Well Location (See map)		
Flange to Concrete (>300mm)		
Construction Materials		
Well Dimensions		
Well Chamber	Material:	Dimensions:
Chamber Floor	Material	Floor Level:
Depth to Watertable		
Type of pump		
Basic Construction Details		
Distance to Usage Points		
Wellhead Assessment		

Step 6 (1) **On-Site Analyses**

Source of Sample	
Depth of Sample	
Colour	
Odour	
Taste	
Turbidity (FTU)	
Temperature (^{0}C)	
Conductivity (µS/cm)	
TDS (mg l^{-1})	
pH	

Step 6 (2)	Laboratory Analyses
Nitrates (mg l^{-1})	
Arsenic (mg l^{-1})	
Manganese (mg l^{-1})	
Fluoride (mg l^{-1})	
Free Chloride (mg l^{-1})	
Cryptosporidium parvum (oocysts/litre)	
Rotavirus (presence/absence)	
Iron (mg l^{-1})	
Dissolved O_2 (mg l^{-1})	
TTC Volume Filtered	
Number of Colonies	
TTC (CFU) per 100ml	

Appendix D4

Sample size for ±3%, ±5%, ±7% and ±10% Precision Levels Where Confidence Level is 95% and P=.5.				
Size of	Sample Size (n) for Precision (e) of:			
Population	±3%	±5%	±7%	±10%
500	a	222	145	83
600	a	240	152	86
700	a	255	158	88
800	a	267	163	89
900	a	277	166	90
1,000	a	286	169	91
2,000	714	333	185	95
3,000	811	353	191	97
4,000	870	364	194	98
5,000	909	370	196	98
6,000	938	375	197	98
7,000	959	378	198	99
8,000	976	381	199	99
9,000	989	383	200	99
10,000	1,000	385	200	99
15,000	1,034	390	201	99
20,000	1,053	392	204	100
25,000	1,064	394	204	100
50,000	1,087	397	204	100
100,000	1,099	398	204	100
>100,000	1,111	400	204	100
a = Assumption of normal population is poor (Yamane, 1967). The entire population should be sampled.				

Appendix D5

SPSS Codebook

Awareness Survey

Variable 1: Questionnaire Code
SPSS Variable Code: SurveyNum
Data Type: Scale
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 1 - 610

Variable 2: Respondent Gender
SPSS Variable Code: Gender
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "Female"; 1 = "Male"

Q1. "What is the current source of drinking water for the household"
Variable 3: Source Type
SPSS Variable Code: SourceType
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "HD SPS/Spring Well"; 1 = "Bored SPS"; 2 = "Don't know"; 3 = "PuGWS"; 4 = "PrGWS"; 5 = "Other"

Q2. "Were you (respondent) living at this address when the well was built"
Variable 4: Residency
SPSS Variable Code: LivingBuilt
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Q3 & Q4. "When was the well built"; "Are you aware of when the well was built"
Variable 5: Age of groundwater source
SPSS Variable Code: KnowledgeWellAge
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"
Special Instruction: Responses to Questions 3 and 4 amalgamated to give one category

Variable 5: Age of groundwater source
SPSS Variable Code: KnowledgeWellAge
Data Type: Scale
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: All positive numbers

Q5. "Are you aware of the name of contractor/persons that constructed the well"
Variable 7: Contractor Awareness
SPSS Variable Code: ContractorAware
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Q6. "To the best of your knowledge, were any of the following involved in the construction of the well"
Variable 8: Design/Construction Involvement
SPSS Variable Code: InvolveHydro
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 9: Design/Construction Involvement
SPSS Variable Code: InvolveDiv
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 10: Design/Construction Involvement
SPSS Variable Code: InvolveLocAu
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 11: Design/Construction Involvement
SPSS Variable Code: InvolveKnow
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"
Special Instruction: All respondents included in this filter variable; only those answering "Yes" to Q6 are included in variables 6-8

Q7. "Do you have any records of the drilling/construction process i.e. borehole log, hydrogeologists report, contract receipts"
Variable 12: Well Records
SPSS Variable Code: WellRecKnowledge
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 13: Well Records
SPSS Variable Code: WellRec
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"
Special Instruction: This is a multiple dichotomy filtering question i.e. only those answering "Yes" to Q7 are included, with all respondents included in variable 10

Q8. "Do you know any of the following details regarding your well"
Variable 14: Well Details
SPSS Variable Code: WellDetDia
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 15: Well Details
SPSS Variable Code: WellDetDepth
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 16: Well Details
SPSS Variable Code: WellDetWT
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 - 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 17: Well Details
SPSS Variable Code: WellDetLiner
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 - 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 18: Well Details
SPSS Variable Code: WellDetSeals
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 - 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 19: Well Details
SPSS Variable Code: WellDetPump
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 - 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 20: Well Details
SPSS Variable Code: WellDetNumber
Data Type: Scale
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 - 64
Valid Input Codes: 0 – 6

Q9. "Is your water currently being treated"
Variable 21: Water Treatment
SPSS Variable Code: Treatment
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 - 64
Valid Input Codes: 0 = "No"; 1 = "Yes"; 2 = "Don't know"

Q11. "Has your well ever been cleaned or pumped out"
Variable 22: Well Maintenance
SPSS Variable Code: Maintenance
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 - 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 23: Well Maintenance
SPSS Variable Code: MaintenanceKnow
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 - 64
Valid Input Codes: 0 = "No"; 1 = "Yes"
Special Instruction: All respondents answering Q11 included in variable 20, only those answering "Yes" or "No" included in variable 21

Q12. "What are the factors you would or did consider when having a new well constructed"
Variable 24: Factors of importance
SPSS Variable Code: FactorCost
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 25: Factors of importance
SPSS Variable Code: FactorYield
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 26: Factors of importance
SPSS Variable Code: FactorQual
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 27: Factors of importance
SPSS Variable Code: FactorDepth
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 28: Factors of importance
SPSS Variable Code: FactorLoc
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 29: Factors of importance
SPSS Variable Code: FactorOther
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Q13. "From Q. 12, what in your opinion is the most important factor"
Variable 30: Factors of importance
SPSS Variable Code: ImpFactor
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "Quality"; 1 = "Not quality"
Special Instruction: If respondent answered "Quality" to Q.13, coded "0", all alternative answers coded "1"

Variable 31: Factors of importance
SPSS Variable Code: MostImpFactor
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 1 = "Quality"; 2 = "Yield"; 3 = "Cost"; 4 = "Depth"; 5 = "Location"

Q14. "Has your water source been tested at any time by a public or private authority"
Variable 32: Water Quality Testing
SPSS Variable Code: WaterQualTestKnow
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "Don't know"; 1 = "Know"

Variable 33: Water Quality Testing
SPSS Variable Code: WaterQualTest
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"
Special Instruction: Only respondents coded "1" in variable 30 are included in variable 31

Q14. "If yes to Q. 14, are you aware of the results of testing"
Variable 34: Water Quality Testing
SPSS Variable Code: WaterQualResults
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"
Special Instruction: Only respondents coded "1" in variable 31 are included in variable 32

Q16. "Which of the following best describes the level of importance that you assign to regular testing of your well water"
Variable 35: Water Quality Testing
SPSS Variable Code: TestImp
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "Undecided"; 1 = "Low"; 2 = "Medium"; 3 = "High"

Q17. "Would you be prepared to pay for regular water testing"
Variable 36: Water Quality Testing
SPSS Variable Code: PrepRegTesting
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"; 2 = "Undecided"

Q18. "Are you aware of any of the following with regard to wells or groundwater pollution"
Variable 37: Groundwater Contaminants
SPSS Variable Code: ContamColi
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 38: Groundwater Contaminants
SPSS Variable Code: ContamRota
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 39: Groundwater Contaminants
SPSS Variable Code: ContamCrypt
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 40: Groundwater Contaminants
SPSS Variable Code: ContamGiar
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 41: Groundwater Contaminants
SPSS Variable Code: ContamMn
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 42: Groundwater Contaminants
SPSS Variable Code: ContamLi
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 43: Groundwater Contaminants
SPSS Variable Code: ContamNit
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 44: Groundwater Contaminants
SPSS Variable Code: ContamAs
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 45: Groundwater Contaminants
SPSS Variable Code: ContamFl
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 46: Groundwater Contaminants
SPSS Variable Code: ContamNumber
Data Type: Scale
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 - 9

Q19. "Do you know of any potential health problems (symptoms or illnesses) which may be caused by any of the above contaminants (Q.19)?"
Variable 47: Health Hazard Awareness
SPSS Variable Code: HealthHazAware
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Q20. "Do you know of any activities in the surrounding area that may potentially affect your water quality"
Variable 48: Hazard Source Awareness
SPSS Variable Code: PotSourceHazAware
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Q21. "Do any of the following occur within 100m of your well"
Variable 49: Hazard Source Awareness
SPSS Variable Code: PotHazFarm
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 50: Hazard Source Awareness
SPSS Variable Code: PotHazSlurry
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 51: Hazard Source Awareness
SPSS Variable Code: PotHazSilage
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 52: Hazard Source Awareness
SPSS Variable Code: PotHazGraze
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 53: Hazard Source Awareness
SPSS Variable Code: PotHazST
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 54: Hazard Source Awareness
SPSS Variable Code: PotHazNumber
Data Type: Scale
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 - 5

Q22. "How many people use water in the household"
Variable 55: Household Composition
SPSS Variable Code: ResInfant
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"
Special Instruction: Variables 48 – 58 dichotomised due to large proportion of non-numerical responses

Variable 56: Household Composition
SPSS Variable Code: Res1to5
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 57: Household Composition
SPSS Variable Code: Res5to18
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 58: Household Composition
SPSS Variable Code: ResAdult
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 59: Household Composition
SPSS Variable Code: ResElderly
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 60: Household Composition
SPSS Variable Code: HouseholdTot
Data Type: Scale
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: All positive integers

Q23. "What is the main source of drinking water for the household"
Variable 61: Water Consumption
SPSS Variable Code: DrinkingSource
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "Household Tap"; 1 = "Bottled/Other"

Q24. "Do you (respondent) drink water from the household tap i.e. "raw water", not including hot beverages or boiled water"
Variable 62: Water Consumption
SPSS Variable Code: RespTap
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Q25. "Approximately how often do you drink from the household tap"
Variable 63: Water Consumption
SPSS Variable Code: ConFreq
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 1 = "1/day"; 2 = "2-3/day"; 3 = "4-5/day; 4 = ">5/day"

Q26. "Approximately how much water would you (respondent) drink from the household tap per day"
Variable 64: Water Consumption
SPSS Variable Code: ConVol
Data Type: Scale
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0.125 = "<0.5 pint"; 0.375 = "0.5-1 pint"; 0.75 = "1-2 pint; 1.25 = "2-3 pint"; 1.75 = "3-4 pint"; 2.25 = ">4 pint"

Q27. "Has anyone in the household suffered from a water related illness in the past 12 months"
Variable 65: Waterborne Illness
SPSS Variable Code: DiagGastro
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Q28. "Has anyone in the household shown symptoms of gastroenteritis in the last 12 months in the absence of a positive diagnosis or seeking medical advice"
Variable 66: Waterborne Illness
SPSS Variable Code: PotGastro
Data Type: Nominal
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 0 = "No"; 1 = "Yes"

Q29. "How satisfied are you with your current water supply"
Variable 67: Supply Satisfaction
SPSS Variable Code: SupplySat
Data Type: Scale (Likert)
Row Number: RA Survey Datasheet 1 – 245; Private SA Datasheet 1 – 300; Public SA Datasheet 1 – 64
Valid Input Codes: 1 - 5

Appendix E1



(A)Appropriately finished borehole with steel wellhead cover in Castledaly, Co. Westmeath; (B) Borehole finished above ground with 220 mm flange clearance, surrounded by poor on-site conditions, Castledaly, Co. Westmeath; (C) Hand-dug well finished above ground, with 1.3 m flange clearance and concrete liner cover (SA5); (D) Poorly finished hand-dug well, finished "at ground" level with insufficient wellhead cover, with high likelihood of surface ingress, outside Moate, Co Westmeath



(A) Steel liner cap fitted on borehole, with poor well chamber conditions (SA1); (B) Borehole finished "at ground" level with no liner cap or wellhead cover, Ballynahowan, Co Westmeath; (C) Typical below ground well chamber finish, with borehole liner protruding over chamber floor, (SA3); (D) Poorly finished borehole finished "at ground" level, with no liner cap or wellhead cover apparent (SA3)



(A) Cracking apparent in below ground well chamber (SA1); (B) Typical borehole configuration with steel liner, rising main and rope for submersible pump suspension, in Rosemount, Co. Westmeath; (C) Typical below ground wellhead configuration with water level dip-meter in left foreground, located on grazing

land in Rosemount, Co. Westmeath; (D) Steel outer liner (180 mm diameter), with HDPE inner liner (100 mm diameter) in newly constructed well (SA1)



(A) Hand-dug well (1 m diameter) located in shallow well chamber in garage, with concrete liner apparent (SA3); (B) Hand-dug wellhead finished above ground level with 375 mm flange clearance (SA4); (C) Spring well outside Gort, Co Galway; (D) Traditional stone-lined well (7 m depth) outside Moate, Co Westmeath

Appendix E2

Bedrock Type Vs. Mean Sample Watertable

Bedrock Type	Sample Number	Mean watertable depth (m)	Std. Deviation
Limestone	54	5.08	7.7
Granite	40	3.27	2.83
Sandstone	39	6.14	4.32
Shale	13	9.38	9.88
Namurian	7	3.33	4.07
Silurian Volcanics	3	3.1	2.59

Appendix E3

Septic **Tank**

STDist

	Sum of Squares	df	Mean Square	F	Sig.
Between Groups	4235.776	2	2117.888	1.383	.253
Within Groups	304762.247	199	1531.469		
Total	308998.023	201			

Distance/Gradient Analysis

Bonferroni

Post-Hoc Test

(I) STGrad	(J) STGrad	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Lower Bound	Upper Bound
UG	OG	13.28826	9.70568	.518	-10.1450	36.7215
	DG	.29624	7.25616	1.000	-17.2229	17.8154
OG	UG	-13.28826	9.70568	.518	-36.7215	10.1450
	DG	-12.99201	8.00438	.318	-32.3177	6.3337
DG	UG	-.29624	7.25616	1.000	-17.8154	17.2229
	OG	12.99201	8.00438	.318	-6.3337	32.3177

Appendix E4

Area By Area Sample Sources

Study Area	Sample Programme Duration	Source Type		Sample Collection Source		
		Bored	Hand-dug	Outdoor	Indoor	Well direct
SA1: *Belmont*, Co. Offaly	July 08 – April 09	36 (72)	14 (28)	24 (48)	22 (44)	4 (8)
SA2: Crossbridge, Co. Wicklow	January 08 – November 09	43 (86)	7 (14)	32 (64)	17 (34)	1 (2)
SA3: Paulstown, Co. Kilkenny	November 09 – February 10	44 (88)	6 (12)	30 (60)	18 (36)	2 (4)
SA4: Kilclone, Co. Meath	July 09 – April 10	40 (77)	12 (23)	30 (58)	16 (31)	6 (11)
SA5: Birdhill, Co. Tipperary	March 10	6 (67)	3 (33)	3 (33)	2 (22)	4 (44)

(percentages in parentheses)

Appendix E5

Area By Area Mean Watertable

Study Area	Mean Watertable (m)	Range	Boreholes (Mean, m)	Hand-dug wells (Mean, m)
SA1: *Belmont*, Co. Offaly	5.34	0.1 – 10.9	6.51	2.56
SA2: Crossbridge, Co. Wicklow	3.29	0 – 12.17	3.75	0.61
SA3: Paulstown, Co. Kilkenny	10.4	0 – 48.66	12.2	0.3
SA4: Kilclone, Co. Meath	2.29	0.29 – 9.29	2.58	1.31
SA5: Birdhill, Co. Tipperary	3.58	0.2 – 6.8	4.81	1.52

Appendix E6

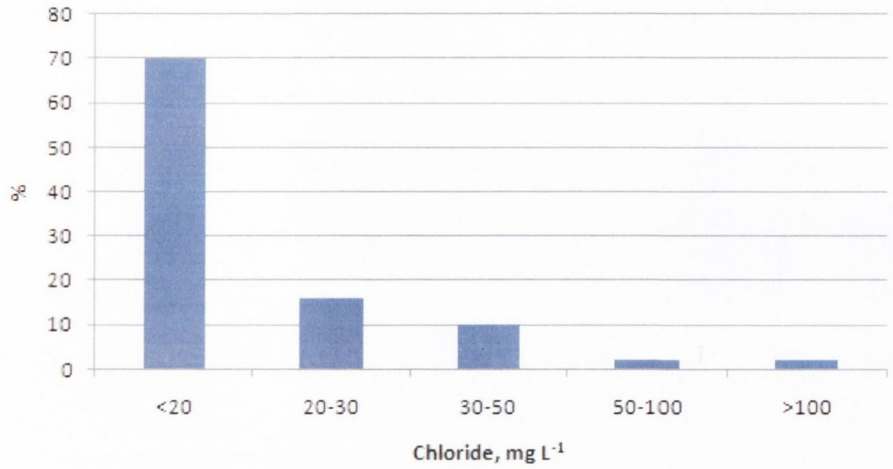
Correlation matrix of main physicochemical parameters

	WT Depth	GW Temperature	GW pH	GW EC
WT Depth	-			
GW Temperature	-0.169*	-		
GW pH	0.113	0.2**	-	
GW EC	0.043	0.417**	0.509**	-

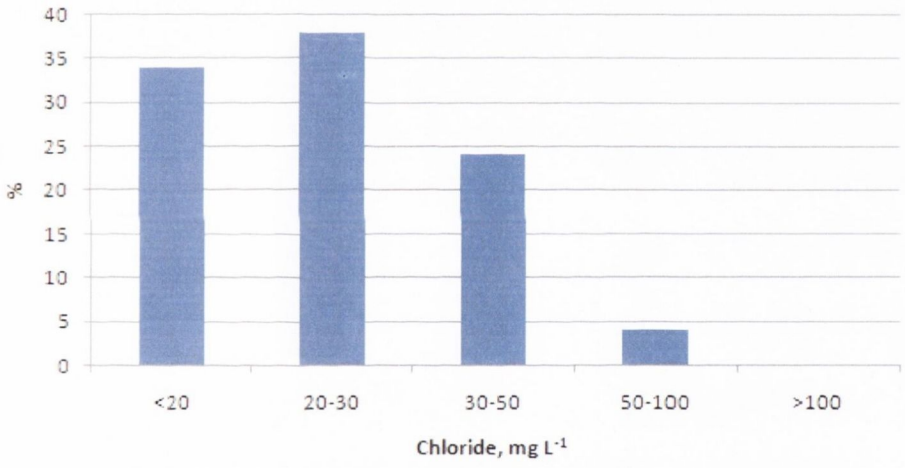
*. Correlation is significant at the 0.05 level

**. Correlation is significant at the 0.01 level

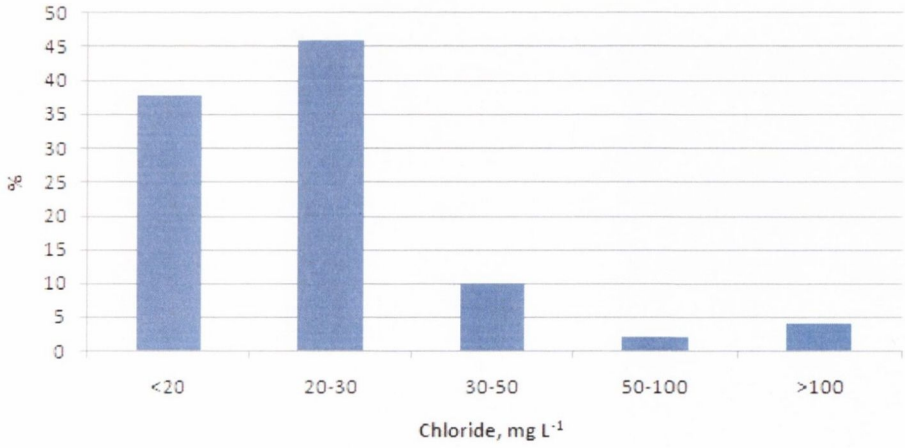
Appendix E7



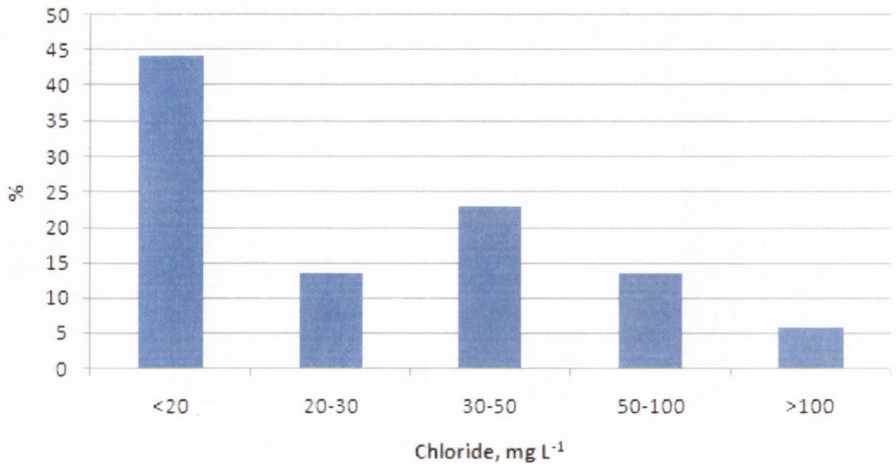
Study Area 1, Percentage of wells with mean Chloride concentrations in the ranges indicated



