Report No. 251

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The Impact of On-site Domestic Wastewater Effluent on Rivers and Wells

Authors: Laurence Gill, Vincent O'Flaherty, Bruce Misstear, Laura Brophy, Christopher Fennell, Donata Dubber, David O'Connell, Kathryn Kilroy, Maria Barrett, Paul Johnston, Francesco Pilla and Phillip Geary

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Rialtas na hÉireann
Government of Ireland

ENVIRONMENTAL PROTECTION AGENCY

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- Office of Environmental Enforcement
- Office of Evidence and Assessment
- Office of Radiation Protection and Environmental Monitoring
- Office of Communications and Corporate Services

The EPA is assisted by an Advisory Committee of twelve members who meet regularly to discuss issues of concern and provide advice to the Board.

EPA RESEARCH PROGRAMME 2014–2020

The Impact of On-site Domestic Wastewater Effluent on Rivers and Wells

(2012-W-MS-12)

EPA Research Report

Prepared for the Environmental Protection Agency

by

Trinity College Dublin and National University of Ireland Galway

Authors:

Laurence Gill, Vincent O'Flaherty, Bruce Misstear, Laura Brophy, Christopher Fennell, Donata Dubber, David O'Connell, Kathryn Kilroy, Maria Barrett, Paul Johnston, Francesco Pilla and Phillip Geary

ENVIRONMENTAL PROTECTION AGENCY
An Ghníomhaireacht um Chaomhnú Comhshaoil
PO Box 3000, Johnstown Castle, Co. Wexford, Ireland

Telephone: +353 53 916 0600 Fax: +353 53 916 0699
Email: info@epa.ie Website: www.epa.ie

ACKNOWLEDGEMENTS

This report is published as part of the EPA Research Programme 2014-2020. The EPA Research Programme is a Government of Ireland initiative funded by the Department of Communications, Climate Action and Environment. It is administered by the Environmental Protection Agency, which has the statutory function of co-ordinating and promoting environmental research.

The authors wish to acknowledge the assistance of Martin Danaher (Teagasc) and Patrice Behan (Dublin Institute of Technology) and all the members of the EPA steering committee: Brian Donlon (EPA), Aisling O'Connor (EPA), Margaret Keegan (EPA), Matthew Craig (EPA), Conor Quinlan (EPA), Lisa Sheils (EPA), Niall Cussen (DECLG, now DHPLG), Adrian Butler (Imperial College London), Julie Heslin (HSE) and Martin Cormican (NUIG).

The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

EPA RESEARCH PROGRAMME 2014–2020
Published by the Environmental Protection Agency, Ireland

ISBN: 978-1-84095-780-8 May 2018

Price: Free Online version

Project Partners

Laurence Gill, Principal Investigator
Department of Civil, Structural and
Environmental Engineering
Trinity College Dublin
Ireland
Tel: +353 1 896 1047
Email: laurence.gill@tcd.ie

Donata Dubber
Department of Civil, Structural and
Environmental Engineering
Trinity College Dublin
Ireland
Tel: +353 1 896 3199
Email: dubberd@tcd.ie

Laura Brophy
Department of Civil, Structural and
Environmental Engineering
Trinity College Dublin
Ireland
Email: labrophy@tcd.ie

Bruce Misstear
Department of Civil, Structural and
Environmental Engineering
Trinity College Dublin
Ireland
Tel: +353 1 896 2800
Email: bmisster@tcd.ie

Christopher Fennell
Department of Civil, Structural and
Environmental Engineering
Trinity College Dublin
Ireland
Email: fennelca@tcd.ie

Paul Johnston
Department of Civil, Structural and
Environmental Engineering
Trinity College Dublin
Ireland
Email: pjhnston@tcd.ie

Francesco Pilla
Department of Civil, Structural and
Environmental Engineering
Trinity College Dublin
Ireland
Email: fpilla@tcd.ie

Vincent O'Flaherty, Co-Investigator
Microbiology Department
School of Natural Sciences
National University of Ireland Galway
Ireland
Tel: +353 9 149 3734
Email: vincent.oflaherty@nuigalway.ie

Kathryn Kilroy
Microbiology Department
School of Natural Sciences
National University of Ireland Galway
Ireland
Email: k.kilroy1@nuigalway.ie

Maria Barrett
Microbiology Department
School of Natural Sciences
National University of Ireland Galway
Ireland
Email: maria.barrett@nuigalway.ie

David O'Connell
Department of Civil, Structural and
Environmental Engineering
Trinity College Dublin
Ireland
Email: oconnedw@tcd.ie

Phillip Geary
School of Environmental and Life Sciences
University of Newcastle
Callaghan
Australia
Email: phil.geary@newcastle.edu.au

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Executive Summary

This research project evaluated the impact of domestic wastewater treatment system (DWWTS) effluent as a health hazard to consumers of groundwater from private wells and also a source of nutrient water pollution to surface water in small catchments. This has been achieved by (1) surveying and sampling over 200 wells in four areas, comprising two areas of Extreme groundwater vulnerability (Counties Wexford and Kilkenny) and two areas of Low vulnerability (Counties Wicklow and Cavan), and (2) sampling river water quality upstream and downstream of clusters of DWWTSs on small streams in four catchments underlain by low-permeability subsoils (in Counties Wexford, Wicklow, Longford and Cavan). In addition, research has focused on the use of specific chemical, biochemical and microbiological compounds as tracers of DWWTS effluent contamination in both private wells and rivers, leading to the development of appropriate strategies for fingerprinting sampling of both surface water and wells.

The research on private wells found that 15% of wells were contaminated by *Escherichia coli* in a one-off sampling round. Site assessments also revealed that just 35% of wells assessed were adequately finished and protected against contamination. Through the development of a multiple logistic regression statistical model, the site assessment and water quality data were analysed to identify the specific factors associated with private well contamination. While geological setting and siting with respect to a DWWTS were not found to be significantly associated with the occurrence of contamination, poor wellhead condition and proximity to a potential point source of agricultural contamination were statistically significantly associated with the presence of *E. coli* in groundwater.

A representative subset of 24 wells was monitored over a 14-month period to assess temporal variation in well water quality. This monitoring period identified significant temporal variation in quality, with 58% of the wells testing positive for *E. coli* on at least one occasion. These wells were then used to evaluate a range of chemical and microbiological fingerprinting techniques. Fingerprinting methods evaluated include ionic ratios [potassium/sodium (K/Na) and chloride/bromide (Cl/Br)], fluorescent whitening compounds (FWCs), faecal sterol profiles, artificial sweeteners, caffeine, pharmaceuticals (carbamazepine and sulfamethoxazole) and human-specific Bacteroidales 16S ribosomal RNA (rRNA) genetic markers. The performance of each fingerprinting technique as well as the resources (and hence costs) required for their application varied considerably. Three methods/compounds (K/Na, Cl/Br and acesulfame) showed promise in identifying and tracing sources of well contamination.

The research on the four small river catchments, all with low-permeability subsoils and a high concentration of septic tanks (from 16 to 33 per km^2), revealed rapid responses to rainfall events, indicative of shallow hydraulic pathways. Water quality and flow monitoring over the course of a year revealed varying contributions of phosphorus attributable to DWWTS in the catchments as follows: Wicklow 13.8%, Wexford 8.5%, Cavan 1.3% and Longford 11.3%. Fingerprinting techniques were also trialled in the surface water studies leading to a recommended tiered approach for short-term, low-flow monitoring, starting with the more common analytical techniques of measuring *E. coli* and ammonia, but then moving to more confirmatory tests for FWCs, followed by faecal sterols and then artificial sweeteners, caffeine and pharmaceuticals if needed.

The results obtained from these trials have been used to feed into the Source Load Apportionment Modelling carried out by the Catchments Unit within the Environmental Protection Agency (EPA) to assess well construction and siting with respect to contamination from DWWTSs and to propose appropriate strategies for fingerprinting sampling of both surface water and private wells.

1 Introduction

The domestic wastewater of approximately 30% of the population in Ireland (~490,000 dwellings) is treated on site by domestic wastewater treatment systems (DWWTSs), of which 90% are septic tanks (CSO, 2017). The potential impact of such on-site effluent is pollution of groundwater and/or surface water. If the effluent loading on the subsoil is too high, the permeability of the subsoil is excessive or there is an insufficient depth of subsoil, then the groundwater beneath a percolation area is at risk of pollution, in particular from microbiological pathogens and/or nutrients. Alternatively, if there is insufficient permeability in the subsoil to take the effluent load, surface ponding may occur, with associated health risks, and there will be a risk of effluent discharge/runoff of pollutants (particularly nutrients) to surface water and wells, especially where the latter have unprotected wellheads.

Approximately 172,000 households in Ireland obtain their water supply from private sources, such as a well (CSO, 2017). Private wells in Ireland are largely unmonitored, and they may be poorly sited and constructed and are thus susceptible to contamination. Previous sampling studies of private wells in Ireland found that approximately 30% tested positive for microbial contaminants (Hynds *et al*., 2012), with the main sources believed to be DWWTSs, farmyard run-off, grazing animals and the land-spreading of manure or slurry (EPA, 2010; Hynds *et al.*, 2012).

All on-site systems should discharge their effluent into the ground via designed percolation/infiltration areas (see EPA, 2009) or, for older systems, via soakaways (Keegan *et al*., 2014). In low-permeability areas, much of this effluent is likely to make its way quickly to the nearest surface water course as a result of ponding via near-surface flow or, in many cases, being directly channelled there by a pipe or drainage channel. DWWTSs are therefore thought to have an impact on surface waters in some geological settings, particularly with respect to phosphorus (P). Indeed, some previous research in three small sub-catchments with low-permeability subsoils (3–5 km^2) in the River Blackwater in Northern Ireland has attributed significant P impacts on river water quality to septic tanks, particularly during low-flow conditions (Arnscheidt *et al*., 2007).

The EU Water Framework Directive (2000/60/EC) requires an integrated view of the water cycle and its components: groundwater and surface water should be viewed not in isolation, but as components in the overall water balance of a river basin. Such an integrated approach now requires water management to be considered on a River Basin District (RBD) catchment scale. Risk assessments for different generic pollutant sources are now being carried out by the Environmental Protection Agency (EPA). The EPA has recently developed a modelling tool linked to available geographic information system (GIS) data for catchment nutrient characterisation called the Source Load Apportionment Model (SLAM), which is used to help with these risk assessments (Mockler and Bruen, 2018). Given that DWWTSs have long been suspected of contributing to the problem of contamination of both groundwater and surface water bodies, the EPA wants to be able to assess the risk of nutrients reaching rivers via both groundwater and surface water pathways in different geological and geomorphological settings.

There is a need to investigate rapid and cost-effective ways of determining the presence or absence of human-specific faecal contamination in ground and surface waters. As reported in a number of studies, it is important to correlate different chemical tracers (e.g. Cl/Br ratios, caffeine) with microbial markers for definitive conclusions to be made regarding the tracking and health impacts of on-site effluent.

The aims of this project were to evaluate the impact of DWWTS effluent as a health hazard to consumers of groundwater from private wells and as a source of nutrient water pollution to surface water in small catchments. These aims were achieved by (1) surveying and sampling over 200 wells in four areas, comprising two areas of Extreme groundwater vulnerability (Counties Wexford and Kilkenny) and two areas of Low vulnerability (Counties Wicklow and Cavan), and (2) sampling river water quality upstream and downstream of clusters of DWWTSs on small streams in catchments covered in low-permeability subsoils (in Counties Wexford, Wicklow, Longford and Cavan). The results obtained from these trials have been used to feed into the SLAM modelling carried out by the Catchments Unit of the EPA, and to assess well construction and siting with respect to contamination from on-site systems. In addition, research has focused on the use of specific chemical, biochemical and microbiological compounds as tracers of DWWTS effluent contamination in both private wells and rivers in order to propose appropriate strategies for fingerprinting sampling of both surface water and wells.

2 Private Wells

2.1 Site Selection

Four geologically distinct study areas were chosen: two were characteristic of Extreme groundwater vulnerability areas (DELG *et al.*, 1999), as groundwater underlying these areas is regarded as being vulnerable to contamination (SA1 – Clogh, County Wexford, and SA3 – Gowran, County Kilkenny); the remaining two, in contrast, were areas of Low groundwater vulnerability where there is potentially inadequate percolation of effluent (SA2 – Prospect, County Wexford, and SA4 – Kilnacrott, County Cavan). These were selected in parallel with the study investigating the impact of DWWTSs on surface water, as described in Chapter 3 (see Figure 2.1). All four selected areas (details given in Table 2.1) had a relatively high density of private wells and DWWTSs. A total of 212 wells – approximately 50 private wells per study area – were chosen for a site assessment survey.

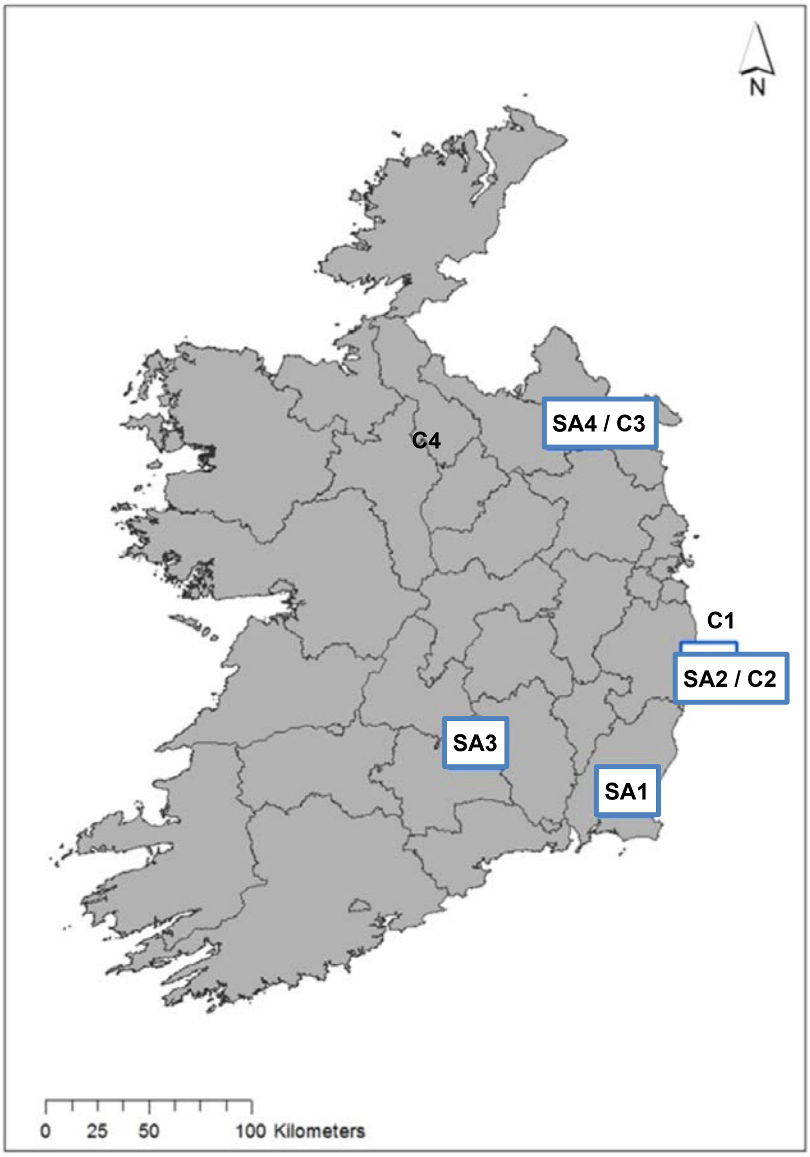

Figure 2.1. Location of sampling areas for wells (SA1, SA2, SA3 and SA4) and stream catchments (C1, C2, C3 and C4).

Table 2.1. Details of four study areas

Study area	Bedrock	Groundwater vulnerability	Aquifer category	Soil type	Subsoil	Land use
SA1: Clogh (11.5 km^2) [61 wells]	OM	Extreme X and Extreme E	Locally Important	Acid well-drained mineral soil (non-calcareous)	Shale till (or bedrock)	Pasture land for beef, dairy and sheep, and arable land for tillage crop
SA2: Prospect (14 km^2) [53 wells]	OM and CV	Low vulnerability	Locally Important	Poorly drained mineral topsoil (mainly calcareous)	Limestone till (with a matrix of Irish Sea Basin origin)	Mainly pasture-based agriculture with some tillage, woodland and golf courses
SA3: Gowran (25 km^2) [48 wells]	BF and CF	Extreme X and Extreme E	Regionally important karstified (diffuse)	Well-drained mineral soils, grey brown podzolics and poorly drained mineral soils	Sandstone and shale till (with some limestone till)	Pasture and arable agriculture
SA4: Kilnacrott (19 km^2) [50 wells]	OF, CF, BS, LA, SF	Low vulnerability	Poorly productive	Acid poorly drained and well-drained mineral soils	Sandstone and shale till	Pasture-based agriculture

BF, Ballyadams Formation (crinoidal wackestone/packstone limestone); BS, Ballysteen Formation (dark muddy limestone, shale); CgF, Clogrenan Formation (cherty, muddy, calcarenitic limestone); CtF, Castlerahan Formation (dark quartz greywacke, conglomerate); CV, Campile Formation (rhyolitic or felsic volcanics and grey and brown slates); LA, Lough Avaghon Formation (massive sandstone and microconglomerate); OF, Oghill Formation (massive sandstone and micro-conglomerate); OM, Ordovician Metasediments; SF, Stackallen Formation (micrite and mudstone).

