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Assessment of disposal options for treated waste water from single houses in low permeability subsoils

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Environment, Community and Local Government

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Assessment of disposal options for treated wastewater from single houses in low-permeability subsoils

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EPA Research Report

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DISCLAIMER

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Executive summary

This research project focused on two areas related to on-site wastewater treatment: (1) the performance of existing soakaway systems in a range of different soil permeability settings; and (2) solutions for the treatment and discharge of on-site domestic wastewater in low-permeability soils. Six existing soakaway systems, all more than 20 years old, were instrumented and monitored with respect to the fate and transport of both chemical and microbiological pollutants. This showed that existing soakaway systems in low-permeability soils are likely to be causing shallow lateral flow of effluent into the nearest surface depression, promoting a risk of surface water pollution. However, the natural attenuation of bacteria and phosphorus in such lateral flow pathways does seem to be significant. In higher permeability soils, the effluent from soakaway systems is more likely to percolate downwards and cause a risk of localised groundwater pollution if the water table is shallow. However, no negative influences were detected at any of the on-site wells in deeper groundwater at those sites. In addition, the use of microbial source tracking methods, using human-specific Bacteriodales bacteria, was tested during these trials and showed promise.

The two sites located on low-permeability soil were upgraded by installing alternative pressure-dosed distribution [low-pressure pipe (LPP) and drip dispersal (DD)] systems. This resulted in a decrease in the faecal contamination of groundwater, as well as the prevention of surface ponding of effluent, at both sites. Furthermore, the field results and calibrated models of the unsaturated zone show that the LPP system could be a solution for sites with T-values of less than 90, and the DD system could be a solution for sites with T-values of less than 120 after secondary treatment.

A series of field trials were also carried out on evapotranspiration systems using willow trees, with the aim of creating a zero-discharge solution for areas of low-permeability soils. However, the results showed that such systems are unlikely to be able to act as completely zero-discharge systems in the Irish climate if the *in situ* low-permeability subsoil is used as a backfill. The systems were, however, shown to promote excellent pollutant attenuation and significantly reduce net effluent discharge to the environment, so should be considered as a viable passive treatment for either existing (legacy) and/or new developments. It should be noted that this would require a change in current consent procedures to allow for such a controlled discharge to surface water.

A decision support tool was also developed, on the basis of geospatial modelling, to identify possible solutions to the discharge of on-site effluent in low-permeability areas. This tool, which could be used for strategic assessment at a local-authority level, has shown that the concept of clustered decentralised systems could target a significant proportion of potentially poor sites in low-permeability areas and could lower the burden of monitoring associated with discharge consents. Furthermore, analyses indicate that such systems could be economically favourable compared with single-house systems. Hence, this option should be investigated and developed further from a technical, social, economic and legal (e.g. ownership, liability, etc.) perspective.

The research has also highlighted that surface water discharge will need to be reconsidered in areas where discharge to ground is simply not possible. Most packaged wastewater treatment plants, if correctly installed and regularly maintained, are able to achieve effluent concentrations lower than the usual minimum required discharge limits. For discharge to nutrient-sensitive areas, however, improvements in the removal of total nitrogen and total phosphorus will be needed. New technologies in this area should be as passive and low maintenance as possible. In addition, the use of water-saving devices can increase the feasibility and sustainability of most effluent disposal options and should be considered where possible.

1 Introduction

The domestic wastewater of approximately one-third of the population in Ireland (≈ 500,000 dwellings) is treated on site by domestic wastewater treatment systems (DWWTSs), of which more than 87% are septic tanks (CSO, 2011). It is estimated that the overall proportion of the country with inadequate conditions for DWWTSs, that can arise all year round or intermittently during wet weather conditions, is 39% (EPA, 2013). If situated and constructed incorrectly, the potential impacts of such on-site effluent treatment systems include the pollution of groundwater and/or surface water. In particular, areas with (1) inadequate percolation because of low-permeability subsoils and/or (2) insufficient attenuation because of high water tables and shallow subsoils present the greatest challenge in Ireland for dealing with effluent from DWWTSs. If there is insufficient permeability in the subsoil to take the effluent load, ponding and breakout of untreated or partially treated effluent at the surface may occur and this is associated with serious health risks. There will also be a risk of effluent discharge/runoff of pollutants to surface waters and to wells that lack proper headworks or sanitary grout seals (Hynds et al., 2012). The nutrient load in the effluent (either as direct discharge to surface water or via the groundwater pathway) can contribute to eutrophication in sensitive water bodies, whilst contamination of water sources by human enteric pathogens can promote the outbreak of disease. Alternatively, if (1) the permeability of the subsoil is excessive, (2) the effluent loading on the subsoil is too high or (3) there is an insufficient depth of unsaturated subsoil (e.g. a high water table or shallow bedrock), then the groundwater beneath a percolation area is at risk of pollution, in particular from microbiological pathogens and/or nutrients.

The specification (EPA, 2009) of a lower limit to subsoil permeability (defined according to the on-site percolation T-test at T = 90) for effluent discharge to ground, in conjunction with surface water discharges generally not being licensed for one-off housing, will probably mean that many areas will be deemed unsuitable for single house development. To address these problematic areas and allow development, while protecting water resources from the risk of pollution by existing septic tanks in these areas, the so-called legacy sites that are now starting to be assessed under the National Inspection Plan, alternative wastewater treatment and disposal options are needed. Hence, the aims of this project were to (1) assess the performance of and the risk of pollution from existing septic tank soakaway systems in a range of subsoil permeabilities and (2) identify alternative disposal options and investigate their suitability for areas of low-permeability subsoils. The options considered and assessed by the project in a series of field studies were (1) pressurised distribution systems and (2) sealed basin evapotranspiration (ET) systems. The results obtained from these trials have been used to propose design criteria and to determine the operating limits for these systems, in an Irish context, for consideration by the Department of the Environment, Community and Local Government (DECLG)/Environmental Protection Agency (EPA). Other possible effluent disposal options were investigated strategically using Geographic Information Systems (GISs) with collated information to assess their feasibility and overall sustainability in areas of inadequate percolation. This information was also used to develop a web-based GIS decision support toolset that will allow environmental planners and managers to evaluate alternative strategies from both cost–benefit and environmental sustainability perspectives.

2 Field studies

2.1 Impact of traditional septic tank soakaway systems on water quality

Six different single houses with existing on-site septic tanks discharging into soakaways (all more than 20 years old) were chosen after an extensive site assessment process across a range of different subsoil permeabilities and types (Table 2.1). Soil moisture and groundwater instrumentation were installed at each site; each site was then monitored over the course of a year to delineate the effluent plume and to determine the pollutant attenuation within the plumes en route to the receiving groundwater and surface water.

As well as the chemical analyses of the soil moisture and groundwater samples for organics and nutrients, the presence of faecal contamination was determined. This was done, first, by testing samples for faecal indicators (i.e. total coliforms and *Escherichia coli*) using culture-based techniques. The presence of coliforms was then confirmed by molecular methods targeting host-specific Bacteroidales bacteria. In addition, the potential for nitrate removal was evaluated by targeting the microbial community capable of carrying out denitrification [the microbial reduction of nitrate to nitrogen (N) gas]. The indicator targets employed in this study are outlined in Table 2.2.

2.1.1 Site A (County Kilkenny)

Suction lysimeters were installed at a range of depths, as shown in Figure 2.1, to sample the downstream soil moisture in the vicinity of the soakaway system on this low-permeability soil every 2–3 weeks. Although there was no evidence of surface ponding, the results

Table 2.1. Characteristics of soakaway sites across a range of subsoil permeabilities

Site	No of residents	K_{fs}[a] (m/d)	Permeability classification	Subsoil classification[b]	Groundwater vulnerability[c]	Years since desludge
A	3	0.059 (T = 75)	Low	Sandy silt/clay	Moderate	2
B	4	0.061 (T = 73)	Low	Silt/clay	Moderate	10
C	3	0.21 (T = 21)	Moderate	Gravelly sand and silt/clay	High	2–3
D	3	0.37 (T = 12)	Moderate	Gravelly sand	High	2–3
E	2/3	1.31 (T = 3.2)	High	Gravelly sand	High	3
F	5	0.56 (T = 8)	High	Silty sand	Extreme/high	> 10

[a]**Field-saturated hydraulic conductivity.**
[b]**Carried out in accordance with BS 5930 (British Standards Institution, 1999).**
[c]**DELG/EPA/GSI (1999).**

Table 2.2. Faecal indicator (traditional and host-specific) and N-cycle functional gene targets employed

Indicator	Reference	Why use it?
Total coliforms	IDEXX Laboratories	Faecal indicator organism
E. coli	IDEXX Laboratories	Faecal indicator organism
Human-specific Bacteroidales (BacHum)	Kildare et al. (2007)	Indicates human faecal source
Bovine-specific Bacteroidales (BacBov)	Kildare et al. (2007)	Indicates bovine faecal source
nosZ	Henry et al. (2006)	Denitrification potential

Note: further information relating to these methods is available in Kilroy et al., 2014.
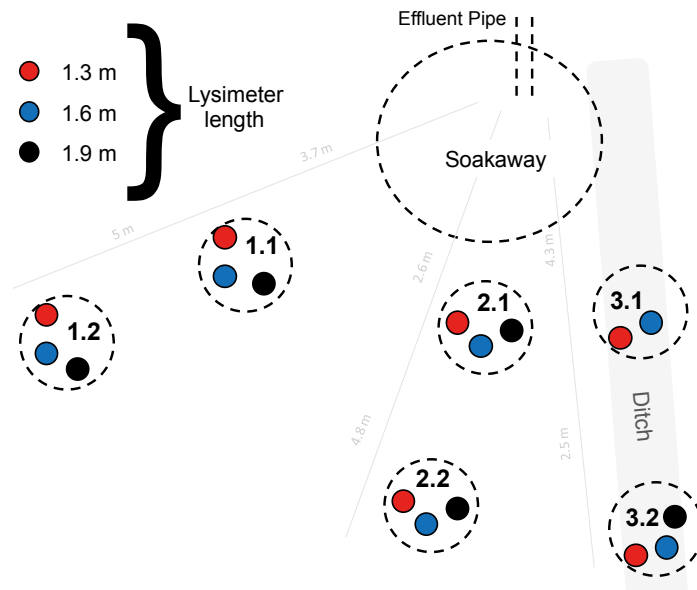
Figure 2.1. Plan of instrumentation at Site A (pre-remediation).

of monitoring over 12 months showed that the effluent appeared to be moving laterally to the adjacent ditch. This was picked up particularly during wet weather conditions in lysimeter nests 3.1 and 3.2. However, there was also evidence of downward percolation of effluent from other lysimeters, particularly in nest 2.1, during drier periods.

It was not possible to quantify the fraction of effluent moving laterally compared with that percolating vertically. For the effluent percolating downwards and sampled in the suction lysimeters, there was good overall removal of the *E. coli* indicator bacteria (2.9 log average), as well as efficient phosphorus (P) removal (>93%) by the biomat and/or shallow subsoil. The ammonia in the effluent was converted to nitrate with depth, although during periods of heavy rainfall in winter this process was supressed. Moreover, with respect to the lateral migration of the effluent, the results from the suction lysimeter samplers installed in the ditch indicated significant attenuation of indicator bacteria, P and N over a relatively short distance (<20 m).

At this site, no correlation was observed between the *E. coli* and BacHum results in the soil moisture samples. The lateral movement of effluent, however, was supported by the results obtained from groundwater sampled in piezometers 20 m downstream of the soakaway system, at depths of 3 to 4 m; the origins of total coliforms and *E. coli* in these groundwater samples were confirmed by molecular analysis. The presence of livestock at this site during the sampling period

may have contributed to the *E. coli* levels found in the groundwater samples, as suggested by the microbial source tracking analysis, which revealed the presence of the bovine-specific Bacteroidales bacteria (BacBov) target in the upstream groundwater sampling piezometer (again at depths of 3 to 4 m) and confirmed the origin of that faecal signal (Figure 2.2).

At the end of the period of monitoring the soakaway, the site was excavated to install a packaged treatment plant, as well as alternative effluent distribution systems, as discussed in Section 2.2. This excavation revealed a flooded pool of effluent in the soakaway approximately 300 mm below ground level (Figure 2.3), which gives a good visualisation of why lateral migration into the adjacent ditch was occurring.

2.1.2 Site B (County Monaghan)

The soakaway system on this site had a continual presence of standing water at ground level, as well as a distinctive variation in grass growth in the soakpit area and along the length of a hollow depression (Figure 2.4). This was assumed to be indicative of lateral effluent migration along the hollow. Lysimeters were installed around the soakaway system and additional piezometers were installed along the hollow, up to a distance of approximately 50 m away.

Again (as per Site A) the results from the lysimeter samples indicated significant lateral effluent movement along the field in the slight hollow under wet weather
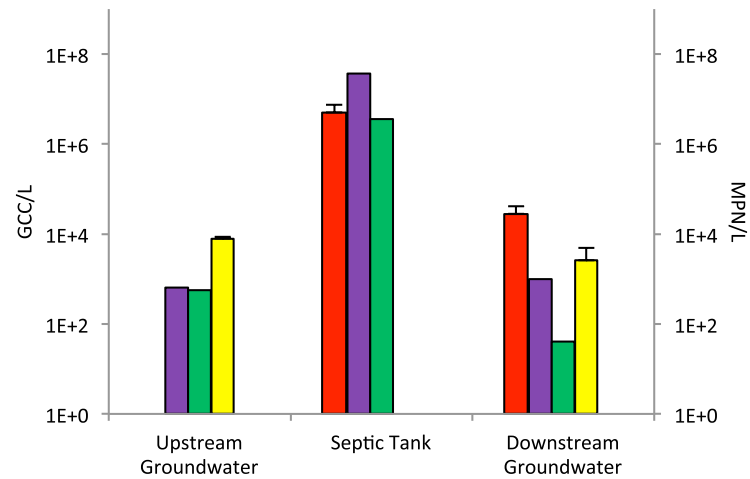
Figure 2.2. Faecal indicator gene copy concentrations, in gene copies per litre (GC/l), and most probable number (MPN/l) determined from septic tank effluent and groundwater samples in Site A pre-remediation. BacHum (red), BacBov (yellow), total coliforms (purple), *E. coli* (green).