Study Area 2, Percentage of wells with mean Chloride concentrations in the ranges indicated

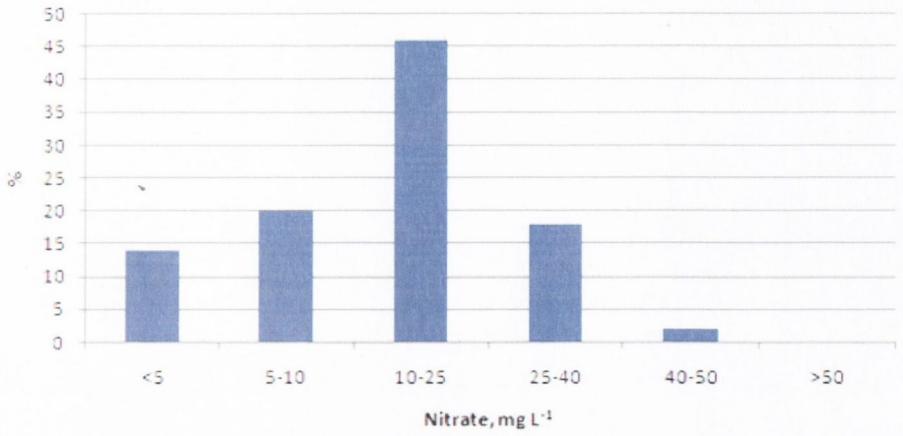


Study Area 3, Percentage of wells with mean Chloride concentrations in the ranges indicated

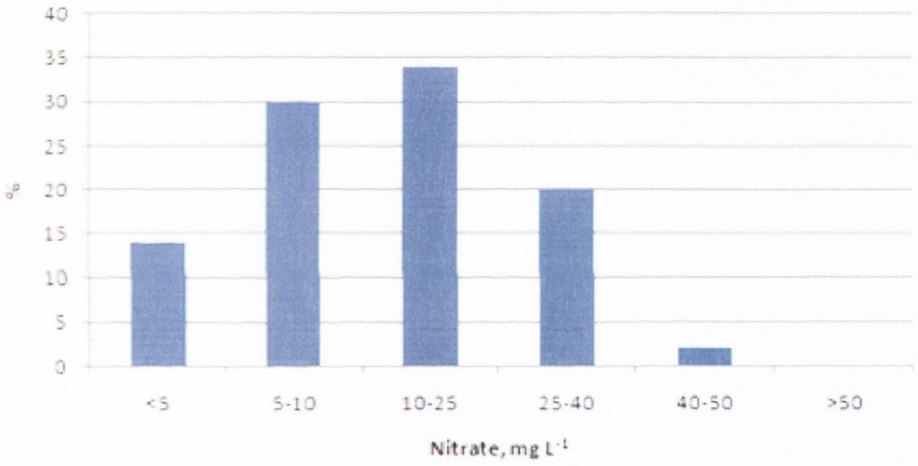


Study Area 4, Percentage of wells with mean Chloride concentrations in the ranges indicated

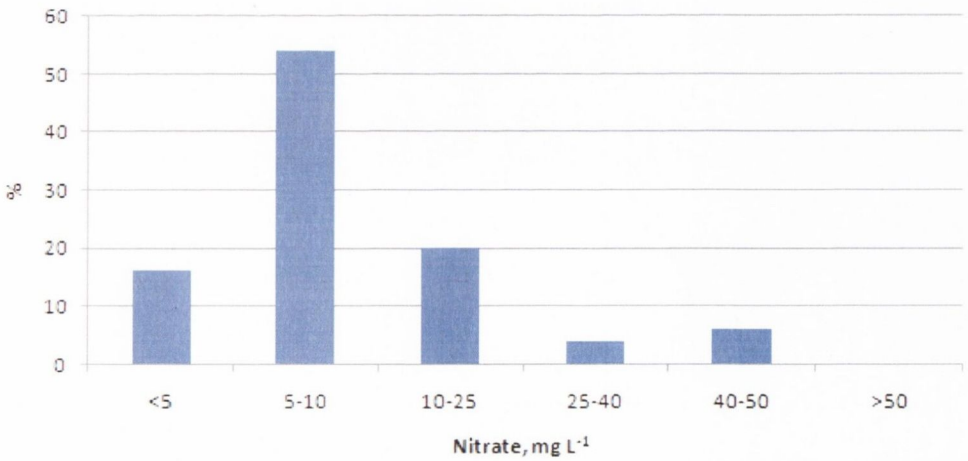
Appendix E8



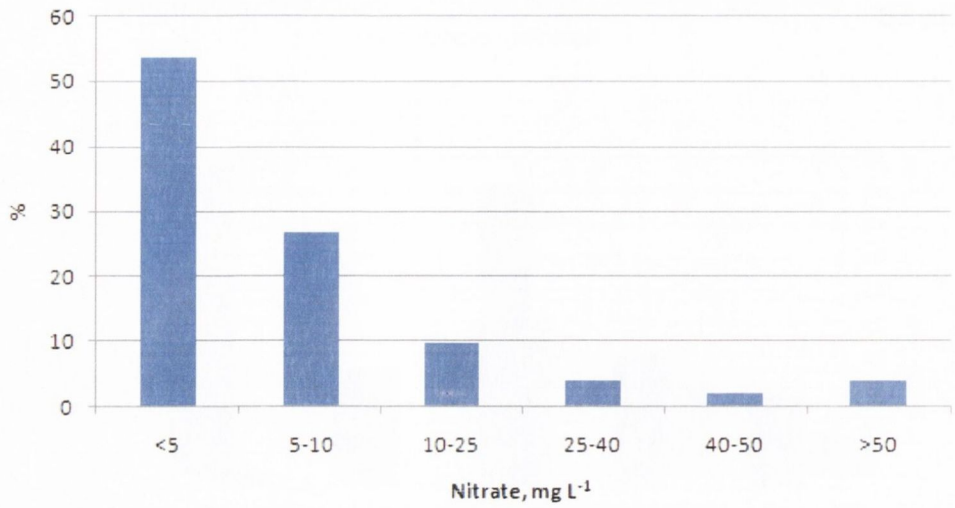
Study Area 1, Percentage of wells with mean nitrate concentrations in the ranges indicated



Study Area 2, Percentage of wells with mean nitrate concentrations in the ranges indicated

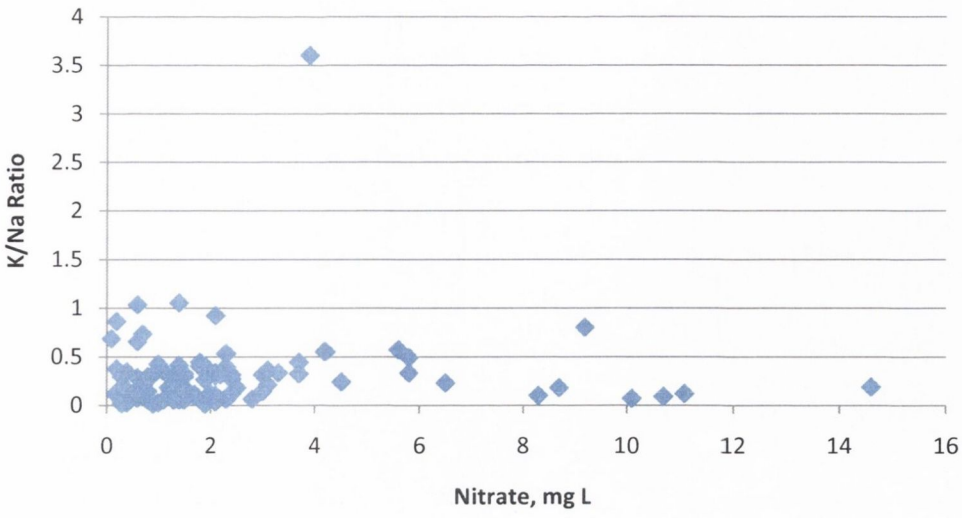


Study Area 3, Percentage of wells with mean nitrate concentrations in the ranges indicated



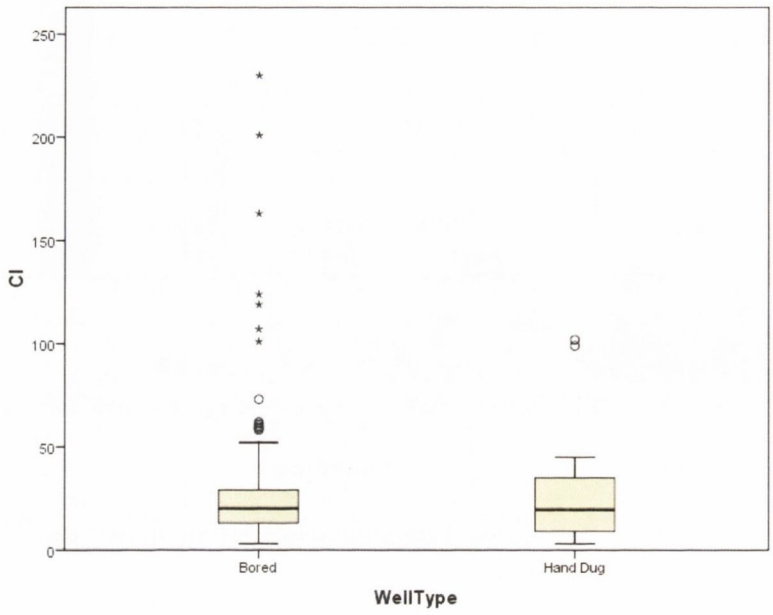
Study Area 4, Percentage of wells with mean nitrate concentrations in the ranges indicated

Appendix E9



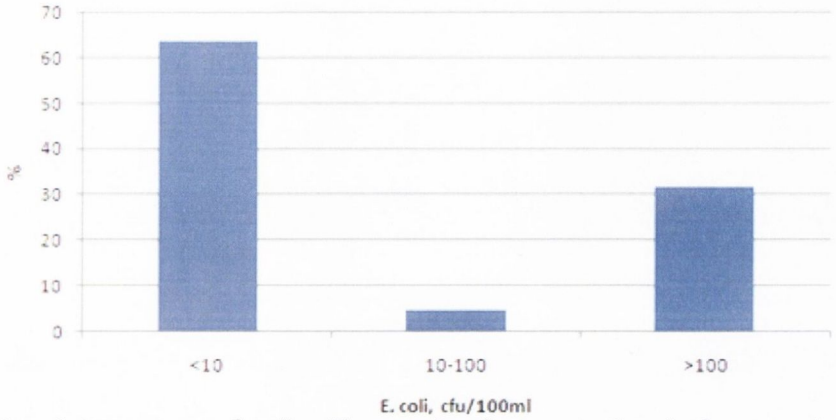
K/Na Ratio vs Groundwater Nitrate (n = 100)

Appendix E10

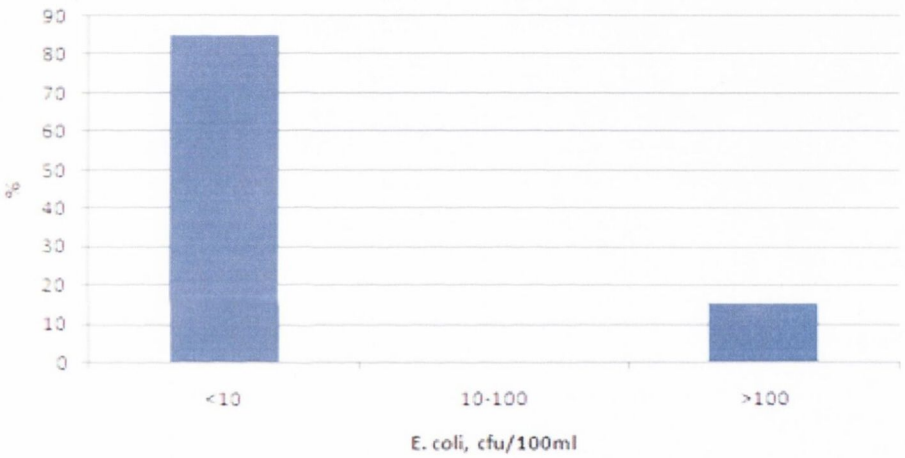


Summary results showing average (centre line) 25-75% (box) and minimum and maximum (error bars) of groundwater chloride concentrations among both source types

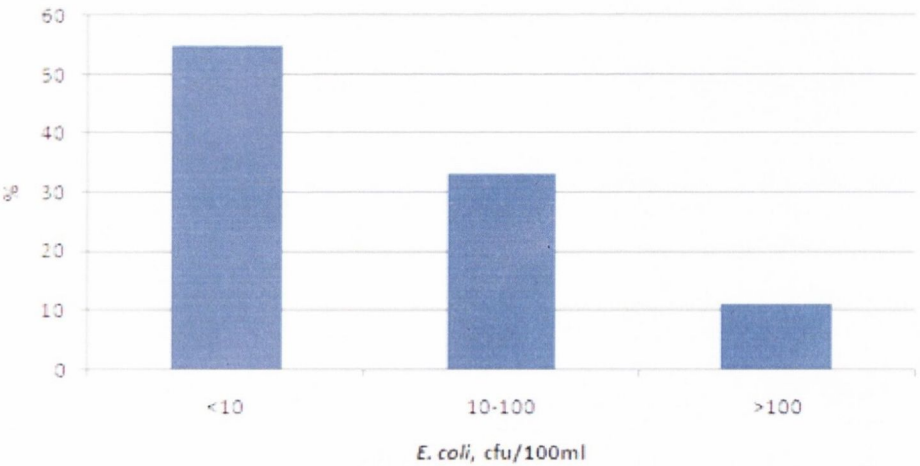
Appendix E11



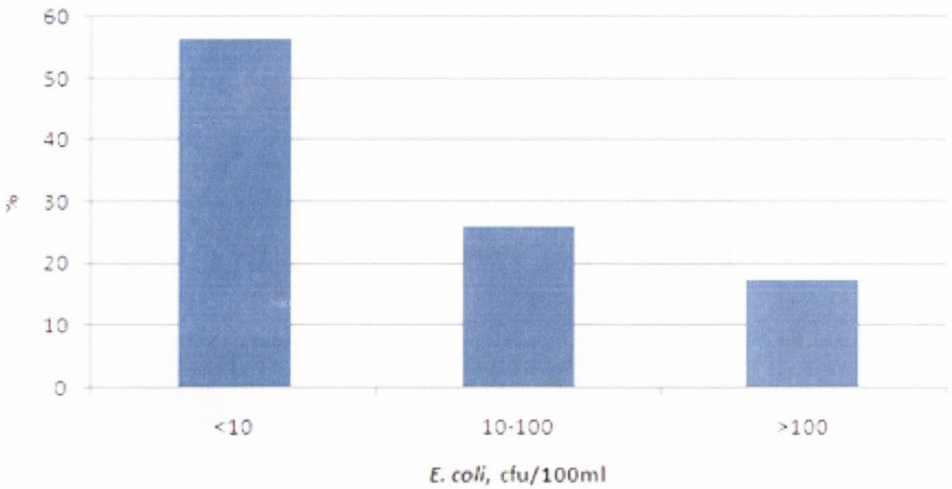
Study Area 1, Percentages of wells with mean *E. coli* concentrations in the ranges indicated



Study Area 2, Percentages of wells with mean *E. coli* concentrations in the ranges indicated



Study Area 3, Percentages of wells with mean *E. coli* concentrations in the ranges indicated



Study Area 4, Percentages of wells with mean *E. coli* concentrations in the ranges indicated

Appendix E12

Presence/Absence of faecal streptococci among source types

Well type	Faecal *streptococci*	
	Present	Absent
Bored	27 (61.3)	17 (38.7)
Hand-dug	9 (69.2)	4 (30.7)