2.2 Site Susceptibility Assessment

2.2.1 Methodology

The susceptibility of a private groundwater well to contamination is a function of more than its hydrogeological setting. The risk of contamination is also likely to be a function of several localised site-specific factors such as the well's location or construction details. A standardised site susceptibility survey was carried out at each site to assess the main factors affecting well water quality:

- **General site details:** including the slope of the site, as well as the site boundaries and the on-site ground conditions.
- **Well details:** the age of the well, the number of people the well supplies, its depth and the construction materials used in its design. In addition, this section of the survey enquired whether there had been any past water quality problems associated with the well and whether there was a treatment system installed that might have affected the analysis.
- **Wellhead details:** an assessment of how adequately the wellhead works have been completed and protected – is the wellhead above ground or below, is it covered/capped, and is there a concrete apron surrounding the wellhead. See Figure 2.2 for different examples of wellhead finishes.
- **Surrounding land use:** the land use identified within a 100 m radius of the wellhead was subdivided into residential property/grounds, roads, the various types of agricultural land, and agricultural infrastructure such as farm buildings.

2.2.2 Summary of results

The key well site survey results for the four areas are shown in Table 2.2. A large majority (92%, $n = 196$) of the wells used in the study were bored constructions, with the remaining 8% ($n = 16$) being hand-dug wells (there were no hand-dug wells surveyed in areas SA1 and SA2).

As would be expected, there is a significant difference in the median ages and reported depths of the bored versus the dug wells. Dug wells were found to be generally older and shallower, although for a considerable number of wells the depth was not known by the householder.

(1)

(2)

(3)

(4)

(5)

Figure 2.2. Wellhead finishes. (1) Capped and covered wellhead liner extending above ground level with concrete surface apron (as recommended). (2) Well liner cut off at ground level, capped, uncovered, with a missing concrete apron. (3) Above-ground concrete pump house construction. (4) Wellhead liner finished in a below-ground-level manhole. The liner is above the manhole chamber floor but is uncapped. (5) Poorly covered wellhead, finished below ground level, uncapped with potential for ingress of surface water/debris.

Table 2.2. A comparison of the site assessment data for the four study areas

	SA1	SA2	SA3	SA4
No. of wells sampled	61	53	48	50
No. of bored wells	61	53	45	37
No. of dug wells	0	0	3	13
Mean depth – bored wells (m)	41.6	42.8	57.3	58.5
Mean depth – dug wells (m)	n/a	n/a	11.7	4.6
Mean age – bored wells (years)	20	20.9	19.3	20.3
Mean age – dug wells (years)	n/a	n/a	86.6	75
Mean no. of consumers	3.5	3.5	3.7	3.4
Wells used by farms (%)	21	10	36	32
Wells with past quality problems (%)	13	10	19	16
Wells with secure boundary present (%)	77	74	67	70
Wellheads finished above ground (%)	91	90	71	59
Wellheads finished level with ground (%)	4	4	19	38
Wellheads finished below ground (%)	5	8	4	2
Well cover present (%)	84	79	86	87
Well cap present (%)	79	62	62	30
Concrete surface apron present (%)	56	62	62	40
Mean no. of DWWTSs within 100 m of well	2.4	3.3	1.3	1.8
Mean DWWTS age (years)	19.4	18.3	22	26
DWWTSs with septic tanks (%)	75	46	81	73
DWWTSs with packaged treatment systems (%)	25	54	19	27
Mean DWWTS setback distance (m)	36	43	44	40
Mean percolation area setback distance (m)	40	49	46	42
DWWTSs sited downgradient of well (%)	68	69	53	62
DWWTSs sited upgradient of well (%)	9	10	9	22
DWWTSs sited level with well (%)	24	21	37	16
Wells within 100 m of road (%)	80	91	81	78
Mean setback distance from road (m)	30.1	40.8	23.8	31.5
Point agricultural contamination source within 100 m of well (%)	30	11	23	33
Mean setback distance from point agricultural contamination source (m)	25	40	21	32

Overall the surveys revealed that just 35% of wells were adequately finished and protected against contamination (where an adequately finished well is regarded as one in which the liner extends above the ground surface, is capped, and is surrounded by a concrete surface apron that has been well grouted and covered by a manhole/pump house). Associations between poor wellhead design characteristics suggest that poorly finished wells in relation to one design feature are more likely to also be poorly finished with regard to other factors. It is important to note that poor wellhead design can greatly increase the risk of contamination (Misstear *et al.*, 2017).

Some more specific interesting statistics in relation to the wells include:

- The wells that are associated with farms were significantly older than those that solely supply houses (Mann–Whitney $U = 2653$; $p < 0.01$).
- Most houses used no form of water treatment (72%; $n = 152$); none of the dug wells were fitted with a water treatment device (despite four of them having had problems with bacterial contamination in the past).
- 46% of bored wells were missing a simple cap, 44% were missing a concrete surface apron and 17% were uncovered, and 17% of borehole

liners were cut level with or below the ground surface. These wells are all at a greater risk of contamination because of inadequate well finishing that could be rectified relatively easily. Furthermore, 46% of wells were finished in below-ground-level manholes/pump houses.

With respect to the DWWTSs at the sites, 69% ($n = 134$) were septic tanks, with 31% ($n = 61$) being packaged secondary treatment systems. The mean setback distance between the wells and the associated domestic wastewater treatment tank and percolation area was 40 m and 44 m respectively. In 64% of the sites, the DWWTS was favourably situated down gradient of the well. However, in 12% the DWWTS was situated up gradient of the well, and in 24% of the sites the DWWTS was level with the well. Finally, a larger proportion of wells in SA1 and SA4 are within 100 m of a potential agricultural point source of pollution than was found in SA2 and SA3 ($\chi^2 = 5.64$; $p < 0.05$ and $\chi^2 = 8.79$; $p = < 0.01$ respectively).

2.3 One-off Water Quality Sampling Results

2.3.1 *Methodology*

During the site susceptibility survey, water samples were taken from a cold-water tap fed directly from the borehole (i.e. prior to any water treatment) for water quality analysis. Parameters measured directly in the field were temperature, electrical conductivity, total dissolved soils and pH, while parameters analysed back in the laboratory were anions (nitrate, chloride and sulphate), cations (sodium, potassium, calcium, magnesium, iron, manganese and ammonium) and microbiological indicator organisms (total coliforms and *Escherichia coli*).

2.3.2 *Summary of results*

The proportion of wells that tested positive/negative for *E. coli*, as shown in Figure 2.3, did not statistically significantly differ between the four study areas ($\chi^2 = 1.08$; $p = 0.78$). Similarly, no significant difference was found for the proportion of wells that tested positive/negative for total coliforms between the four study areas ($\chi^2 = 3.32$; $p = 0.35$).

With respect to the hydrochemistry of the well waters, the four study areas showed relatively distinct hydrochemical facies, as illustrated in the Piper diagram in Figure 2.4, with the influence of the limestone bedrocks found in SA3 and SA4 evident by the dominance of calcium–magnesium–bicarbonate type groundwaters.

Nitrate concentrations varied considerably between the four sites, as shown in Figure 2.5, with higher concentrations found in SA1 and SA3, and a larger proportion of samples from SA2 and SA4 found to have concentrations below the detection limit of the analytical method used (<1 mg/l N). Other findings included:

- The highest nitrate concentrations were observed in the Extreme groundwater vulnerability areas SA1 and SA3, whereas the lowest nitrate concentrations occurred in the areas of Low vulnerability and high likelihood of inadequate percolation (Figure 2.5).

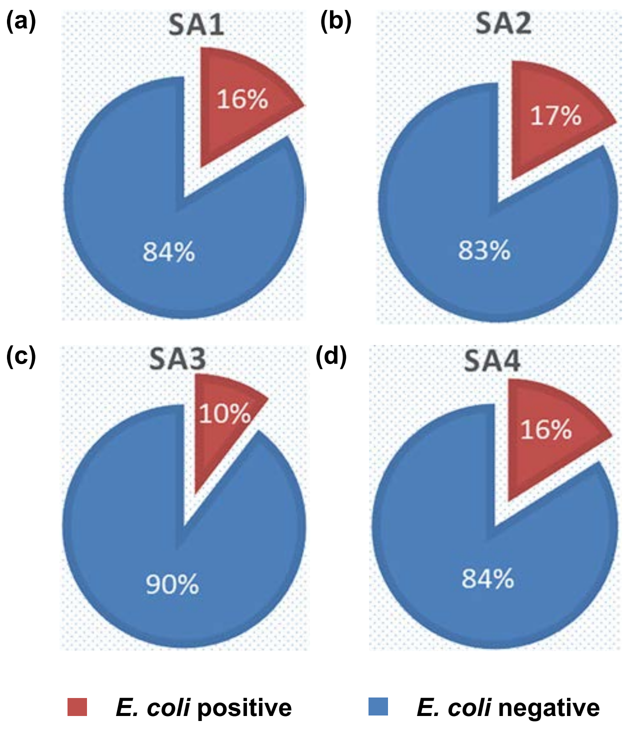

Figure 2.3. The percentages of wells that tested positive/negative for *E. coli*.

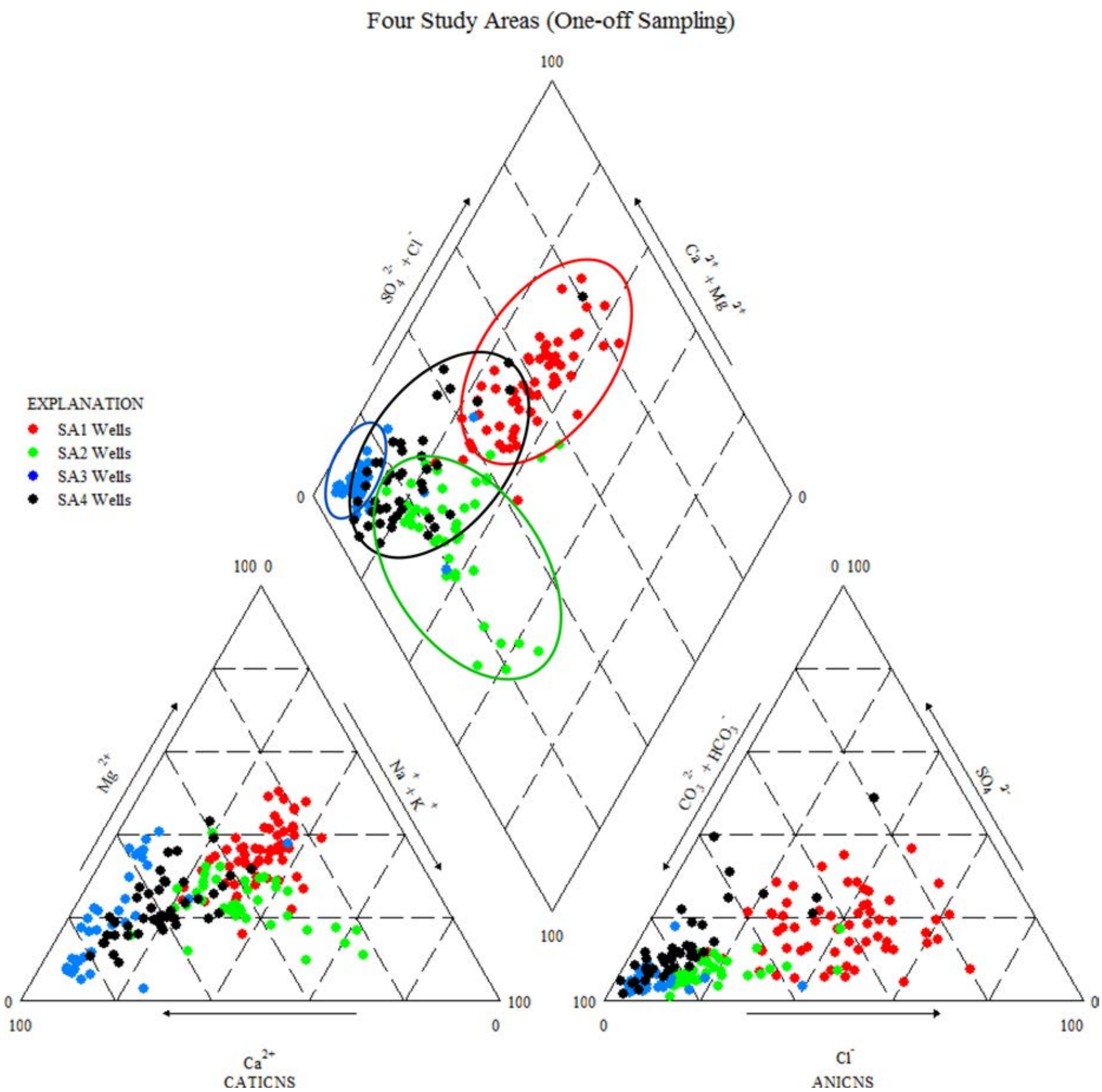

Figure 2.4. Piper diagram of all four study sites highlighting the associated hydrochemical facies.

- Dug wells are more susceptible to nitrate contamination than boreholes.
- Wells on sites served by a secondary treatment system had significantly higher nitrate concentrations than those in sites served by a conventional DWWTS ($t = 3.34$; $p < 0.01$)
- Significant positive relationships were seen between nitrate concentrations and the population equivalent of the associated sites' DWWTSs.
- The mean highest nitrate concentrations were associated with the shallow, well-drained class (Amin SW), followed by the deep, well-drained soils (Amin DW).

2.4 Monitoring Wells

2.4.1 Methodology

Upon completion of the one-off sampling, six wells from each study area were selected for monthly monitoring. Based on the results of the site assessment surveys and one-off sampling, these 24 wells were chosen to represent a range of well, DWWTS and site characteristics, as well as different water quality properties. A conceptual framework of four distinct categories was developed (Figure 2.6), each colour coded to represent a possible scenario that could arise from the well selection criteria. The red category includes wells that performed

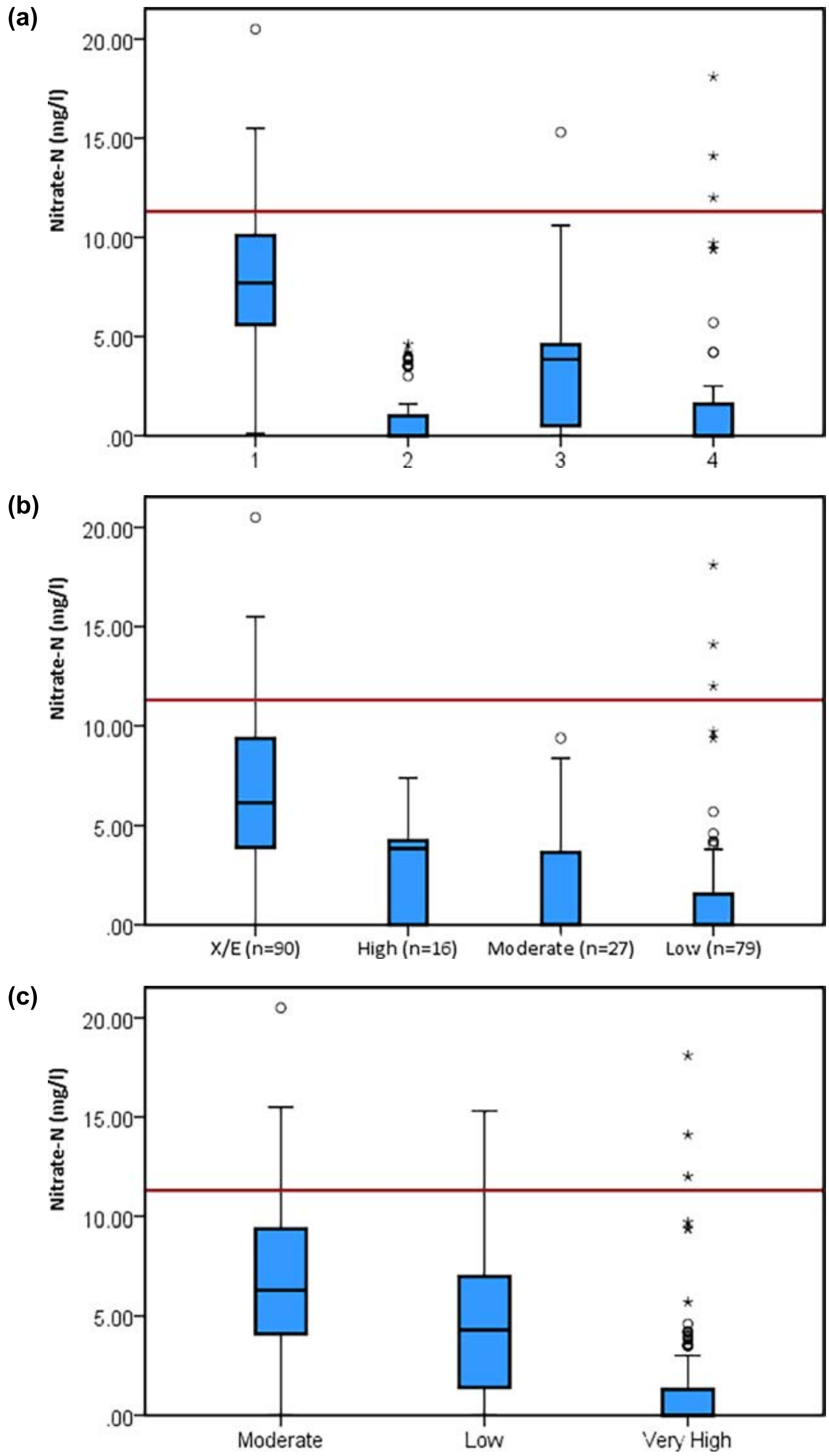

Figure 2.5. Nitrate concentrations by (a) study area, (b) groundwater vulnerability setting and (c) likelihood of inadequate percolation category (EPA, 2013). The legislative limit of 11.3 mg/l of nitrogen present as nitrate (NO_3-N) is shown by the red line (91/676/EC and 98/83/EC).