Figure 2.3. Site A soakaway being excavated and lysimeters in adjacent ditch.

Figure 2.4. Site B soakaway location and direction of shallow lateral effluent flow in field.

conditions, as well as some percolation downwards at the lysimeters surrounding the soakaway. In these lysimeter nests, there generally seemed to be good P removal and good removal of *E. coli*, but one or two incidences of high *E. coli* breakthrough with depth indicated a potential threat to groundwater. Again, it was not possible to determine how much of the effluent was percolating versus how much was flowing laterally across the field at a shallow depth. The analysis of samples in the shallow effluent pathway (lysimeter S3.2B to downstream boreholes D/S B1 and B2) showed a slower attenuation of bacterial indicators (Figure 2.5) but good attenuation (>97%) of P within 40 m. This indicates a potential microbial threat to surface water, but less of a nutrient threat.

A positive correlation was observed between the *E. coli* and BacHum results from the lysimeters installed in the plume at shallow depths, which corroborated the other chemical and microbiological analysis, that is that the effluent was flowing laterally in such low-permeability subsoil. The lateral movement of effluent was also supported by the results from groundwater sampled downstream of the soakaway systems, in which the origins of total coliforms and *E. coli* were confirmed by the molecular analysis (Figure 2.6). The presence of the BacHum target suggested that the majority of faecal contamination detected in the groundwater downstream

of the soakaway systems at Sites A and B was from a human source.

2.1.3   Site C (County Meath)

This site had a stepped surface profile with moderate permeability subsoil and a shallow water table approximately 1.0–1.2 m below ground level B (Figure 2.7), regulated by the water level of a downstream river approximately 25 m from the soakaway. The soil stratification at the soakaway, under the topsoil, comprised three distinct soil horizons: a well-drained gravelly sand layer (0–1.4 m), a silt/clay layer (1.4– 1.8 m) and a further gravelly sand layer (>1.8 m).

The analysis of chloride concentrations across all sampling positions indicated the presence of the effluent plume at all depths, but with evidence of significant dilution once in the downstream groundwater, particularly beneath the silt/clay layer in lower saturated zone B.

Overall, the chemical results suggest a predominantly vertical plume in the free-draining upper horizon above the water table, promoting a highly efficient removal of organics and nitrification of ammonia, as well as the removal of indicator bacteria and P. A limited extension of the plume in a lateral direction along the gradient of the site was also apparent. However, some high concentrations of indicator bacteria (up to 6.6×10^3/100 ml)
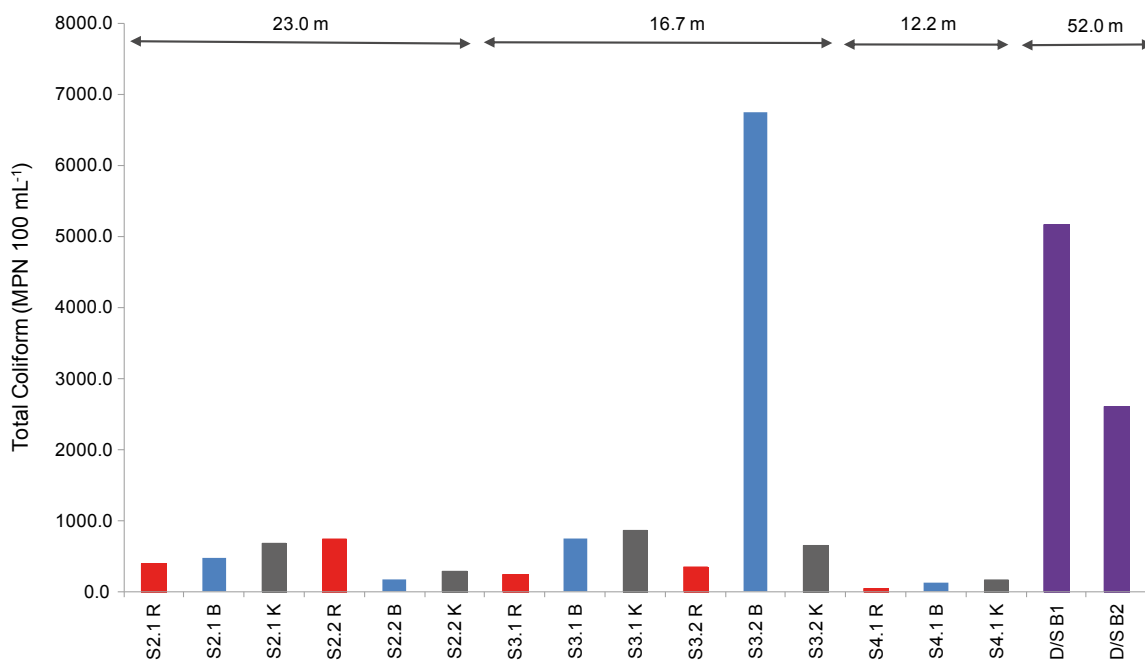
Figure 2.5. Elevated mean total coliform concentrations [most probable number (MPN)/100 ml] at lysimeters (red = 1.3 m depth, blue = 1.6 m depth, black = 1.9 m depth) and downstream boreholes D/S B1 and B2 (purple).
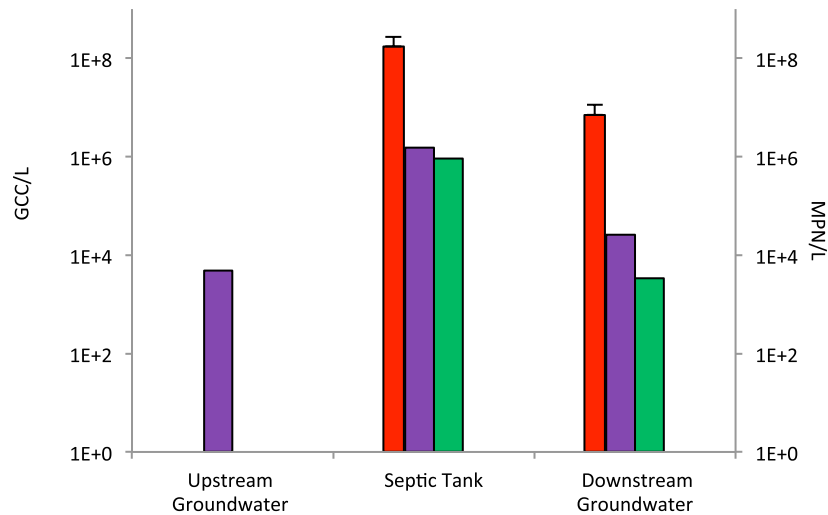
Figure 2.6. Faecal indicator gene copy concentrations in gene copies per litre (GCC/l) and most probable number (MPN/l) determined from effluent and groundwater samples in Site B [BacHum (red), BacBov (yellow), total coliforms (purple), *E. coli* (green)].
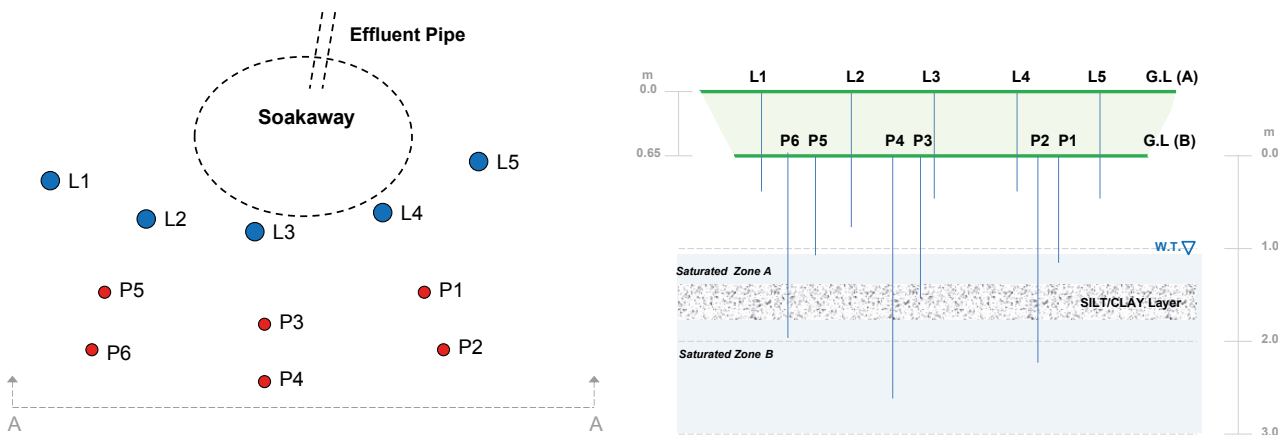
Figure 2.7. Plan view and cross-section of installed instrumentation at Site C (lysimeters: L1 to L5 lysimeters, piezometers: P1 to P6).

and P (up to 6 mg/l) were also found in underlying saturated zone A of this upper gravelly sand layer, indicating the possible presence of preferential flow paths and/or the direct ingress of effluent from the base of the soakaway directly into this zone. Lower down in saturated zone B, there were very low concentrations of nitrate in dry summer conditions (as shown in Figure 2.8) compared with the elevated concentrations in the unsaturated zone, but with more evidence of total N (TN) removal during periods of high soil moisture in the wetter autumn/winter period; this is believed to be the result of the combined effect of denitrification and dilution in the saturated conditions of the phreatic zone. In addition, the fluctuating water table resulted in a reduced attenuation capacity in the unsaturated zone

during periods of increased rainfall recharge, and direct effluent transport to the saturated zone was observed.

Overall, this shows the risks associated with such soakaway systems, particularly in shallow groundwater scenarios, and indicates that the soakaway system is inadequate and does not provide sufficient protection to the underlying groundwater.

2.1.4   Site D (County Meath)

This was another moderate subsoil permeability site located only 2 km from Site C but with different subsoil conditions and a much lower water table. This site also had the greywater (bath, shower, washing machine and kitchen sink water) being discharged to one soakaway
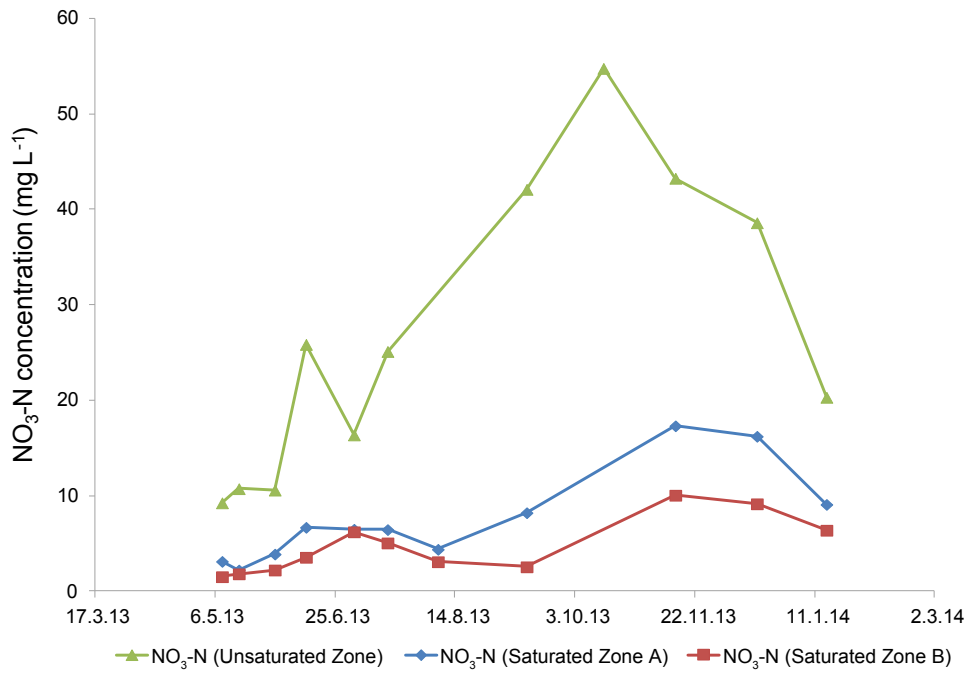
Figure 2.8. Time series plot of nitrogen, present as NO$_3$ (NO$_3$-N), concentrations within the unsaturated and saturated zones at Site C.

and the toilet water being discharged to an adjacent soakaway, as shown in Figure 2.9.

Despite the free-draining conditions observed at the site, an initial hand auger test in the area of the grey-water soakaway found that water levels around this soakaway were approximately 0.8 m below ground level A, even though piezometer P6 at this location revealed unsaturated conditions to a depth of more than 2.6 m; this suggests that the continuous accumulation of

solids and biological clogging of the infiltrative surface had resulted in ponding of the effluent. The analysis of mean chloride concentrations recorded at the site, as per Site C, showed the presence of the effluent plume at all depths.