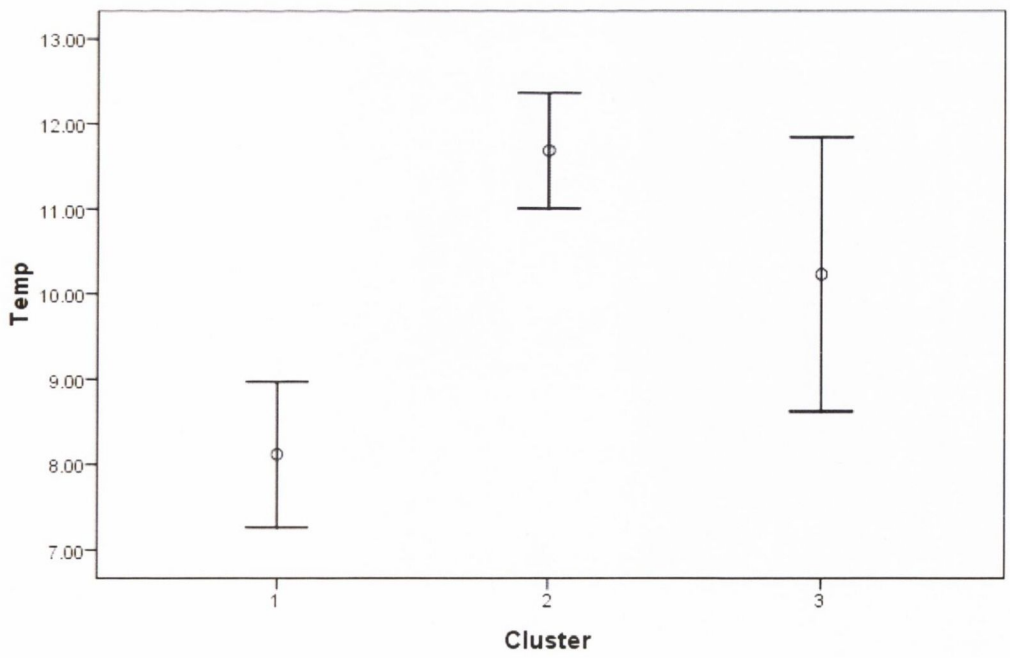
Appendix E13

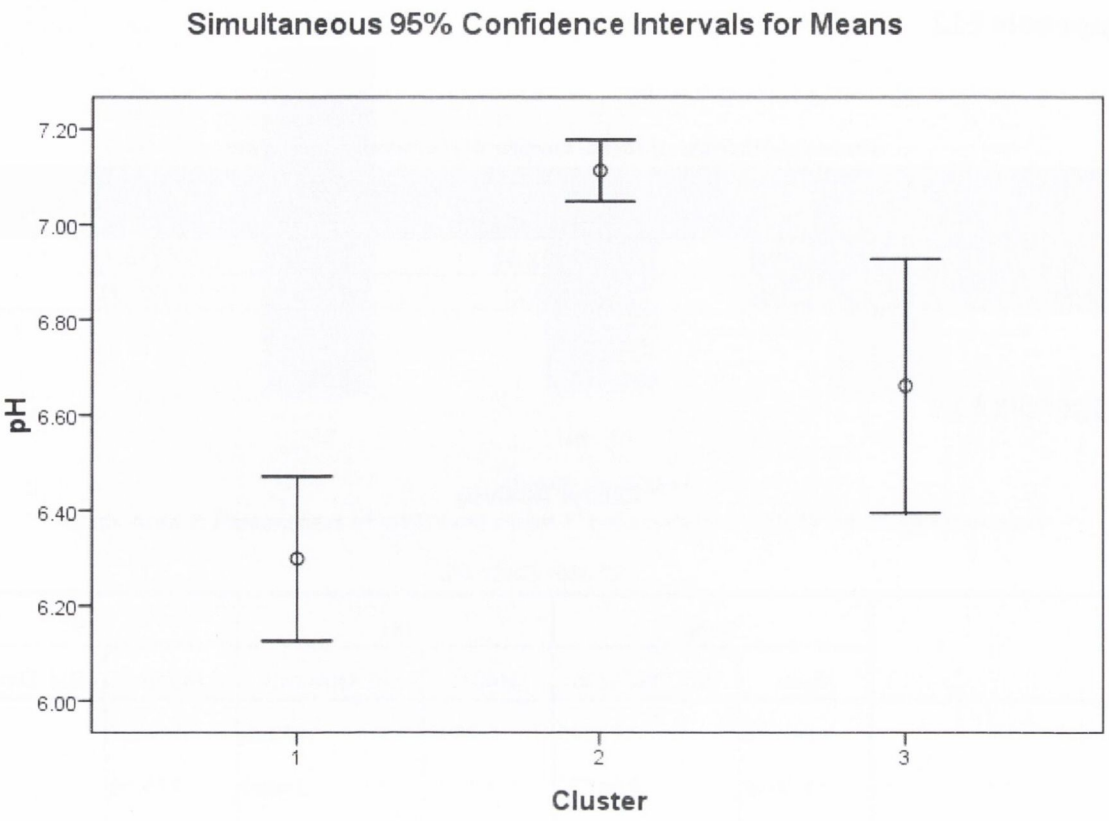
Cluster Analysis

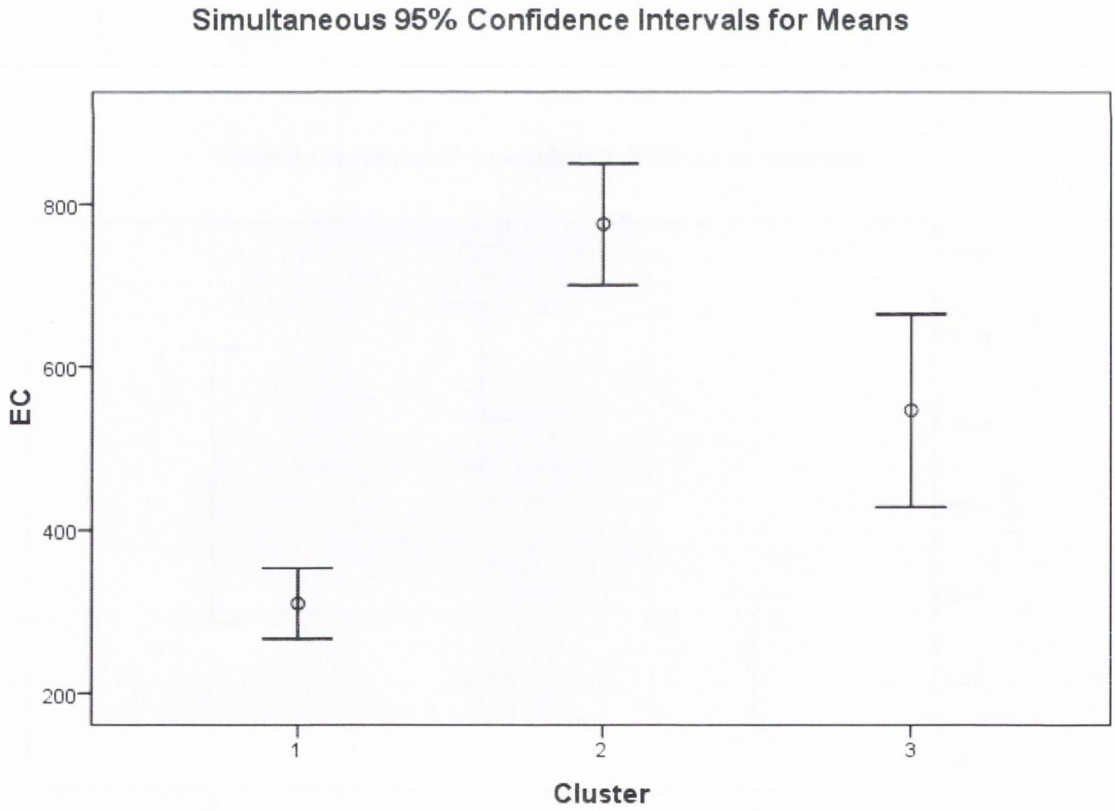
Cluster Centroids

		Temp		pH		EC	
		Mean	Std. Deviation	Mean	Std. Deviation	Mean	Std. Deviation
Cluster	1	8.1173	3.07845	6.2985	.62302	310.22	156.671
	2	11.6858	2.61770	7.1127	.24865	775.55	288.315
	3	10.2293	4.12730	6.6610	.68249	546.60	302.609
	Combined	10.0527	3.50824	6.7164	.62888	554.86	324.873

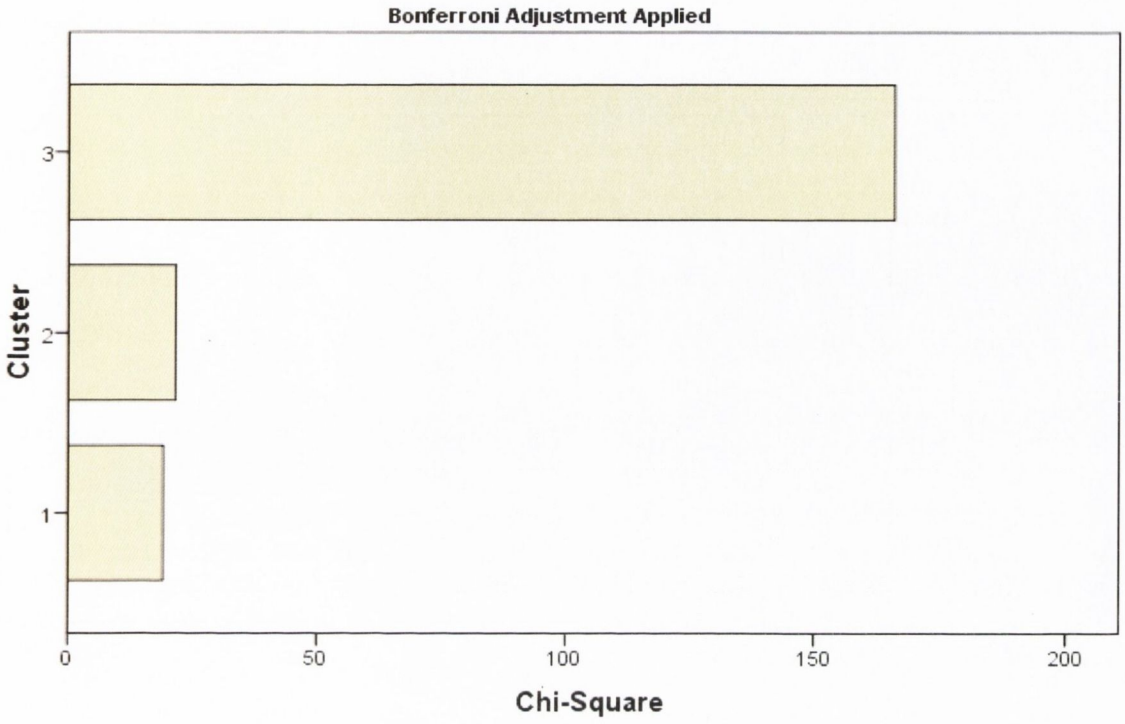
Simultaneous 95% Confidence Intervals for Means



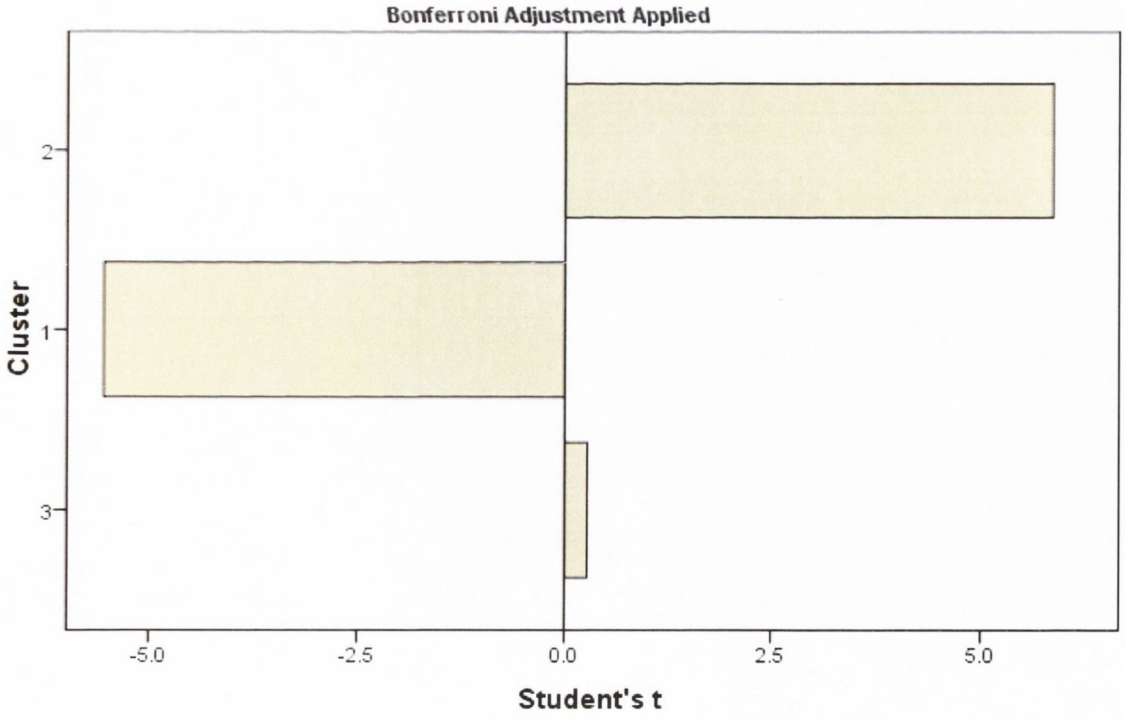

Simultaneous 95% Confidence Intervals for Means
7.20
7.00
6.80
6.60
6.40
6.20
6.00
pH
1
2
3
Cluster


Simultaneous 95% Confidence Intervals for Means
800
600
400
200
EC
1
2
3
Cluster

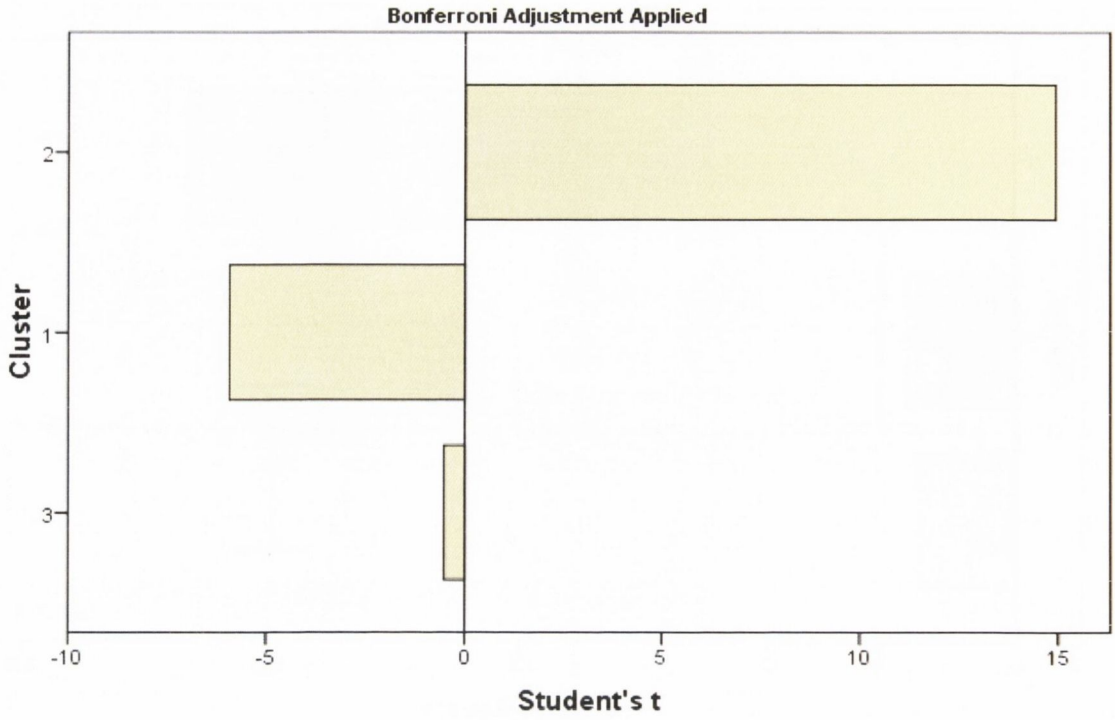
WellType



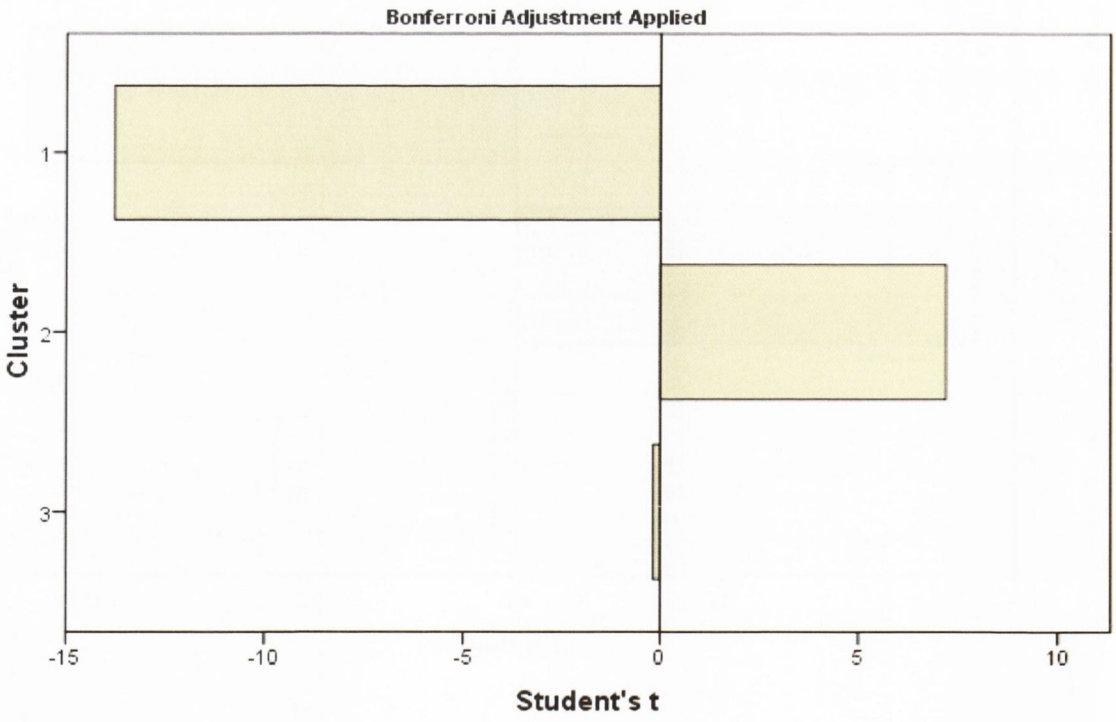
Temp



pH

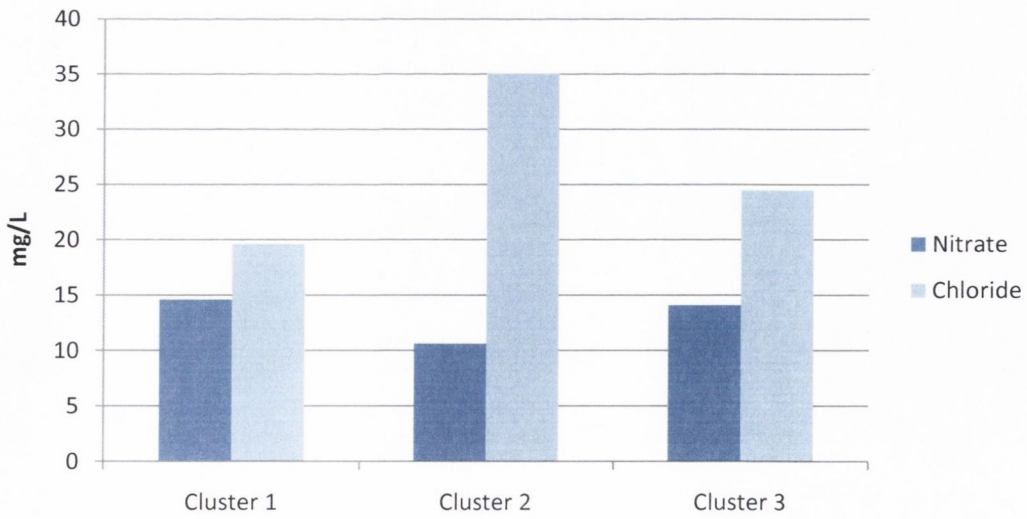

Bonferroni Adjustment Applied
Cluster
2
1
3
-10
-5
0
5
10
15
Student's t

EC


Bonferroni Adjustment Applied
Cluster
1
2
3
-15
-10
-5
0
5
10
Student's t

Cluster 1 & 2 vs Bedrock

Bedrock Type	Cluster 1	Cluster 2
Devonian Kiltorcan Type Sandstone	17	0
Dinantian (Early) Limestones	5	0
Dinantian Pure Unbedded Limestone	3	1
Dinantian Lower Impure Limestone	15	0
Devonian Old Red Sandstone	2	3
Silurian Metasediments and Volcanics	0	1
Dinantian Unbedded Impure Limestone	35	3
Undifferentiated Namurian	7	0
Dinantian Dolomitised Limestone	1	0
Westphalian Sandstones	0	4
Namurian Sandstone	0	13
Namurian Shales	0	10
Dinantian Pure Bedded Limestone	2	4
Granites	1	42
Total	88	81



Mean chloride and nitrate concentrations among clusters

Appendix E14

Ranges, means and standard deviations of water quality parameters among monitored private sources

BM1 (Bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	9.41	7.27	13.34	1.73
Temperature	8.89	1.6	12.8	3.03
pH	7.32	6.67	7.8	0.33
EC	643.2	499	1955	342.8
NO^3-N	1.34	0.3	4.9	1.07
Cl	52	10	228	70.34
NH^4	0.12	0.03	0.22	0.067
K	3.25	1.15	5.85	1.75
Na	43.77	18.45	71.6	26.6
K/Na Ratio	0.101	0.06	0.16	0.03
Mg	7.22	0.08	13.59	6.001

BM2 (Bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	10.01	9.76	10.28	0.157
Temperature	9.23	3.15	12.8	2.48
pH	7.12	6.83	7.35	0.13
EC	710.4	586	795	51.6
NO^3-N	5.24	1.4	10.1	1.75
Cl	22.3	11	55	9.63
NH^4	0.12	0.02	0.28	0.089
K	3	0.26	6.89	3.46

Na	61.8	38.17	80.43	21.57
K/Na Ratio	0.06	0	0.18	0.09
Mg	5.41	0.01	14.76	5.99

BM3 (Hand dug well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	2.42	1.67	3.02	0.37
Temperature	8.8	1.05	12.5	3.01
pH	7.15	6.47	7.6	0.24
EC	739	614	898	58.1
NO^3-N	3.19	0.9	8.1	1.55
Cl	22.2	9	50	9.16
NH^4	0.07	0.03	0.16	0.045
K	4.16	0.73	6.29	2.15
Na	10.62	8.43	13.92	2.16
K/Na Ratio	0.41	0.05	0.57	0.21
Mg	6.76	5.62	11.44	2.08

BM11 (Bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	10.33	9.56	11.34	0.55
Temperature	9.44	2	16	3.36
pH	6.96	6.75	7.22	0.13
EC	877	634	956	67.7
NO^3-N	8.4	1.7	14.9	3.25
Cl	20.57	12	34	5.76
NH^4	0.03	0.02	0.04	0.01

K	11.36	8.99	17.37	3.11
Na	13.87	12.31	15.05	1.01
K/Na Ratio	0.82	0.63	1.41	0.28
Mg	8.25	0.03	13.07	3.87

BM24 (Hand dug well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	2.03	0.8	3.23	0.68
Temperature	8.45	1.9	15	3.96
pH	6.89	6.67	7.15	0.13
EC	865	639	940	86.5
NO^3-N	4.66	1.1	24.1	5.19
Cl	22.5	11	34	6.65
NH^4	0.03	0.01	0.05	0.01
K	5.82	5.3	6.04	0.3
Na	10.56	8.31	12.27	1.43
K/Na Ratio	0.56	0.43	0.72	0.1
Mg	5.77	3.7	9.72	1.72

CB2 (Bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	2.62	1.29	3.7	0.68
Temperature	8.87	1.85	14.5	3.74
pH	5.47	5	6.18	0.31
EC	249	176	282	28.8
NO^3-N	4.59	1.1	8	1.78
Cl	16.3	8	31	5.46
NH^4	0.1	0.05	0.18	0.05

K	7.67	5.37	12.75	2.32
Na	11.19	9.93	13.33	1.42
K/Na Ratio	0.59	0.53	0.7	0.06
Mg	5.17	4.07	7.83	1.36
Mn	0.16	0.1	0.24	0.06
Zn	0.025	0	0.06	0.028

CB3 (Bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	2.89	1.65	4.38	0.79
Temperature	8.69	1	12.5	3.39
pH	5.76	5.04	6.44	0.36
EC	263	207	297	20.2
NO^3-N	3.35	0.2	4.9	1.34
Cl	36.7	20	96	17.55
NH^4	0.15	0.01	0.39	0.15
K	17.16	11.87	35.5	6.88
Na	14.5	13.36	15.82	0.97
K/Na Ratio	0.99	0.88	1.15	0.1
Mg	6.52	4.82	9.63	1.84
Mn	0.68	0.44	0.92	0.22
Zn	0.03	0.02	0.05	0.01

CB5 (Bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	1.19	0.63	1.82	0.3

Temperature	9.05	2.6	15.9	3.42
pH	5.73	5.18	6.26	0.24
EC	156.5	97	171	16.78
NO^3-N	2.14	0.2	3.4	0.79
Cl	14.6	7	31	6.29
NH^4	0.05	0.01	0.11	0.04
K	2.77	1.17	10.57	2.81
Na	8.87	8.4	9.54	0.36
K/Na Ratio	0.21	0.14	0.36	0.07
Mg	4.01	3.32	6.01	0.97
Mn	1.09	0.8	1.59	0.33
Zn	0.06	0.05	0.08	0.01