poorly in all selection criteria. For these wells, site assessment surveys indicated that they are vulnerable to contamination, with the subsequent sampling indicating the presence of contamination. In contrast, the green category represents wells that performed adequately in all criteria. Site assessment surveys would have indicated that the wells were properly sited, constructed and sealed, as well as adequately maintained. Subsequent tests would also have shown good water quality. The two intermediate categories

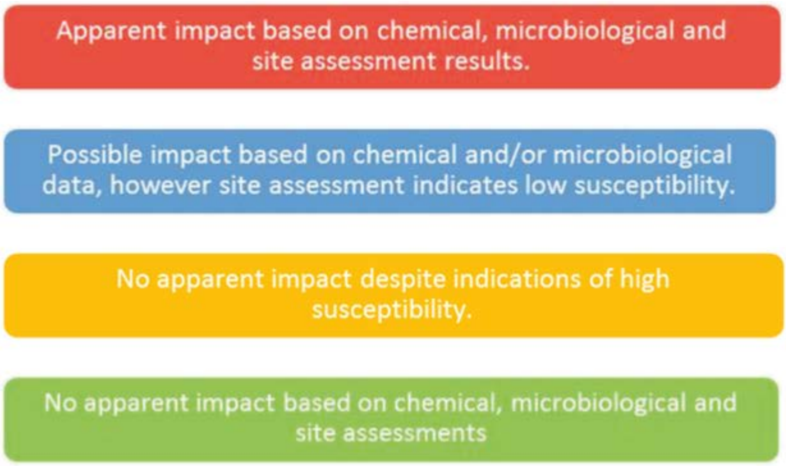

Figure 2.6. Conceptual framework of colour-coded categories for the selection of monitoring wells.

(yellow and blue) represent the one-off microbial and/or chemical water quality results that appear to contradict the findings of the site assessment survey.

Each well was monitored monthly over a 14-month period (M1–M14) using the same field, laboratory and quality control procedures outlined in section 2.3.1.

In addition, the suitability of a range of different additional fingerprinting compounds was examined during two monthly monitoring events (M11 and M13), as discussed later in Chapter 4. This enabled an evaluation of their individual suitability, as well as of the level of agreement between the various methods. Any evidence of contamination indicated by these techniques could then be compared with the site assessment survey, one-off sampling event and 14-month monitoring record for interpretation.

2.4.2 Summary of results

Significant temporal variation in water quality, especially microbial water quality, was observed in the 24 monitoring wells over the 14-month monitoring period.

Physicochemical parameters

The results from physicochemical water quality parameter analyses broadly yielded what would have been expected from the different aquifer geologies, with temperatures varying between 8°C (winter) and 14°C (summer) and alkalinity highest in the samples from SA3 (limestone area).

The seasonal variations in nitrate concentrations provided interesting information relevant to the aims of the study, with the monthly values for SA1 shown in Figure 2.7 as an example. The variability in nitrate concentrations between the six wells (CX52, CX53, CX54, CX45, CX43 and CX23) suggests differences in either nitrate source and/or pathway components. In CX45 (and potentially in CX54), the reduction in nitrate over the winter could be for several reasons or a combination of them. One reason could be dilution as a result of increased recharge. Nitrate in CX45 is strongly and significantly positively correlated with conductivity ($r = 0.969$; $p < 0.01$), calcium ($r = 0.697$; $p < 0.01$), potassium ($r = 0.938$; $p < 0.01$), magnesium ($r = 0.881$; $p < 0.01$) and sodium ($r = 0.916$; $p < 0.01$). All of these parameters are also positively correlated with temperature, indicating a seasonal effect that is consistent with increased recharge and dilution and lower degrees of mineralisation. However, another possible explanation is that the reduction in nitrate concentrations in winter is due to the prohibition of the land spreading of agricultural fertilisers during part of the winter period. The nitrate concentrations in CX45 decrease from November and start to increase from April, while the concentration in CX54 peaks in September before deceasing over the winter months and increasing again in April. However, the isolated in peaks in nitrate concentrations in CX23, CX52 and CX43 over the winter months do not support the same assumption of a steady seasonal increase and decrease due to agricultural inputs, and they may represent specific contamination events.

Microbiological indicators

As illustrated in Figure 2.8, sampling of the 24 wells in October 2015 indicated that 17% were contaminated

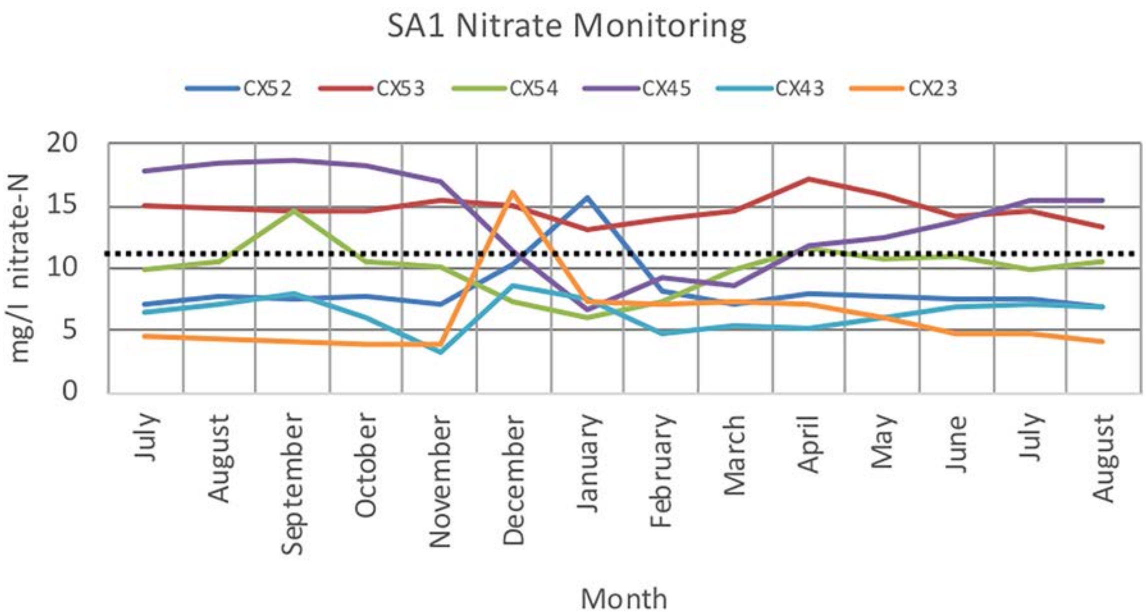

Figure 2.7. SA1 monthly groundwater nitrate concentrations (July 2015 to August 2016). The dotted line shows the legislative limit.

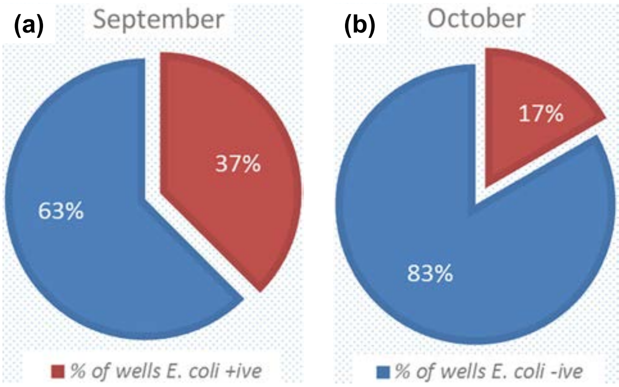

Figure 2.8. Temporal variability in *E. coli* presence/absence encountered during the monitoring of the 24 selected wells (July 2015 to August 2016).

with *E. coli*, whereas the sampling results from the preceding September concluded that a much greater 37% were contaminated. Furthermore, over the 14-month period, 58% of the wells tested positive for *E. coli* at least once. This highlights how the frequency and timing of sampling can greatly influence results. In addition, it highlights the caution with which results from a single sample should be treated.

The monitoring of the selected wells yielded noteworthy results. While some wells have been grossly and consistently contaminated, the contamination of others has been shown to be intermittent and of varying degrees. The considerable variation in *E. coli* occurrence highlights how one-off samples must be treated with caution and how regular well sampling would be necessary to ensure that consumer health is protected. In several cases, this contamination can be related to site-specific aspects recorded by the site surveys. This highlights the potential value of such assessments as tools for identifying susceptible wells and guiding remedial works and also as potential precursors to contamination fingerprinting techniques. Nonetheless, while surveys can be useful, inferences regarding the sources of contamination based on these surveys alone are by no means absolute, and so this section has illustrated several scenarios in which the development of effective contamination fingerprinting techniques would be valuable.

2.5 Microbial Contamination and Source Susceptibility

2.5.1 Bivariate analysis

All variables assessed as part of the site survey were analysed for potential associations with the occurrence of *E. coli*. Taking the full dataset, with respect to bored wells, six variables were statistically significantly associated with *E. coli* at the 0.05 significance level. Many of these variables are related to the wellhead finish. Specific to the wellhead finish, the presence or absence (P/A) of a well cap, cover and concrete surface apron were all significantly associated with *E. coli* P/A. Wells not fitted with a cap were four times more likely to be contaminated with *E. coli* than those adequately sealed with a cap [odds ratio (OR) 3.99; 95% confidence interval (CI) 1.53–0.38]. Wells without a concrete surface apron were 3.4 times more likely to be contaminated than those with an apron (OR 3.4; 95% CI 1.37–8.5). The derived variable (adequately finished or not) was also significantly associated with *E. coli* P/A (OR 3.01; 95% CI 1.24–6.8) (Table 2.3).

Notably, no significant associations were apparent between *E. coli* P/A and any of the geological or DWWTS variables (as listed in Table 2.2).

2.5.2 Multivariable logistic regression

Initially, to illustrate the application of logistic regression (LR) to the data, each site-specific variable was assessed individually for association with *E. coli* P/A using LR. While this is essentially another form of bivariate analysis (section 2.5.1), it is another way of analysing the influence of each variable on well contamination and is a useful precursor to multiple logistic regression (MLR) and the "purposeful selection" method. MLR analysis was then performed. Collinearity was first assessed between the independent variables in the current dataset. Significant collinearity is apparent between well types (dug vs bored) and well design properties. Consequently, the MLR model was developed using only the bored wells. Significant collinearity is also apparent between the P/A of a well cover and a concrete surface apron (tolerance statistic <0.7). This is likely to be due to the construction of one with the other (e.g. a pump house is often constructed with a concrete base). Owing to this collinearity, the P/A of a concrete apron variable was removed from the dataset. Again, this does not mean that the concrete surface apron does not influence contamination.

A hybrid method of parameter selection and model development (a combination of the "purposeful selection" and stepwise procedures) was used. In this hybrid method, variables found to be significantly associated with *E. coli* P/A were entered into the model. A stepwise entry (Forward: Conditional) procedure was used for the remaining predictor variables. Firstly, with respect to the purposeful selection procedure, the three variables significantly associated with *E. coli* in the bivariate analysis (P/A of a point source of agricultural contamination, well cap and well cover) were entered into an MLR model. While all three variables were significant in individual bivariate LR models, only two (P/A of a point source of agricultural contamination and well cap) were significant in the MLR model ($p<0.05$). Well cover P/A was no longer significant ($p=0.25$).

The final MLR model (containing well cap P/A and point agricultural contaminant source P/A) is shown in Table 2.4. The resulting regression equation and coefficients show that wells not fitted with a well cap are 3.5 times more likely to be contaminated with *E.*

Table 2.3. Cross-tabulation of the derived variable (whether or not a wellhead is adequately finished) and the associated *E. coli* P/A test results

E. coli P/A and wellhead adequately finished P/A (derived)				
		Inadequately finished	Adequately finished	Total
E. coli P/A	Present	23	4	27
	Absent	111	58	169
Total		134	62	196

Table 2.4. Final MLR model

	β^a	S.E.	β Sig.	Exp. (β)
Well cap P/A	–1.247	0.508	0.014	0.287
Point agricultural contaminant source P/A	1.812	0.451	0.000	6.120
Constant	–2.041	0.346	0.000	0.130

[a]ln (odds of a given occurrence of the presence of *E. coli*).

Table 2.5. Classification table for the LR model (threshold 0.4)

Observed		Predicted		
		E. coli P/A		Percentage correct
		0.0	1.0	
E. coli P/A	0.0	153	16	90.5
	1.0	14	13	48.1
Overall percentage				84.7

coli. Wells in proximity to a point source of agricultural contamination are 6.1 times more likely to be contaminated with *E. coli.*

To assess the actual probabilities produced by the LR associated with the final MLR model, the cut-off value was adjusted, with predictive performance monitored. Notably, when a cut-off classification of 0.4 is used (i.e. $p > 0.4$ predicts *E. coli* presence while $p < 0.4$ predicts *E. coli* absence), the model becomes much more accurate at predicting *E. coli* presence (Table 2.5), with the capacity to predict *E. coli* presence rising to 48%.

In summary, the MLR analysis suggests that proximity to a point agricultural contamination source and the P/A of a well cap are significant factors in controlling borehole susceptibility to *E. coli* contamination.

3 Streams in Low-permeability Catchments

3.1 Site Selection

A screening process of several small catchments with predominantly low subsoil permeabilities – i.e. high likelihood of inadequate percolation to groundwater (LIPTG) – was carried out across Ireland comparing other GIS-based characteristics, such as subsoil permeability, DWWTS location and density, geology, aquifer type and land use, to identify suitable catchments with a heightened risk of impact on river water quality from DWWTSs. The results of the catchment criteria survey were then converted into a scored hierarchy system based on the criteria that were ranked as most important in the survey and which could be readily mapped using ArcGIS software and available datasets. Site visits were then undertaken to potentially suitable catchments to select the final study catchments, which enabled the location of potential access points to the stream for monitoring, etc. The final four study sites in Counties Wicklow, Wexford, Cavan and Longford are listed in Table 3.1.

3.2 Fieldwork

3.2.1 Instrumentation

In each of the four catchments selected for study, a point on the main stream in the upper catchment upstream of the main cluster of DWWTSs was selected as the upper monitoring point, and a point on the main stream in the lower catchment downstream of the cluster of DWWTSs was selected as the lower monitoring point (see Table 3.1). At every monitoring site, instrumentation was installed in a stilling well to measure river depth, water temperature and electrical conductivity continuously, as shown in Figure 3.1.