High levels of nitrification were shown to be taking place in the unsaturated zone, with 98.2% of the remaining inorganic N of soil moisture samples being in the NO$_3$ form (NO$_3$-N) at depths below and outside the base of
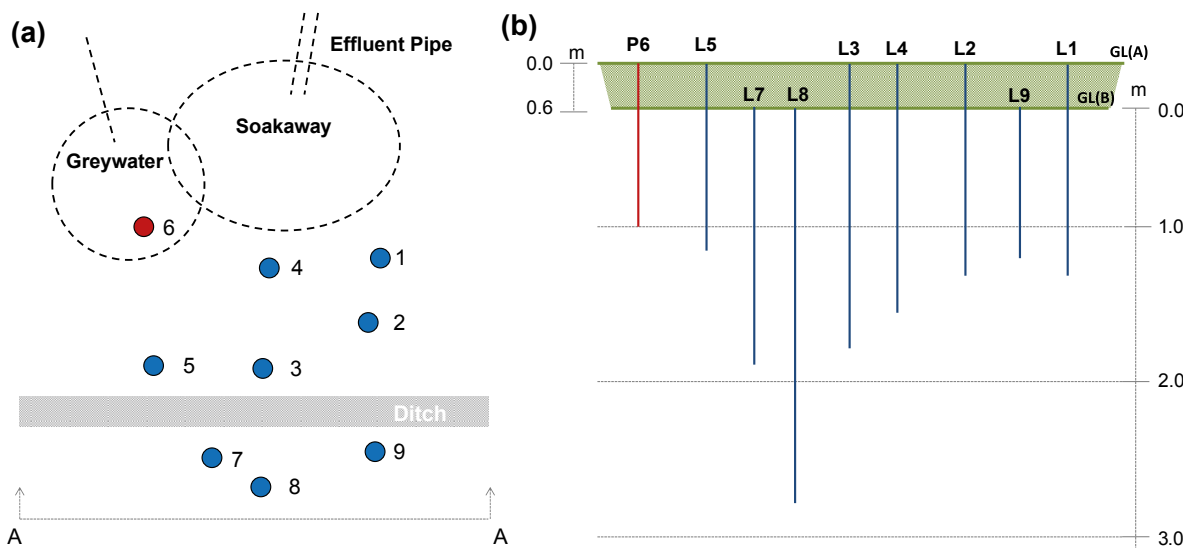
Figure 2.9. Plan view (a) and cross-section (b) of installed instrumentation at Site D (lysimeters: L1 to L9, piezometer: P6).

the septic tank soakaway. Equally, elevated NO_3-N concentrations downstream of both the septic tank effluent (STE) and greywater soakaways indicated a degree of lateral movement of the plume along the site's gradient. A considerable proportion (>97%) of PO_4-P (phosphorus in the PO_4 form) in the percolating effluent was also removed within the unsaturated subsoil.

During the course of the sampling period, *E. coli* concentrations in excess of a most probable number (MPN) of 1000/100 ml were recorded in downstream soil moisture lysimeter samples on six occasions (see Table 2.3). Of these six recorded breakthroughs, two were recorded at sample position L2, two at sample position L7 and two at sample position L8. Breakthroughs of this magnitude are of particular concern as L7 and L8 were the deepest sampling points installed at the site, at 1.85 m and 2.6 m below ground level B, respectively. Despite this concern, groundwater samples taken from a downstream well approximately 30 m from the soakaway showed no presence of *E. coli* throughout the sampling period. Nevertheless, results from the study suggest that the soakaway system in operation at Site D does not provide sufficient protection to underlying groundwater supplies.

2.1.5 Site E (County Westmeath)

Soil moisture lysimeters were installed at a variety of depths downstream of the soakaway in high permeability subsoil on the side of a relatively steep hill, as shown in Figure 2.10. The well used for supplying water to the house was located directly downstream of the septic tank discharge point, and this well was also sampled regularly throughout the monitoring period.

It was apparent from the chloride concentrations in the lysimeters that effluent being discharged to the subsoil moved rapidly in a predominantly vertical direction following the overall gradient of the site. There was little evidence of the effluent plume extending laterally (apart from along the topographic gradient), consistent with the free-draining subsoil characteristics.

Despite the high hydraulic conductivity of the site, most of the organics and P in the effluent were reduced in the biologically active zone within the soakpit, which develops at the infiltrative surface of the wastewater effluent discharge. As with the previously described moderate permeability sites (Sites C and D), the unsaturated conditions at Site E gave rise to high levels of nitrification, which occurred rapidly within the receiving

Table 2.3. Measured concentrations of *E. coli* across Site D

	No of samples	No of samples with concentration (MPN/100 ml)			
		<10	10–100	100–1000	>1000
Lysimeters	30	3	13	8	6
Piezometer	6	0	0	2	4
Groundwater (on-site well)	7	7	0	0	0
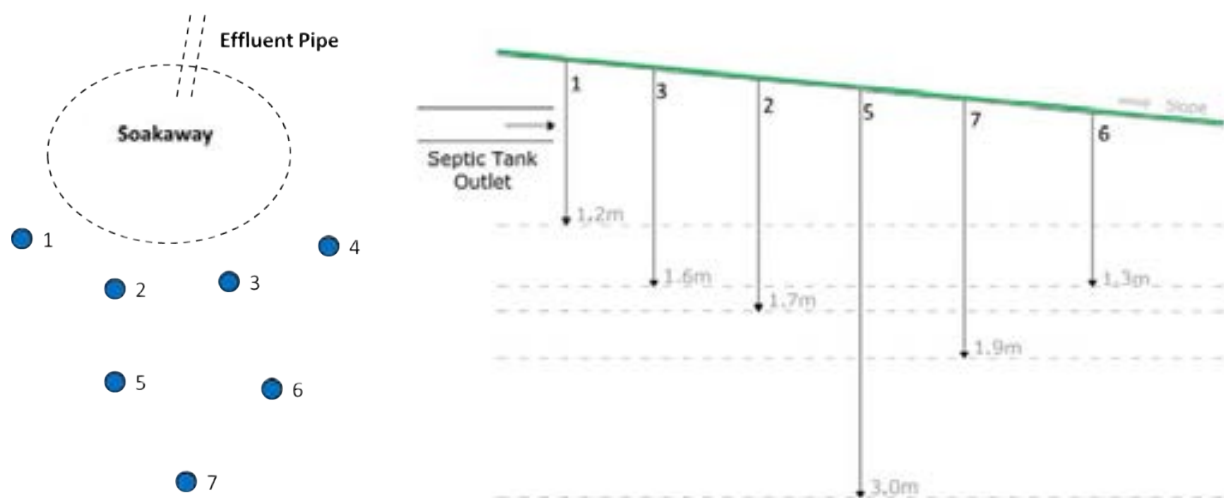
Figure 2.10. Plan view and cross-section of installed lysimeters (1 to 7) at Site E.
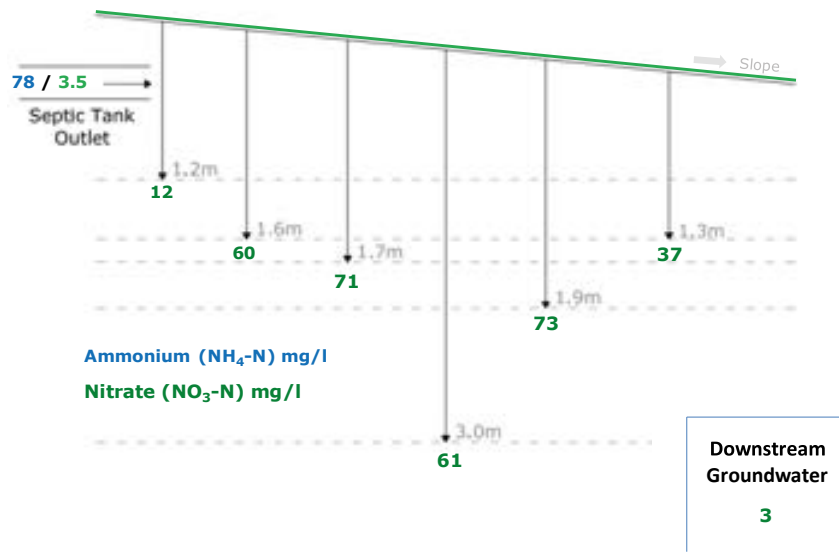
Figure 2.11. Cross-section of mean NO$_3$-N concentrations in relation to subsoil depth.

subsoil, with 98% of the inorganic N present in the soil moisture lysimeter samples as NO$_3$-N (see Figure 2.11). However, only 35% of the overall total inorganic N was removed within the upper subsoil horizons monitored during the study (as a result of a limited availability of saturated micro-sites and thus limited denitrification potential). As such, inorganic N remains in the system as NO$_3$-N, and this is likely to reach the groundwater eventually.

Analysis of the bacteriological results from Site E showed a mean 3.3 log reduction of total coliforms within soil moisture lysimeter samples beneath the infiltrative surface. Despite this, significant concentrations of *E. coli*, exceeding 1000 MPN/100 ml, were observed during the study at the 3.0 m sample depth on two occasions. This is of concern given that the depth to bedrock at Site E is only approximately 4.0 m. As with

the previous moderate permeability sites, no adverse effect on groundwater quality was detected with no *E. coli*, for example, detected throughout the monitoring period; however, the results highlight the vulnerability and substantial threat to groundwater quality, which could pose a serious health risk to drinking water quality in the close vicinity of the site.

2.1.6   Site F (County Cork)

Again, this site was also located on high permeability subsoils on a gently sloping hillside, similar to Site E, with a downstream borehole (not used directly for water supply). The locations of the soil moisture sampling lysimeters are shown in Figure 2.12.

The results provide evidence for an extension of the effluent plume laterally to lysimeter 7, along the gradient
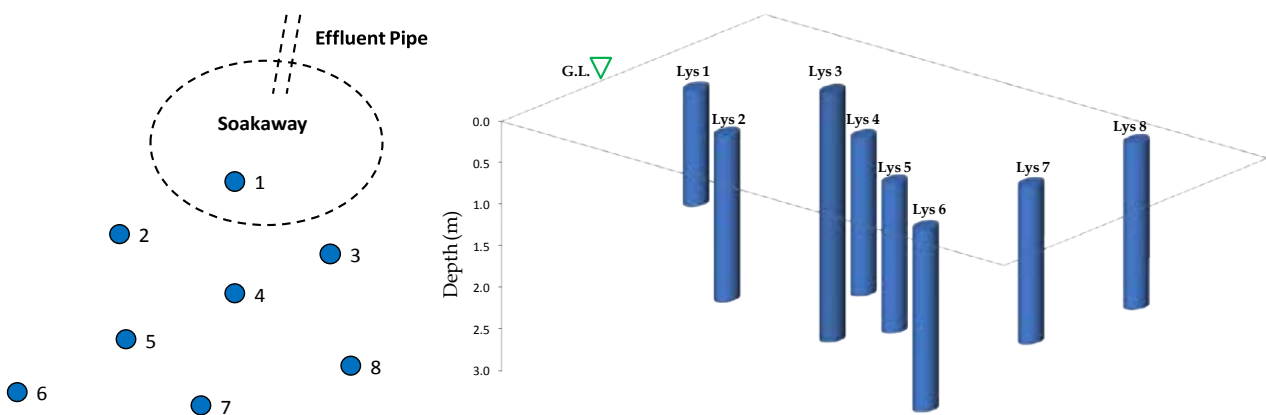
Figure 2.12. Plan and three-dimensional view of installed instrumentation at Site F.

Figure 2.13. Evidence of organic layer (biomat) at Site F from augured core during lysimeter 3 installation.

of the overall site. This is presumably because of the formation of a lower permeability biozone along the gradient of the site. Indeed, a more organic-rich layer was seen at a depth of 0.8 m when auguring holes during the installation of the instrumentation (Figure 2.13). Again, as with the previous higher permeability sites, rapid nitrification of the STE was seen to occur within the subsoil. However, as with the previous sites, the free-draining conditions at Site F reduce the potential for denitrification and thus NO_3-N remains in the subsoil until eventual recharge to groundwater. The apparent high P removal (with subsoil lysimeter values of, on average, 0.5 mg/l at depth) was more effective at this site than at Site E because of the slightly higher levels of clay present in the subsoil.

E. coli concentrations up to 90 MPN/100 ml were recorded within the soil moisture lysimeter samples at Site F, as shown in Figure 2.14. As with Site E, the shallow free-draining subsoil gives rise to concern, particularly in relation to enteric bacteria. However, as with the results outlined previously for Sites C, D and E, despite the high permeability subsoil, the groundwater quality sampled from the downstream well location showed no adverse response to the effluent discharge upstream.

2.2 Performance of pressurised effluent distribution systems

2.2.1 *Site construction and instrumentation*

Low-permeability Sites A and B in Kilkenny and Monaghan were upgraded with pressurised distribution systems and packaged secondary treatment plants. On each site, the effluent was split into two, with half being diverted to a low-pressure pipe (LPP) system and the other half to a drip dispersal (DD) system. The DD systems were designed to have a nominal loading rate of 3 l/m² per day and used 12.7-mm diameter tubing (Geoflow, Corte Madera, CA) with the drippers 600 mm
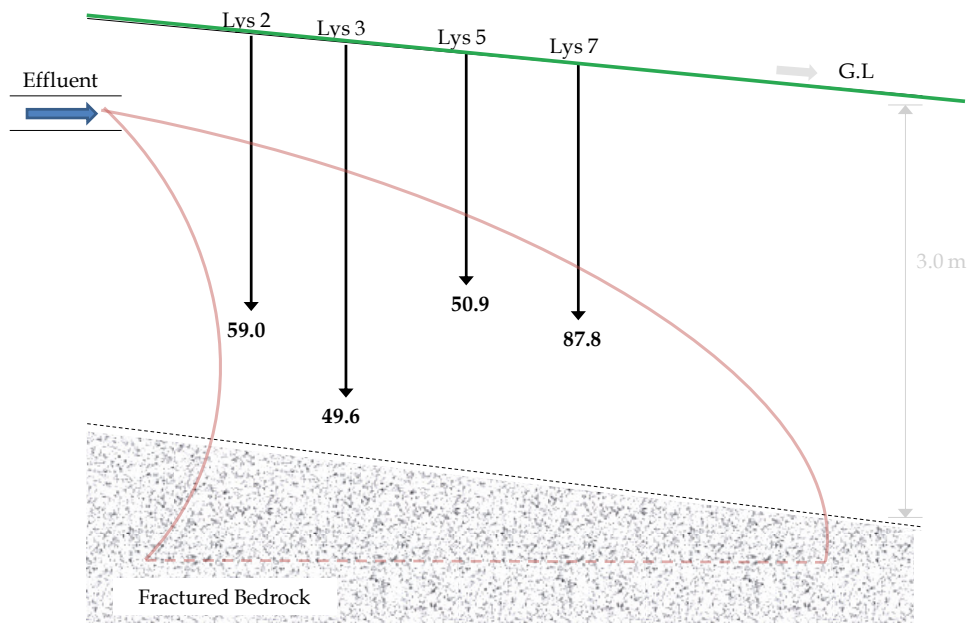
Figure 2.14. Cross-section of geometric mean concentrations of *E. coli* (MPN/100 ml) at selected locations at Site F. Lys, lysimeter.