CB8 (Bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	1.36	0.73	2.1	0.38
Temperature	9.03	4.3	14.1	3.23
pH	5.94	5.5	6.27	0.25
EC	302	262	361	23.2
NO^3-N	3.51	0.3	8.4	2.33
Cl	21.7	11	64	11.36
NH^4	0.12	0.03	0.43	0.17
K	4.11	2.77	6.62	1.31
Na	8.66	8.06	9.45	0.59
K/Na Ratio	0.42	0.34	0.59	0.08
Mg	3.05	2.25	4.23	0.76
Mn	-	-	-	-

Zn	0.29	0.18	0.42	0.12

CB10 (Spring well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	0.6	0.39	1.21	0.21
Temperature	8.53	1	14.5	4.48
pH	5.96	5.2	6.81	0.52
EC	75.4	39	122	16.1
NO^3-N	1.27	0.1	3.4	0.75
Cl	10.26	4	19	3.69
NH^4	0.04	0.03	0.06	0.01
K	1.78	0.53	4.31	1.41
Na	6.79	5.71	7.63	0.71
K/Na Ratio	0.33	0.09	0.6	0.22
Mg	2.5	1.73	3.64	0.7
Mn	0.002	0	0.02	0.007
Zn	0.01	0.02	0.02	0.002

Mea4 (Bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	9.74	7.42	16.37	2.84
Temperature	7.5	4.5	13.2	2.87
pH	7.16	6.95	7.37	0.15
EC	633	366	709	114
NO^3-N	1.13	0.3	2.9	0.8
Cl	27.6	22	33	4.1
NH^4	0.07	0	0.16	0.05
K	4.08	3.26	4.85	0.85

Na	17.3	15.59	18.88	1.64
K/Na Ratio	0.21	0.18	0.25	0.03
Mg	11.38	10.86	11.84	0.4

Mea12 (Bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	2.27	2.04	2.76	0.22
Temperature	8.26	6.1	10.7	1.56
pH	7.06	6.88	7.25	0.15
EC	679	635	727	24.7
NO^3-N	1.51	0.2	4.9	1.69
Cl	27.5	17	44	10.41
NH^4	0.14	0.01	0.53	0.17
K	1.67	1.29	1.84	0.22
Na	15.84	13.84	17.8	1.39
K/Na Ratio	0.13	0.09	0.27	0.06
Mg	10.62	9.77	11.22	0.54

Mea15 (Combined hand-dug/bored well)

Parameter	Mean	Min	Max	Std Dev
Water Table Depth	2.49	1.54	4.02	0.89
Temperature	7.91	5.3	12	2.3
pH	7.09	6.89	7.29	0.14
EC	686	591	753	55
NO^3-N	2.26	0.9	5.5	1.41
Cl	22.1	15	38	8.25
NH^4	0.1	0.01	0.58	0.2

K	2.93	2.38	3.35	0.5
Na	12.37	10.32	14.82	2.27
K/Na Ratio	0.14	0.03	0.21	0.09
Mg	8.02	6.78	9.59	1.43

Mea18 (Hand-dug well)

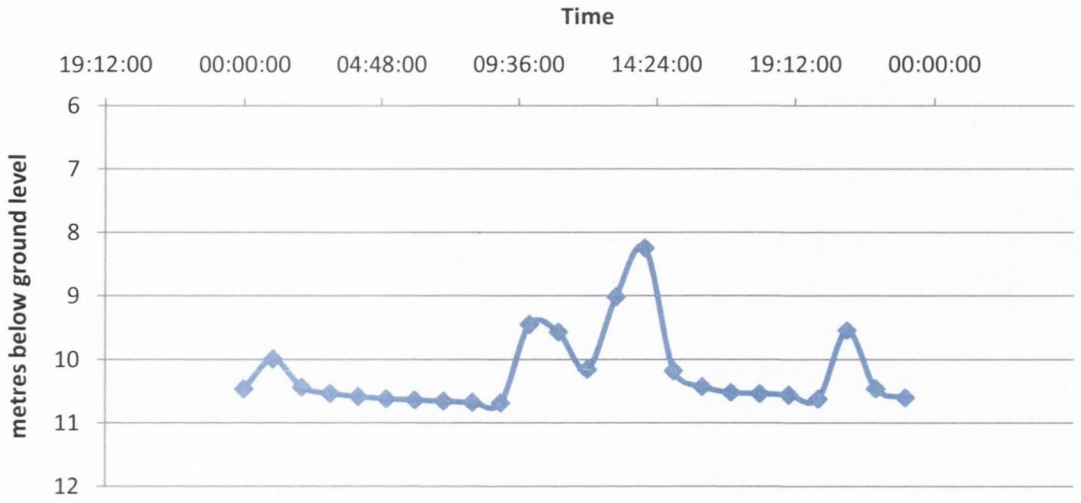
Parameter	Mean	Min	Max	Std Dev
Water Table Depth	1.17	0.93	1.4	0.15
Temperature	7.93	3	14	3.93
pH	7.04	6.76	7.36	0.18
EC	770	517	954	170
NO^3-N	3.01	0.9	7.9	2.57
Cl	31.37	14	44	11.36
NH^4	0.04	0.02	0.09	0.02
K	19.93	1.94	28.77	10.68
Na	15.84	11.42	21.32	4.54
K/Na Ratio	1.24	0.16	1.79	0.64
Mg	7.98	4.52	9.97	2.25

Mea21 (Bored well)

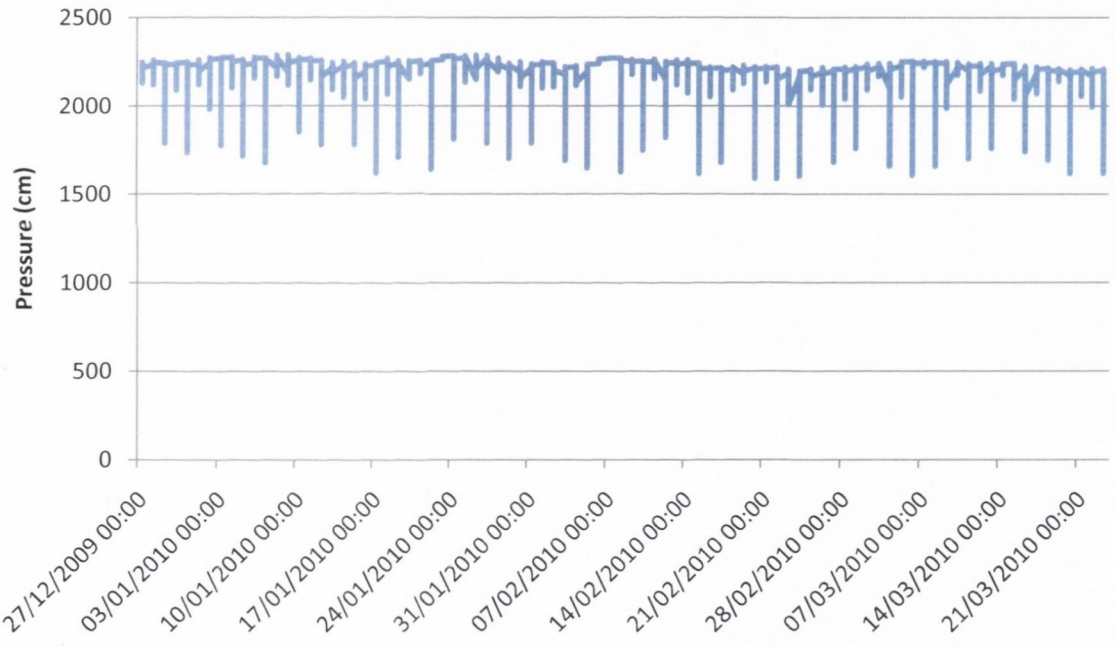
Parameter	Mean	Min	Max	Std Dev
Water Table Depth	1.44	1.03	1.71	0.2
Temperature	6.88	3.2	12.8	3.44
pH	7.15	6.89	7.45	0.2
EC	670	653	716	23
NO^3-N	1.28	0.2	3.7	1.21
Cl	25.3	10	38	9.3
NH^4	0.05	0.01	0.13	0.04

K	2.14	1.17	2.94	0.84
Na	14.73	9.25	18.47	4.19
K/Na Ratio	0.13	0.12	0.16	0.02
Mg	9.5	5.95	11.97	2.56

Appendix E15



Pressure transducer (Diver) output at BM1 during the 24-hour period 02/01/10



Pressure variation at BM1 from 27/12/09 – 21/3/10 (Transducer set at 20 m depth)

Appendix E16

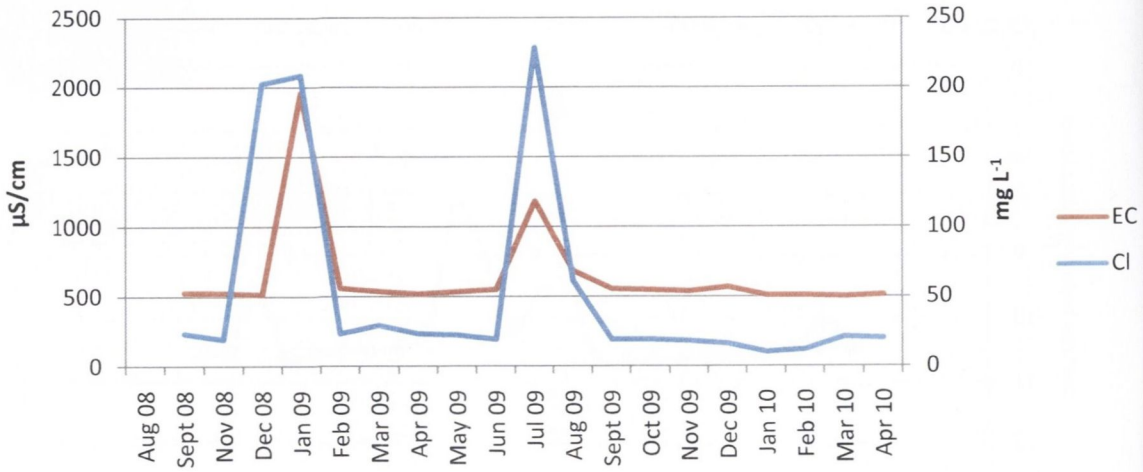


Figure 7.23 Study Area 1, Monthly measure Cl and EC at BM1

Appendix E17

Effective Precipitation (Recharge) Calculations

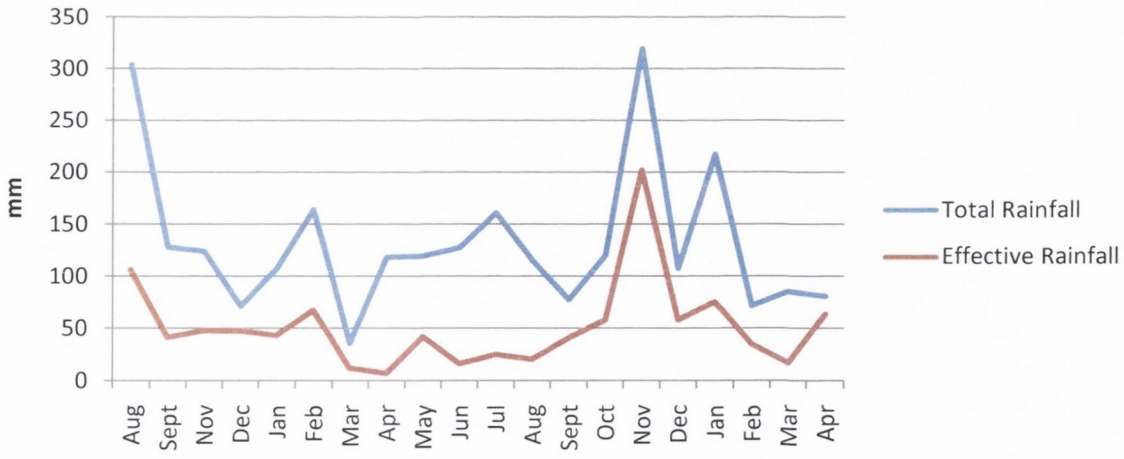
Effective precipitation calculations were done in Excel 2007 using daily total precipitation, evapotranspiration (Penman) and soil moisture deficits (high, medium or low permeability subsoil as appropriate). These datasets were acquired from the closest functioning Met Eireann synoptic station to the respective study areas. Calculations were undertaken using "If Statement" equations, written by the researcher, as follows:

$$SMD = if\ (smd + total\ rainfall - ET < 0,\ SMD + Total\ rainfall - ET,\ 0)$$

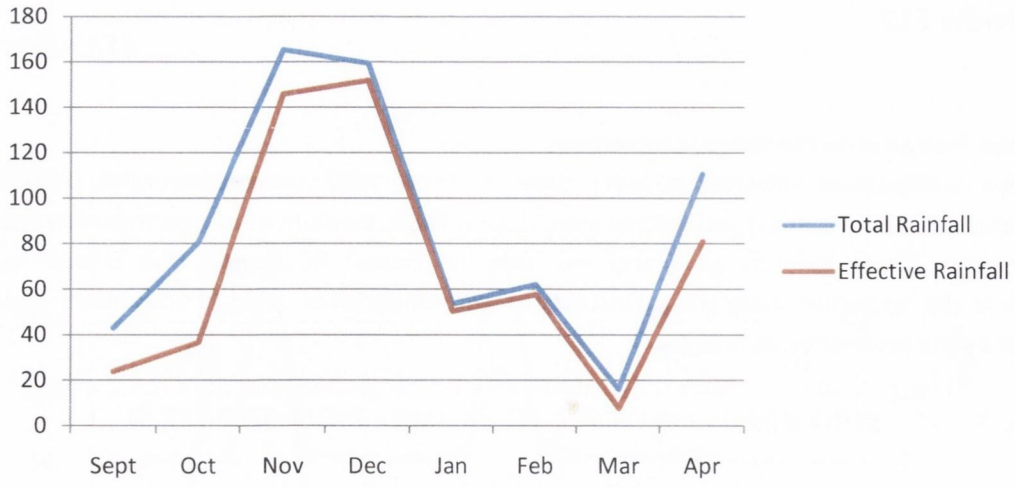
$$Effective\ Precipitation = if\ (SMD + Total\ rainfall - ET > 0,\ SMD + Total\ rainfall - ET,\ 0)$$



Study Area 1, Total vs Effective Precipitation

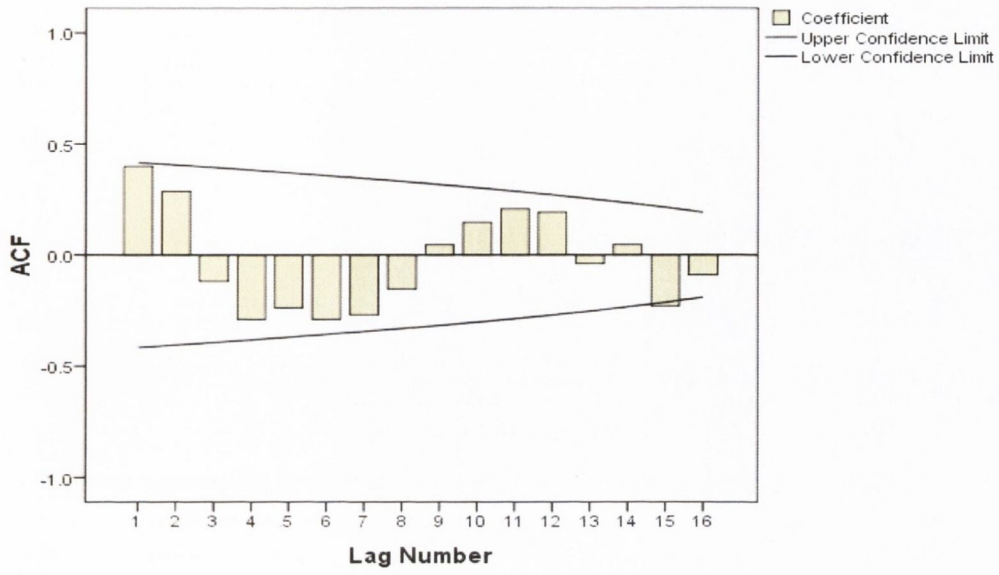


Study Area 2, Total vs Effective Precipitation

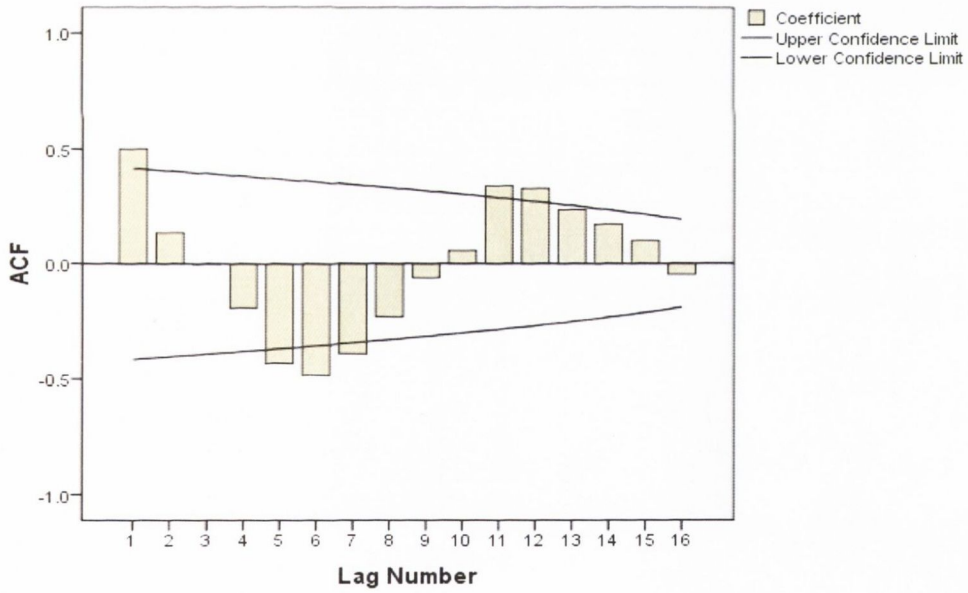


Study Area 4, Total vs Effective Precipitation

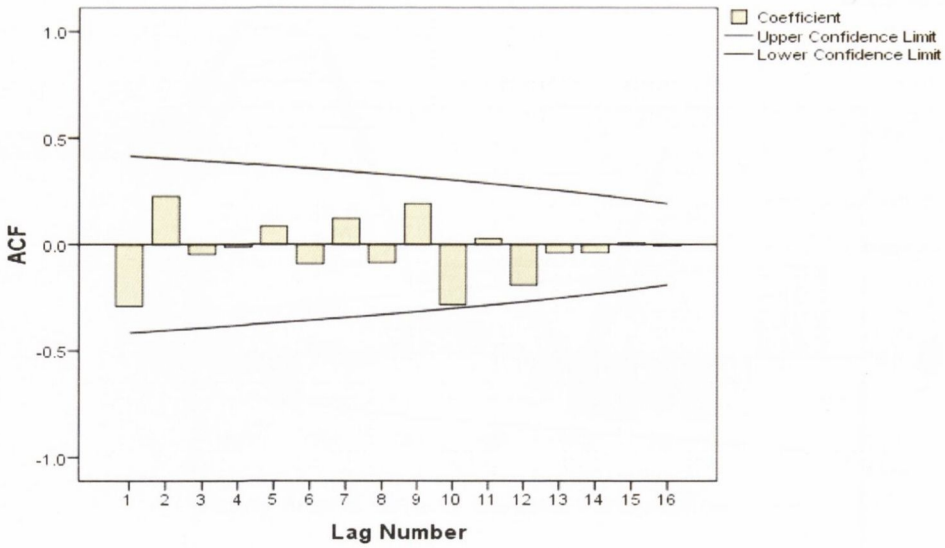
Appendix E18



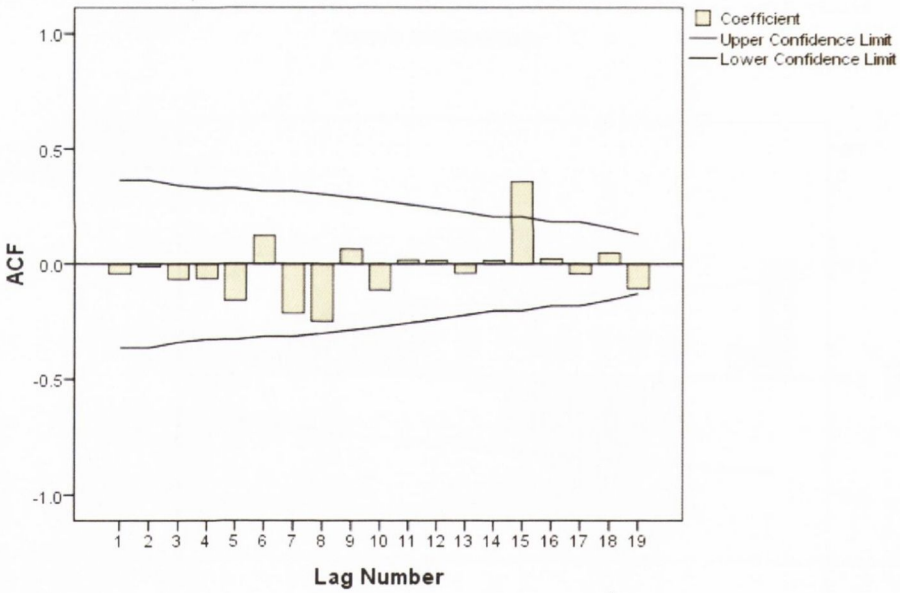
Study Area 2, Autocorrelation function of transformed (ln) mean monthly watertable depth