The continuous depth recordings were converted into discharge data by developing stage/discharge relationships at each monitoring point using the mid-section area/velocity method, or salt dilution gauging in low-flow conditions. Weather stations measuring rainfall, wind speed and direction, air temperature, humidity, air pressure, net radiation and incident

Table 3.1. Four selected study catchments – characteristics

	Area (km^2)	No. of DWWTSs		Density (no./km^2)	LIPTG	Land use
		Upstream	Downstream			
Catchment 1 (C1) Duran, Co. Wicklow	3.07	0	78	25.4	Very high (1.08 km^2) High (1.99 km^2)	Pasture (46%) Arable (10%) Forestry (33%) Other (11%)
Catchment 2 (C2) Prospect, Co. Wexford	2.95	19	78	32.9	Very high (2.80 km^2) Moderate (0.15 km^2)	Pasture (83%) Arable (0%) Forestry (2%) Other (15%)
Catchment 3 (C3) Kilnacrott, Co. Cavan	3.85	2	58	15.6	Very high (3.85 km^2)	Pasture (85%) Forestry (15%) Other (<0.5%)
Catchment 4 (C4) Aghboy, Co. Longford	2.07	4	36	19.3	Very high (2.07 km^2)	Pasture (89%) Arable (0%) Other (11%)

Figure 3.1. (a) Upstream and (b) downstream monitoring points for C3.

radiation were also installed at each site, in proximity to the upper monitoring points. All instrumentation was connected to a Campbell Scientific data logger (Model CS800).

3.2.2 Sampling methodology

Each catchment was studied for a minimum of 1 year, with water quality monitored over a 1-week period every month. Stream water samples upstream and downstream were collected using autosamplers (see Figure 3.1) at each monitoring point. The autosamplers were programmed to collect samples on a 24/7 sampling regime, as outlined by Jordan and Cassidy (2011), in order to achieve a reliable quantification of nutrient patterns with respect to temporal variations. All samples were analysed for dissolved reactive phosphorus (DRP), total dissolved phosphorus, dissolved unreactive phosphorus, total phosphorus (TP), particulate phosphorus, nitrate (NO_3), nitrite (NO_2), ammonia (NH_3) and total nitrogen (TN). Occasional analyses were made for potential tracers such as fluorescent whitening compounds (FWCs), faecal sterols, Bacteroidales, caffeine, artificial sweeteners and selected pharmaceutical compounds (as outlined in section 4.1). In addition, spot sampling campaigns were targeted for microbiological indicators, total coliforms, *E. coli* and enterococci.

3.3 Results of Stream Monitoring

3.3.1 Hydrology

All catchments exhibited a rapid response to rainfall events, as shown in the sample hydrographs for C1 (Figure 3.2). This flashy stream behaviour was expected because of the predominant cover of low-permeability subsoils in all four catchments, with the dominant hydraulic pathways likely to be via shallow subsurface, interflow and artificial drainage networks.

3.3.2 Nutrient and microbiological quality

The results of the in-stream DRP and TP concentrations at the upstream and downstream monitoring sites over the year-long monitoring period generally revealed a trend towards higher in-stream P concentrations at the downstream site, as shown, e.g. for C1 in Figure 3.3, which would be indicative, perhaps, of some impact on stream water quality by the DWWTS. The results of the nitrogen fraction data (NO_3, NO_2 and NH_3) also generally indicated higher concentrations for the TN at the lower monitoring sites on all catchments, as shown, e.g. for C3 in Figure 3.4. Equally, a microbiological analysis for total coliforms, *E. coli* and enterococci from grab samples at the upstream and downstream monitoring sites over the course of the year provided further indication of the

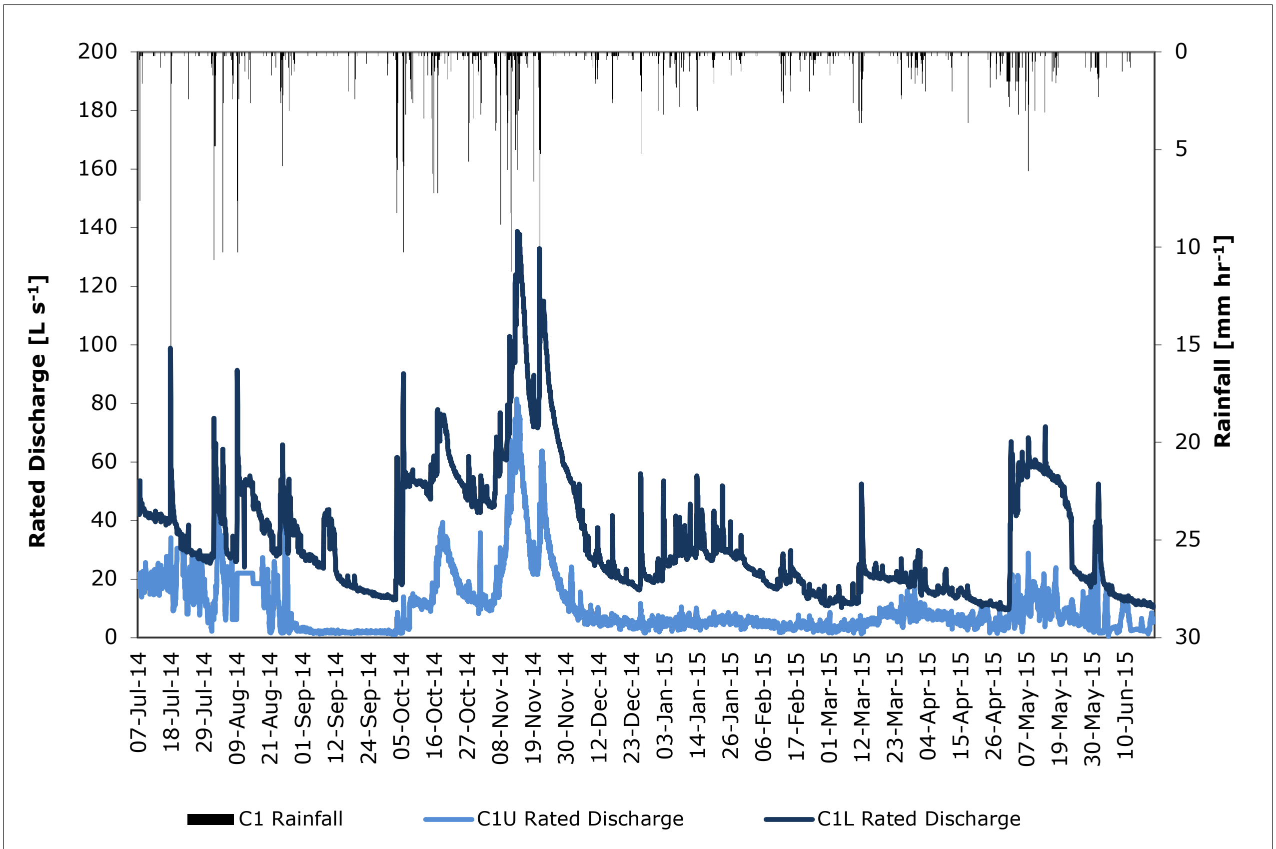

Figure 3.2. Record of the rated stream discharge at the C1U (upper) and C1L (lower) primary monitoring stations, plus hourly cumulative rainfall (mm/hr) from July 2014 to June 2015.

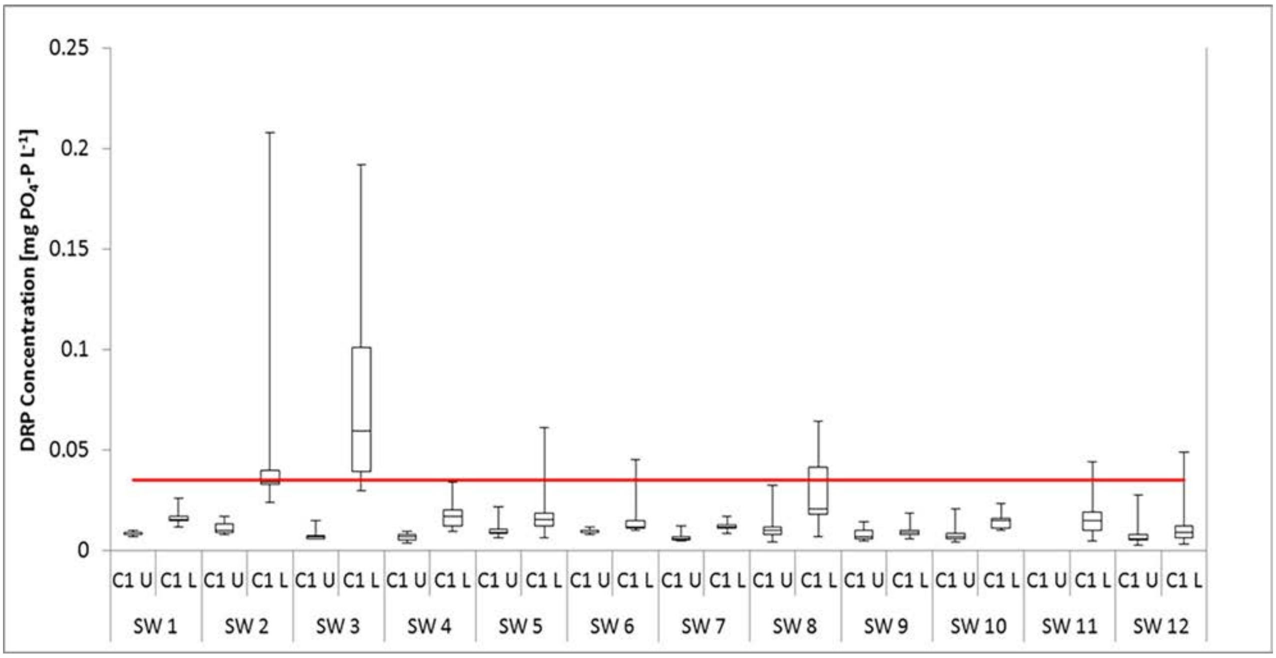

Figure 3.3. Distribution of C1 DRP concentration data at both $C1_U$ and $C1_L$ across the 12 sampling weeks including minimum, maximum, median, first quartile and third quartile data points. The red line represents the 0.035 mg P/l water quality threshold for "good" status.

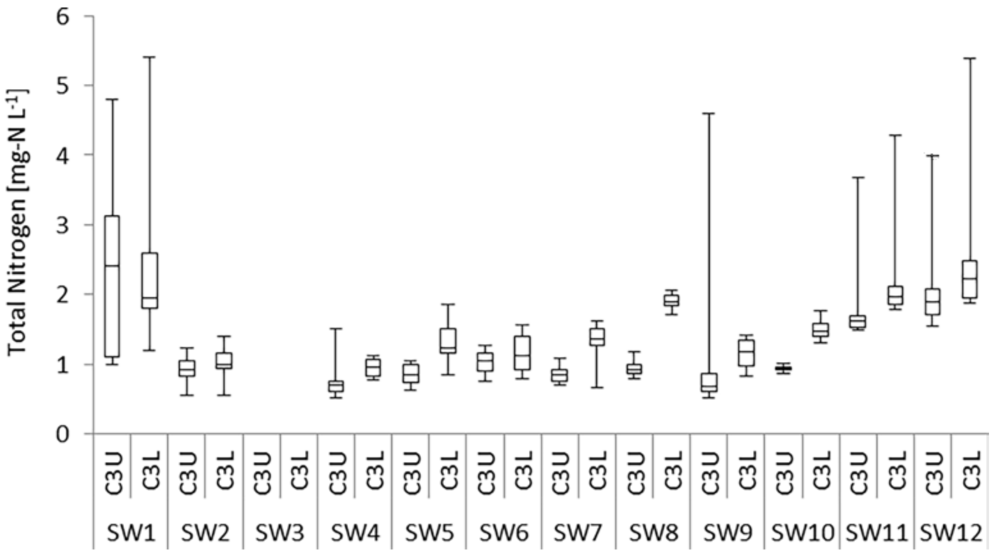

Figure 3.4. Summary of TN concentrations in C3, illustrating median, minimum, maximum, first and third quartile data points.

reduction in water quality downstream of the main cluster of DWWTSs in all catchments.

3.4 Nutrient Contribution from DWWTSs

In order to estimate the net P [and nitrogen (N)] contribution from DWWTSs to the stream water quality in each catchment, the weekly catchment P load was first explored under conditions of low flow, such that it could be assumed that direct P inputs from agricultural sources would be minimal, and that the majority of the observed P loading at the lower monitoring point would therefore be from a combination of natural mobilisation processes, which includes in-stream sediment breakdown, and any groundwater inputs (background P load), plus inputs from the catchment DWWTSs (DWWTS P load). A conceptual model was developed, as shown in Figure 3.5, that could be applied during stable, low-flow monitoring conditions to calculate the nutrient loads attributable to DWWTSs, on the assumption that effluent was being discharged at a constant rate throughout such monitoring periods and was not significantly impacted by changes in meteorological conditions.

To apply the model described in Figure 3.5, sampling weeks were selected during which hydrological conditions were such that the $P_{BACKGROUND}$ term could be applied at the whole catchment (based upon the lowest measured $P_{UPSTREAM}$ values across the sampling weeks). These were assumed to increase linearly all the way down the catchment (linked to the flow to catchment area measurements at the upstream and downstream monitoring points) with no discrete inputs from agriculture along the river, during which DWWTS impacts were likely to be greatest. The DWWTS P load calculated during such periods was then applied to the other SWs during higher flows. Following the estimation of a weekly P loading from the DWWTS (which was assumed to be consistent throughout the year), estimates of the annual P loading from the catchment were then made to examine how much P load potentially arises from the DWWTS. Predictive relationships between $P_{DOWNSTREAM}$ were developed, based on variables for which a continuous record existed throughout the monitoring period (i.e. $Q_{DOWNSTREAM}$ and average 7-day soil moisture deficit) from which the annual TP load from each catchment could be calculated. Assuming more or less constant P loading from DWWTSs into the stream across the year, the net P contribution from DWWTSs to the streams was estimated, as shown in Table 3.2. The same approach was used to determine the net N contribution from DWWTS to the streams across each monitoring year (Table 3.3).

These fieldwork-based predictions have also been compared against the predictions made by the EPA's SLAM, which has been applied to the four study catchments. The comparison between the fieldwork-based predictions and the SLAM-modelled predictions is shown in Figures 3.6 and 3.7 for P and N respectively. These show that SLAM seemed to

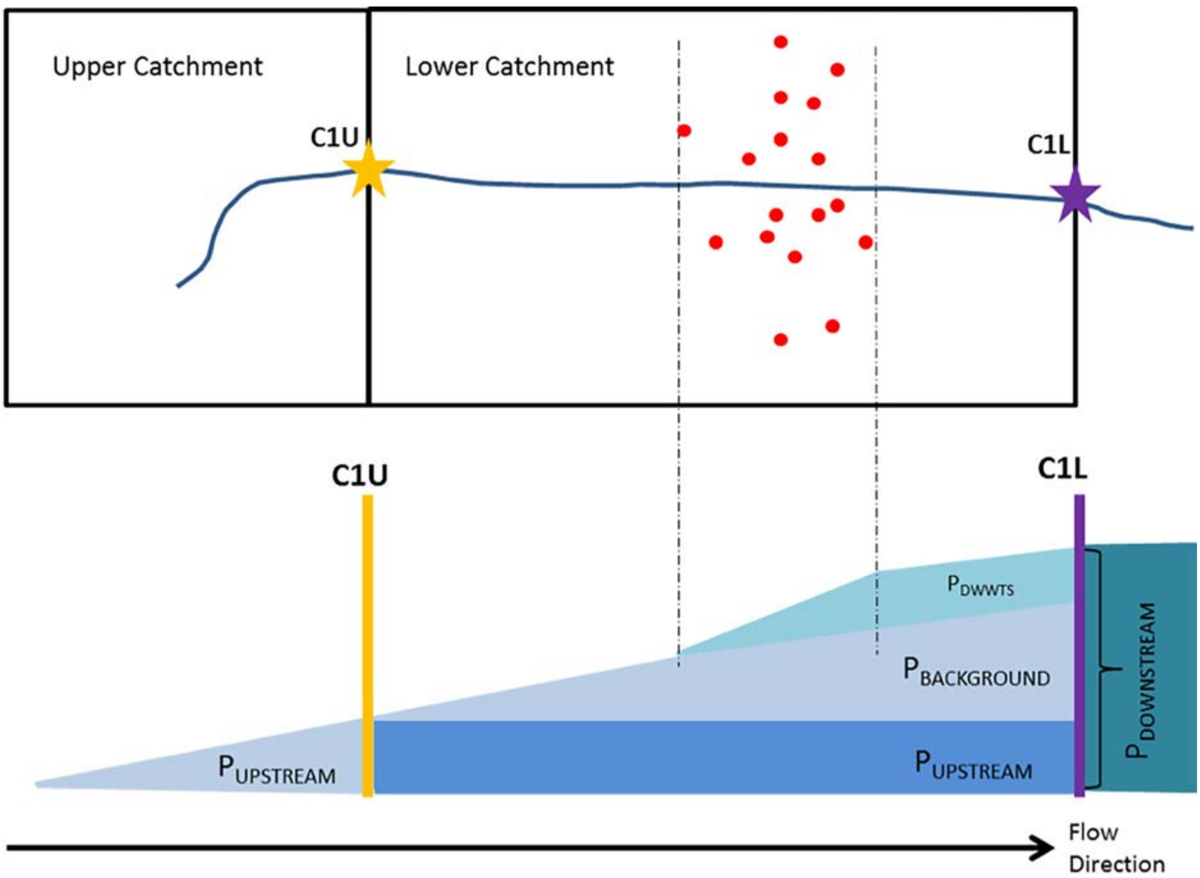

Figure 3.5. Schematic of conceptual model used to estimate DWWTS P loading in C1. Note, this was also applied to DWWTS TN loading.