Figure 2.15. (a) Installation of DD tubing, (b) final DD installation at Site B and (c) installation of LPP at Site A.

apart. The pipes were installed by hand at a shallow depth, of approximately 150 mm, in the root zone (see Figure 2.15). The pumping into the DD system was on a timed basis at four doses per day under normal conditions. The LPP system had 9-m-long trenches that were 300 mm wide at a depth of 400 mm and were filled with washed pea gravel and designed to have a loading rate of 3 l/m^2 per day. Their spacing and layout is shown in Figure 2.17. In the LPP system, the effluent was pumped through 25-mm diameter pipes on a volumetric basis, at approximately 90 l per pumping event discharging through 5.0-mm diameter holes, spaced at 0.95-m intervals.

Site A had an extended aeration activated sludge plant installed, but effluent from the septic tank was first dosed onto the DD and LPP systems, from August 2012, before changing over to the packaged plant on 1 October 2013. At Site B, a coconut husk media filter was added and the secondary treated effluent was divided equally and sent to the pressurised DD and LPP distribution systems from the time of start up (September 2012) (Figure 2.16).

The percolation areas were instrumented with (1) tensiometers and soil moisture probes to establish the temporal soil moisture conditions; (2) suction lysimeters for soil moisture samples for chemical and microbiological analysis; (3) full weather stations to determine effective rainfall at each site; (4) shallow groundwater monitoring boreholes upstream and downstream of the percolation areas; and (5) flow measuring devices to measure the effluent loading (Figure 2.17). The energy usage by the pumps and aerators was also monitored on both sites. This showed an interesting difference between the fixed-film media filter process, which had an annual energy requirement for pumping the effluent up into the DD and LPP systems of 105 kWh/y, and the activated sludge reactor unit, which used 254 kWh/y for continuous aeration and a further 61 kWh/y for subsequent pumping into the DD and LPP systems.

2.2.2 Soil moisture results

The mean hydraulic loadings to the DD and LPP systems across the monitoring periods on both sites are given in Table 2.4. Mean effluent production on the sites

Figure 2.16. Packaged treatment plants: (a) extended aeration process at Site A and (b) media filter at Site B.
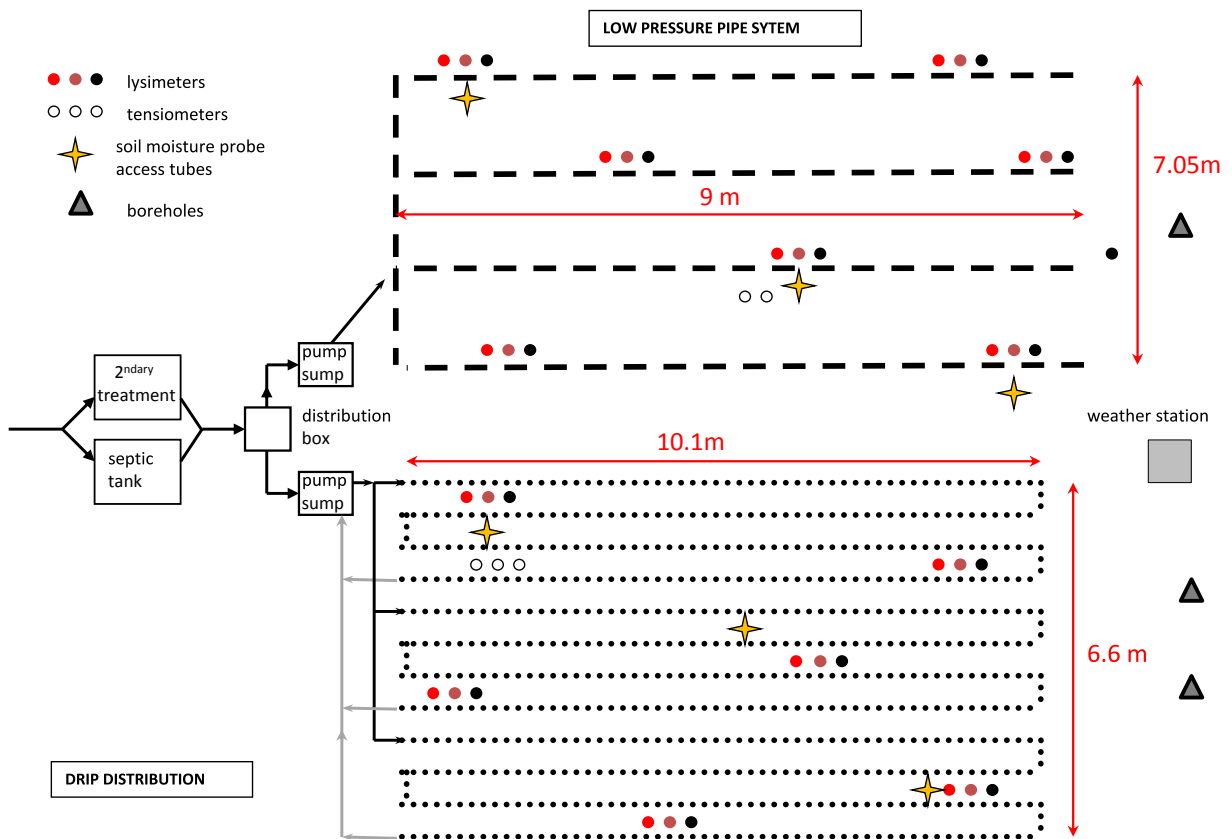
Figure 2.17. Site A (Kilkenny) set-up and instrumentation.

Table 2.4. Mean hydraulic loadings at Sites A and B

	Site A		Site B	
	DD	*LPP*	*DD*	*LPP*
Daily flow (l/d)	193.5	193.5	253.9	171.2
Hydraulic load (l/m²/d)	2.90	2.29	2.79	1.71
Hydraulic load (l/m²/d)[a]	–	18.0	–	13.4
Pumping frequency (per d)	4	2.05	4	2.50

[a]On trench base.

equated to 129.0 l per capita (l/c) per day on Site A and 104.1 l/c per day on Site B, both of which are lower than the EPA Code of Practice design figure of 150 l/c per day (EPA, 2009), but in line with several other detailed on-site studies of effluent production in Ireland (Dubber and Gill, 2014).

The soil moisture conditions beneath the different distribution systems at Site B over the monitoring period are shown in Figures 2.18 and 2.19, which reveal clear differences between the LPP and DD systems. Beneath the LPP trenches, there is a significant increase in soil moisture below the 400-mm-deep infiltration plane, whilst more uniform (and more unsaturated) soil moisture conditions were found beneath the DD systems, as corroborated by the modelling described later (in Section 2.2.4). The lower soil moisture levels beneath the DD systems were particularly evident in summer periods because of the higher ET losses from the shallow level into which the effluent is introduced. It should be noted that these same trends were also evident between the LPP and DD systems at Site A.
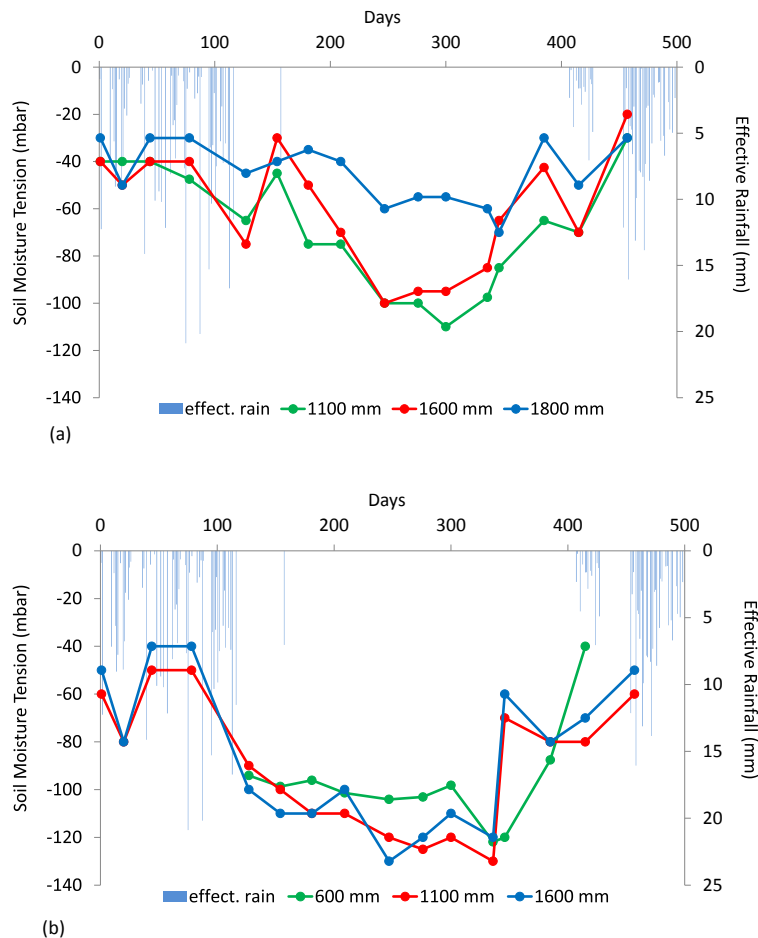
Figure 2.18. Soil moisture tension beneath (a) the LPP system and (b) the DD system at Site B.
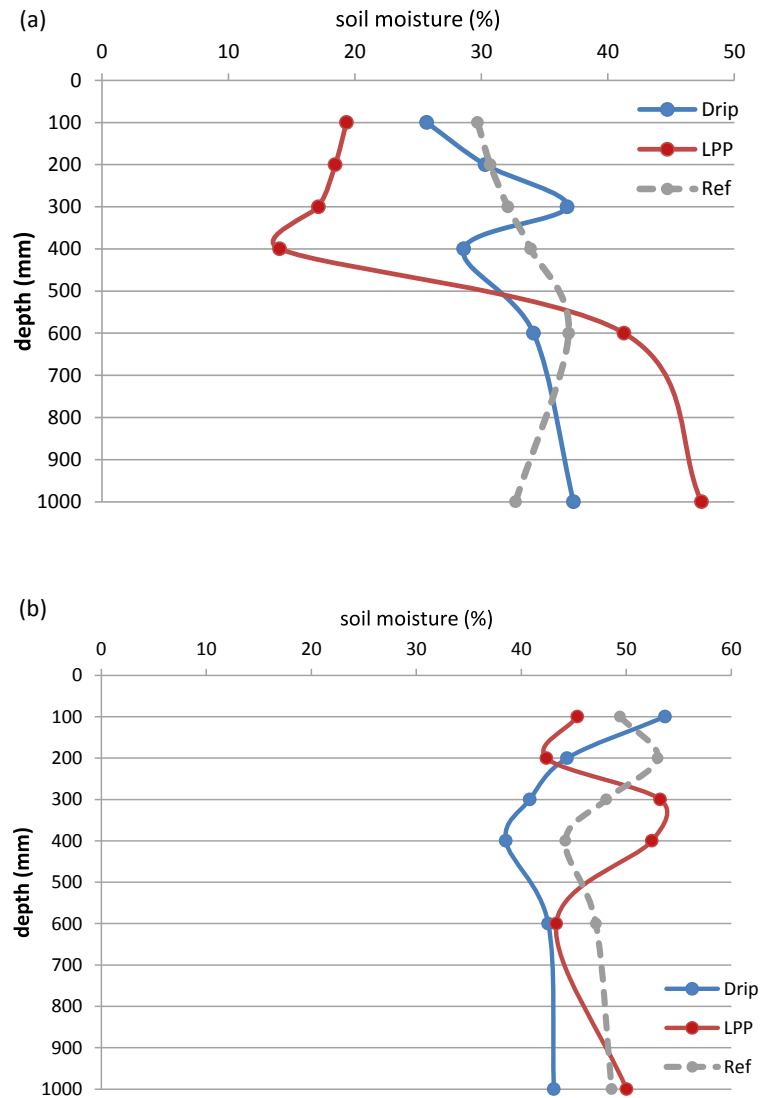
Figure 2.19. Site B soil moisture profiles beneath DD (drip) and LPP systems in (a) summer and (b) winter.

No surface water ponding was evident on either site above either distribution system throughout the monitoring period; however, the soil moisture results did show that the subsoil did become saturated at times in winter, particularly after heavy rainfall events. At both sites, overloading trials were carried out whereby, for 6-week periods, all of the effluent was diverted to either the DD system or the LPP system in order to effectively double the hydraulic loading. Although a mild effect was picked up by the soil moisture monitoring, the subsoil conditions were similar to the results under normal loading conditions and were more related to the meteorological conditions at that time of year.

2.2.3 Chemical and microbiological analysis

The analysis of chloride, as a tracer, in the soil moisture lysimeter samples from both sites revealed that the effluent under all four systems was fairly evenly distributed, as would be expected with such pressurised systems.

In general, the effluent quality below a 1-m depth of subsoil was very good beneath all systems. The N coming from both packaged treatment plants was almost totally nitrified when going into the soil and remained in nitrate form down through the subsoil with very little attenuation. In contrast, the soil moisture results at Site A when STE was being dosed did show nitrification and then some subsequent N removal (attributed to denitrification). This difference between the fate of secondary treated and STE in the subsoil concurred with previous results from gravity flow distribution systems on other sites in Ireland (Gill et al., 2009). Detection and quantification of the *nosZ* gene was carried out on soil sample extracts taken from underneath the distribution lines of

the LPP and DD systems at Site B. This aimed to determine whether or not denitrifying bacteria, which could potentially reduce nitrate to N gas, were present. The abundance of the *nosZ* gene was then correlated with nitrate concentrations retrieved from the soil moisture lysimeter samples; an inverse correlation was observed for the DD system (Figure 2.20), but not for the LPP system.