Study Area 2, Autocorrelation function of transformed (ln) mean monthly groundwater pH



Study Area 2, Autocorrelation function of transformed (ln) mean monthly groundwater Cl



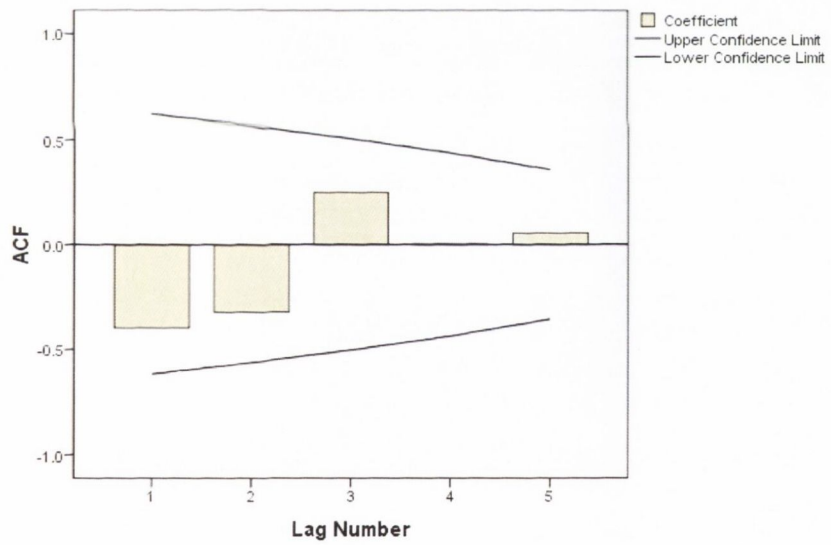
Study Area 2, Autocorrelation function of transformed (ln) mean monthly groundwater E. coli

Appendix E19

Study Area 2, K:Na Ratio vs *E. coli*

Source Reference	Mean K:Na Ratio where *E. coli* present	Mean K:Na Ratio where *E. coli* absent
CB2	0.7	0.56
CB3	1.06	0.96
CB8	0.45	0.39
CB11	0.33	0.25

Appendix E20



Study Area 4, Autocorrelation function of transformed (ln) mean monthly groundwater Cl

Appendix E21

Study Area 4, K:Na Ratio vs *E. coli*

Source Reference	Mean K:Na Ratio where *E. coli* present	Mean K:Na Ratio where *E. coli* absent
Mea4	0.195	0.25
Mea12	0.14	0.11
Mea15	0.03	0.2
Mea18	1.43	1.19
Mea21	0.13	0.14

Appendix F1

Survey Modality

Selection of data collection techniques is generally based on cost, completion time and response rate (Yann-Yann, 2003). As outlined in *Chapter 6 (Section 6.8.7)*, two differing survey completion modalities were used throughout the study in order to increase the overall response rate and in so doing, increase the significance and inferential power of associated findings. As shown throughout Chapter 8, many similar trends and patterns were established among both RA and SA datasets. However, as shown in *Section 8.6*, RA survey respondents typically exhibited higher overall awareness scores. This may be due to the generally younger respondent demographic associated with the SA survey, for example, residency during construction period was shown to be an important factor with regard to source awareness in both datasets (RA; $t(243) = -7.95$, $p < 0.001$. SA; $t(296) = -2.814$, $p = 0.005$). Therefore, due to the lower level of residency among SA respondents, a lower overall source awareness is to be expected and consequently found. However, demographic variance would not be expected to affect groundwater contamination awareness and therefore, were survey modality not an issue, both datasets are expected to display comparable contamination awareness. As shown (Table F1a), mean contamination awareness is substantially difference, with an independent samples t-test finding this mean difference to be statistically significant ($t(543) = -8.674$, $p < 0.001$).

This would seem to suggest that similar trends may be evident throughout both datasets and therefore these trends are likely to exist in the total population in addition to the sample population, and therefore inference may be made based on these datasets. However, due to differing levels of awareness, which cannot be accounted for based on demographic variation alone, the author believes that the two datasets should remain independent of each other and has therefore treated them accordingly throughout Chapter 8.

Table F1a. *Group statistics relating to RA and SA source satisfaction and awareness scores*

	Survey Type	N	Mean	Std. Deviation	Std. Error Mean
Supply Satisfaction	SA	294	4.53	0.853	0.050
	RA	245	4.53	0.880	0.056
Source Awareness	SA	300	5.36	2.162	0.125
	RA	245	7.93	2.325	0.149
Contamination Awareness	SA	300	2.36	1.484	0.086
	RA	245	3.43	1.385	0.086

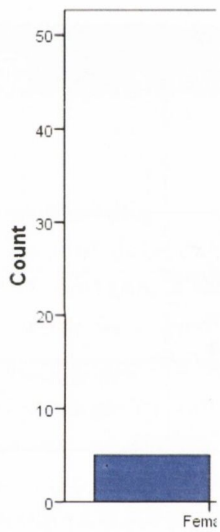
Appendix F2

Statistical procedures used in Chapter 8 (From Blalock Jr., 1960)

	Univariate Procedures	Bivariate Procedures			
		Dichotomous	Nominal	Ordinal	Interval
Dichotomous	Proportions, %, ratios	χ^2, difference of proportions, Fishers exact test	-	-	-
Nominal	Proportions, %, ratios	χ^2	χ^2	-	-
Ordinal	Median, quartiles, deviates	Mann-Whitney, Smirnov ranks	ANOVA, ANOVA with ranks	Rank order correlations, tau, gamma	-
Interval	Mean, median, St. Dev.	Difference of means (t-tests)	ANOVA, E^2, intraclass correlations	-	Correlation, regression

Appendix F3

Outline statistics for PuGWS awareness questionnaire respondents



Gender * Living Built (n = 63)

Count

		LivingBuilt		
		No	Yes	Total
Gender	Female	5	0	5
	Male	49	9	58
	Total	54	9	63

Gender * Contractor Awareness (n = 64)

Count

Gender * Knowledge of Well Age (n = 64)

Count

Gender		KnowledgeWellAge		
		No	Yes	Total
Gender	Female	5	0	5
	Male	56	3	59
	Total	61	3	64

Gender * Knowledge of Involvement (n = 64)

Count		InvolveKnow		
		Dont know	Know	Total
Gender	Female	5	0	5
	Male	45	14	59
	Total	50	14	64

Gender * Well Records Awareness (n = 64)

Count		WellRecKnowledge		
		Dont know	Know	Total
Gender	Female	4	1	5
	Male	42	17	59
	Total	46	18	64

Gender * Well Diameter Awareness (n = 64)

Count		WellDetDia		
		No	Yes	Total
Gender	Female	5	0	5
	Male	57	2	59
	Total	62	2	64

Gender * Well Depth Awareness (n = 64)

Count		WellDetDepth		
		No	Yes	Total
Gender	Female	5	0	5

	Male	55	4	59
	Total	60	4	64

Gender * Average Water Table Awareness (n = 64)

Count				
		WellDetWT		
		No	Yes	Total
Gender	Female	5	0	5
	Male	57	2	59
	Total	62	2	64

Gender * Well Liner Awareness (n = 64)

Count				
		WellDetLiner		
		No	Yes	Total
Gender	Female	5	0	5
	Male	58	1	59
	Total	63	1	64

Gender * Well Sealing Awareness (n = 64)

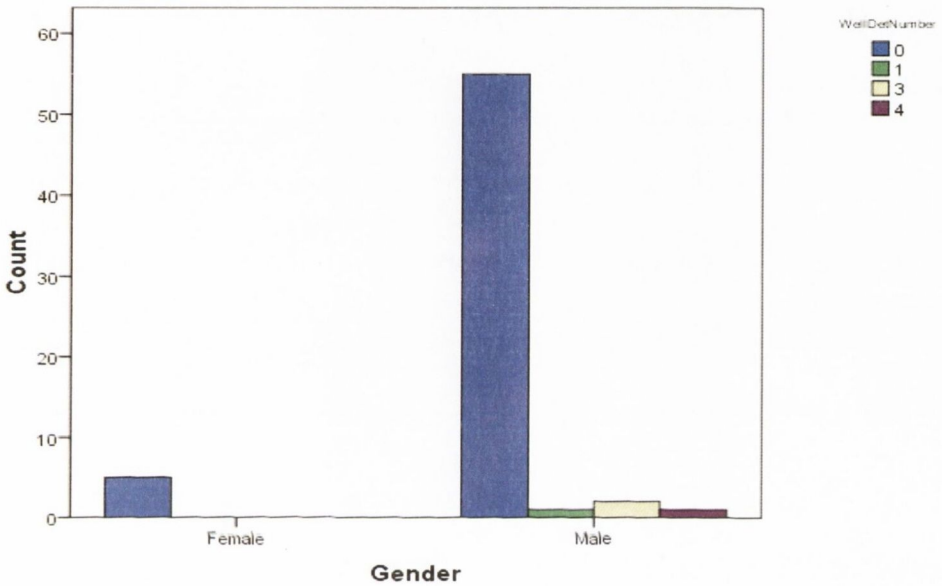
Count				
		WellDetSeals		
		No	Yes	Total
Gender	Female	5	0	5
	Male	58	1	59
	Total	63	1	64

Gender * Well Pump Type Awareness (n = 64)

Count				
		WellDetPump		
		No	Yes	Total
Gender	Female	5	0	5
	Male	58	1	59

Gender * Well Pump Type Awareness (n = 64)

Count				
		WellDetPump		
		No	Yes	Total
Gender	Female	5	0	5
	Male	58	1	59
	Total	63	1	64



Gender * Treatment Awareness (n = 58)

Count					
		Treatment			
		No	Yes	Dont know	Total
Gender	Female	1	2	2	5
	Male	7	23	23	53
	Total	8	25	25	58

Gender * Maintenance Awareness (n = 64)

Count				
		MaintenanceKnow		
		Dont know	Know	Total
Gender	Female	3	2	5

	Male	48	11	59
	Total	51	13	64

Gender * Source Maintenance (n = 13)

Count				
		Maintenance		
		No	Yes	Total

Gender * Source Cost (n = 50)

Count				
		FactorCost		
		No	Yes	Total
Gender	Female	3	2	5
	Male	19	26	45
	Total	22	28	50

Gender * Source Yield (n = 47)

Count				
		FactorYield		
		No	Yes	Total
Gender	Female	4	1	5
	Male	18	24	42
	Total	22	25	47

Gender * Source Quality (n = 50)

Count				
		FactorQual		

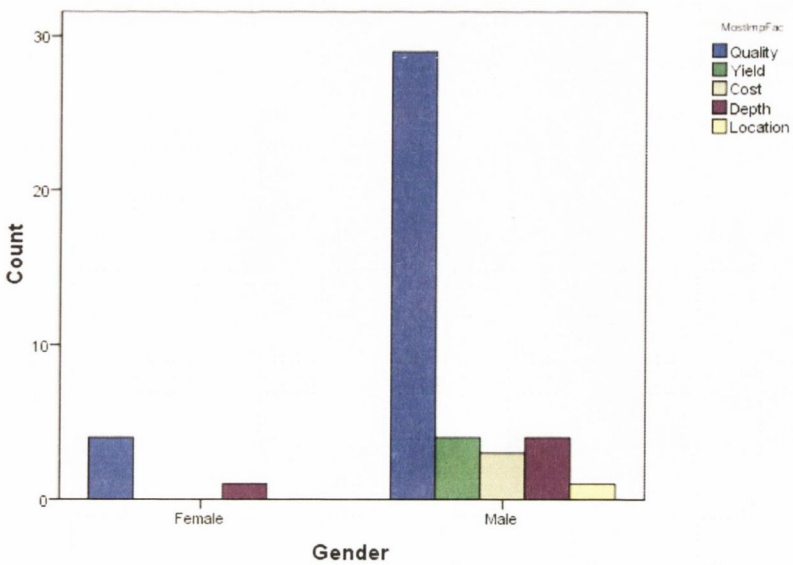
		No	Yes	Total
Gender	Female	1	4	5
	Male	10	35	45
	Total	11	39	50

Gender * Source Depth (n = 50)

Count				
		FactorDepth		
		No	Yes	Total
Gender	Female	4	1	5
	Male	27	18	45
	Total	31	19	50

Gender * Source Location (n = 50)

Count				
		FactorLoc		
		No	Yes	Total
Gender	Female	3	2	5
	Male	23	19	42
	Total	26	21	47



Gender * Water Test Awareness (n = 57)

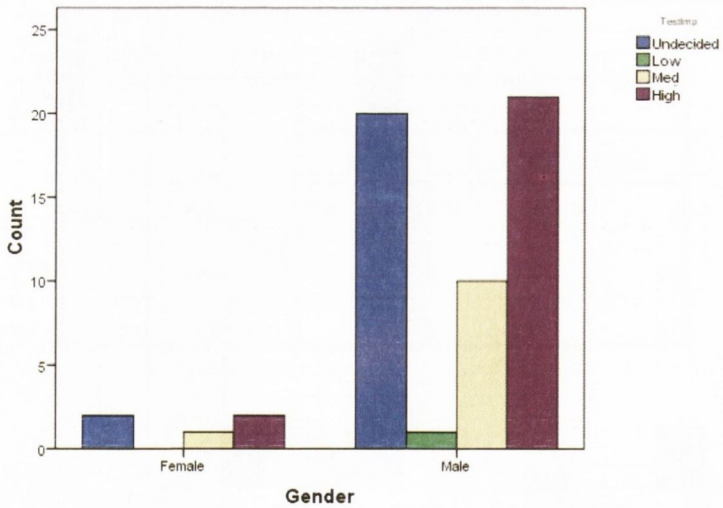
Count				
		WaterQualTestKnow		
		Dont know	Know	Total
Gender	Female	3	2	5
	Male	25	27	52
	Total	28	29	57

Gender * Water Quality Test (n = 27)

Count				
		WaterQualTest		
		No	Yes	Total
Gender	Female	0	2	2
	Male	9	16	25
	Total	9	18	27

Gender * Test Result Awareness (n = 16)

Count				
		WaterQualResults		
		Dont know	Know	Total
Gender	Male	13	3	16
	Total	13	3	16



Gender * Regular Testing (n = 56)

Count				
		PrepRegTesting		
		No	Yes	Total
Gender	Female	2	3	5
	Male	33	18	51
	Total	35	21	56

Gender * *E. coli* Awareness (n = 55)

Count				
		ContamColi		
		No	Yes	Total
Gender	Female	1	4	5
	Male	15	35	50
	Total	16	39	55

Gender * Rotavirus Awareness (n = 55)

Count				
		ContamRota		
		No	Yes	Total
Gender	Female	3	2	5
	Male	33	17	50
	Total	36	19	55

Gender * *Cryptosporidium* Awareness (n = 55)

Count				
		ContamCrypt		
		No	Yes	Total
Gender	Female	1	4	5
	Male	15	35	50
	Total	16	39	55

Gender * Lime Awareness (n = 55)

Gender * Manganese Awareness (n = 55)

Count		ContamMn		
		No	Yes	Total
Gender	Female	2	3	5
	Male	30	20	50
	Total	32	23	55

Gender * Nitrate Awareness (n = 55)

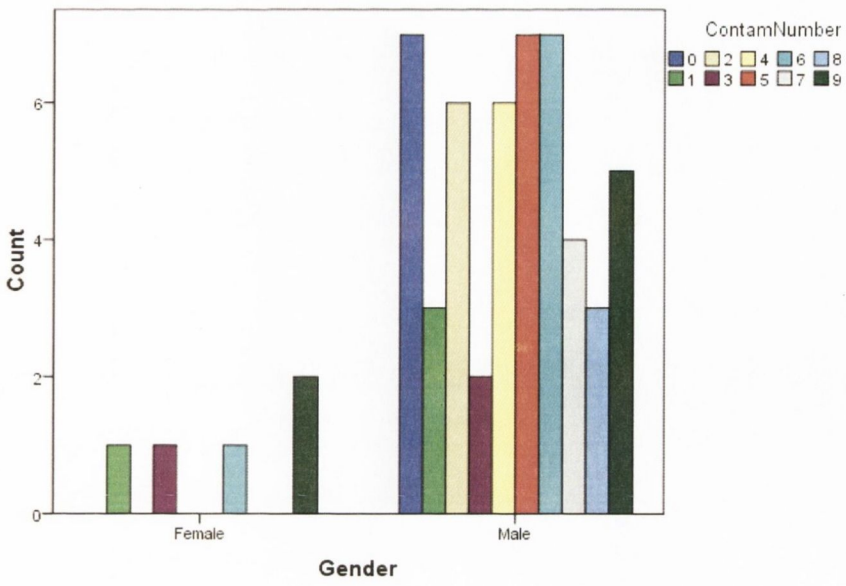
Count		ContamNit		
		No	Yes	Total
Gender	Female	1	4	5
	Male	22	28	50
	Total	23	32	55

Gender * Arsenic Awareness (n = 55)

Count		ContamAs		
		No	Yes	Total
Gender	Female	3	2	5
	Male	38	12	50
	Total	41	14	55

Gender * Fluoride Awareness (n = 55)

Count		ContamFl		
		No	Yes	Total
Gender	Female	1	4	5
	Male	20	30	50
	Total	21	34	55



Gender * Health Hazard Awareness (n= 56)

Count				
		HealthHazAware		
		No	Yes	Total
Gender	Female	3	2	5
	Male	27	24	51
	Total	30	26	56

Gender * Source Hazard Vicinity (n = 56)

Count				
		PotSourceHazAware		
		No	Yes	Total
Gender	Female	1	4	5
	Male	31	20	51
	Total	32	24	56

Gender * Farm within 100m (n = 54)

Count				
		PotHazFarm		
		No	Yes	Total
Gender	Female	4	1	5
	Male	30	19	49
	Total	34	20	54

Gender * Slurry/Fertiliser Spreading (n = 54)

Count

		PotHazSlurry		
		No	Yes	Total
Gender	Female	5	0	5
	Male	39	10	49
	Total	44	10	54

Gender * Silage Pit/Storage within 100m (n = 54)

Count

		PotHazSilage		
		No	Yes	Total
Gender	Female	5	0	5
	Male	41	8	49
	Total	46	8	54

Gender * Animal Grazing within 100 m (n = 54)

Count

		PotHazGraze		
		No	Yes	Total
Gender	Female	4	1	5
	Male	28	21	49
	Total	32	22	54

Gender * Septic Tank within 100 m (n = 54)

Count

		PotHazSeptic		
		No	Yes	Total
Gender	Female	5	0	5
	Male	37	12	49
	Total	42	12	54

Gender * Household Drinking Source (n = 55)

Count		DrinkingSource		
		Household Tap	Other	Total
Gender	Female	4	1	5
	Male	41	9	50
	Total	45	10	55

Gender * Average Daily Consumption Frequency (n = 46)

Count		ConFreq				
		1/day	2-3/day	4-5/day	>5/day	Total
Gender	Female	1	1	2	0	4
	Male	12	20	5	5	42
	Total	13	21	7	5	46

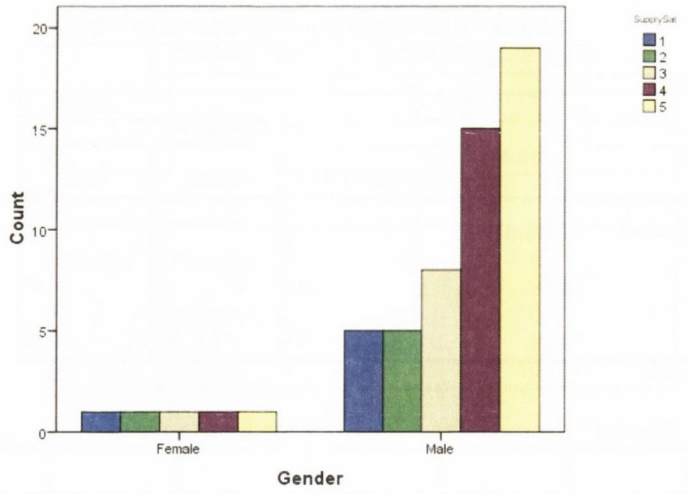
Gender * Average Daily Consumption Volume (n = 47)

Count		ConVo						
		<0.5	0.5-1	1-2	2-3	3-4	>4	Total
Gender	Female	1	0	1	1	1	0	4
	Male	4	10	15	7	3	4	43
	Total	5	10	16	8	4	4	47

There were no positive diagnosis of waterborne illnesses over the previous 12 months within the surveyed PuGWS population (N = 53)

Gender * Potential AGI (n = 52)

Count		PotGastro		
		No	Yes	Total
Gender	Female	4	1	5
	Male	36	11	47
	Total	40	12	52



Appendix F4

Source Type vs Hydrogeologist Involvement (n = 90)

Count				
		InvolveHydro		
		No	Yes	Total
SourceType	HD SPS	19	1	20
	Bored SPS	59	2	61
	PrGWS	8	0	8
	Other	1	0	1
	Total	87	3	90

Source Type vs Water Diviner Involvement (n = 90)