Table 3.2. Final results of the estimation of the contribution of DWWTS to the total TP loading from the four catchments

Catchment	Study period	Total annual P load *(kg P/year)*	Total annual DWWTS P load *(kg P/year)*	Contribution of DWWTS-P to total *(%)*
C1 (Wicklow)	Jul 2014 to Jun 2015	51.2	7.1±1.2	13.8 ±2.4
C2 (Wexford)	Jul 2014 to Jun 2015	257	21.7±3.4	8.5±1.3
C3 (Cavan)	Aug 2015 to Jul 2016	179	2.4±2.1	1.3±1.2
C4 (Longford)	Oct 2015 to Sep 2016	117	13.2±4.1	11.3±3.6

Table 3.3. Final results of the estimation of the contribution of DWWTS to the total N loading from the four catchments

Catchment	Study period	Total annual N load *(kg N/year)*	Total annual DWWTS N load *(kg N/year)*	Contribution of DWWTS N to total *(%)*
C1 (Wicklow)	Jul 2014 to Jun 2015	3313	945±163	28.5±4.9
C2 (Wexford)	Jul 2014 to Jun 2015	2528	306±41	12.1±1.6
C3 (Cavan)	Aug 2015 to Jul 2016	1071	73±33	6.8±3.1
C4 (Longford)	Oct 2015 to Sep 2016	1520	132±86	8.7±5.8

provide reasonable agreement for annual catchment TP loads for C1 and C4 (given the uncertainties), while it under-predicted the TP loads for C2 and C3. For the annual DWWTS P loads, SLAM shows reasonable agreement for C2 and C3 while markedly over-predicting for C1 but under-predicting for C4. For N, the predictions were much closer between the model and the research data, although the substantially higher N load from DWWTSs for C1 compared with both the SLAM predictions and the other catchments is striking. C1 had the greatest number of DWWTSs located within 100 m of the stream out of the four study catchments; the insensitivity of SLAM below such resolution (see Gill and Mockler, 2016) may explain this difference.

Finally, to gain further insights into the attenuation of nutrients from DWWTSs en route to the river, a function has been developed on the basis of an attenuation factor applied to every DWWTS, dependent on its distance from the stream. The most consistent relationship across all catchments was based upon a power inverse relationship, as follows:

$$\text{Load to stream per tank} = \text{DWWTS load}/\text{dist}^{\delta} \quad (3.1)$$

Table 3.4 shows the percentage reduction in the potential weekly DWWTS P loading (0.026 kg P/week per DWWTS) as a function of distance, such that the P loading contribution to a stream in any poorly drained catchment may be estimated. Figure 3.8 illustrates this relationship based on the findings of this research project as an average value (approximated to the C2 data), an upper bound (the C4 data) and a lower bound (represented as an average of the C1 and C3 data).

Table 3.4. Percentage reduction in the potential weekly DWWTS P loading (0.026 kg P/week per DWWTS) as a function of distance, predicted using an inverse power method

Catchment	Percentage of initial DWWTS P (0.026 kg P/week per DWWTS) reaching the stream			
	10 m	100 m	500 m	1000 m
C1 (Wicklow)	20.4	4.31	1.46	0.91
C2 (Wexford)	42.3	19	10.7	8.3
C3 (Cavan)	25.3	6.38	2.45	1.62
C4 (Longford)	53	28	18	15
Average	35%	14%	8%	6%

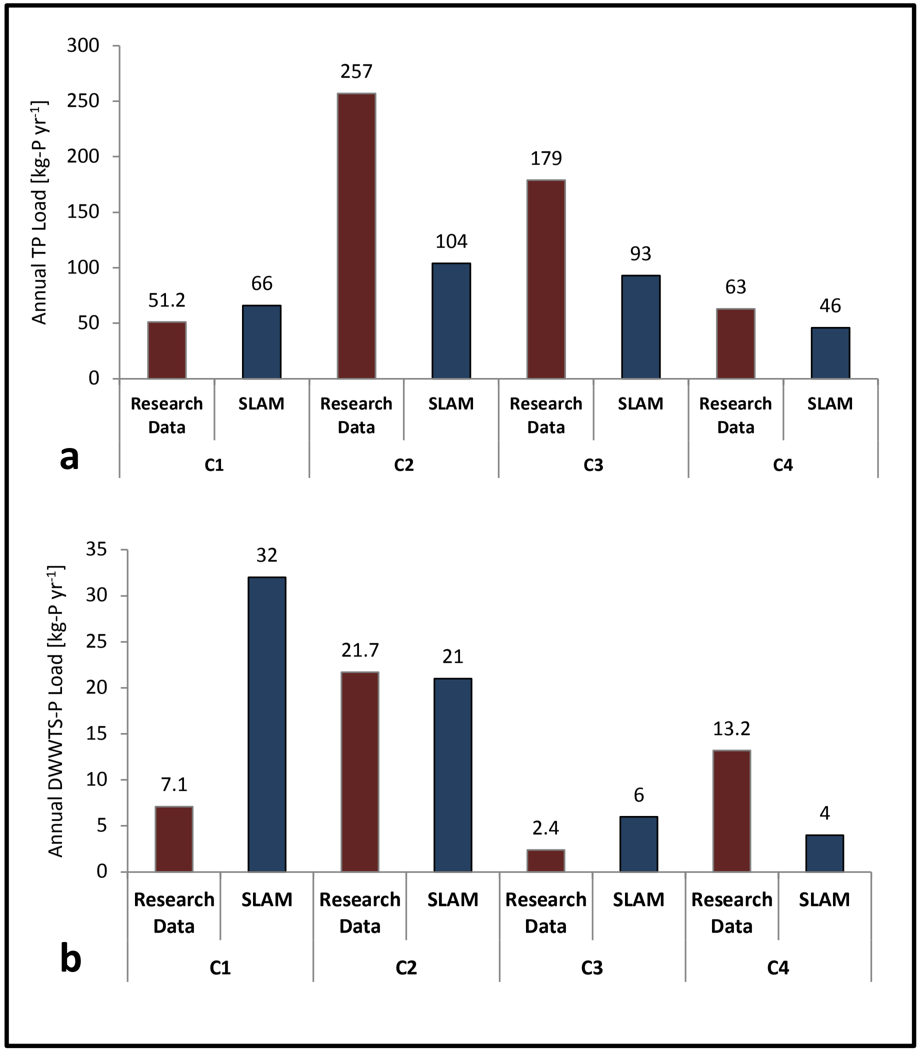

Figure 3.6. Summary chart of (a) annual catchment TP load and (b) DWWTS TP loading in each catchment, as estimated by the research data and by SLAM.

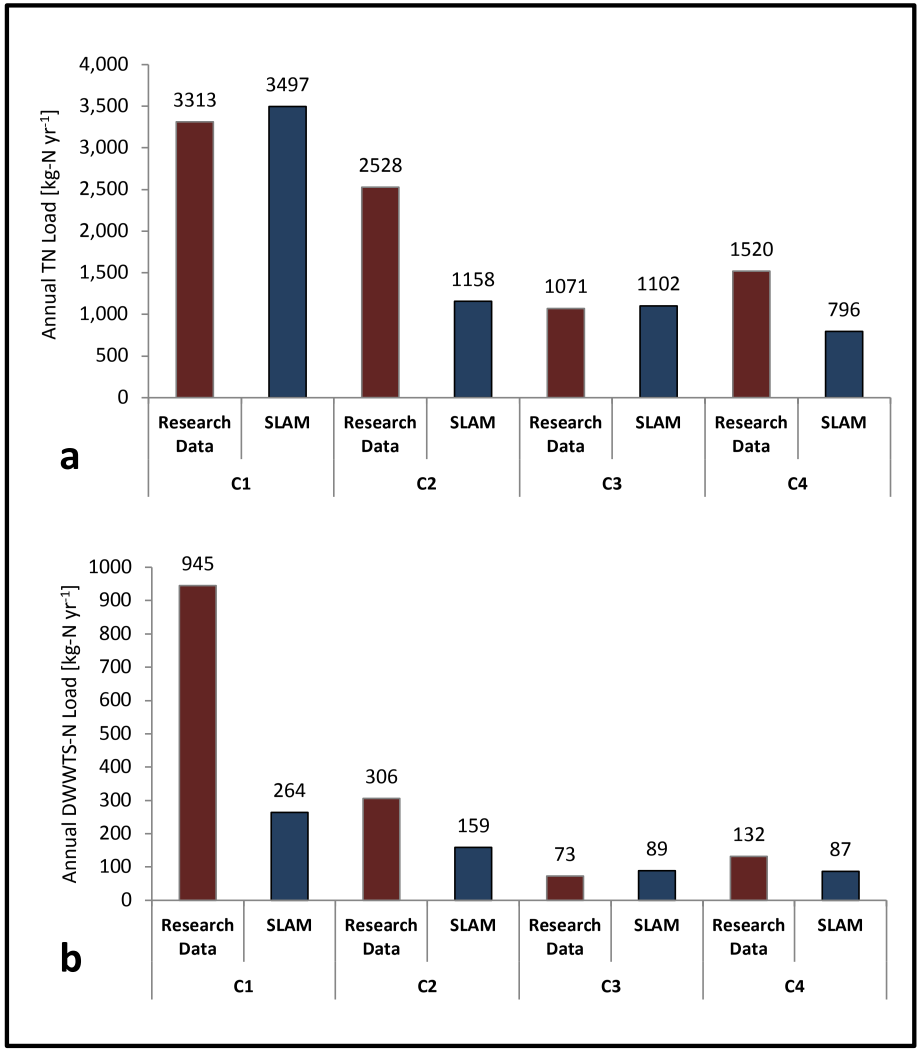

Figure 3.7. Summary chart of (a) annual catchment TN load and (b) DWWTS TN loading in each catchment, as estimated by the research data and by SLAM.

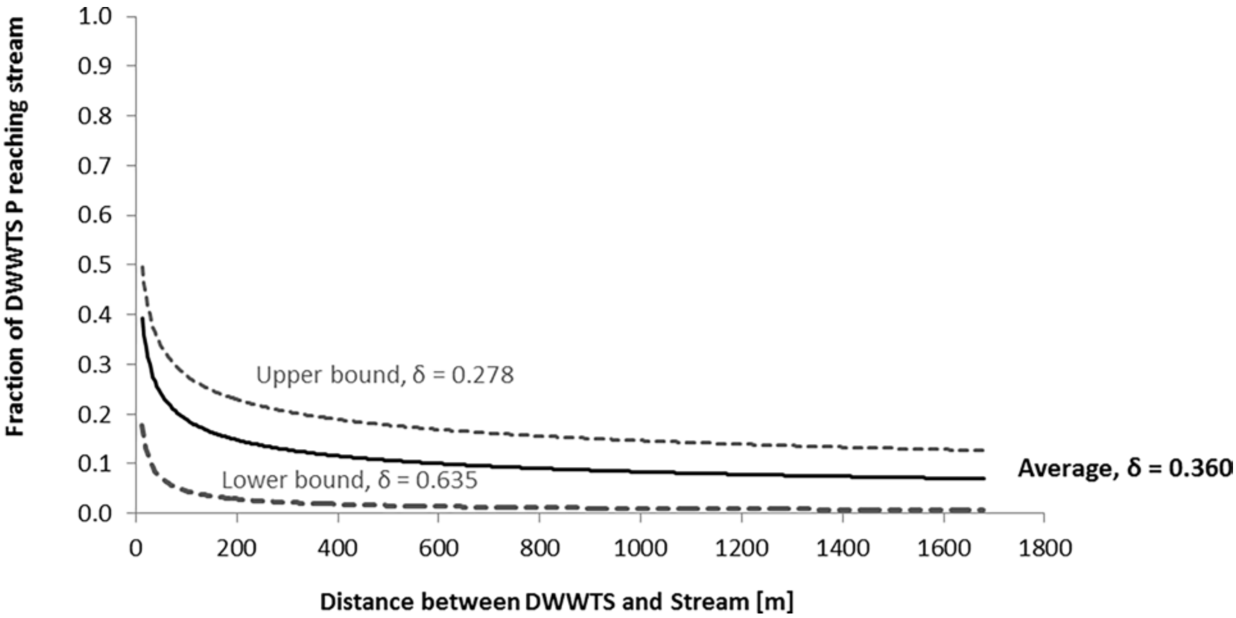

Figure 3.8. Power inverse relationship describing the reduction of DWWTS P load with increasing distance of the DWWTS from the stream.

4 Fingerprinting for Contamination in Wells and Rivers

4.1 Fingerprinting Compounds

Indicator microorganisms (particularly total and thermotolerant coliforms) have traditionally been used to assess microbial water quality. More specifically, *E.coli*, an indigenous member of the intestinal flora of healthy humans and warm-blooded animals, has been shown to be the only coliform almost exclusively associated with a faecal source. However, while the presence of faecal bacteria indicates potential contamination from a faecal source such as a DWWTS (Collins, 2000; De Simone *et al.*, 2009), they are not source-specific (Sinton *et al.*, 1998; Hagedorn and Weisberg, 2009). Many techniques have been developed in recent decades to attempt to find a reliable method for attributing faecal contamination in surface water and groundwater to its specific source. Collectively referred to as faecal source tracking techniques, these methods can be broadly divided into chemical- and microbial-based approaches. Chemical-based approaches rely on the presence of chemical compounds in faeces/wastewater that are specifically related to the host or source. The subsequent detection and quantification of these chemicals in aquatic environments can be used to indicate an impact from the source with which they are associated. Chemical-based markers include simple ionic ratios, faecal sterols, FWCs, artificial sweeteners, caffeine, pharmaceuticals and personal care products. Microbial-based approaches rely on detecting subtle differences present within different groups of microorganisms that can subsequently be used to identify the host or environment from which the organisms were derived. Microbial techniques have included the use of enteric viruses, F-specific RNA coliphages and host-specific genetic markers (Tran *et al.*, 2015).

The tracers applied and evaluated in both the private wells and the rivers include simple ionic ratios, faecal sterol profiles, caffeine, artificial sweeteners, pharmaceuticals, FWCs and genetic Bacteroidales markers.

4.1.1 Potassium/sodium ratio

Effluents from known sources of contamination, such as farmyards, often have K concentrations and K/Na ratios distinctly different from those normally found in natural uncontaminated groundwater. The findings from several studies support the lower ratios associated with DWWTS effluent (Arienzo *et al.*, 2009).

4.1.2 Chloride/bromide ratio

The use of Cl/Br ratios as a tracer of septic tank leachate in groundwater is described by Katz *et al.* (2011), where a threshold ratio of 400 is suggested. Wells with a Cl/Br ratio below this value were considered “non-impacted”, while wells with a Cl/Br ratio in the range of 400–1100 were classed as being influenced by septic tank systems.

4.1.3 Faecal sterol profiles

All faecal material contains sterols and their breakdown products, stanols. However, the types and distribution of sterols and stanols in faeces vary between animals, which gives rise to distinct faecal sterol profiles or fingerprints. This level of specificity between an animal and its faeces enables the potential use of faecal sterol profiles as a tracer of faecal contamination. Distinguishable faecal sterol characteristics for humans, herbivores and birds have been found to be sufficiently distinctive to be of diagnostic value in identifying sources of faecal contamination in waters (Lucas *et al.*, 2007; Derrien *et al.*, 2012; Leeming *et al.*, 2015). In humans, cholesterol, which is a constituent of animal cell membranes and is therefore found in human diets, is preferentially reduced to coprostanol. In contrast, in the environment, this cholesterol is predominately reduced to cholestanol. Similarly, plant-derived 24-ethylcholesterol (which is common in the diets of herbivores) is reduced to 24-ethylcoprostanol and 24-ethylepicoprostanol in the gut of herbivores, whereas in the environment it is primarily reduced to 24-ethylcholestanol.

4.1.4 Microbial source tracking

The use of Bacteroidales as a potential faecal indicator was initially proposed in the mid-1980s because of their greater abundance in faeces than traditionally used coliform bacteria (USEPA, 2005). Early work by Bernhard and Field (2000) demonstrated that bacteria of the order Bacteroidales, and specifically the genus *Bacteroides* isolated from ruminant and humans, were host specific. The specific assays applied as part of this current research were developed for the detection and enumeration of inputs of faecal pollution to catchments by Kildare *et al*. (2007); analysis of samples was carried out by the Department of Microbiology in the National University of Ireland Galway (NUIG).