Both the LPP and DD systems in Site B had, on average, a similar level of *nosZ* abundance [3.81×10^6 gene copies (GC)/g for the LPP system and 4.60×10^6 GC/g for the DD system] (Figures 2.21a and b) indicating a similar potential for denitrification in each, when conditions are appropriate. Nitrate concentration gradients (Figures 2.21c and d) (and the correlation analysis), however, suggest that the DD system provided more
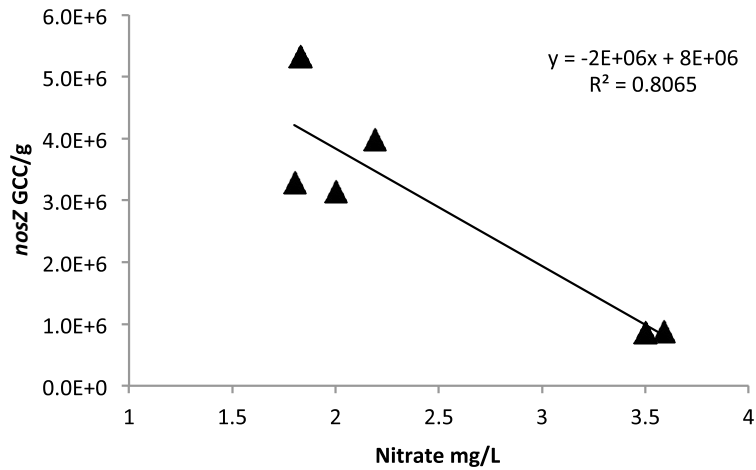
Figure 2.20. Scatter plot showing the inverse regression between lysimeter nitrate and *nosZ* gene copy soil extract concentrations (mg/l and GC/g, respectively) in samples collected from the DD system at Site A.
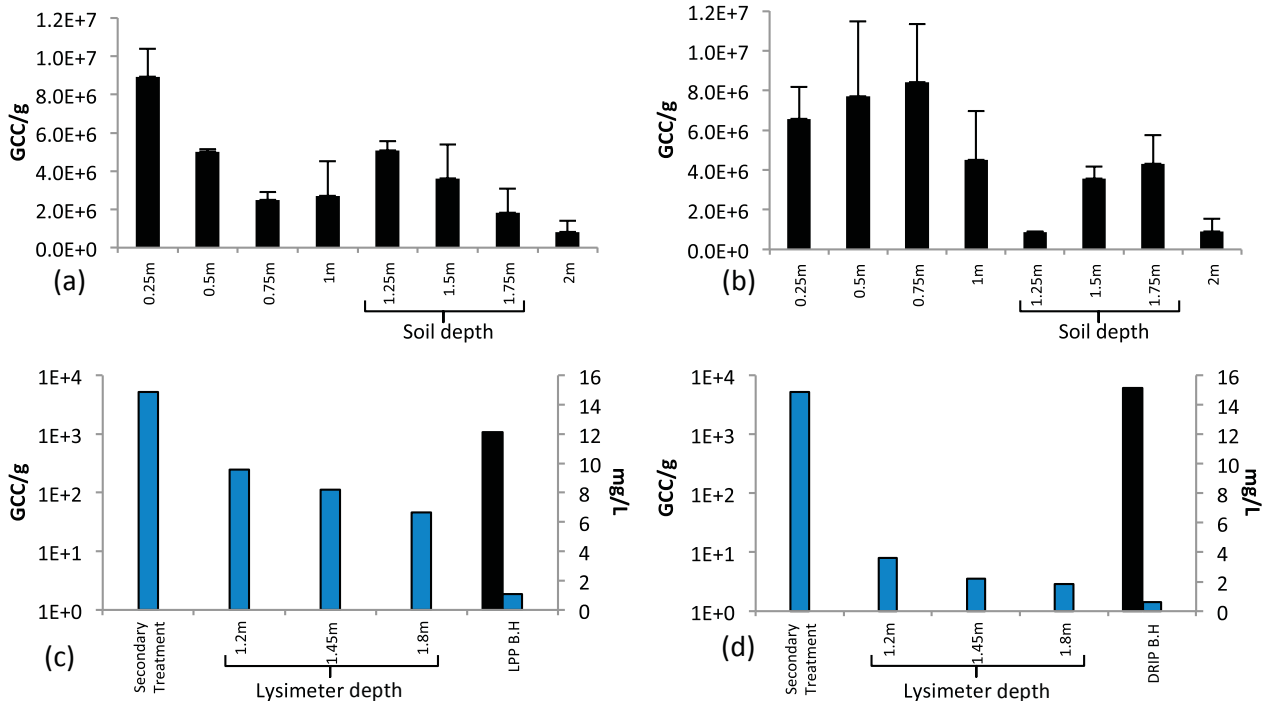
Figure 2.21. Soil, soil moisture and groundwater samples retrieved from under the LPP (left) and DD (right) distribution lines showing *nosZ* gene copy concentrations (GC/g of soil) (black) and nitrate levels in mg/l of soil moisture (blue). B.H., borehole.

favourable environmental conditions for actual denitrification to occur, and thus reduced nitrate losses to groundwater more effectively than the LPP system. Hence, in areas of particular nutrient concern, the DD system would be a preferable treatment solution than the LPP distribution system.

Phosphorus was almost totally removed in the lysimeter samples within the first 1 m of subsoil, beneath both systems on both sites. Equally, *E. coli* concentrations within these samples were around the detection limit (< 1 MPN/100 ml) at the lowest sampling plane beneath all sites (including throughout the time on Site A when the STE was being dosed into the distribution systems), with the exception of an isolated situation at one sampling point that was attributed to a preferential flow path during very wet conditions. In addition, a series of bacteriophage spiking trials were also carried out (using phages PR772, ΦX174 and MS2) to investigate the potential fate of viruses through the subsoil beneath these distribution systems, using these indicator phages. The lysimeter results showed excellent removal of all phages under all distribution systems.

The lysimeter sample results from overloading trials, whereby all the effluent was diverted to either the LPP or the DD for a period in order to effectively double the effluent hydraulic loading, revealed very little difference from the percolating effluent quality down through the subsoils under more normal effluent loading conditions.

It should be noted that, although the effluent quality at depth in the subsoil was very good when STE was being discharged, the DD system did need significant maintenance every 2 months to clear blocked filters, which was not the case once it had been switched to secondary treated effluent.

Site B samples showed evidence of human effluent still reaching the groundwater, downstream of the new systems, 2 months after installation (Figure 2.22a). Site B was receiving secondary treated effluent and, based on the available results, it appears that it may have taken longer to develop an effective biomat here than at Site A (Figure 2.22b and d), which was still receiving effluent from the existing septic tank post remediation. BacHum was not detected downstream of either the DD or the LPP system from Site A at any time post
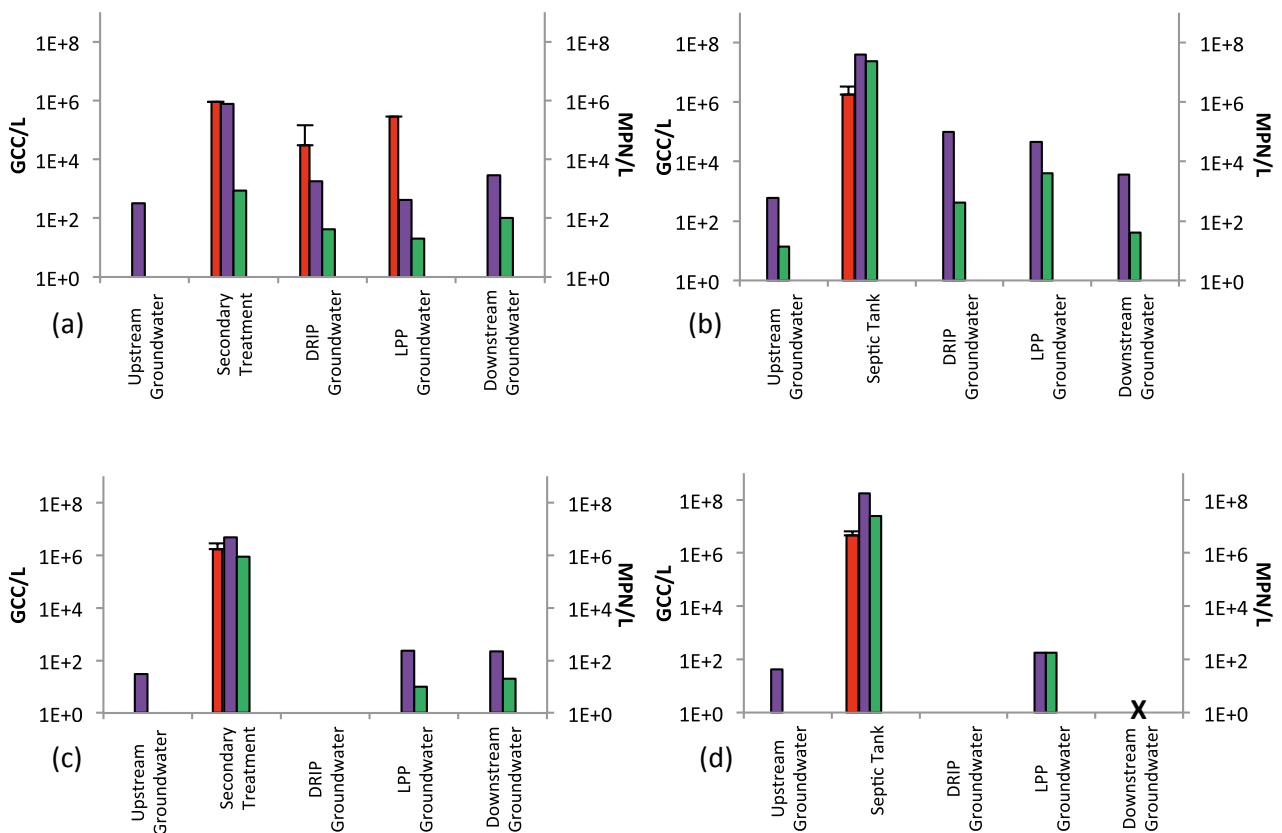
Figure 2.22. Faecal indicator concentrations determined from effluent and groundwater samples at Site B (left) and Site A (right) 2 months, (a) and (b), and 5 months, (c) and (d), post remediation. BacHum (red), total coliforms (purple), *E. coli* (green). X, no sample retrieved.

remediation, indicating a more rapid and effective development of biomat due to the high organic loading from the STE. Finally, 5 months after remediation, BacHum was no longer detected in the groundwater in the borehole downstream of either site (Figure 2.22c), while decreases in the total coliform and *E. coli* levels were also recorded, indicating a successful remediation.

2.2.4 Modelling

Hydrus 2D (PC-Progress, Prague, Czech Republic) was used to calibrate the soil moisture results from the two sites. Hydrus 2D solves the one-dimensional Richards' equation for time varying unsaturated flow. The van Genuchten (1980) equation is used to describe the pressure head versus water content relationships in the soil. Four separate models were built in two-dimensional cross-section through the DD and LPP systems at Sites A and B, according to the exact dimensions for each system constructed on site, information on soil texture and layering, effluent loading and meteorological conditions. Each model was then calibrated to the field data on soil moisture tension and moisture content at different depths and across the period of monitoring by changing the hydraulic conductivity (K_s). The model

simulations show that the subsoil conditions directly beneath the LPP trenches were much more saturated than below the drip irrigation pipes, as indicated from the field results particularly in summer conditions (Figure 2.23). However, as mentioned previously, there did not seem to be any significant difference with respect to treatment at depth.

Once satisfactorily calibrated, the models were rerun with reduced hydraulic conductivity values to try to establish minimum soil permeability values at which the systems would still be able to take the hydraulic loading without ponding at the surface or excessive saturation occurring. Examples of the simulated soil moisture conditions across the monitoring periods in the DD and LPP percolation areas at Site B are shown in Figure 2.24.

In parallel to this work, another research study has taken the field results from more than 800 falling-head percolation tests (T-tests) from across the country and modelled them using Hydrus 2D to indicate what the equivalent K_s values and soil textures would be. From this, a generic relationship between T-value [as used as part of the site assessment in Ireland (EPA, 2009)] and the more rigorous K_s value (as used in the modelling) was derived. However, because of the asymptotic nature of the T-value, the relationship is very sensitive