Count				
		InvolveDiv		
		No	Yes	Total
SourceType	HD SPS	3	17	20
	Bored SPS	7	54	61
	PrGWS	5	3	8
	Other	1	0	1
	Total	16	74	90

Source Type vs Local Authority Involvement (n = 90)

Count				
		InvolveLocAu		
		No	Yes	Total
SourceType	HD SPS	19	1	20
	Bored SPS	58	3	61
	PrGWS	2	6	8
	Other	0	1	1
	Total	79	11	90

Appendix F5

Source Type vs Knowledge of existing well records (n = 299)

Count				
		WellRecKnowledge		
		Dont know	Know	Total
SourceType	HD SPS	27	33	60
	Bored SPS	115	70	185
	Dont know	0	2	2
	PrGWS	30	15	45
	Other	6	1	7
	Total	178	121	299

Source Type vs Possession of well records (n = 121)

Count				
		WellRec		
		No	Yes	Total
SourceType	HD SPS	33	0	33
	Bored SPS	47	23	70
	Dont know	2	0	2
	PrGWS	14	1	15
	Other	1	0	1

Source Type vs Possession of well records (n = 121)

Count				
		WellRec		
		No	Yes	Total
SourceType	HD SPS	33	0	33
	Bored SPS	47	23	70
	Dont know	2	0	2
	PrGWS	14	1	15
	Other	1	0	1
	Total	97	24	121

Appendix F6

Source Type vs Knowledge of previous maintenance (n = 299)

Count				
		MaintenanceKnow		
		Dont know	Know	Total
SourceType	HD SPS	9	51	60
	Bored SPS	35	150	185
	Dont know	0	2	2
	PrGWS	28	17	45
	Other	5	2	7
	Total	77	222	299

Source Type vs Previous Maintenance (n = 222)

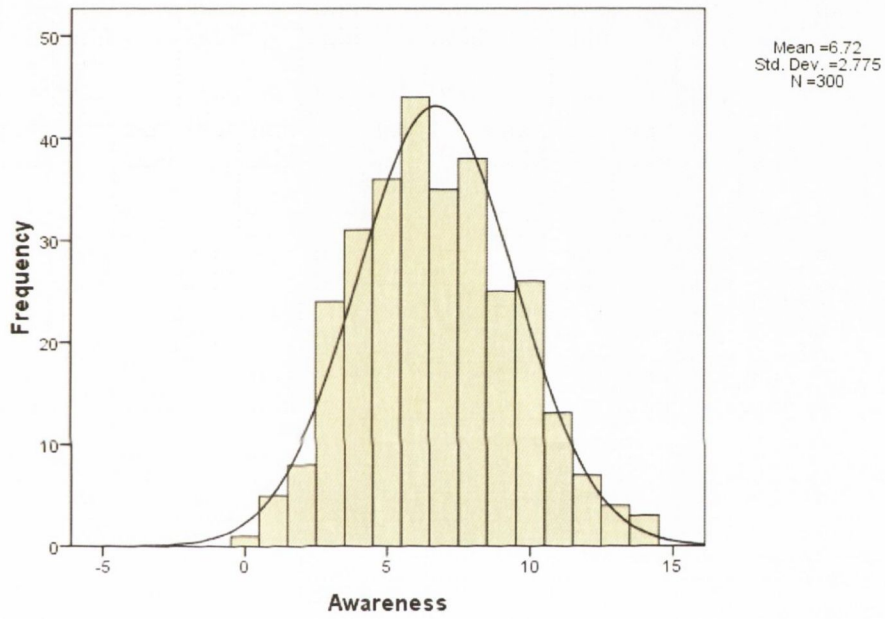
Count				
		Maintenance		
		No	Yes	Total
SourceType	HD SPS	26	25	51
	Bored SPS	101	49	150
	Dont know	2	0	2
	PrGWS	8	9	17

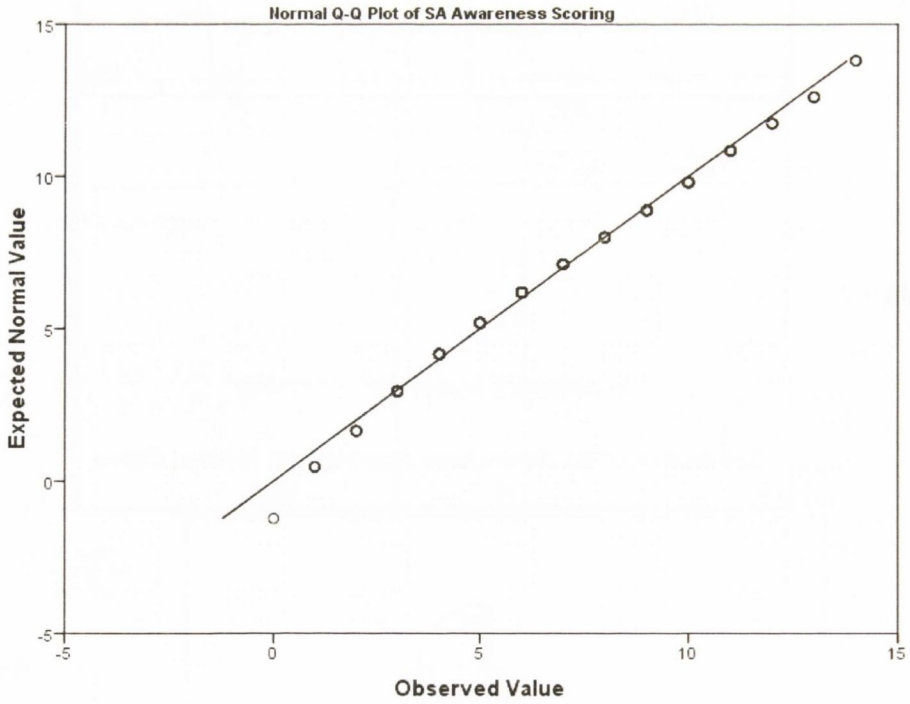
Other	2	0	2
Total	139	83	222

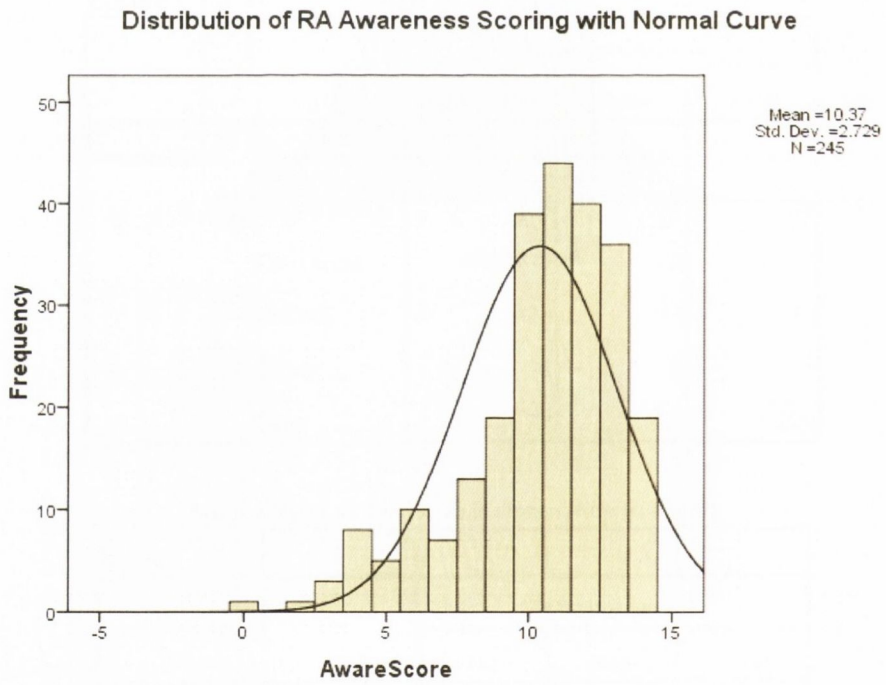
Appendix F7

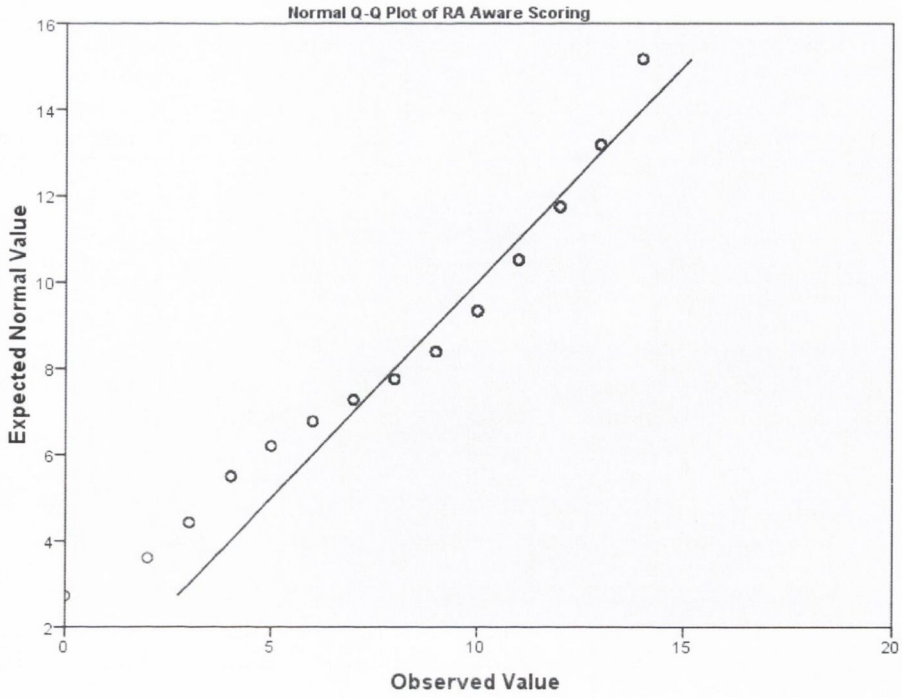
Exploratory analysis of normality

Distribution of SA Awareness Scoring with Normal Curve




Normal Q-Q Plot of SA Awareness Scoring
Expected Normal Value
Observed Value
15
10
5
0
-5
-5
0
5
10
15


Distribution of RA Awareness Scoring with Normal Curve
Mean =10.37
Std. Dev. =2.729
N =245
Frequency
AwareScore
50
40
30
20
10
0
-5
0
5
10
15



Appendix F8

Source Awareness vs Contamination Awareness Correlations

			SourceAwareness	ContamAwareness
Spearman's rho	SourceAwareness	Correlation Coefficient	1.000	-.073
		Sig. (2-tailed)	.	.258
		N	245	245
	ContamAwareness	Correlation Coefficient	-.073	1.000
		Sig. (2-tailed)	.258	.
		N	245	245

Health Hazard Awareness vs Source/Contamination Awareness

	HealthHazAware	N	Mean	Std. Deviation	Std. Error Mean
SourceAwareness	No	74	7.59	2.847	.331
	Yes	171	8.07	2.051	.157
ContamAwareness	No	74	2.12	1.122	.130

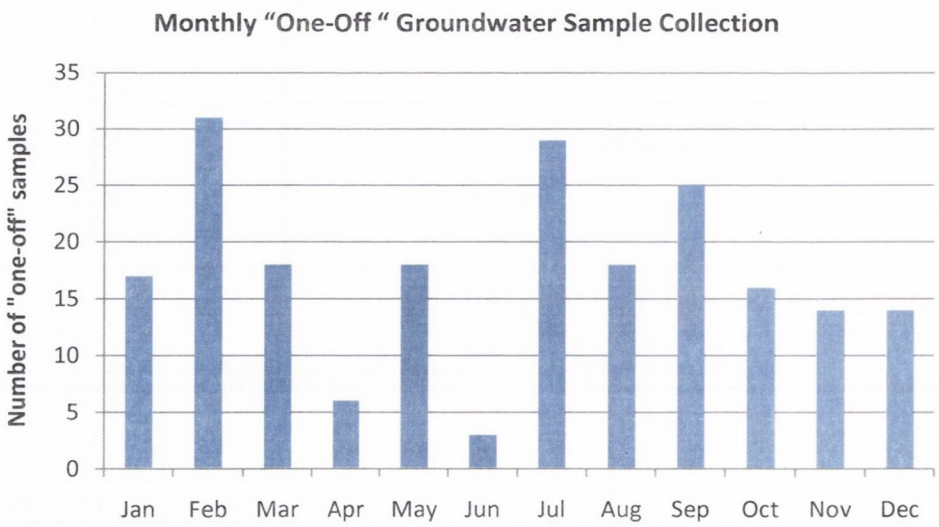
Health Hazard Awareness vs Source/Contamination Awareness

	HealthHazAware	N	Mean	Std. Deviation	Std. Error Mean
SourceAwareness	No	74	7.59	2.847	.331
	Yes	171	8.07	2.051	.157
ContamAwareness	No	74	2.12	1.122	.130
	Yes	171	4.00	1.068	.082

Source Satisfaction vs Source Awareness

	Levenes test for equality of variance		t-test for equality of means		
	F	Sig.	t	df	Sig.
Equal Variance Assumed	17.264	.000	-1.474	243	.142
Equal Variance Not Assumed			-1.298	107	.197

Appendix G1



Appendix G2

SPSS Codebook - Susceptibility Model

Variable 1: Source SPSS Code
SPSS Variable Code: SPSSRef
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: 1 – 211

Variable 2: Supply Type
SPSS Variable Code: SupplyType
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 0 = "HD SPS"; 1 = "Bored SPS"; 2 = "PrGWS"; 3 = "Spring Well"; 4 = "Combined Well"

Variable 3: Resident Number
SPSS Variable Code: ResNumber
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers

Variable 4: Dominant Bedrock
SPSS Variable Code: BedrockType
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "DKS"; 2 = "DESSL"; 3 = "DPUL"; 4 = "DLIL"; 5 = "DORS"; 6 = "SMV"; 7 = "DUIL"; 8 = "NU"; 9 = "DDL"; 10 = "WSA"; 11 = "WSH"; 12 = "NSA"; 13 = "NSH"; 14 = "DPBL"; 15 = "GII"

Variable 5: Dominant Subsoil
SPSS Variable Code: SubsoilType
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Limestone Tills"; 2 = "Peat"; 3 = "Sands and Gravels"; 4 = "Sandstone Tills"; 5 = "Shale/Sandstone Tills"; 6 = "Namurian Shale/Sandstone Tills"; 7 = "Granite Tills"

Variable 6: Non-Dominant Subsoil
SPSS Variable Code: SubsoilMix
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No"; 1 = "Yes"

Variable 7: Subsoil Thickness Range
SPSS Variable Code: SubsoilLayer
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "<3m"; 2 = ">3m"; 3 = "3-5m"; 4 = "5-10m"; 5 = "3-10m"; 6 = ">10m"

Variable 8: Aquifer Importance
SPSS Variable Code: AquiferImp
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Lm", 2 = "Ll", 3 = "Pl", 4 = "Rkd", 5 = Pu"

Variable 9: Groundwater Vulnerability Category
SPSS Variable Code: Vulnerability
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Low"; 2 = "Moderate"; 3 = "High"; 4 = "Extreme"; 5 = "Extreme Sur/Kar"

Variable 10: Gravel Aquifer
SPSS Variable Code: GravelAq
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 11: Source Age
SPSS Variable Code: SourceAge
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive Integers

Variable 12: Source Treatment
SPSS Variable Code: Treatment
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 13: Source Depth
SPSS Variable Code: SourceDepth
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive Integers

Variable 14: Site Gradient
SPSS Variable Code: SiteGrad
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Flat (>1:20)", 2 = "Shallow (1:20-1:10)", 3 = "Steep (<1:10)"

Variable 15: Number of residences supplied
SPSS Variable Code: HousesSupplied
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive Integers

Variable 16: Nearest road distance
SPSS Variable Code: RdDist
Data Type: Scale

Row Number: 1 – 211
Valid Input Codes: All positive Integers (m)

Variable 17: Gradient from source to road
SPSS Variable Code: RdGrad
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Up Gradient", 2 = "On Gradient/Level", 3 = "Down Gradient"

Variable 18: Road Type
SPSS Variable Code: RdType
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Primary", 2 = "Secondary", 3 = "Tertiary/Link"

Variable 19: Dominant Landuse Type in source proximity
SPSS Variable Code: Landuse
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Residential", 2 = "Agricultural", 3 = "Residential/Agricultural", 4 = "Forestry/Residential/Agricultural", 5 = "Other/Commercial/Industrial"

Variable 20: Surface Ponding Evident On-Site
SPSS Variable Code: SitePonding
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 21: Rivers within 300m
SPSS Variable Code: Rivers
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 22: Drainage Courses within 300m
SPSS Variable Code: Drains
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 23: Surface Water Body within 300m distance
SPSS Variable Code: SurfaceWaterDist
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (m)

Variable 24: Working Wells within 100m
SPSS Variable Code: WorkingWells
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 25: Abandoned Wells within 100m
SPSS Variable Code: AbandonedWells
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 26: Surface Springs within 100m
SPSS Variable Code: SurSpring
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 27: Vegetation Indicators
SPSS Variable Code: VegIndicators
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 0 = "None", 1 = "Phragmites", 2 = "Mosses"

Variable 28: Ground Condition within 100m
SPSS Variable Code: GrCondition
Data Type: Ordinal
Row Number: 1 – 211
Valid Input Codes: 1 = "Poor", 2 = "Poor/Med", 3 = "Med", 4 = "Med/Good", 5 = "Good"

Variable 29: Number of Septic Tanks within Approx 100m
SPSS Variable Code: STNumber
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers

Variable 30: Distance from source to closest septic tank
SPSS Variable Code: STDist
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (m)

Variable 31: Gradient from source to closest septic tank
SPSS Variable Code: STGrad
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Up Gradient", 2 = "On Gradient/Level", 3 = "Down Gradient"

Variable 32: Number of On-Site Treatment Systems within Approx 100m
SPSS Variable Code: OSTNumber
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers

Variable 33: Distance from source to closest On-Site System
SPSS Variable Code: OSTDist

Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (m)

Variable 34: Gradient from source to closest On-Site System
SPSS Variable Code: OSTGrad
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Up Gradient", 2 = "On Gradient/Level", 3 = "Down Gradient"

Variable 35: Grazing/Agriculture Type
SPSS Variable Code: AnimalType
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Beef", 2 = "Mixed", 3 = "Dairy", 4 = "Sheep", 5 = "Horses"

Variable 36: Distance from source to closest grazing animals/agricultural land
SPSS Variable Code: AnimalDist
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (m)

Variable 37: Gradient from source to closest animals/agricultural land
SPSS Variable Code: AnimalGrad
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Up Gradient", 2 = "On Gradient/Level", 3 = "Down Gradient"

Variable 38: Other potentially contaminating activities ongoing within approximately 100m
SPSS Variable Code: OtherActiv
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 39: Other potentially contaminating activities ongoing within approximately 100m
SPSS Variable Code: ActivType
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Farmyard", 2 = "Silage Pit", 3 = "Animal Housing incl Stables", 4 = "Garage/Residential Fuel Storage", 5 = "Other commercial/Construction"

Variable 40: Distance from source to other potentially polluting activities
SPSS Variable Code: ActivDist
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (m)

Variable 41: Gradient from source to other potentially polluting activities
SPSS Variable Code: ActivGrad
Data Type: Nominal
Row Number: 1 – 211

Valid Input Codes: 1 = "Up Gradient", 2 = "On Gradient/Level", 3 = "Down Gradient"

Variable 42: Presence/Absence of Secure Boundary
SPSS Variable Code: SecBound
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "Absent", 1 = "Present"

Variable 43: Area of secure boundary (if present)
SPSS Variable Code: SecBoundArea
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (m^2)

Variable 44: Wellhead Finish
SPSS Variable Code: WellheadFinish
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Above Ground", 2 = "At Ground Level", 3 = "Below Ground"

Variable 45: Presence/Absence of Wellhead Cover
SPSS Variable Code: WellheadCover
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "Absent", 1 = "Present"

Variable 46: Wellhead Cover Material (if present)
SPSS Variable Code: CoverMat
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Steel", 2 = "Concrete", 3 = "PVC/HDPE", 4 = "Other"

Variable 47: Presence/Absence of Liner Cap
SPSS Variable Code: LinerCap
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "Absent", 1 = "Present"

Variable 48: Liner Cap Material (if present)
SPSS Variable Code: CapMat
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Steel", 2 = "Concrete", 3 = "PVC/HDPE", 4 = "Other"

Variable 49: Surface Water Ponding within 10m of Wellhead (incl chamber)
SPSS Variable Code: WellPonding
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "Not Evident", 1 = "Evident"

Variable 50: Surface Water Ingress (chamber and/or liner)

SPSS Variable Code: Ingress
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 0 = "Not Evident", 1 = "Possible/Probable", 2 = "Evident"

Variable 51: Evidence of waterproof seals
SPSS Variable Code: Waterproof
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "Not Evident", 1 = "Evident"

Variable 52: Evidence of chamber/liner cracking
SPSS Variable Code: Cracking
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "Not Evident", 1 = "Evident"

Variable 53: Ground Condition within 10m of source
SPSS Variable Code: RadGrCond
Data Type: Ordinal
Row Number: 1 – 211
Valid Input Codes: 1 = "Poor", 2 = "Poor/Med", 3 = "Med", 4 = "Med/Good", 5 = "Good"