4.1.5 Emerging organic contaminants

The introduction of a vast number of organic compounds, now used in everyday life, provides opportunities in the field of contamination fingerprinting. These emerging organic compounds, as well as their metabolites and transformation products, can originate from a range of sources.

Fluorescent whitening compounds

FWCs, also known as optical brighteners, are added to common laundry detergents because of their favourable ability to make fabrics appear brighter in colour and to prevent yellow staining and fading over time. As the focus of this study was to assess potentially cost-effective tracers, the fluorometric method has been evaluated here. A new method, modified from the method developed by Cao *et al*. (2009), has been developed in parallel to this study, as detailed in Dubber and Gill (2017). This method considers photo-decay curves for both DSBP and DAS1 FWCs, either of which can be found in almost all detergents used today. The threshold limit for the photo decay method was determined as 0.3 µg/l; samples below this threshold were deemed to be negative with respect to FWCs. All samples with fluorescence concentrations greater than 0.3 µg/l were analysed further using the photo-decay method.

Pharmaceuticals and personal care products

The emergence of pharmaceuticals and personal care products (PPCPs) as contaminants of aquatic environments has received increasing attention in recent years, yet a great deal regarding their occurrence and toxicity is still poorly understood (Lapworth *et al*., 2012). Pharmaceuticals can enter aquatic environments from various different sources, including from the direct disposal of pharmaceuticals down drains, from landfills and from the direct discharges from pharmaceutical manufacturing companies. Importantly, however, the main source of pharmaceuticals is believed to be the discharge of both treated and untreated domestic wastewater. All samples taken during the mass tracer evaluation events were analysed for carbamazepine and sulfamethoxazole in the Teagasc Food Research Centre, Ashtown, County Dublin.

Caffeine and artificial sweeteners

Caffeine, while usually associated with coffee products, is also found in tea, chocolate, tobacco, medication and energy drinks (Chen *et al*., 2002). Of the caffeine consumed by humans, a small proportion is not metabolised and is excreted as urine and enters wastewater. Furthermore, caffeine can enter wastewater from the disposal of unconsumed caffeine products and the washing of coffee pots and cups. Numerous studies have detected caffeine in human wastewater, both internationally and in Ireland (Seiler *et al*., 1999; Gill *et al*., 2009; Daneshvar *et al*., 2012).

Artificial sweeteners are commonplace in modern human diets because of their presence in many foods and drinks; there is some, more limited, use of artificial sweeteners in animal feeds. Six artificial sweeteners are approved for use in the USA by the United States Food and Drug Administration (FDA): acesulfame-K, aspartame, neotame, saccharin, advantame and sucralose. The use of artificial sweeteners in Europe is regulated under EU Regulation No. 1333/2008, and a full list of approved sweeteners is identified therein (1333/2008/EC). Notably, cyclamate is approved for use in Europe.

All samples taken during the mass tracer evaluation events were analysed for caffeine, acesulfame, aspartame, cyclamate, saccharin and sucralose in the Teagasc Food Research Centre, Ashtown, County Dublin.

4.2 Results of Sampling for Fingerprinting Compounds

Some examples of the results of the sampling for DWWTS fingerprinting compounds are presented in this section from the private well water research.

4.2.1 Chloride/bromide ratio

The Cl/Br ratios for the 24 wells sampled during the M11 and M13 monitoring events are shown in Figure 4.1, alongside the results for the corresponding P/A of *E. coli*. The Cl/Br ratios for all wells analysed during these events have also been plotted against the corresponding chloride concentrations in Figure 4.2, with the different ranges that have been developed and used as diagnostic tools for indicating anthropogenic influences by various researchers overlaid.

4.2.2 Faecal sterol profiles

For the M11 sampling event, 7 out of the 16 wells had a total sterol content below the 2 µg/l benchmark and so no conclusion could be drawn regarding an impact (or otherwise) from faecal contamination. The remaining nine wells did have a total sterol content greater than 2 µg/l and so the next step in the process (examining the ratios of coprostanol:cholestanol and 24-ethylcopsrostanol:24-ethylcholestanol) could be applied. However, in three out of the nine wells none of these sterols were detected. In the remaining six wells, there were isolated detections of coprostanol, cholesterol, cholestanol and 24-ethylcoprostanol. Three had quantifiable levels of one of these sterols, while the remaining three had quantifiable levels of two sterols. This was not sufficient to apply the ratio techniques. The results from the M13 event led to similar conclusions. None of the wells had a total sterol content greater than the 2 µg/l threshold. The results from this therefore suggest that faecal sterols analysis is not a suitable human wastewater contamination fingerprinting technique for domestic wells. This is probably due to the hydrophobic nature of the faecal sterols and stanols, which means that they are easily adsorbed onto particulate phases of wastewater, sediments and soils during the passage from the surface to subsurface environment.

4.2.3 Microbial source tracking

The results from the Bacteroidales assays from the M13 monitoring event are shown in Figure 4.3. Also shown are the corresponding results for the conventional faecal indicator bacteria (FIB) (total coliforms). While an assessment of these results is provided, the small sample size inhibits meaningful statistical tests.

The human specific marker, BacHum-UCD, was quantified in three out of the eight wells, with gene copy concentration in gene copies per litre (GCC/l) values ranging from 2.73×10^3 to 7.37×10^3, indicating that these wells are impacted by a human faecal source, such as a DWWTS. The BacBov-UCD marker was quantified in two out of the eight well samples, with values ranging from 2.13×10^3 to 2.73×10^3 GCC/l, indicating an impact from bovine-based agriculture. The two wells that tested positive for the BacBov marker also tested positive for BacHum, which indicates mixed sources of contamination. The BacUni-UCD marker was detected in all eight wells, with values ranging from 2.16×10^2 to 1.32×10^3 GCC/l. With respect to conventional FIB, two wells (KX10 and CX52) tested positive for *E. coli*, while four wells tested positive for total coliforms.

The results show that the assays developed by Kildare *et al*. (2007) can detect BacHum-UCD, BacBov-UCD and BacUni-UCD markers in water samples from Irish private wells. However, interpreting the results is challenging for several reasons. Notably, there is a significant lack of agreement between the Bacteroidales markers and the conventional FIB. This is potentially due to the culture-independent and culture-dependent nature of the respective tests along with the fact that little has been documented on the decay of Bacteroides cells and DNA in aquatic environments internationally (Ahmed *et al*., 2016).

4.2.4 Fluorescent whitening compounds

Most well water samples returned FWC values relatively close to the methods limit of detection (0.3 µg/l). Furthermore, all the values were below the photo-decay threshold and so the photo-decay method could not be applied [see Dubber and Gill (2017) for more detail]. Subsequently, it was concluded that none of the wells in SA1 or SA2 tested positive for FWCs using this method. Initial fluorescence values were, however, notably higher for many of the wells in SA3 and SA4 (Figure 4.4). During the M11 sampling event, five of the wells (in SA3 and SA4) had concentrations below the photo-decay threshold, which again indicates a negative result. However, the remaining seven wells had concentrations above the threshold and so the photo-decay method was applied to these samples. No noticeable reduction of fluorescence was observed for any sample after 1 minute's exposure, while a relatively consistent drop was seen after 10

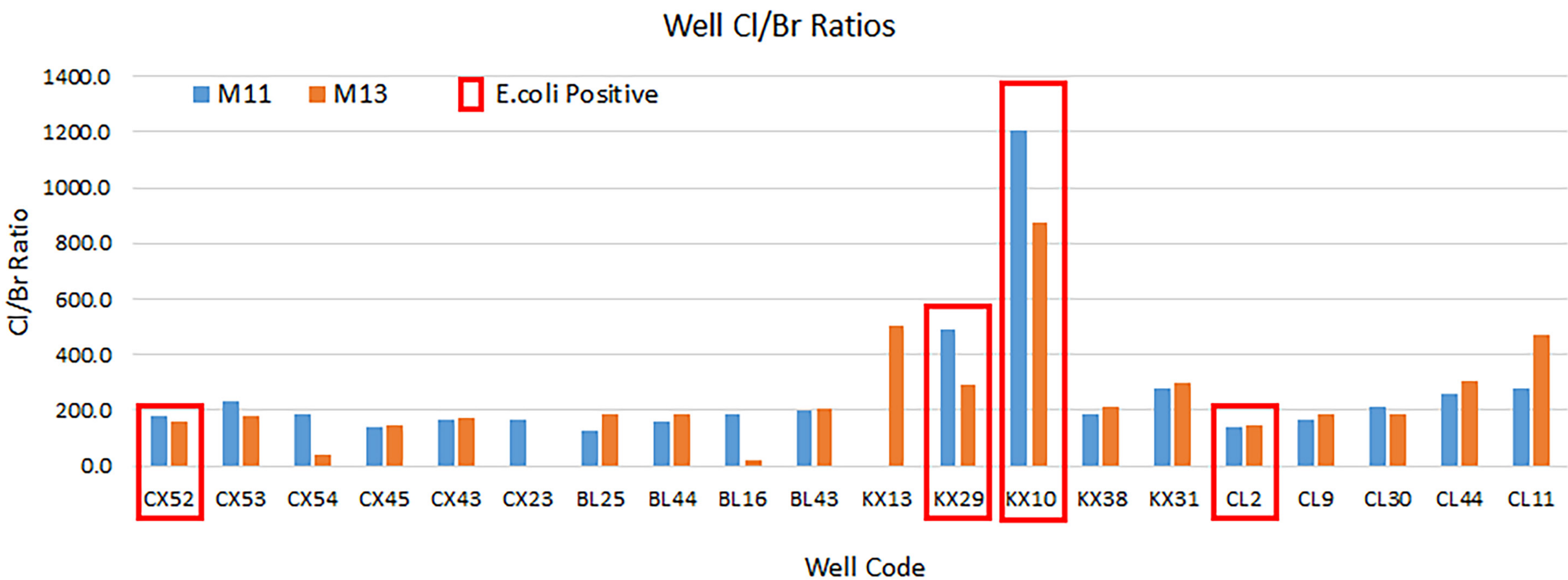

Figure 4.1. Cl/Br ratios for all wells in M11 and M13 monitoring periods. Also shown are the wells that tested positive for *E. coli*. Samples treated by water softeners are excluded.

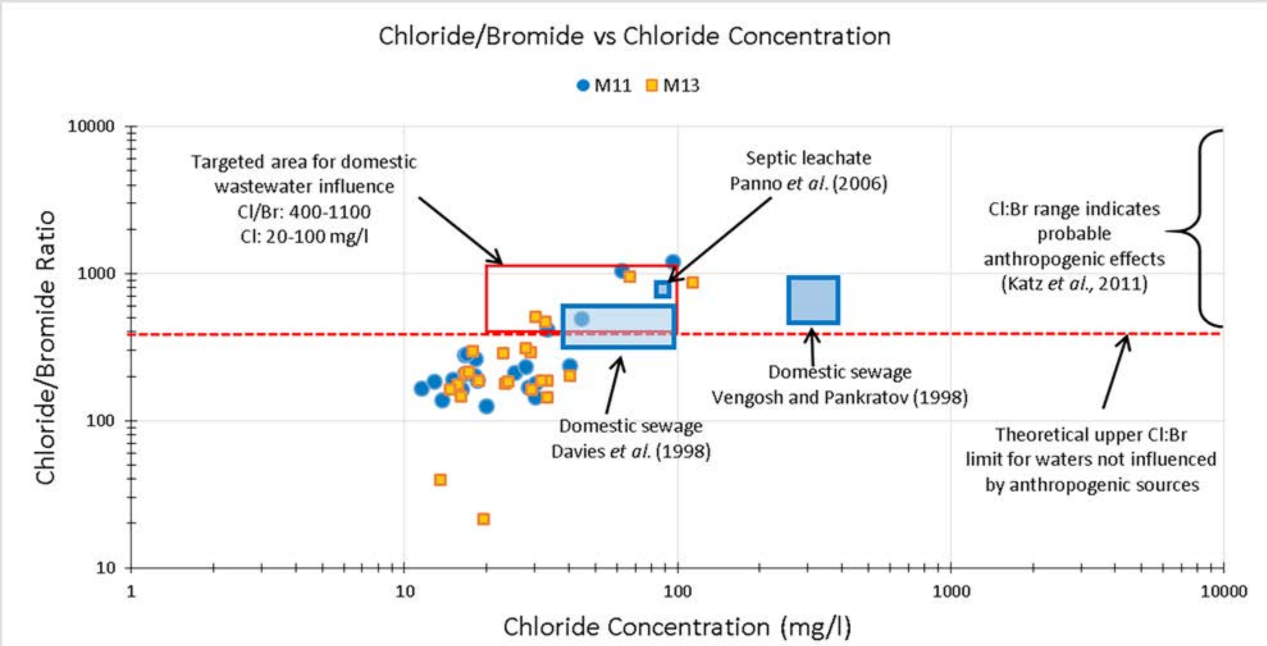

Figure 4.2. Cl/Br ratio versus chloride concentration for M11 and M13 monitoring event (*n* = 24 wells). Included are previous measured ranges for domestic wastewater and thresholds for indicating potential influences from anthropogenic activities.

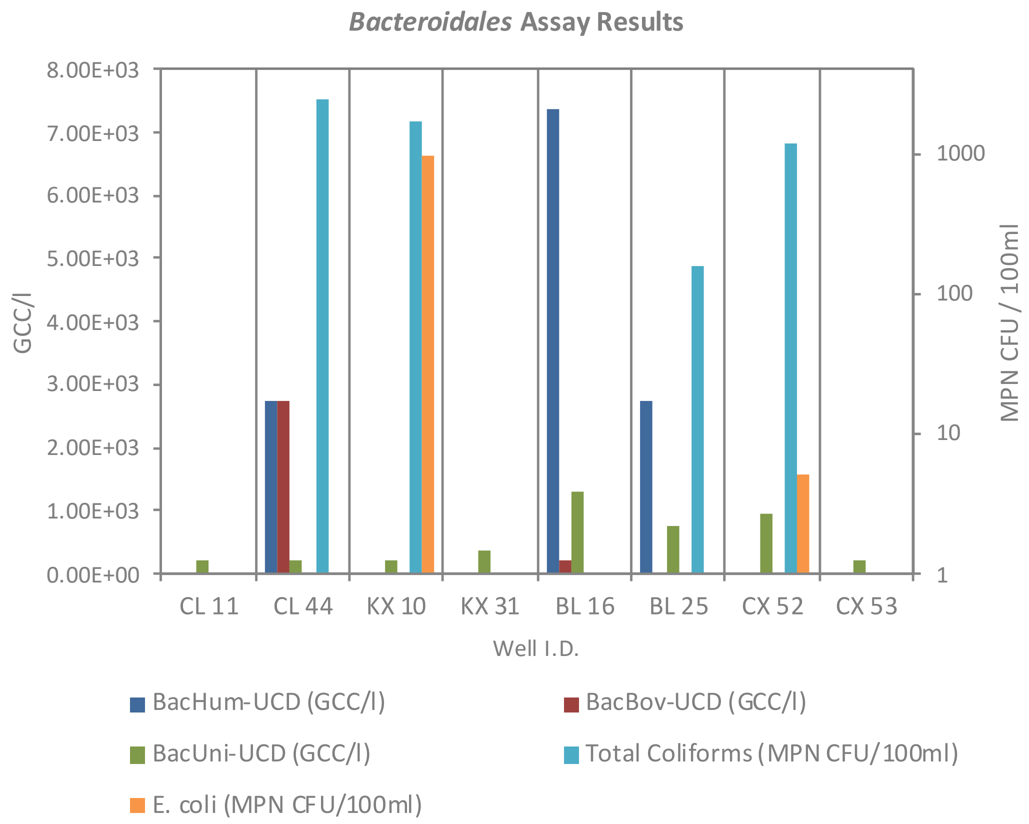

Figure 4.3. Results from the Bacteroidales assays carried out on eight well samples from M13. Note the difference in scales between the results for Bacteroidales assays (GCC/l) and conventional FIB [MPN (CFU)/100ml]. MPN, most probable number; CFU, colony-forming unit.