0.313 0.322 0.332 0.341 0.350 0.360 0.369 0.378 0.388 0.397 0.407 0.416

Water Content - [%] |, Min=0.313, Max=0.416
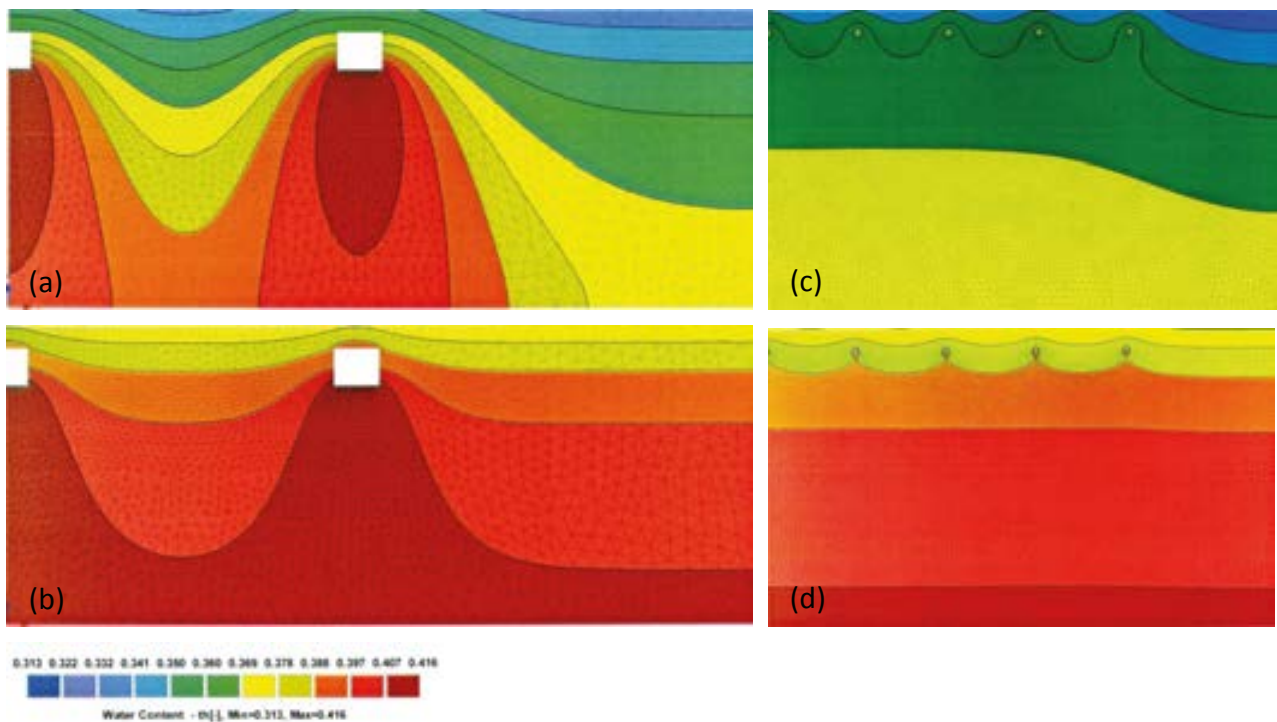
Figure 2.23. Hydrus 2D soil moisture simulations from calibrated 2-m depth cross-sections through (a) LPP distribution (summer), (b) LPP distribution (winter), (c) DD distribution (summer) and (d) DD distribution (winter).
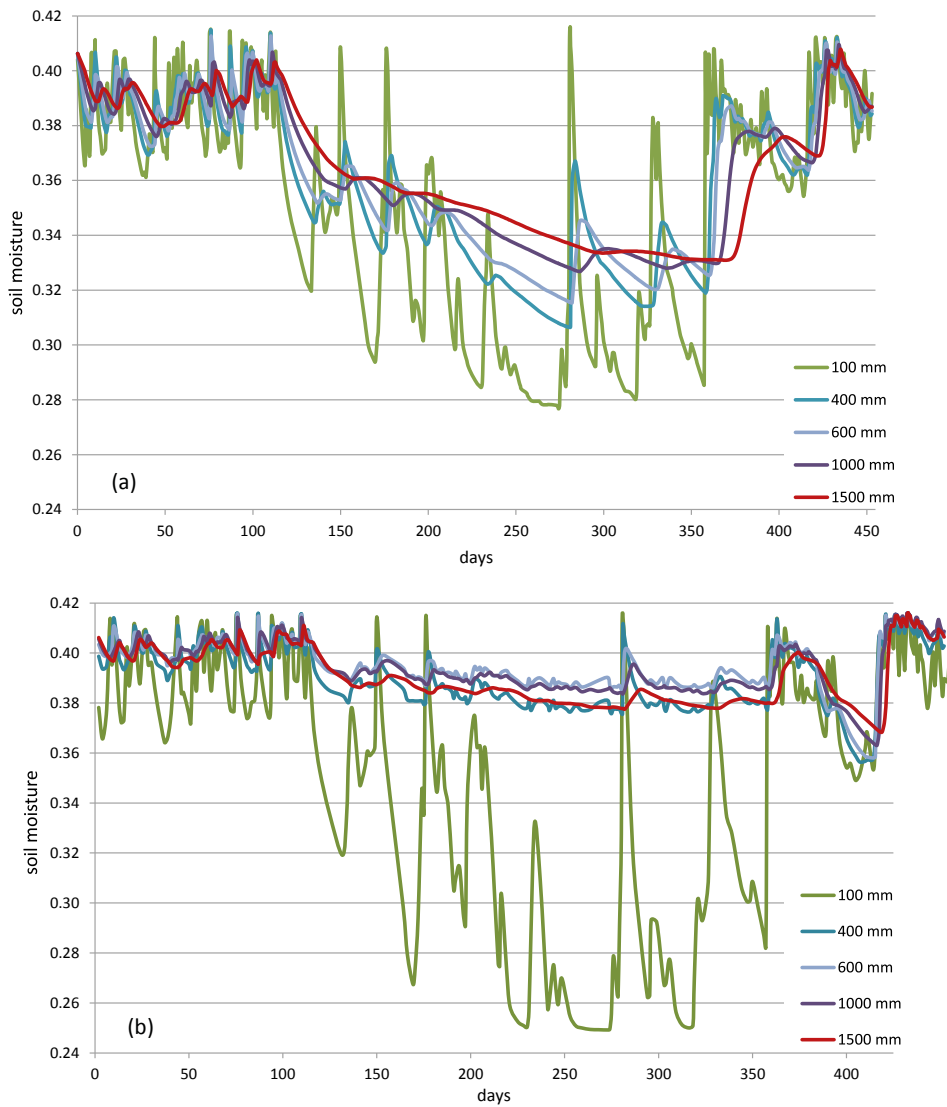
Figure 2.24. Simulated soil moisture profiles for $K_s = 50$ mm/day at Site B: (a) DD distribution and (b) LPP distribution.

at high T-values. Nevertheless, from modelling both systems at both sites, it was concluded that DD systems could be used in subsoils with T-values of up to 120 using secondary treated effluent (but not STE) and designed to have an areal loading rate of 2.8 l/m²/d with the required depth of unsaturated subsoil being a minimum of 600 mm. For LPP systems, the conclusion is that STE (from septic tanks with effluent filters) can be used with subsoils with T-values of less than 75 and then secondary treated effluent can be used for soils with T-values of less than 90, based on a trench loading rate of 18 l/m²/d but requiring 900 mm of unsaturated subsoil. For both designs, an average effluent production of 120 l/c per day should be used and, to ensure this hydraulic loading, the house should be fitted with water-saving devices such as dual flush

toilets (which are now a mandatory part of Building Regulations), as discussed in detail in Dubber and Gill (2013).

2.3   Evapotranspiration systems

The concept of discharging the on-site wastewater effluent into a sealed basin and relying on the net ET from willow trees to exceed the rainfall and effluent hydraulic loads across a year, was tested using both mesocosm and full-scale trials. This technology has been introduced in Denmark with some success (Gregersen and Brix, 2001; Arias, 2012). The study was initially funded by Wexford County Council and then additional funding was provided by EPA to continue the monitoring and extend it to other parts of the country.

2.3.1 *Mesocosm trials*

Four different willow varieties (*Tordis*, *Sven*, *Inger* and *Torhild*), planted in 75-l barrels, were monitored for a period of 4 years at the Trinity College Dublin (TCD) Botanic Gardens in Dublin while receiving three different effluent types: primary effluent (STE), secondary effluent and rainwater (Curneen and Gill, 2014). In summary, there appeared to be no statistical difference between the ET performances of the different species, but there was a statistically significant difference between ET in relation to effluent loading, with ET rates being highest for those cultivars receiving primary effluent, followed by those receiving secondary treated effluent, which, in turn, had much higher ET rates than those receiving just rainfall. Hence, the results obtained show that the addition of effluent has a positive effect on ET. In addition, water quality monitoring showed that the willows could also take up a high proportion of N and P from the primary and secondary treated effluents added each year.

2.3.2 *Full-scale trials*

Thirteen different full-scale systems were constructed and monitored during the project: 10 in County Wexford, two in County Leitrim and one in County Limerick. Most sites were designed and constructed by the TCD research team, although three sites had pre-existing ET systems with willows designed and constructed by a commercial consultant, with input from a Danish specialist. All sites were constructed with an impermeable liner, distribution piping laid in gravel and then backfilled with the *in situ* soil from the ground, as shown in Figure 2.25. Each site had a minimum of three different varieties of willows planted at a density of three plants per m². The site designs were varied between the sites in order to compare different design issues, such as size [plan area per population equivalent (PE)], aspect ratio, effluent type (septic tank versus secondary treated) and willow variety. At all sites, meteorological conditions were monitored, as well as the water levels in the willow basins, and effluent and rainfall loading. From this, the monthly ET could be calculated as well as the crop factors (K_c).

Figure 2.25. Construction and initial growth of willows.
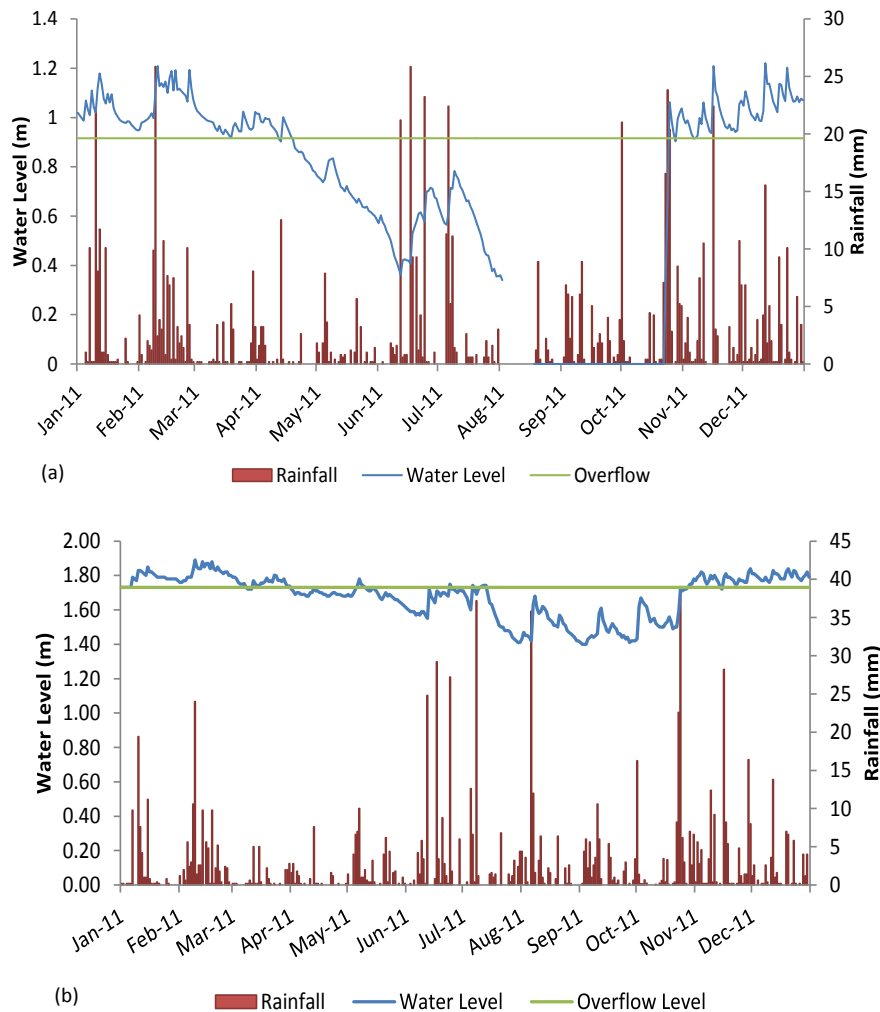
Figure 2.26. Rainfall and water levels in 2011 for (a) Site A and (b) Site C.

Typical ET results from two of the longest running systems are shown in Figures 2.26 and 2.27. Annual rainfall to effluent loadings for the fully inhabited sites varied from 1.3 to 6.5. The results showed that no system managed to achieve zero discharge in any year, with the water surface remaining at maximum level for many of the winter months as well as periodically at other times of the year, indicating some loss of water by lateral exfiltration at the surface. This was attributed to the fact that the low-permeability soil that had been dug out of the ground to form the basin was then used to backfill the systems and had a much lower effective (useful) porosity, only about 10% to 20% (mean = 14.9%), than had been assumed in the design, namely about 35% to 40%. In addition, the typical high relative humidity that occurs in Ireland meant that wind did not come out as a significant variable with respect to ET performance. However, chemical and microbiological sampling of the water in the sumps and ponded water over the winter periods showed improved water quality, equivalent to

surface runoff from the other systems that had not been fed any effluent (Table 2.5). Hence, the systems were acting as excellent pollutant attenuation devices, even if they could not operate as fully zero-discharge systems on an annual basis.

On the basis of monitoring 13 systems, different generic designs based on realistic crop factors for Ireland have been simulated across different climatic conditions. This has led to the optimised proposed guideline designs for the Irish climate (Table 2.6), which have been developed to inform policy decisions regarding the suitability of the use of these systems in Ireland. These guidelines have been developed in order to minimise the overflow of water from the systems in winter (that will really be more akin to rainfall runoff rather than effluent), in conjunction with other practical and financial considerations. It should be noted, however, that such treatment systems would not be suitable for all single-house scenarios as they do require a large
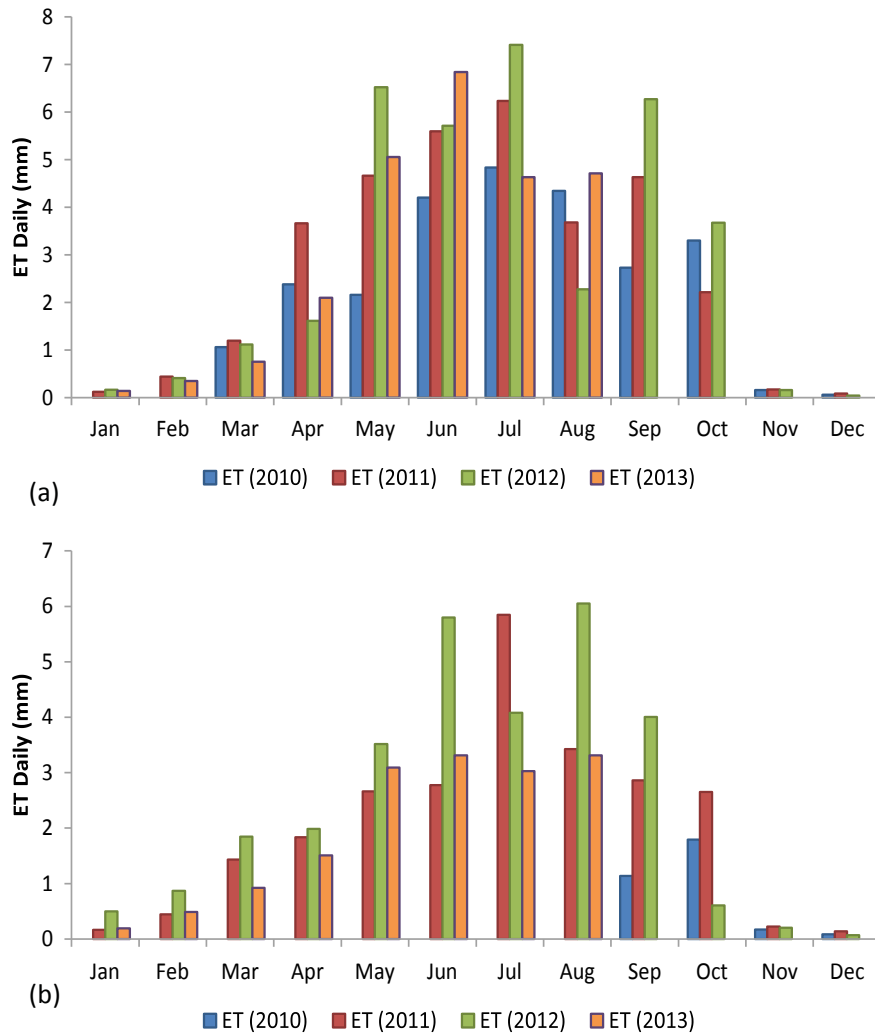
Figure 2.27. Monthly average ET rates for (a) Site A and (b) Site C for 2010 to 2013.