Variable 54: On-Site Ground Condition
SPSS Variable Code: SiteCond
Data Type: Ordinal
Row Number: 1 – 211
Valid Input Codes: 1 = "Poor", 2 = "Poor/Med", 3 = "Med", 4 = "Med/Good", 5 = "Good"

Variable 55: Flange to ground/chamber floor clearance
SPSS Variable Code: FlangeConc
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (mm)

Variable 56: Source Construction Materials
SPSS Variable Code: ConsMat
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Concrete/Steel, 2 = "Concrete", 3 = "Stone", 4 = "Steel", 5 = "PVC/HDPE"

Variable 57: Liner Material
SPSS Variable Code: LinerMat
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Steel, 2 = "Concrete", 3 = "PVC/HDPE", 4 = "No Liner", 5 = "Stone"

Variable 58: Liner Diameter
SPSS Variable Code: LinerDia
Data Type: Scale
Row Number: 1 – 211

Valid Input Codes: All Positive Integers (mm)

Variable 59: Chamber Presence/Absence
SPSS Variable Code: ChamberPres
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "Absent", 1 = "Present"

Variable 60: Chamber Construction Material (if present)
SPSS Variable Code: ChamberMat
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Concrete Block", 2 = "Concrete Solid", 3 = "PVC", 4 = "Steel", 5 = "Dug Hole (Soil)"

Variable 61: Chamber Area (if present)
SPSS Variable Code: ChamberArea
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (m^2)

Variable 62: Chamber Depth (if present)
SPSS Variable Code: ChamberDepth
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (m)

Variable 63: Chamber Floor Finish (concrete base, level, drained)
SPSS Variable Code: ChamberFloor
Data Type: Dichotomous
Row Number: 1 – 211
Valid Input Codes: 0 = "No", 1 = "Yes"

Variable 64: Pump Type
SPSS Variable Code: PumpType
Data Type: Nominal
Row Number: 1 – 211
Valid Input Codes: 1 = "Submersible", 2 = "Surface", 3 = "No Pump (likely gravity feed)"

Variable 65: Water Table upon sample collection
SPSS Variable Code: SampleWT
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (m)

Variable 66: Overall Source Assessment
SPSS Variable Code: OverallAss
Data Type: Ordinal
Row Number: 1 – 211
Valid Input Codes: 1 = "Poor", 2 = "Poor/Med", 3 = "Med", 4 = "Med/Good", 5 = "Good"

Variable 67: Previous 24-hour precipitation
SPSS Variable Code: Hr24Precip
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (mm)

Variable 68: Previous 48-hour precipitation
SPSS Variable Code: Hr48Precip
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (mm)

Variable 69: Previous 120-hour precipitation
SPSS Variable Code: Hr120Precip
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (mm)

Variable 70: Previous month precipitation
SPSS Variable Code: MonthPrecip
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (mm)

Variable 71: Maximum temperature during 24-hour sampling period
SPSS Variable Code: MaxTemp
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (°C)

Variable 72: Minimum temperature during 24-hour sampling period
SPSS Variable Code: MinTemp
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (°C)

Variable 73: Average temperature during 24-hour sampling period
SPSS Variable Code: AveTemp
Data Type: Scale
Row Number: 1 – 211
Valid Input Codes: All positive integers (°C)

Variable 74: Supply Satisfaction
SPSS Variable Code: SupplySat
Data Type: Ordinal
Row Number: 1 – 211
Valid Input Codes: 1 = "Very Dissatisfied", 2 = "Somewhat dissatisfied", 3 = Neither satisfied nor dissatisfied", 4 = "Somewhat satisfied", 5 = "Very satisfied"

Variable 75: Source Awareness Score
SPSS Variable Code: SourceAware

Data Type: Ordinal
Row Number: 1 – 211
Valid Input Codes: 0-10

Variable 76: Contamination Awareness Score
SPSS Variable Code: ContamAware
Data Type: Ordinal
Row Number: 1 – 211
Valid Input Codes: 0-5

Variable 77: Total Awareness Score (SourceAware + ContamAware – 1)
SPSS Variable Code: TotalAware
Data Type: Ordinal
Row Number: 1 – 211
Valid Input Codes: 0-14

Appendix G3

Worked example for source with no contamination present

From *Equation 9.3*:

$$P = \frac{e^{(a+bx)}}{(1 + e^{(a+bx)})}$$

a = -1.015

b_1 = -3.49; X_1 = 0 if point agricultural source >100m, 1 if point agricultural source <100m

b_2 = -5.007; X_2 = 0 if grazing land not up-gradient, 1 if grazing land up-gradient

b_3 = -5.076, X_3 = 0 if no liner clearance, 1 if liner clearance

b_4 = -3.159, X_4 = 0 if wellhead uncovered, 1 if wellhead covered

b_5 = 0.272, X_5 = Precipitation (mm) over 120-hour duration prior to sampling

b_6 = 0.024, X_6 = Precipitation (mm) over 30-day duration prior to sampling

b_7 = 3.687, X_7 = 0 if roadway not down gradient, 1 if roadway down gradient

Therefore;

$$P = \frac{[\exp(a + b_1X_1 + b_2X_2 + \ldots\ldots b_7X_7)]}{1 + \exp(a + b_1X_1 + b_2X_2 + \ldots\ldots b_7X_7)}$$

CW1 (0 cfu/100ml):
No point agricultural source (i.e. farmyard, feed storage, silage pit) within 100 m; X_1 = 0
Grazing land not up-gradient; X_2 = 0
65 mm liner clearance; X_3 = 1
Wellhead covered with well-fitted concrete slab; X_4 = 1
120-hour precipitation – 0.6 mm
30-day precipitation – 97.8 mm

Roadway measured 46 m down gradient; $X_7 = 1$

The regression equation for SA1BM13 is:

$$P = \frac{\exp(-3.1994)}{1 + \exp(-3.1994)} = \frac{0.0407}{1.0407} = 0.039$$

As previously outlined, a classification cutoff of 0.5 was used and therefore the resulting SA1 regression model correctly predicts that SA1 BM13 has no *E. coli* present i.e. 3.9% probability of contamination.

Appendix G4

TABLE B.3. RECOMMENDED MINIMUM DISTANCE BETWEEN A RECEPTOR AND A PERCOLATION AREA OR POLISHING FILTER

T/P-value[1]	Type of soil/subsoil[2]	Depth of soil/subsoil (m above bedrock) (see Notes 1, 2, 3, 6)	Minimum distance (m) from receptor to percolation area or polishing filter[5]				
			Public water supply	Karst feature	Down-gradient domestic well or flow direction is unknown (see Note 5)	Domestic well alongside (no gradient)	Up-gradient domestic well
>30	CLAY; sandy CLAY (e.g. clayey till); SILT/CLAY	1.2 >3.0	60	15	40 30	25	15
10–30	Sandy SILT; silty SAND; silty GRAVEL (e.g. sandy till)	1.2 >8.0	60	15	45 30	25	15
<10	SAND; GRAVEL; silty SAND	2.0[3] 2.0[4] >8.0[4]	60	15	60 40 30	25	15

[1]The T-value (expressed as min/25 mm) is the time taken for the water level to drop a specified distance in a percolation test hole. For shallow subsoils the test hole requirements are different and hence the test results are called P-values. For further advice see Annex C.
[2]BS 5930 descriptions.
[3]Water table 1.2–2.0 m.
[4]Water table >2.0 m.
[5]The distance from the percolation area or polishing filter means the distance from the periphery of the percolation area or polishing filter and not from the centre.

From EPA, 2009d

Appendix G5

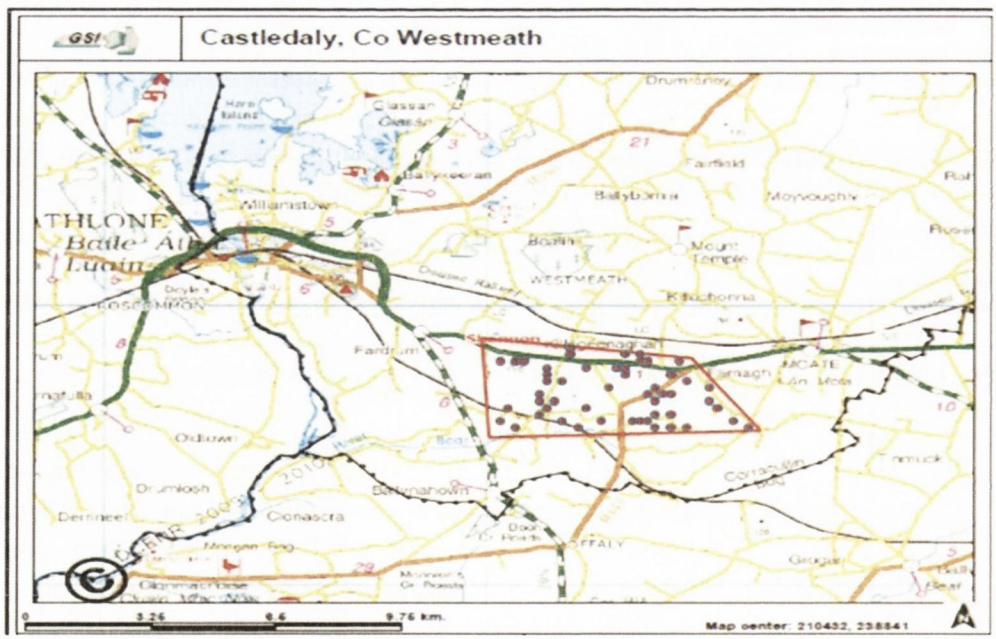
Validation Set: Castledaly, Co Westmeath

Castledaly, Co Westmeath is located in the southwest of Co Westmeath, approximately 110km west of Dublin City and covers an area of approximately 19.2 km^2, with a central point of Castledaly village, situated approximately 3 km from the Co Offaly border. In all, 51 private groundwater sources were assessed and sampled, resulting in a sample source density of 2.6 km^{-2}. The area is situated in the Shannon River Basin District (RBD).

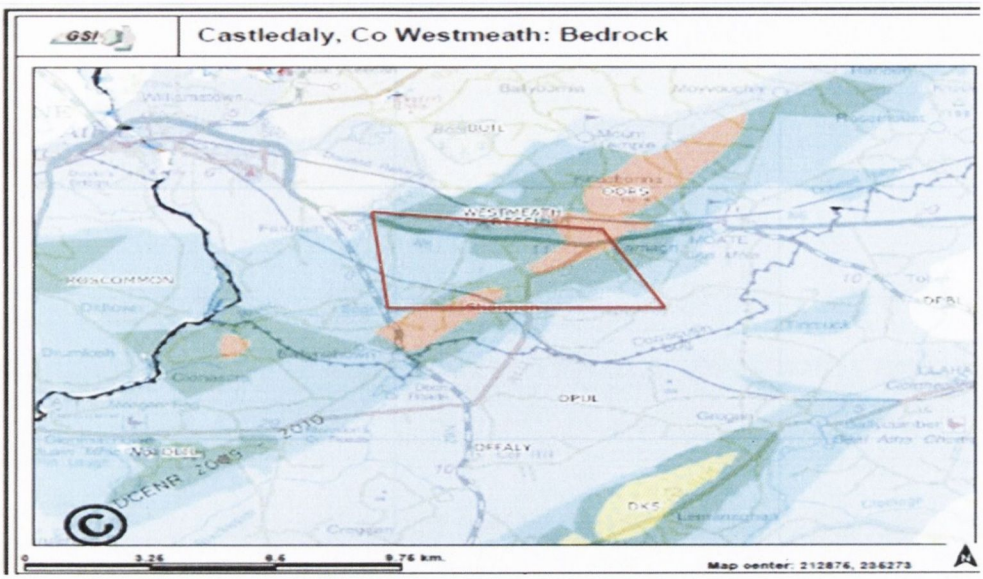
The superficial geology in the validation set area isrelatively uniform with Carboniferous limestone tills predominant (>90% of area). Small pockets of alluviums and glaciofluvial (Carboniferiou) limestone sands and gravels are dispersed throughout. The bedrock geology is predominantly Dinantian lower impure limestones (40%), Dinantian early limestones (40%), with approximately 15% of the area underlain by Devonian Old Red sandstones, primarily to the west.

Approximately 85% of the validation set area is designated as being of *High* vulnerability, with the remainder composed *Moderate* vulnerability, located to the south west of the validation area. The aquifer in the area is entirely characterised as being locally important, with bedrock being moderately productive only in local zones (Ll).

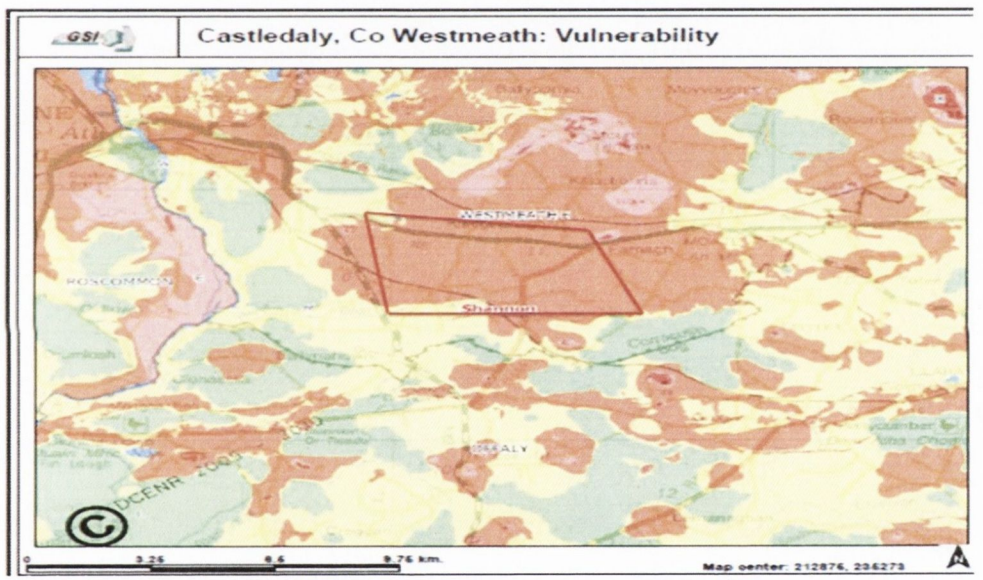
As for SA1, the validation set study area is entirely un-serviced by public water supply, all residences in the vicinity were reliant upon private groundwater supplies, with both HD SPS and Bored SPS being used. The primary landuse in the area was agricultural with mixed grazing of beef cattle and sheep being the dominant agricultural activity.



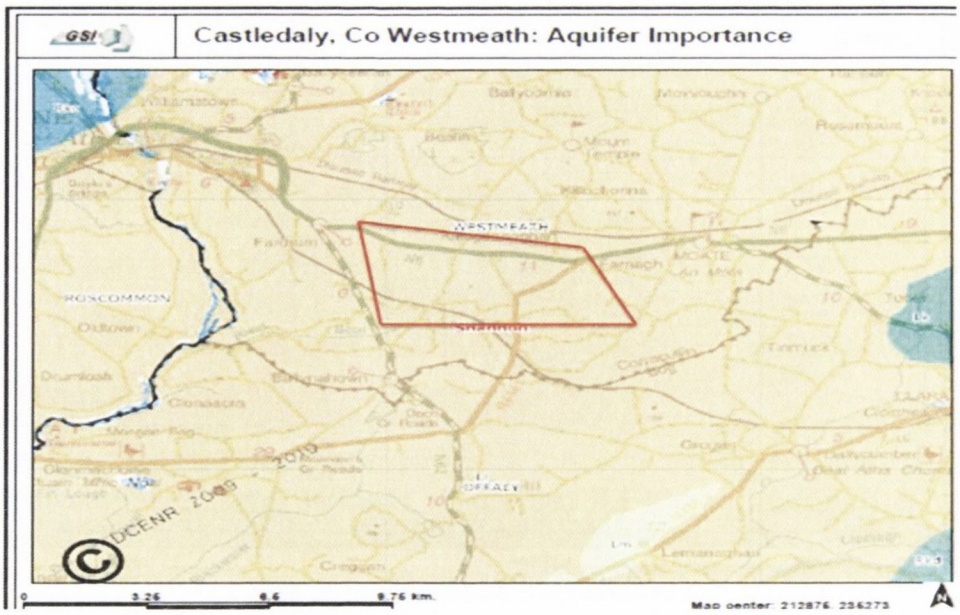
Validation Set: Location Map



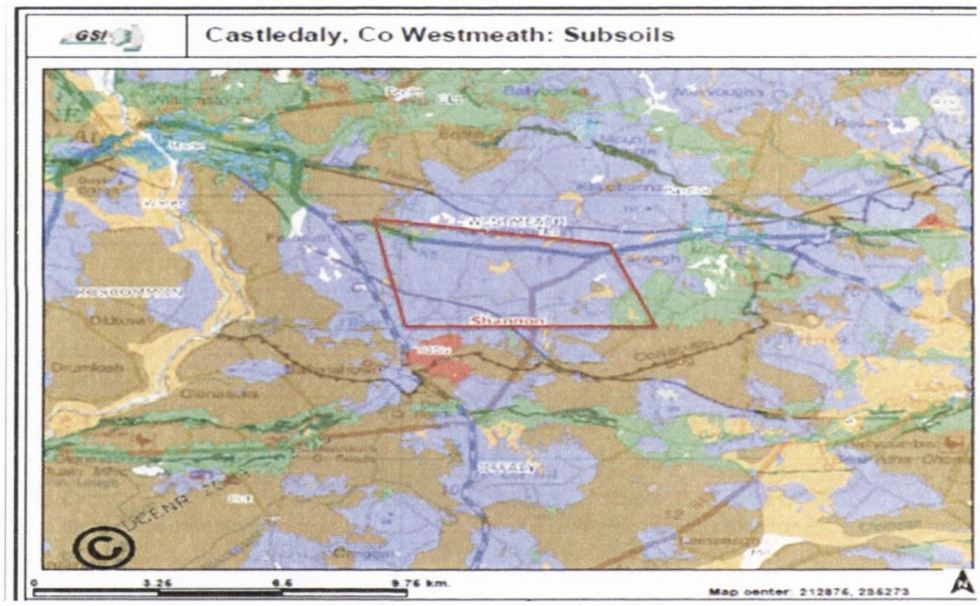
Validation Set: Bedrock Geology Map



Validation Set: Groundwater Vulnerability Map



Validation Set: Aquifer Importance Map



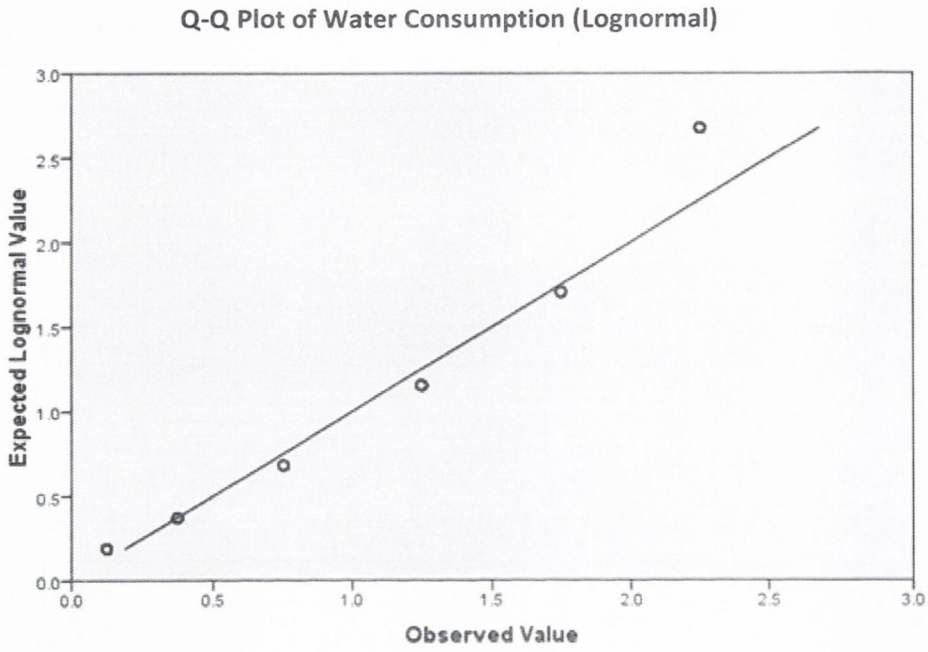
Validation Set: Subsoils Map

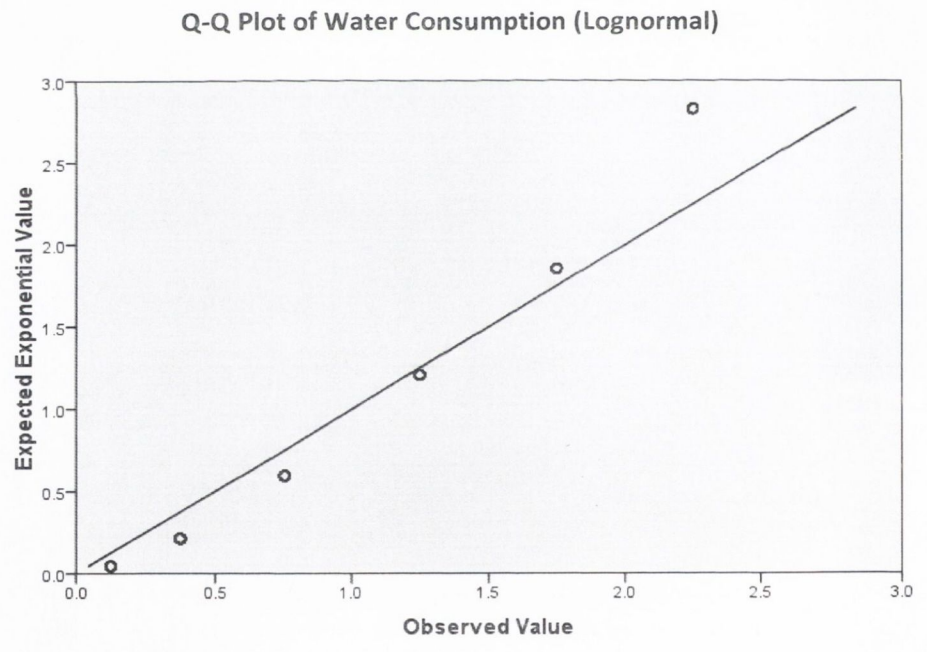
Appendix H1

Summary of drinking water consumption studies

Country	Sample (n)	Study Type	Mean Daily Consumption (L day^{-1})	Reference
Australia	234	D	0.89	Robertson *et al*, 2000
Bangladesh	38	Q	3	Watanabe *et al*, 2004
Canada	1,757	Q	1	Jones *et al*, 2006
Central Europe	1,392	Q	1.2 ± 0.5	Lloyd Hough *et al*, 2010
France	427	D	0.9	Gofti-Laroche *et al*, 2001
Isreal	5m	D	1.1 ± 0.1	Kristal-Bonch *et al* 1995
Korea	2,240	Q	1.016	Ji *et al*, 2010
UK	416	Q	0.7	Hunter *et al*, 2004
USA	73	Q	0.9 ± 0.6	Ryan *et al*, 2000
USA	26,081	Q	1.1	Roseberry & Burmaster, 1992
Sweden	11,189	Q & D	0.86 ± 0.48	Westrell *et al*, 2006
Canada	2,332	T	1.39	Pintar *et al*, 2009
Canada	1,757	T	1	Jones *et al*, 2006
The Netherlands	-	Q	0.153	Teunis *et al*, 1997
Canada	125	D	1.5	Levallois *et al*, 1998
UK	3,564	D	0.2	Hopkins & Ellis, 1980
USA	1,183	Q	1.9	Williams *et al*, 2001
Germany	195	Q	0.5	Dagendorf, 2003