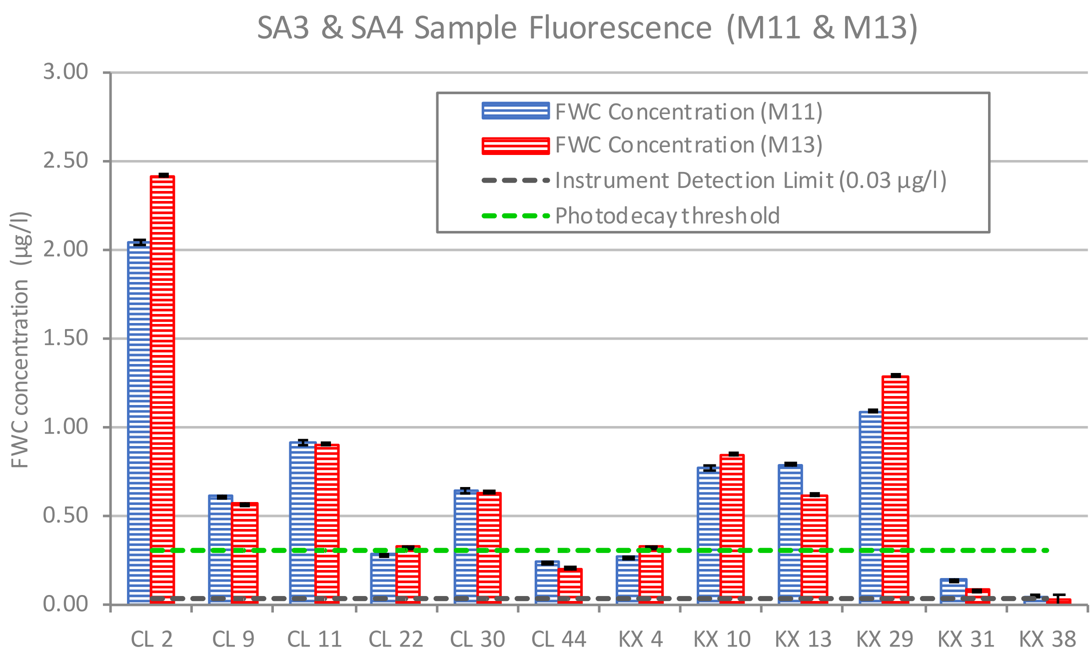

Figure 4.4. SA3 and SA4 initial sample fluorescence (i.e. pre-UV light exposure). Fluorescence readings expressed in terms of FWC concentration calculated from DSBP calibration curve.

minutes' exposure. Correspondingly, according to the predefined criteria (whereby a photo-decay signal reduction ratio of >0.25 after 10 minutes indicates the presence of FWCs), no wells tested positive for FWCs.

4.2.5 Pharmaceuticals and personal care products

The sampling results found carbamazepine and sulfamethoxazole to be of limited use as tracers of DWWTS contamination in private wells. This could potentially be due to the domestic scale of the research and the assumed relatively low likelihood of these PPCPs being in the wastewater. Nonetheless, as the presence of PPCPs in a private groundwater supply does indicate a human influence, potentially from a DWWTS, the testing for PPCPs in private wells could yield useful results and may still be warranted. However, the high cost of analysis and the advanced technology required must be considered.

4.2.6 Caffeine and artificial sweeteners

The results indicated that acesulfame, as a tracer of wastewater contamination, is applicable to the DWWTS and private well scale. The isolated presence of cyclamate in a single well suggests that it might also be potentially useful, although less so than for acesulfame. The absence of saccharin and sucralose suggests that they have limited use as a groundwater tracer.

4.2.7 Conclusions for fingerprinting compounds

In conclusion, the evaluation of various contamination fingerprinting techniques for private wells yielded mixed results. Simple ionic ratios (K/Na and Cl/Br) yielded interesting results. Elevated Cl/Br ratios appear to be consistent with an impact from a DWWTS, while elevated K/Na ratios appear to be consistent with an impact from decaying organic matter, such as farmyard effluent. While neither method appears absolutely conclusive in identifying a contaminant source, given the positive results and low cost of application, the method appears to have value. Further positive results were seen with respect to the sweetener acesulfame. Consistent with the literature, acesulfame was detected in several wells, suggesting that it is a useful tracer of DWWTS contamination. In contrast, faecal sterol profiling, FWCs, carbamazepine and caffeine analysis yielded negative results, with an apparently limited applicability to the private wells and DWWTSs in Ireland. While the pharmaceutical product sulfamethoxazole was detected in a single well, this provides little information regarding its overall applicability as a contamination tracer. Microbial source tracking results using the Bacteroidales assays developed by Kildare *et al*. (2007) did yield positive results, although interpretation of these is difficult for several reasons.

Targeted sampling events were carried out to assess similar fingerprinting techniques in the surface water studies, leading to a recommended tiered approach for short-term, low-flow monitoring (see section 4.3.2). The results showed that the more common analytical techniques of *E. coli* and ammonia were useful as an initial screening tool to determine whether a stream was impacted. However, more confirmatory fingerprinting tests were then required to determine whether the impact was from DWWTS effluent. The results showed that FWCs, contrary to the findings on the groundwater samples, could be useful indicators of localised recent DWWTS discharge into streams. Equally, artificial sweeteners, caffeine and pharmaceuticals were all picked up downstream of DWWTSs on the catchments, which must have originated from a human source. Faecal sterols also showed promise as a means of indicating the contribution of human versus herbivore contamination. However, microbial source tracking results from the combined monitoring events left many questions and uncertainties with respect to the applicability of the method at the current stage of development of this technique.

4.3 Tiered Approach

Tiered approaches for the application of the potentially useful fingerprinting techniques were developed. These tiered approaches provide a logical structure of steps that could be applied to help identify the source(s) of any contamination that is present in either a private well or a river, particularly if is linked to human wastewater effluent. The philosophy behind such tiered approaches is to start by using less resource-intensive techniques, such as site assessment and/or more standard water quality parameters, and then move on to more resource-intensive analyses for confirmatory specific fingerprints of human wastewater effluent.

4.3.1 Tiered investigative approach for wells

The recommended tiered approach for the investigation of groundwater contamination of a well is shown in Figure 4.5. The first stage involves a rigorous site assessment carried out in tandem with water quality analysis for conventional water quality microbial and chemical parameters (*E. coli*, ammonia, etc.) in addition to more specific tracer techniques to determine K/Na and Cl/Br ratios. If these procedures are deemed unsuccessful (i.e. if they do not indicate a clear source or if the remedial works they recommend do not rectify the contamination), then the more resource-intensive second stage in the tiered approach should be applied, whereby the well is tested for artificial sweeteners. However, as the site assessment and ionic ratios were found to accurately assess susceptibility to contamination in a considerable number of wells, it is likely that the first stage in the tiered approach will be able to identify the probable sources and pathways of observed contamination. If remedial work based on the findings of stage one is successful in improving well quality and preventing future contamination, then the application of stage two should not be necessary.

4.3.2 Tiered investigative approach for surface water

The recommended tiered approach for the investigation of suspected surface water contamination of a stream/river is shown in Figure 4.6. In such a scenario, short-term, high-frequency monitoring should be set up upstream and downstream of the suspected point of contamination and samples taken during low-flow conditions. These should first be analysed for standard water quality microbial and chemical parameters: *E. coli* and ammonia. If these show a significant difference between the upstream and downstream samples, then analysis should be carried out for FWCs using the Dubber and Gill (2017) methodology to confirm whether the contamination is from domestic wastewater effluent. If the FWC method suggests no human impact, but such a source is still suspected, then additional confirmatory analysis should be carried out by targeting caffeine, artificial sweeteners and pharmaceutical compounds. Equally, further confirmatory tests can be made by analysing for faecal sterols, which will give an additional indication of the percentage contribution of human effluent contamination versus that of other sources. The final confirmation of a failing individual DWWTS could then be attempted by the use of a high concentration of fluorescent dye added to the system itself to see if it ends up in the stream.

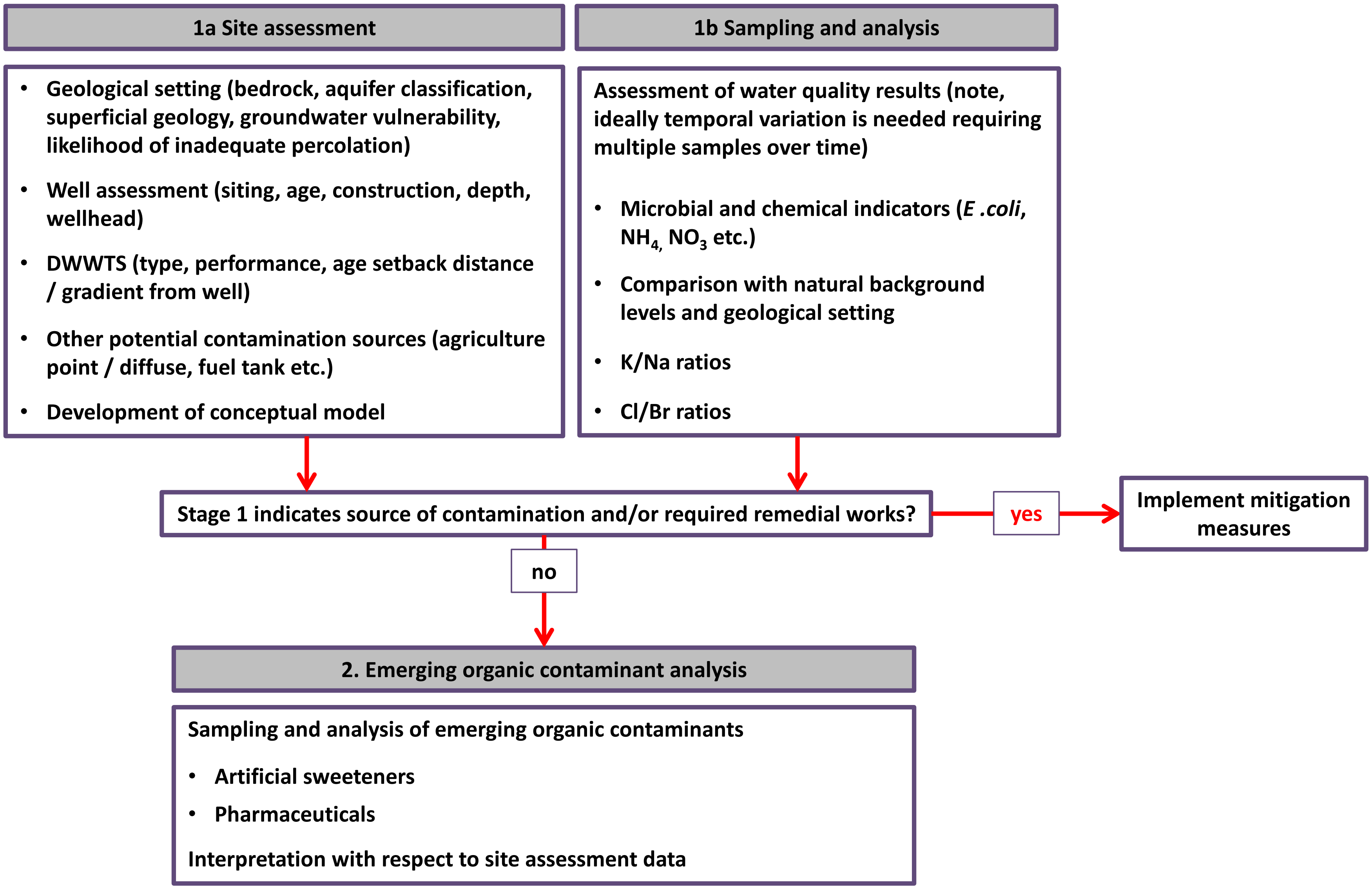

Figure 4.5. Tiered approach for the application of contamination fingerprinting techniques for wells.

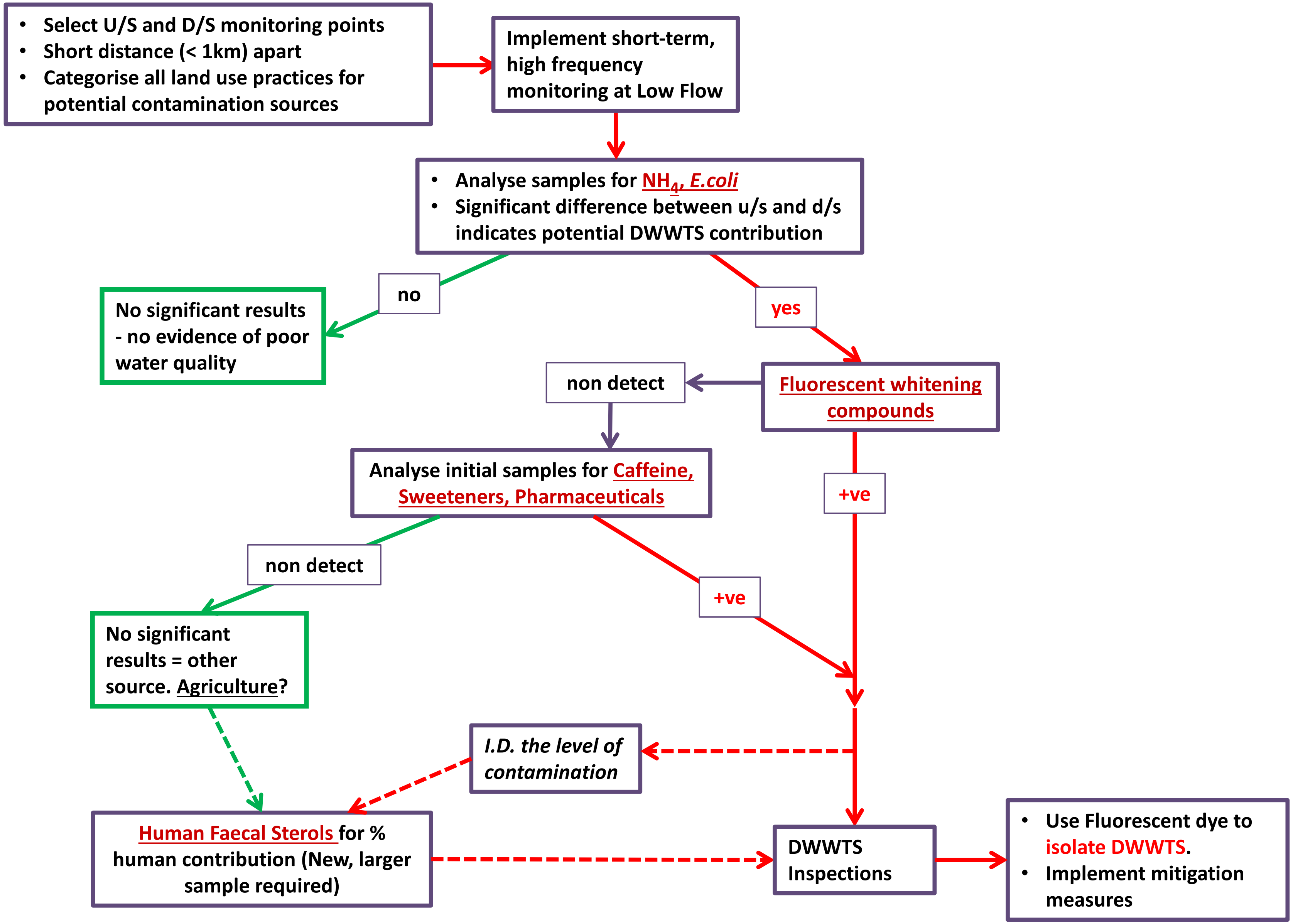

Figure 4.6. Tiered approach for the application of contamination fingerprinting techniques for streams.

5 Conclusions and Recommendations

5.1 Conclusions

The 212 site assessments carried out on private wells have shown that the standard of wells in Ireland (and hence their susceptibility to contamination) is highly variable. This research has demonstrated that only 35% of the wells assessed were adequately finished/sealed, defined as where the liner extends above the ground surface and where the well is capped, covered and surrounded by a concrete surface apron. A considerable proportion of wells were therefore found to be inadequately sealed, leaving them potentially vulnerable to contamination by the direct ingress of contaminants. Subsequent sampling showed that a proportion of wells in each of the study areas were indeed contaminated with *E. coli* (16% in SA1, 17% in SA2, 10% in SA3 and 16% in SA4).

Several site-specific factors were found to be associated with incidences of *E. coli* contamination. Bivariate tests and the development of a multiple logistic regression model demonstrated that poor wellhead completion, coupled with proximity to a potential point source of agricultural contamination, significantly increased the likelihood of microbial contamination. This research also illustrated the extent to which susceptibility to contamination depends on the well design (i.e. dug vs bored), whereby dug wells were several times more likely to be contaminated with *E. coli* and/or nitrates than bored wells. Conversely, no geological variables were found to be statistically significantly related to the presence of *E. coli* in any of the sites studied by bivariate or MLR analysis. Although this does not discount geological variables as key factors in controlling private well susceptibility to contamination, it does suggest that other site-specific factors (i.e. wellhead finish, the local properties of the unsaturated zone, etc.) are more prominent controls. It was also notable that there was a lack of any statistically significant associations between private well contamination and any of the DWWTS variables recorded by the site assessment (DWWTS age, type, setback distance, gradient, etc.). While this does not rule out DWWTSs as sources of private well contamination, as concluded with respect to geology, it does suggest that other site-specific factors are more strongly related to contamination.

The subsequent monitoring of 24 specially selected wells over a 14-month period identified significant temporal variation in well water quality. For instance, while 37% of the monitoring wells tested positive for *E. coli* in September, only 17% tested positive in the following month, October. Overall, 58% of the wells tested positive on at least one occasion over the 14-month sampling period. This highlights the caution with which a one-off sample must be regarded. As discussed, based on these results, frequent sampling would be recommended to accurately assess the quality of well water.