Table 2.5. Mean water quality results from willow systems with and without effluent.

	STE	Willow system (receiving effluent)	Willow system (not receiving effluent)
COD (mg/l)	385	29.5	32.9
TN (mg/l)	52.3	7.6	8.1
Orthophosphates (mg/l)	8.2	0.19	0.30
E.coli/100 ml (median)	1.7×10^6	29	< 10

Table 2.6. Willow system area (m²) for different population equivalents

Persons	4	5	6	8	10
Area (m²)	500	625	750	1000	1200

Wastewater production assumed to be 100 l/c per day.

land area, are expensive to construct, require a construction/commissioning time of 1 year and also require regular annual maintenance in the form of coppicing. It should also be noted that this would require a change in current consent procedures to allow for such a controlled discharge to surface water.

3 Strategic assessments

3.1 Advanced treatment with consented surface water discharge

For sites experiencing significant problems with low-permeability subsoils, the consented discharge of biologically treated on-site wastewater to suitable water courses needs to be reconsidered. In particular, in areas of relatively dense settlement which may have been identified by the National Inspection Plan as having significant issues for many on-site systems, it could be economically feasible to connect single houses via a small bore sewer system and treat the wastewater at a decentralised plant before discharge to a nearby watercourse. This would have the advantage [particularly from a local authority's (LA's) perspective] of having only a single consent, covering several houses, to manage. However, it should be noted that the financing, management and other socio-economic logistics of setting up such a decentralised clustered system will need considerable motivation and careful planning between many different actors in the field if such systems are to be successful in the future. For such a solution, appropriate treatment technologies need to be available to ensure compliance with EU and national water quality discharge limits (EC, 1991; EPA, 2009). Hence, treatment technologies currently available for small-scale wastewater treatment systems were reviewed with respect to treatment performance, maintenance requirements and costs.

3.1.1 Treatment performance and effluent quality

In order to assess each system's ability to meet surface water discharge limits, performance results were collected from official tests carried out according to the International/European Standard I.S. EN 12566-3.

About 43 packaged wastewater treatment systems were found to be available in Ireland, including conventional activated sludge (CAS) systems, submerged aerated filters (SAFs), rotating biological contactors (RBCs), media filters, moving bed bioreactors (MBBRs), sequencing batch reactors (SBRs) and membrane bioreactors (MBRs). In addition, a new fixed-film horizontal flow bioreactor (HFBR) (Clifford et al., 2010) will soon be launched onto the market and a pumped flow biofilm reactor (PFBR) is available for centralised and decentralised treatment (PE of 150–5000+) of municipal wastewater (O'Reilly et al., 2011).

Figure 3.1 shows the average effluent concentrations for 5-day biochemical oxygen demand (BOD_5), suspended solids (SS) and N present as NH_4 (NH_4-N) that are achieved by package plants using different generic treatment processes. It can be seen that the average effluent concentration obtained from most plants is well below the current national surface water discharge limits of 20:30:20 mg/l for BOD_5 to SS to NH_4-N (EPA, 2009). Over 60% of available treatment systems achieved average effluent BOD_5 concentrations of 12 mg/l or less with SS concentrations usually being less than 20 mg/l. Because of the temperature sensitivity of nitrifying bacteria, and to provide a better comparison of the nitrification potential of packaged treatment plants, only average ammonia removal rates determined for temperatures ≥12°C were considered. Most plants (about 65%) obtained average effluent NH_4-N concentrations of between 0.2 and 8 mg/l and even at winter temperatures (typically between –5°C and +6°C) ammonia effluent concentrations usually remained below the discharge limit of 20 mg/l. The MBRs achieve the best nitrification rates, with NH_4-N effluent concentrations below 1 mg/l (Figure 3.1c); this is explained by the high sludge retention times resulting from the filtration process (Melin et al., 2006; Dvořák et al., 2013). Media filters and MBBRs also achieved good ammonia removal with most plants producing effluents with NH_4-N concentrations of 5 mg/l or less. Effluent concentrations from other processes vary largely between different plants (Figure 3.1c).

Several packaged treatment plants already achieve good N removal by incorporating anoxic zones; however, removal efficiencies exceeding 65% are rarely achieved. Typically, TN-removal efficiencies range between 60% and 65%, resulting in effluent concentrations of 12–24 mg/l. Only two of the reviewed packaged treatment plants, one SBR and one media filter, have not exceeded the discharge limit of 15 mg/l given by the EU Council Directive for the treatment of urban wastewater (EC, 1991); however, no secondary treatment
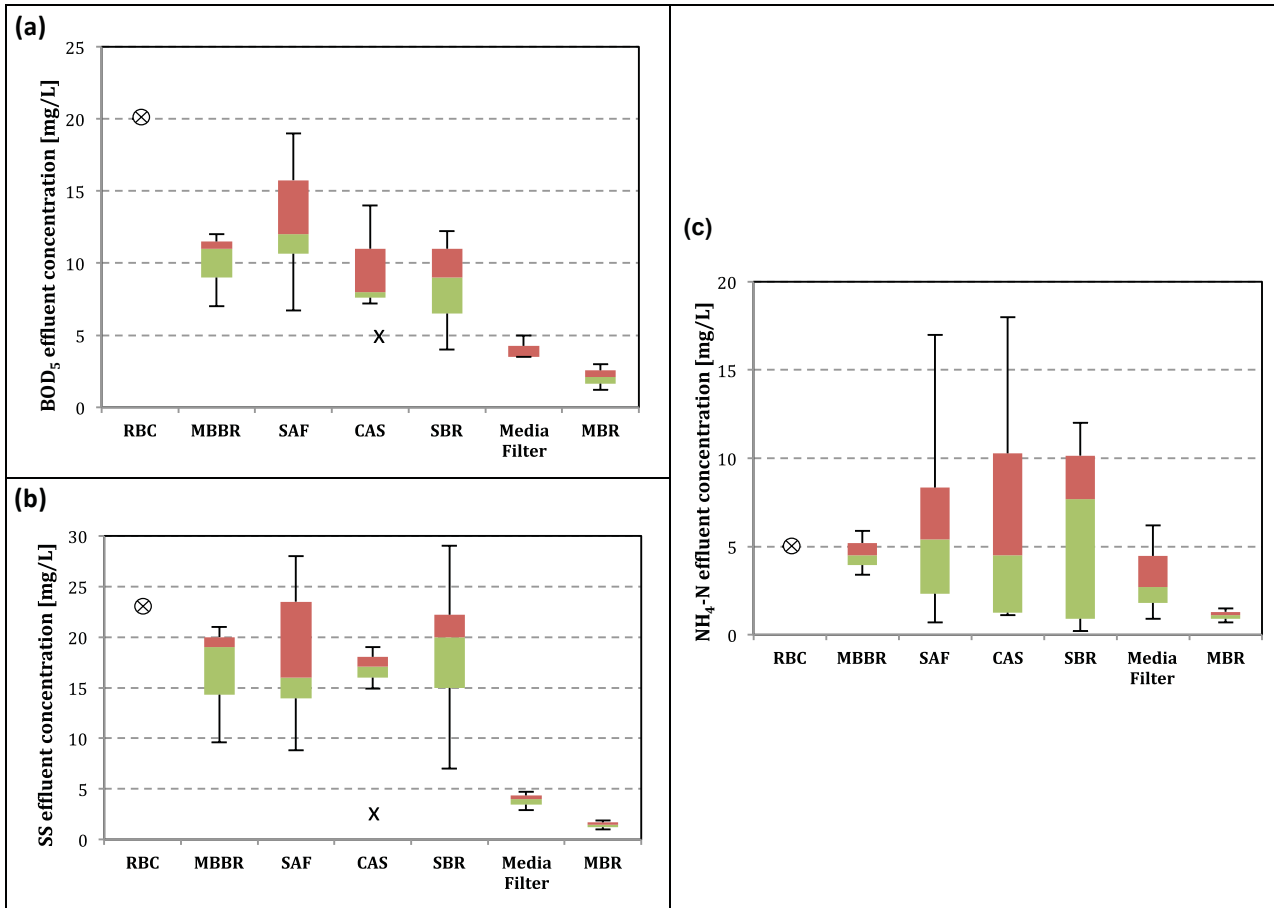
Figure 3.1. Distribution of average effluent concentrations for (a) BOD$_5$, (b) SS and (c) NH$_4$-N, according to EN 12566-3 test results obtained from package plants using different treatment processes. The graphs represent performance data from one RBC, three MBBRs, 12 SAFs, three CAS systems, seven SBRs, three media filters and two MBRs.

system is able to reach the required TN limit of 5 mg/l (EPA, 2009) to allow direct discharge to surface waters in nutrient-sensitive areas. To enhance denitrification, step feeding or effluent recirculation of nitrates into the anoxic zones is needed. However, this adds further complexity to the system, increasing costs and maintenance requirements.

Phosphate removal is normally achieved by chemical precipitation using salts of iron, aluminium or lime as coagulants (Metcalf and Eddy, 2003). While it has proven to be very effective in large-scale plants, secure chemical storage and the incorporation of an efficient dosing system could prove difficult for DWWTSs. Furthermore, additional costs for chemicals and more frequent desludging brought about by increased sludge production have to be considered. Biological P removal requires a high level of process control and is not a realistic option for automated and isolated DWWTSs. As a consequence, most of the package treatment plants do

not incorporate any P removal in their standard models, although some suppliers offer chemical P removal for customised systems. Some packaged treatment plants achieve up to 50% P removal even without any chemical dosing. However, discharge concentrations of effluent are still above 3 mg/l so that additional removal (e.g. using amended sand filters or other P-adsorbing materials) is needed if effluent discharge to surface waters in sensitive areas is to be considered.

For all technologies available on the market, it should be noted that their reliable treatment efficiency is highly dependent on regular maintenance. All available plants require at least one service visit per year and need to be desludged regularly. Significant operational problems appear as a result of user malpractice (e.g. disposal of wipes, nappies and greases, or overuse of antibacterial cleaners) resulting in insufficient treatment performance with effluent concentrations exceeding discharge limits. Information on existing surface water discharge licences

obtained from three LAs indicated that the maintenance of existing treatment systems is very often neglected by the licensee, frequently resulting in non-compliance with the licence's discharge limits. Results from effluent quality analyses were collected from 29 plants in three LAs. Samples originated from single houses (16), and small- (two) and medium-scale (11) treatment systems comprising RBCs, SAFs, MBBRs, SBRs and peat filters that either have a direct discharge or are followed by a soil/sand filter. The data evaluation showed that, of 189 water samples, 74% breached the given discharge limit for at least one water quality parameter. The majority of breaches were observed for TN and SS. Nearly half of the water samples breached the site-specific SS discharge limits of 10, 15 or 30 mg/l, with 20% even exceeding the national limit of 30 mg/l (Figure 3.2b). However, the licence monitoring results also showed that only 18% of the samples breached the given BOD_5 limits, whilst 48% and 84% of effluent samples had a BOD_5 of 3 mg/l or less and 10 mg/l or less, respectively (Figure 3.2a). As low effluent BOD_5 values were observed in connection with high solid concentrations, it is assumed that inorganic solids or non-biodegradable organics are washed out from soil and peat filters

causing the breaches. Sand filters, however, have proven to be effective for additional solid removal.

Nearly 80% of effluent samples from existing licences had ammonium concentrations of less than or equal to 1 mg/l (Figure 3.2c), indicating reliable nitrification processes with current technologies. However, it should be noted that these low concentrations were mainly obtained after treatment by downstream soil and sand polishing filters. For those licences that had to monitor TN and NO_3, 77% and 32% of samples breached the limits, respectively. The high numbers of breaches due to TN in combination with findings from the EN standard test results show the need for further developments to enhance denitrification in existing treatment technologies. P is currently removed either by chemical dosing and precipitation within the biological treatment system or by adsorption processes in subsequent soil or sand filters. In 79% of samples for monitoring, total phosphorus (TP) concentrations were 2 mg/l or less (Figure 3.2d) which meets the discharge limits for sensitive water bodies. However, observations indicated that the removal by filters was more reliable than removal by chemical dosing, which is probably because of the
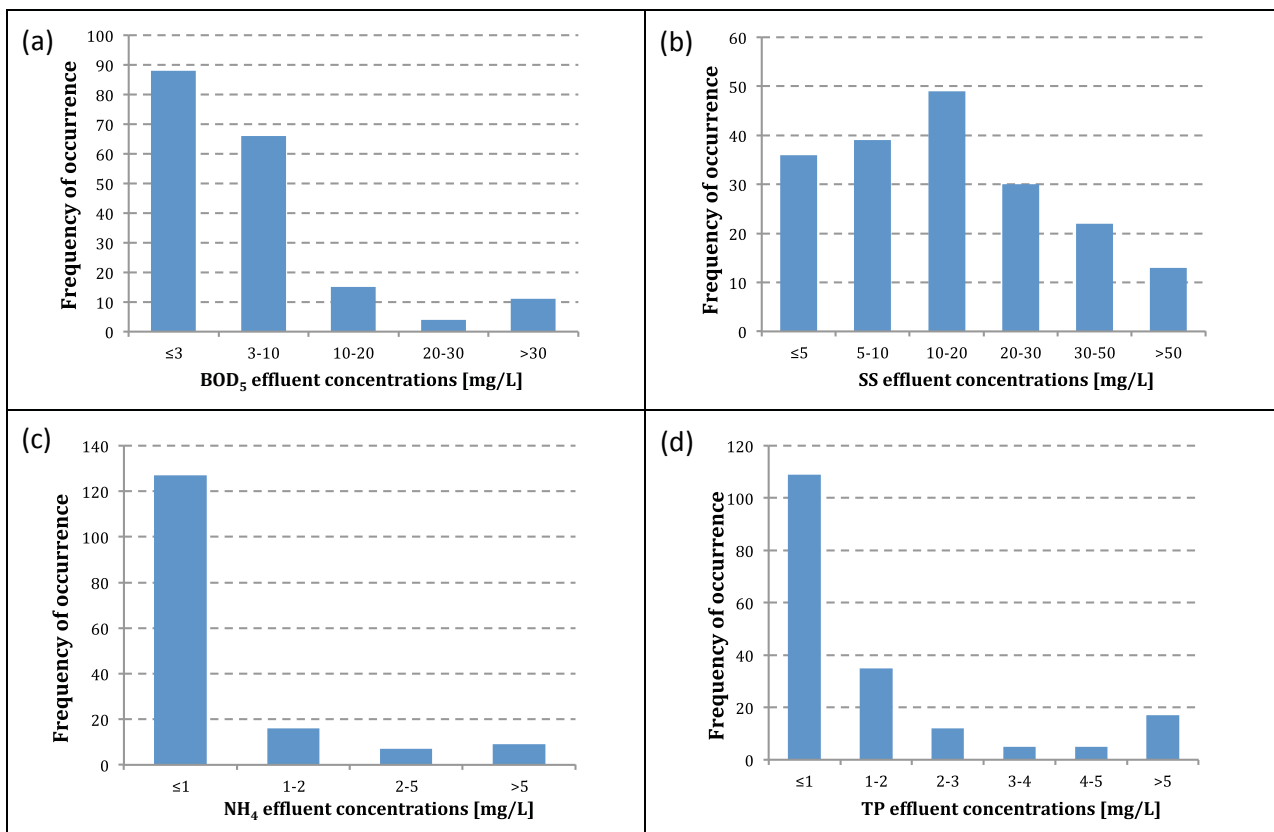
Figure 3.2. Water quality monitoring results from existing discharge licences in three local authorities: (a) biochemical oxygen demand; (b) suspended solids; (c) ammonium; (d) total phosphorus.