Appendix H2


Q-Q Plot of Water Consumption (Lognormal)
Expected Lognormal Value
Observed Value
0.0
0.5
1.0
1.5
2.0
2.5
3.0


Q-Q Plot of Water Consumption (Lognormal)
Expected Exponential Value
Observed Value
0.0
0.5
1.0
1.5
2.0
2.5
3.0

Appendix H3

Deterministic Model 1 (Teunis et al., 1997)

***E. coli* Calculations Modules 2-4**

Daily dose and probability of infection

Module 2 (Untreated HD SPS) $Dose = 5\frac{cfu}{100ml} \times \frac{1}{1} \times 0.5 \times 10^{0} \times 9.26$

Dose = 23.15 cfu/day

$P_i(day^{-1}) = 2.2 \times 10^{-2}$

Module 3 (Bored Bored SPS) $Dose = 3\frac{cfu}{100ml} \times \frac{1}{1} \times 0.5 \times 10^{-1} \times 9.26$

Dose = 1.38 cfu/day

$P_i(day^{-1}) = 1.3 \times 10^{-3}$

Module 4 (Untreated HD SPS) $Dose = 5\frac{cfu}{100ml} \times \frac{1}{1} \times 0.5 \times 10^{-1} \times 9.26$

Dose = 2.3 cfu/day

$P_i(day^{-1}) = 2.2 \times 10^{-3}$

Frequency of contamination (E. coli)

Module 2 (Hand-dug; Untreated): $\left(\frac{4.9}{12} \times 5\right) 2.2 \times 10^{-2} = 0.045$

Module 3 (Bored; Treated): $\left(\frac{3.6}{12} \times 5\right) 1.3 \times 10^{-3} = 0.0019$

Module 4 (Hand-dug; Treated): $\left(\frac{4.9}{12} \times 5\right) 2.2 \times 10^{-3} = 0.0045$

Infections per year

Module 2:

$[14{,}500 \times 2.95 \times 0.951 \times 0.893] \times 0.476 \times 0.08 \times 0.045 \times 1.15 = 72\ infections/year$

(CIR; 168/100,000)

Module 3:

$[130{,}800 \times 2.95 \times 0.173 \times 0.893] \times 0.248 \times 0.08 \times 0.0019 \times 1.15 = 2.58\ infections/year$
(CIR; 3.86/100,000)

Module 4:

$[14{,}500 \times 2.95 \times 0.049 \times 0.893] \times 0.476 \times 0.08 \times 0.0045 \times 1.15 = 0.37\ infections/year$ (CIR; 17.6/100,000)

- Total Infections; 210/annum
- HUS Rate (2.1/100 illnesses; Rangal *et al*, 2005) – 4.4 HUS cases /annum, leading to 3-5% mortality rate (Siegler, 1997) – 0.13-0.22 HUS mortality/annum
- General Mortality Rate (From Macler & Merkle, 2000):

Vulnerable population (<5 &>65 years): 20.25% - 43 illnesses in vulnerable population; $254 \times 0.005 = 0.21\ deaths/annum$

Non-vulnerable" population: 79.75% - 167 illnesses in non-vulnerable population; $167 \times 0.0016 = 0.26\ deaths/annum$

Appendix H4

Deterministic Model 1 (Teunis et al., 1997)

E. coli Model Inputs

	Module 1	Module 2	Module 3	Module 4
C	3 cfu/100ml	5 cfu/100ml	3 cfu/100ml	5 cfu/100ml
Rr	1 (100%)	1 (100%)	1 (100%)	1 (100%)
I	0.5	0.5	0.5	0.5
DR	0	0	-1	-1
V	926 ml/day	926 ml/day	926 ml/day	926 ml/day
Private Groundwater Population	130,800 x 2.95	14,500 x 2.95	130,800 x 2.95	14,500 x 2.95
Level of Treatment	1 – 0.173	1 – 0.049	1 – 0.827	1 – 0.951
Household Consumption	0.893	0.893	0.893	0.893
Contamination Rate	0.248	0.476	0.248	0.476
Pathogenicity	0.08	0.08	0.08	0.08
Secondary Infection Rate	1.15	1.15	1.15	1.15
r	$1x10^{-3}$	$1x10^{-3}$	$1x10^{-3}$	$1x10^{-3}$

Appendix H5

Deterministic Model 1 (Teunis et al., 1997)

***Cryptosporidium* spp.Calculations Module 2**

$$Dose = 0.922 \times {}^{1}/_{0.7} \times 1 \times 10^{-1} \times 0.926$$

Dose = 0.12 oocysts/person/day

$$P_i(day^{-1}) = 1 - e^{-4x10^{-3}\,x\,0.12} = 4.79\,x\,10^{-4}$$

Assuming a 30-day period of contamination (Taylor *et al*, 2004), once per year:

$$P_{i(annual)} = 4.79 \times 10^{-4} \times 30 = 1.44 \times 10^{-2}$$

No. of illnesses

$[145{,}300 \times 2.95 \times 0.147 \times 0.893] \times 0.049 \times [1.44 \times 10^{-2}] \times 1.25 = 49\ infections/year$ (CIR; 11.4/100,000)

- Total Infections; 2,968/annum
- General Mortality Rate: *Cryptosporidium* – 0.1% of elderly and infants, as much as 50% in immuno-suppressed individuals (Haas *et al*, 1999; Gerba & Haas, 1998; Gerba *et al*, 1996)

  Vulnerable population (<5 &>65 years): 20.25% - 601 illnesses in vulnerable population; $155 \times 0.001 = 0.6\ mortalities/annum$

Appendix H6

Deterministic Model 1 (Teunis et al., 1997)

Cryptosporidium spp. Model Inputs

	Module 1	Module 2
C	1.22	0.012
Rr	0.7	0.7
I	1	1
DR	0	-1
V	926 ml/day	926 ml/day
Private Groundwater Population	145,300 × 2.95	145,300 × 2.95
Level of Treatment	1 – 0.147	1 – 0.853
Household Consumption	0.893	0.893
Contamination Rate	0.049	0.049
Pathogenicity	1	1
Secondary Infection Rate	1.25	1.25

Appendix H7

Deterministic Model 2 (Macler & Merkle, 2000)

***E. coli* Calculations Module 2**

$$[14{,}500 \times 2.95 \times 0.893] \times 0.476 \times 0.5 \times 1 \times [0.028] \times 1.15 = 293\ infections/year$$

Vulnerable population (<5 &>65 years): 0.2025 x 293 = 59 illnesses in vulnerable population

$$59 \times 0.005 = 0.29\ deaths/annum$$

"Non-vulnerable" population: 0.7975 x 293 = 233 illnesses in non-vulnerable population

$$233 \times 0.0016 = 0.37\ deaths/annum$$

$$Total\ (Module\ 1 + Module\ 2)$$

Total waterborne *E. coli* illnesses attributable to private domestic wells = 1128/annum

HUS Rate (2.1/100 illnesses; Rangal *et al*, 2005) – 24 HUS cases /annum, leading to 3-5% mortality rate (Siegler, 1997) – Approximately 1 HUS mortality/annum

Total Crude Incidence Rate (CIR) = 263/100,000 (Groundwater Population)

Bored CIR = 216/100,000

HD CIR = 686/100,000

Appendix H8

Deterministic Model 2 (Macler & Merkle, 2000)

E. coli Model Inputs

	Module 1	Module 2
P_{gw}	130,800 × 2.95 × 0.893	14,500 × 2.95 × 0.893
R_{contam}	0.248	0.476
C_{path}	1	1
R_{inf}	0.5	0.5
R_{recov}	1	1
r^*	0.017	0.028
R_{secinf}	1.15	1.15

r 1 = Based on EPEC Beta-Poisson curve (100 cfu/L = 0.06/day; Powell *et al*, 2000), median contamination magnitude of 3 cfu/100ml, 926 ml/day consumption = 0.017/day

r 2 = Based on EPEC Beta-Poisson curve (100 cfu/L = 0.06/day; Powell *et al*, 2000), median contamination magnitude of 5 cfu/100ml, 926 ml/day consumption = 0.028/day

Appendix H9

Deterministic Model 2 (Macler & Merkle, 2000)

***Cryptosporidium* spp.Calculations Module 2**

$$[14500 \times 2.95 \times 0.893] \times [0.0493] \times 0.508 \times \frac{1}{0.7} \times [0.028] \times 1.25 = 48\ infections/year$$

Vulnerable population (<5 &>65 years): 0.2025 x 48 = 10 illnesses in vulnerable population

$$10 \times 0.001 = 0.01\ deaths/annum$$

Appendix H10

Deterministic Model 2 (Macler & Merkle, 2000)

***Cryptosporidium* spp. Model Inputs**

	Module 1	Module 2
P_{gw}	130,800 × 2.95 × 0.893	14,500 × 2.95 × 0.893
R_{contam}	0.0493	0.0493
C_{path}	1	1
R_{inf}	0.508	0.508
R_{recov}	0.7	0.7
r	0.028	0.028
R_{secinf}	1.25	1.25

D/R 1 and D/R 2 = Using a low-dose approximation i.e. at low doses, *Cryptosporidium* dose-response is approximately linear due to exponential dose-response curve, dose response at 0.926 ml/day and 0.922 oocyst/L = 0.028/oocyst (Messner *et al*, 2006)

Appendix H11

Deterministic Model 3 (Howard & Pedley, 2006)

***E. coli* Calculations Modules 2-4**

Step 1: CD

$$CD_{Handdug,untreated} = 50\frac{cfu}{L} \times (1-0) = 50cfu/L$$

$$CD_{Bored,untreated} = 30\frac{cfu}{L} \times (1-10^{-1}) = 3cfu/L$$

$$CD_{Bored,untreated} = 50\frac{cfu}{L} \times (1-10^{-1}) = 5cfu/L$$

Step 2: E

$$E_{Handdug,untreated} = 50\frac{cfu}{L} \times 0.926 = 46\ cfu/day$$

$$E_{Bored,treated} = 3\frac{cfu}{L} \times 0.926 = 2.7\ cfu/day$$

$$E_{Handdug,treated} = 5\frac{cfu}{L} \times 0.926 = 4.6\ cfu/day$$

Step 3:$P_{inf,d}$

(using linear approximation of $1x10^{-3}$ per organism (O157:H7) from Howard & Pedley, 2006 based on *Shigella*, From Rose & Gerba, 1991; Haas *et al*, 1999).

$$P_{inf,d;Handduguntreated} = 46\frac{cfu}{day} \times 0.08 \times 1x10^{-3} = 0.0037/day$$

$$P_{inf,d;Bored,treated} = 2.7\frac{cfu}{day} \times 0.08 \times 1x10^{-3} = 2.1 \times 10^{-4}/day$$

$$P_{inf,d;Handdugtreated} = 4.6\frac{cfu}{day} \times 0.08 \times 1x10^{-3} = 3.7 \times 10^{-4}/day$$

Step 4: $P_{inf,y}$

(using 30% and 41% seasonal contamination rates and 5-day assumption as used in deterministic model 1)

$$P_{inf,y;Handduguntreated} = 0.0037 \times \left[\frac{4.9}{12} \times 5\right] = 7.55\ x\ 10^{-3}/year$$

$$P_{inf,y;Bored,treated} = 2.1 \times 10^{-4} \times \left[\frac{3.6}{12} \times 5\right] = 3.15 \times 10^{-4}/year$$

$$P_{inf,d;Handdugtreated} = 3.7 \times 10^{-4} \times \left[\frac{4.9}{12} \times 5\right] = 7.55 \times 10^{-4}/year$$

Step 5:P_{ill}

(Where $P_{ill/inf} = 0.25$ ($From$ Haas *et al*, 1999; Morbidity data from *Shigella*)

$$P_{ill,Handduguntreated} = 7.55\ x\ 10^{-3} \times 0.25 = 1.88\ x\ 10^{-3}/year$$

$$P_{ill,Boredtreated} = 3.15 \times 10^{-4} \times 0.25 = 7.87 \times 10^{-5}/year$$

$$P_{ill,Handdugtreated} = 7.55 \times 10^{-4} \times 0.25 = 1.88\ x\ 10^{-4}/year$$

Risk Characterisation

Module 2:

$$[14{,}500 \times 2.95 \times 0.951 \times 0.893] \times 0.476 \times 1.88\ x\ 10^{-3} \times 1.15 = 37.4\ infections/year$$

(CIR; 91.9/100,000)

Vulnerable population (<5 &>65 years): 20.25% - 8 illnesses in vulnerable population

$$88 \times 0.005 = 0.04\ deaths/annum$$

"Non-vulnerable" population: 79.75% - 30 illnesses in non-vulnerable population

$$348 \times 0.0016 = 0.05\ deaths/annum$$

HUS Rate (2.1/100 illnesses; Rangal *et al*, 2005) – 0.8 HUS cases /annum, leading to 3-5% mortality rate (Siegler, 1997) – 0.04/annum

Module 3:

$$[130{,}800 \times 2.95 \times 0.173 \times 0.893] \times 0.248 \times 7.87 \times 10^{-5} \times 1.15 = 1.34\ infections/year$$

(CIR; 2/100,000)

Module 4:

$$[14{,}500 \times 2.95 \times 0.049 \times 0.893] \times 0.476 \times 1.88\ x\ 10^{-4} \times 1.15 = 0.2\ infections/year$$

(CIR; 9.5/100,000)

Appendix H12

Deterministic Model 3 (Howard & Pedley, 2006)

E. coli Model Inputs

	Module 1	Module 2	Module 3	Module 4
CD	30	50	3	5
Cr	30	50	30	50
PT	0	0	10^{-1}	10^{-1}
E	27.8	46	2.7	4.6
V	0.926	0.926	0.926	0.926
$P_{inf,d}$	0.0022	0.0037	2.1×10^{-4}	3.7×10^{-4}
r	$1x10^{-3}$	$1x10^{-3}$	$1x10^{-3}$	$1x10^{-3}$
$P_{inf,y}$	0.0396	0.09	3.7×10^{-3}	9×10^{-3}
P_{ill}	0.01	0.022	9.45×10^{-4}	2.25×10^{-3}
P_{gw}	284,961	36,326	56,611	1,871
R_{contam}	0.248	0.476	0.248	0.476
R_{secinf}	1.15	1.15	1.15	1.15

Appendix H13

Deterministic Model 3 (Howard & Pedley, 2006)

***Cryptosporidium* spp.Calculations Modules 2-4**

Step 1: CD

$$CD_{Handdug,untreated} = 0.922\frac{oocyst}{L} \times (1 - 0) = 0.922\ oocyst/L$$

$$CD_{Bored,untreated} = 0.922\frac{oocyst}{L} \times (1 - 10^{-1}) = 0.09\ oocyst/L$$

$$CD_{Bored,untreated} = 0.922\frac{oocyst}{L} \times (1 - 10^{-1}) = 0.09\ oocyst/L$$

Step 2: E

$$E_{Handdug,untreated} = 0.922\frac{oocyst}{L} \times 0.926 = 0.85\ cfu/day$$

$$E_{Bored,treated} = 0.009\frac{oocyst}{L} \times 0.926 = 0.08\ oocyst/day$$

$$E_{Handdug,treated} = 0.009\frac{oocyst}{L} \times 0.926 = 0.08\ cfu/day$$

Step 3:$P_{inf,d}$

(using dose-response point estimate of $4x10^{-3}$ per organism from WHO (2004))

$$P_{inf,d;Handduguntreated} = 0.85\frac{oocyst}{day} \times [4 \times 10^{-3}] = 0.0034$$

$$P_{inf,d;Bored,treated} = 0.008\frac{oocyst}{day} \times [4 \times 10^{-3}] = 3.2 \times 10^{-4}$$

$$P_{inf,d;Handdugtreated} = 0.008\frac{oocyst}{day} \times [4 \times 10^{-3}] = 3.2 \times 10^{-4}$$

Step 4: $P_{inf,y}$

(Assuming a 30-day period of contamination (Taylor *et al*, 2004), once per year)

$$P_{inf,y;Handduguntreated} = 0.0034 \times 30 = 0.102$$

$$P_{inf,y;Bored,treated} = 3.2 \times 10^{-4} \times 30 = 9.6 \times 10^{-3}$$

$$P_{inf,d;Handdugtreated} = 3.2 \times 10^{-4} \times 30 = 9.6 \times 10^{-3}$$

Step 5:P_{ill}

Where $P_{ill/inf} = 0.30$ $(FromHavelaar\&Melse, 2003)$

$$P_{ill,Handduguntreated} = 0.102 \times 0.3 = 0.03$$

$$P_{ill,Boredtreated} = 9.6 \times 10^{-3} \times 0.3 = 2.88 \times 10^{-3}$$

$$P_{ill,Handdugtreated} = 9.6 \times 10^{-3} \times 0.3 = 2.88 \times 10^{-3}$$

Risk Characterisation

Module 2:

$$[14{,}500 \times 2.95 \times 0.951 \times 0.893] \times 0.049 \times 0.03 \times 1.25 = 66\ infections/year$$

(CIR; 162.2)

Vulnerable population (<5 &>65 years): 20.25% - 13 illnesses in vulnerable population

$$106 \times 0.001 = 0.01\ mortalities/annum$$

Module 3:

$$[130{,}800 \times 2.95 \times 0.173 \times 0.893] \times 0.049 \times 2.88 \times 10^{-3} \times 1.25 = 10\ infections/year$$

(CIR; 2.6/100,000)

Module 4:

$$[14{,}500 \times 2.95 \times 0.049 \times 0.893] \times 0.049 \times 2.88 \times 10^{-3} \times 1.25 = 0 - 1\ (0.3)\ infection/year$$

(CIR; 0.7/100,000)

Appendix H14

Deterministic Model 3 (Howard & Pedley, 2006)

Cryptosporidium spp. Model Inputs

	Module 1	Module 2	Module 3	Module 4
CD	0.922	0.922	0.09	0.09
Cr	0.922	0.922	0.922	0.922
PT	0	0	10^{-1}	10^{-1}
E	0.85	0.85	0.08	0.08
V	0.926	0.926	0.926	0.926
$P_{inf,d}$	0.0034	0.0034	3.2×10^{-4}	3.2×10^{-4}
r	$4x10^{-3}$	$4x10^{-3}$	$4x10^{-3}$	$4x10^{-3}$
$P_{inf,y}$	0.102	0.102	9.6×10^{-3}	9.6×10^{-3}
P_{ill}	0.03	0.03	2.88×10^{-3}	2.88×10^{-3}
P_{gw}	284,961	36,326	56,611	1,871
R_{contam}	0.049	0.049	0.049	0.049
R_{secinf}	1.25	1.25	1.25	1.25