The research on the four small river catchments, all with low-permeability subsoils as well as high concentrations of septic tanks (from 16 to 33 per km^2), revealed rapid responses to rainfall events indicative of shallow hydraulic pathways. Water quality and flow monitoring over the course of a year revealed varying contributions of P attributable to DWWTS in the catchments as follows: Wicklow 13.8%, Wexford 8.5%, Cavan 1.3% and Longford 11.3%. These fieldwork-based predictions have also been compared against the predictions made by the EPA's SLAM, which has been applied to the four study catchments. This has provided some useful insights on catchment modelling using coarser data (such as following the SLAM approach). In addition, functions have been developed on the basis of nutrient attenuation factors applied to every DWWTS, dependent on their distance from a stream, that can be incorporated into catchment modelling approaches.

The contamination fingerprinting techniques applied and evaluated during this study include ionic ratios (K/Na and Cl/Br), library independent microbial source tracking, faecal sterol profile analysis and the analysis of a range of emerging organic compounds of specific anthropogenic origin (including FWCs, caffeine, artificial sweeteners and pharmaceuticals). For the groundwater samples it was deemed that three methods/compounds (K/Na, Cl/Br and acesulfame) performed adequately as tracers of well contamination.

The K/Na ionic ratio technique showed potential in identifying impacts from decaying organic matter (such as farmyard waste) while the Cl/Br ionic ratio technique showed potential in identifying impacts from DWWTSs. With respect to artificial sweeteners, the results from the 24 monitoring wells are consistent with this previous research. Acesulfame appears to show significant potential as an indicator of an impact from a DWWTS, as it was detected in several faecal-contaminated monitoring wells. The isolated presence of cyclamate in a single well suggests that it is also potentially useful, albeit probably less so than acesulfame. The absence of saccharin and sucralose suggests that they have limited use as groundwater tracers.

Two tiered approaches for utilising the potentially useful fingerprinting techniques were developed for both groundwater and surface water, based on starting with the less resource-intensive tracer techniques in combination with rigorous site assessments. If these procedures were deemed unsuccessful (i.e. they did not indicate a clear source), then the more resource-intensive second stage in the tiered approach, which tests for human-specific compounds (artificial sweeteners, faecal sterols, etc.), should be applied. Such a tiered approach provides a logical and efficient framework that could be utilised to improve the quality of wells and rivers.

Overall, this research provided valuable insights into the impact of DWWTS effluent as a health hazard to consumers of groundwater from private wells, and also as a source of nutrient water pollution to surface water in small catchments with low-permeability soils. The research also tested various different chemical, biochemical and microbiological compounds as tracers of DWWTS effluent contamination in both private wells and rivers, leading to recommended tiered approaches for testing suspected contamination in both surface water and wells. More specific recommendations arising from this research are given in Table 5.1.

5.2 Recommendations

Table 5.1. Recommendations for implementation and uptake of research findings

Issue	Recommendation	Target	Time frame (years)
Suspected contamination of private wells by DWWTS	Disseminate the recommended tiered approach for groundwater contamination to the EPA, LAs, government departments and environmental consultancies	EPA, LAs, DAFM, HSE, DCCAE	< 1
Suspected contamination of private wells by DWWTS	Introduce tighter regulations for private well construction, including post-construction sign-off by a regulatory body	DCCAE, LAs	1–3
Suspected contamination of private wells by DWWTS	Introduce mandatory well sampling and analysis by homeowner at a set frequency (e.g. once every 5 years) to assess quality.	DCCAE, LAs	1–3
Suspected contamination of rivers by DWWTS	Disseminate the recommended tiered approach for groundwater contamination to the EPA, LAs, government departments and environmental consultancies	EPA, LAs, DAFM, HSE, DCCAE	< 1
Nutrient input from DWWTS into surface waters (re. WFD)	Incorporation of findings from the project into the EPA's SLAM catchment management tool	EPA	1–2
Strategic assessment for more decentralised cluster systems for wastewater treatment in areas of low-permeability subsoils	This research strengthens the recommendation from previous projects (2010-W-LS-3) for pilot schemes to be funded by DHPLG to determine the socio-economic challenges associated with such systems	DCCAE, LAs, IW	2–5
Risk of well contamination from DWWTS	Further research into the use of microbial source tracking techniques and other fingerprinting methods with larger scale links to health statistics	EPA, DoH, HSE	1–3
Contamination of surface waters by DWWTS in areas of low-permeability soils	This research strengthens the recommendations from previous projects (2010-W-LS-3; Gill *et al.*, 2015) to use alternative treatment systems in low-permeability areas (e.g. willow-based evapotranspiration systems, drip distribution systems and low-pressure pipe systems) as well as encouraging water-saving practices	EPA, DCCAE, LAs	1–2
Private well susceptibility to contamination due to poor design and construction	Disseminate the findings from this research to the most likely causes of contamination linked to poor well design/construction, e.g. poor wellhead completion, coupled with proximity to point sources of agricultural contamination	EPA, GSI	< 1

LAs, local authorities; DAFM, Department of Agriculture, Food and the Marine; HSE, Health Service Executive; DCCAE, Department of Communications, Climate Action and Environment; DHPLG, Department of Housing, Planning and Local Government; IW, Irish Water; DoH, Department of Health; GSI, Geological Survey of Ireland; WFD, Water Framework Directive.

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Abbreviations

DHPLG	Department of Housing, Planning and Local Government
DRP	Dissolved reactive phosphorus
DWWTS	Domestic wastewater treatment system
EPA	Environmental Protection Agency
FIB	Faecal indicator bacteria
FWC	Fluorescent whitening compound
GCC/l	Gene copy concentration in gene copies per litre
GIS	Geographic information system
GSI	Geological Survey of Ireland
LIPTG	Likelihood of inadequate percolation to groundwater
LR	Logistic regression
MLR	Multiple logistic regression
MPN	Most probable number
N	Nitrogen
NH_3	Ammonia
NO_2-N	Nitrogen present as nitrite
NO_3-N	Nitrogen present as nitrate
OR	Odds ratio
P	Phosphorus
P/A	Presence/absence
PPCPs	Pharmaceuticals and personal care products
SLAM	Source Load Apportionment Model
TN	Total nitrogen
TP	Total phosphorus
WFD	Water Framework Directive

AN GHNÍOMHAIREACHT UM CHAOMHNÚ COMHSHAOIL
Tá an Ghníomhaireacht um Chaomhnú Comhshaoil (GCC) freagrach as an gcomhshaol a chaomhnú agus a fheabhsú mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaol a chosaint ó éifeachtaí díobhálacha na radaíochta agus an truaillithe.

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

Rialú: *Déanaimid córais éifeachtacha rialaithe agus comhlíonta comhshaoil a chur i bhfeidhm chun torthaí maithe comhshaoil a sholáthar agus chun díriú orthu siúd nach gcloíonn leis na córais sin.*

Eolas: *Soláthraímid sonraí, faisnéis agus measúnú comhshaoil atá ar ardchaighdeán, spriocdhírithe agus tráthúil chun bonn eolais a chur faoin gcinnteoireacht ar gach leibhéal.*

Tacaíocht: *Bímid ag saothrú i gcomhar le grúpaí eile chun tacú le comhshaol atá glan, táirgiúil agus cosanta go maith, agus le hiompar a chuirfidh le comhshaol inbhuanaithe.*

Ár bhFreagrachtaí

Ceadúnú
Déanaimid na gníomhaíochtaí seo a leanas a rialú ionas nach ndéanann siad dochar do shláinte an phobail ná don chomhshaol:
- saoráidí dramhaíola *(m.sh. láithreáin líonta talún, loisceoirí, stáisiúin aistrithe dramhaíola)*;
- gníomhaíochtaí tionsclaíocha ar scála mór *(m.sh. déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta)*;
- an diantalmhaíocht *(m.sh. muca, éanlaith)*;
- úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe *(OGM)*;
- foinsí radaíochta ianúcháin *(m.sh. trealamh x-gha agus radaiteiripe, foinsí tionsclaíocha)*;
- áiseanna móra stórála peitril;
- scardadh dramhuisce;
- gníomhaíochtaí dumpála ar farraige.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil
- Clár náisiúnta iniúchtaí agus cigireachtaí a dhéanamh gach bliain ar shaoráidí a bhfuil ceadúnas ón nGníomhaireacht acu.
- Maoirseacht a dhéanamh ar fhreagrachtaí cosanta comhshaoil na n-údarás áitiúil.
- Caighdeán an uisce óil, arna sholáthar ag soláthraithe uisce phoiblí, a mhaoirsiú.
- Obair le húdaráis áitiúla agus le gníomhaireachtaí eile chun dul i ngleic le coireanna comhshaoil trí chomhordú a dhéanamh ar líonra forfheidhmiúcháin náisiúnta, trí dhíriú ar chiontóirí, agus trí mhaoirsiú a dhéanamh ar leasúchán.
- Cur i bhfeidhm rialachán ar nós na Rialachán um Dhramhthrealamh Leictreach agus Leictreonach (DTLL), um Shrian ar Shubstaintí Guaiseacha agus na Rialachán um rialú ar shubstaintí a ídíonn an ciseal ózóin.
- An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaol.

Bainistíocht Uisce
- Monatóireacht agus tuairisciú a dhéanamh ar cháilíocht aibhneacha, lochanna, uiscí idirchriosacha agus cósta na hÉireann, agus screamhuiscí; leibhéil uisce agus sruthanna aibhneacha a thomhas.
- Comhordú náisiúnta agus maoirsiú a dhéanamh ar an gCreat-Treoir Uisce.
- Monatóireacht agus tuairisciú a dhéanamh ar Cháilíocht an Uisce Snámha.

Monatóireacht, Anailís agus Tuairisciú ar an gComhshaol
- Monatóireacht a dhéanamh ar cháilíocht an aeir agus Treoir an AE maidir le hAer Glan don Eoraip (CAFÉ) a chur chun feidhme.
- Tuairisciú neamhspleách le cabhrú le cinnteoireacht an rialtais náisiúnta agus na n-údarás áitiúil *(m.sh. tuairisciú tréimhsiúil ar staid Chomhshaol na hÉireann agus Tuarascálacha ar Tháscairí)*.

Rialú Astaíochtaí na nGás Ceaptha Teasa in Éirinn
- Fardail agus réamh-mheastacháin na hÉireann maidir le gáis cheaptha teasa a ullmhú.
- An Treoir maidir le Trádáil Astaíochtaí a chur chun feidhme i gcomhair breis agus 100 de na táirgeoirí dé-ocsaíde carbóin is mó in Éirinn.

Taighde agus Forbairt Comhshaoil
- Taighde comhshaoil a chistiú chun brúnna a shainaithint, bonn eolais a chur faoi bheartais, agus réitigh a sholáthar i réimsí na haeráide, an uisce agus na hinbhuanaitheachta.

Measúnacht Straitéiseach Timpeallachta
- Measúnacht a dhéanamh ar thionchar pleananna agus clár beartaithe ar an gcomhshaol in Éirinn *(m.sh. mórphleananna forbartha)*.

Cosaint Raideolaíoch
- Monatóireacht a dhéanamh ar leibhéil radaíochta, measúnacht a dhéanamh ar nochtadh mhuintir na hÉireann don radaíocht ianúcháin.
- Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as taismí núicléacha.
- Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta.
- Sainseirbhísí cosanta ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

Treoir, Faisnéis Inrochtana agus Oideachas
- Comhairle agus treoir a chur ar fáil d'earnáil na tionsclaíochta agus don phobal maidir le hábhair a bhaineann le caomhnú an chomhshaoil agus leis an gcosaint raideolaíoch.
- Faisnéis thráthúil ar an gcomhshaol ar a bhfuil fáil éasca a chur ar fáil chun rannpháirtíocht an phobail a spreagadh sa chinnteoireacht i ndáil leis an gcomhshaol *(m.sh. Timpeall an Tí, léarscáileanna radóin)*.
- Comhairle a chur ar fáil don Rialtas maidir le hábhair a bhaineann leis an tsábháilteacht raideolaíoch agus le cúrsaí práinnfhreagartha.
- Plean Náisiúnta Bainistíochta Dramhaíola Guaisí a fhorbairt chun dramhaíl ghuaiseach a chosc agus a bhainistiú.

Múscailt Feasachta agus Athrú Iompraíochta
- Feasacht chomhshaoil níos fearr a ghiniúint agus dul i bhfeidhm ar athrú iompraíochta dearfach trí thacú le gnóthais, le pobail agus le teaghlaigh a bheith níos éifeachtúla ar acmhainní.
- Tástáil le haghaidh radóin a chur chun cinn i dtithe agus in ionaid oibre, agus gníomhartha leasúcháin a spreagadh nuair is gá.

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil
Tá an ghníomhaíocht á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóirí. Déantar an obair ar fud cúig cinn d'Oifigí:
- An Oifig um Inmharthanacht Comhshaoil
- An Oifig Forfheidhmithe i leith cúrsaí Comhshaoil
- An Oifig um Fianaise is Measúnú
- Oifig um Chosaint Radaíochta agus Monatóireachta Comhshaoil
- An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag comhaltaí air agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair imní agus le comhairle a chur ar an mBord.

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The Impact of On-site Domestic Wastewater Effluent on Rivers and Wells

Authors: Laurence Gill, Vincent O'Flaherty, Bruce Misstear, Laura Brophy, Christopher Fennell, Donata Dubber, David O'Connell, Kathryn Kilroy, Maria Barrett, Paul Johnston, Francesco Pilla and Phillip Geary

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Climate - Water - Sustainability
Identifying pressures • Informing policy • Developing solutions

Identifying Pressures

The domestic wastewater of approximately one-third of the population in Ireland (~490,000 dwellings) is treated on site by domestic wastewater treatment systems (DWWTS), of which approximately 90% are septic tanks. This research project has evaluated the impact of DWWTS effluent (i.e. the liquid waste or sewage that is discharged into the sea or a river) as a health hazard to consumers of groundwater from private wells and also a source of nutrient water pollution to surface water in small catchments. This has been achieved by (1) surveying and sampling 212 wells in four areas, comprising two areas of Extreme and two areas of Low groundwater vulnerability and (2) sampling river water quality upstream and downstream of clusters of DWWTS on small streams in four catchments underlain by low-permeability subsoils. The research on private wells has found that 15% of wells were contaminated by E. coli in a one-off sampling round. Site assessments also revealed that just 35% of wells assessed were adequately finished and protected against contamination. The research on the four small river catchments, all of which were situated in low-permeability subsoils and had a high density of septic tanks (from 16 to 33 per km2), revealed rapid hydrological responses to rainfall events indicative of shallow hydraulic pathways. Water quality and flow monitoring over the course of a year revealed varying contributions of phosphorus attributable to DWWTS in the catchments, from 1.3 to 13.8%.

Informing Policy

The results obtained from these trials have been used to inform the Source Load Apportionment Modelling carried out by the Catchments Unit within the Environmental Protection Agency. This model is being used to help characterise sources of aquatic pollution in catchments, from which programmes of measures are being developed in accordance with Ireland's commitments under the Water Framework Directive. In addition, the 212 site assessments carried out on private wells have shown that the design, construction and condition, and therefore overall standard, of wells in Ireland (and hence their susceptibility to contamination) is highly variable. This highlights the fact that tighter regulation for private well construction needs to be introduced, including post-construction sign-off by a regulatory body. This research strengthens recommendations from previous research projects that alternative treatment systems/approaches are required in low-permeability areas to prevent contamination of surface waters.

Developing Solutions

The research also focused on the use of specific chemical, biochemical and microbiological compounds as tracers of DWWTS effluent contamination in both private wells and rivers, leading to the development of appropriate strategies for fingerprinting sampling of both surface water and wells. Fingerprinting methods evaluated include ionic ratios (potassium/sodium and chloride/bromide), fluorescent whitening compounds, faecal sterol profiles, artificial sweeteners, caffeine, pharmaceuticals (carbamazepine and sulfamethoxazole) and human-specific Bacteroidales 16S rRNA genetic markers. The performance of each fingerprinting technique as well as the resources (and hence costs) required for their application varied considerably. Three methods/compounds showed promise in identifying and tracing sources of well contamination. For surface water studies, a recommended tiered approach is proposed based on short-term, low-flow monitoring, starting with the more common analytical techniques for E. coli and ammonia, but then moving to more confirmatory tests for fluorescent whitening compounds, followed by faecal sterols and then artificial sweeteners, caffeine and pharmaceuticals if needed.

EPA Research: McCumiskey House,
Richiew, Clonskeagh, Dublin 14.

Phone: 01 268 0100
Twitter: @EPAResearchNews
Email: research@epa.ie

www.epa.ie

EPA Research Webpages
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