lack of maintenance in adjusting and refilling dosing systems. Hence, further research on P-adsorbing materials that could be suitable for use in DWWTSs would be useful.

3.1.2 Energy consumptions, capital and operational costs

Cost analyses show that for a single house with five inhabitants, MBRs and MBBRs are the most expensive, while SAF and CAS systems have the lowest capital costs (Table 3.1). While some media filters are in the upper price range there are systems that are cost competitive with SAF and CAS plants. Generally, a decrease in capital costs can be observed when larger models are used to serve small communities and with economies of scale per person realised (Table 3.1). For an SAF plant, for instance, the plant cost per person is reduced by 50%, down to €200, for a plant serving a PE of 40 compared with five.

SAF, MBBR and CAS systems require an air blower running continuously to supply oxygen to the aeration chamber. For a single household with five inhabitants, the annual electricity costs for such a system are estimated to range between €20 and €30 per person. An MBR needs an additional vacuum pump for the filtration process; therefore, running costs can be about twice as high. The rotor of an RBC system uses less energy than an air blower and annual electricity costs are estimated to be around €13 per capita in a single

household. Again, for most of the treatment processes, there are economies of scale with respect to per capita operational electricity costs. For an MBBR system, for instance, running costs can fall from €20 per capita per year for a single-house system to below €10 per capita per year for a small decentralised system serving more than 20 people (Table 3.1). In some media filters, the effluent is distributed by gravity so that no electricity is needed. Other systems use a pump with float switch to apply the effluent over the media and the associated electricity costs are less than €5 per capita per year. Similarly, both HFBRs and PFBRs do not need aeration devices and use only hydraulic pumps to operate. The aeration in an SBR system does not run on a continuous basis and treatment cycles are started only when enough wastewater has been collected in the primary chamber.

3.1.3 Potential for clustered decentralised systems in Ireland

To analyse the potential for a decentralised treatment solution, a GIS was used to identify clusters of houses in areas of high and very high likelihood of inadequate percolation (LIP) within four test counties (Leitrim, Limerick, Sligo and Wexford). An iterative buffering and clipping sequence was used to identify houses close enough to each other to be clustered. To keep the sewer length (and therefore costs) to a minimum, a maximum distance between houses was defined (80, 100 or 150 m) and a maximum distance to a river (150,

Table 3.1. Per capita capital and annual operational expenditures for single house and small decentralised systems in Ireland

	Single-house system costs (€/c)[a]		Small decentralised system costs (€/c)[b]	
	Capital	Operational	Capital	Operational
MBR	1600–1800	40–60	600–1200	<30
MBBR	1200	20–30	600–800	<10
Media filter	700–1000	0–5	350–700	0–5
SBR	500–750	4–7	300–500	4–7
SAF	400–700	20–30	150–250	<18
CAS	450–550	20–30	250–450	<15
RBC	n/a	13	420–600	<5
HFBR	n/a	4–8	n/a	4–8
PFBR	nr	nr	n/a	4–5

[a]**Based on a single-house system serving five inhabitants.**

[b]**Serving small communities with a PE of >20.**

c, capita; n/a, data not available during project duration; nr, not relevant.

200 or 250 m) was used to exclude clusters that do not have a potential surface water discharge point nearby.

For a distance between houses and a distance to a river of 100 m and 200 m, respectively, the analysis identified 124, 159 and up to 253 clusters (of more than four houses) in Leitrim, Sligo and Limerick, respectively, while up to 494 clusters were found in Wexford, which is densely populated with large areas classified as unsuitable for effluent percolation to ground. These clusters include 1445, 1799, 2403 and 7167 sites representing 15.5%, 23.5%, 21.1% and 40% of all legacy sites located in areas of inadequate percolation in Leitrim, Limerick, Sligo and Wexford, respectively (Figure 3.3). The average cluster size varies from 14 houses in Leitrim to 18 houses in Wexford; this would increase to 18 houses and up to 27 houses in Leitrim and Wexford, respectively, if the allowable maximum distance between houses was extended to 150 m. However, nearly half of the identified clusters would be rather small, consisting of only 4–10 houses.

It has been observed that large clusters are usually found close to sewered areas, with some of them not being considered for decentralised treatment because of the absence of a suitable surface water discharge point. In these cases, a decentralised sewer network that feeds into the adjacent centralised sewer system could be considered (if the treatment plant has enough spare capacity). This would provide a solution to a further 364 (3.9%), 434 (4.2%), 335 (3.9%) and 1177 (6.6%) legacy

sites in high LIP areas in Leitrim, Limerick, Sligo and Wexford, respectively.

Overall, these results show that decentralised treatment could provide a solution, particularly for more densely populated counties, such as Wexford, and in areas close to urban centres and large rural towns. However, the assimilative capacity of the relevant water body would need to be assessed first and compared with the expected discharge from such a decentralised wastewater treatment plant (WWTP). Moreover, the legal, social and economic aspect of the ownership and management of such clustered wastewater infrastructure would need to be addressed before this could become a more widely accepted solution. In this respect, lessons can be learned from the experience of the group water schemes throughout Ireland.

3.2 Connection to an existing sewer system

For sites where a centralised WWTP is close enough to treat and dispose of wastewater, the feasibility of connecting to that network should be investigated. An existing bylaw in County Cavan, for instance, states that any dwelling within 100 m of mains sewerage must be connected. Results from GIS analyses (using the buffering tool as an estimation for actual road distance) for the four test counties show that 4% to 8% of all potential legacy sites in areas of high LIP lie within
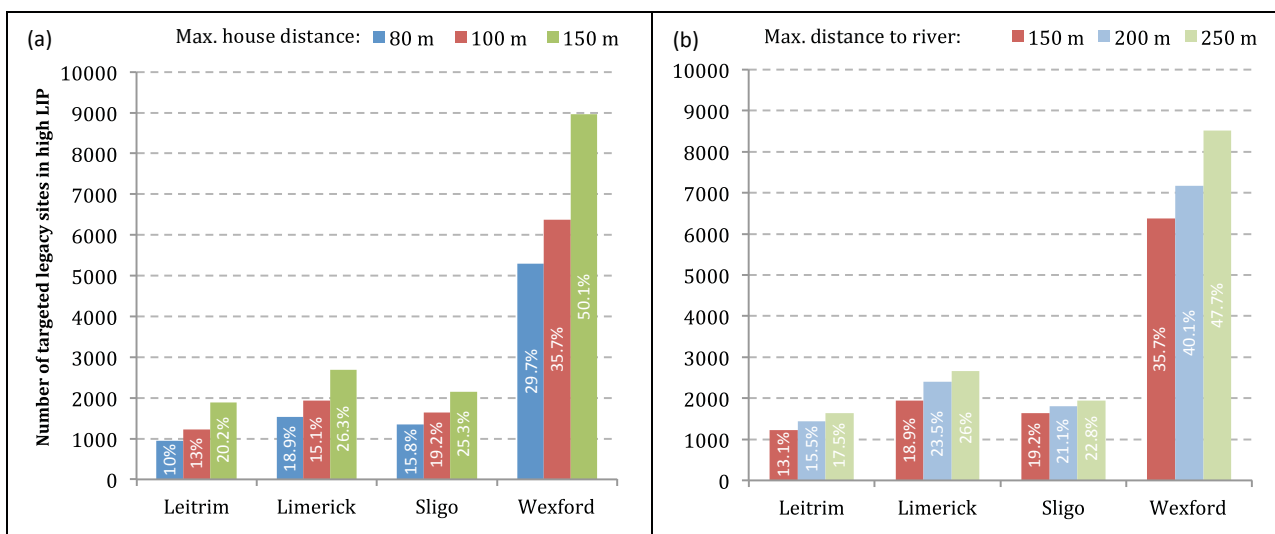
Figure 3.3. Number and proportion of legacy sites in areas with a high LIP that would be part of a cluster of ≥4 houses defined by (a) a maximum distance to river of 150 m and varying distances between houses and (b) a maximum distance between houses of 100 m and varying distances to a river.

a 100-m radial distance of an existing sewer network (Figure 3.4), indicating that effluent disposal problems could be solved by extending urban and rural sewered areas. These proportions increase accordingly when a larger radial distance around sewered areas is considered (Figure 3.4) providing a potential solution for 3400 (100 m radius) up to 6000 houses (250 m radius) within the four counties. However, the additional expenses for extending the sewerage network would need to be taken into account, as well as the available network and treatment capacity for the specific treatment facility before this is considered a viable solution.

3.3 Discharge through imported media filter into bedrock zones

Where the low-permeability subsoil layer is sufficiently shallow, such that it could be excavated and replaced by a more suitable imported media (soil or sand), the discharge of secondary treated effluent through this imported media filter into more permeable subsoil or bedrock zones could be considered. A maximum depth of 3 m is considered economically realistic for the excavation of the existing subsoil. GIS analyses carried out for four test counties show that this solution could potentially apply to 478, 142, 216 and 870 houses in Leitrim, Limerick, Sligo and Wexford, respectively. This would represent 4.1% and 4.7% of the legacy sites in areas of high LIP in Counties Leitrim and Wexford, respectively, but only 2.1% and 1% in Sligo and Limerick, respectively. However, in addition to a maximum subsoil

depth, a minimum depth (≈ 1 m) of unsaturated zone is needed to ensure sufficient treatment within the soil/sand filter before the effluent reaches the water table; this would need to be confirmed by an on-site assessment. Furthermore, an assessment of the bedrock permeability would be required to ensure that it would be able to take the hydraulic load. GIS-based maps of the bedrock types within Ireland can give an indication of the expected hydraulic properties. Figure 3.5 shows the predominant bedrock types that are found in Ireland in areas of high LIP and shallow subsoils (< 3 m depth). Generally, sedimentary rocks (e.g. limestone and sandstone) will have a higher hydraulic conductivity than igneous rocks, such as granite; however, the actual permeability is largely affected by the depth and extent of weathering and interconnected fracturing. Poorly productive aquifers, which are found in over 90% of the relevant areas, are usually of low transmissivity in the main bedrock, but percolation into the shallow transition and bedrock zones can be observed where the effluent would then be transported horizontally to the next river.

In order to identify bedrock types for which this could be a potentially feasible solution, available bedrock permeability data were consulted. Relevant data were obtained from the Geological Survey of Ireland (GSI)/EPA Aquifer and Subsoils Parameters Database, which is currently being compiled by Tobin Consulting Engineers (Dublin, Ireland) and contains a variety of hydrogeological parameters, such as well-specific capacity, transmissivity and hydraulic conductivity. The
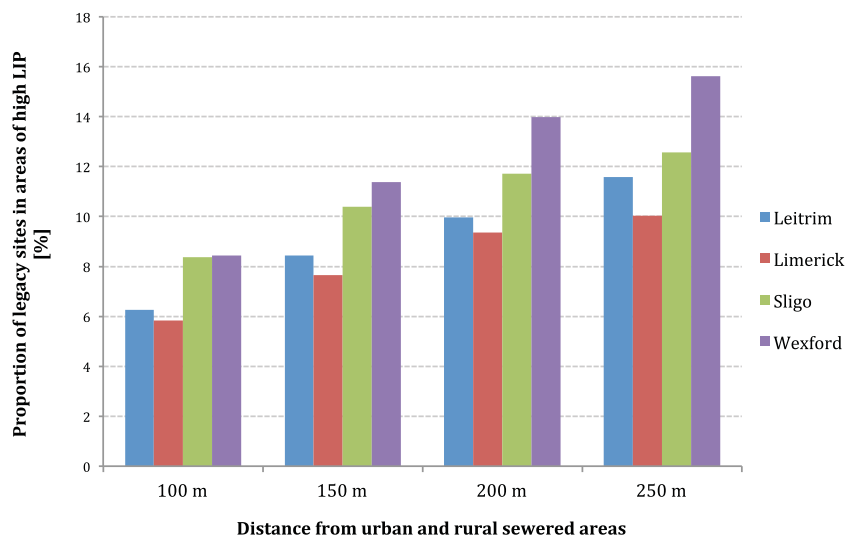
Figure 3.4. The proportion of legacy sites in areas of inadequate percolation that are within close distances of urban and rural sewered areas in the four test counties.
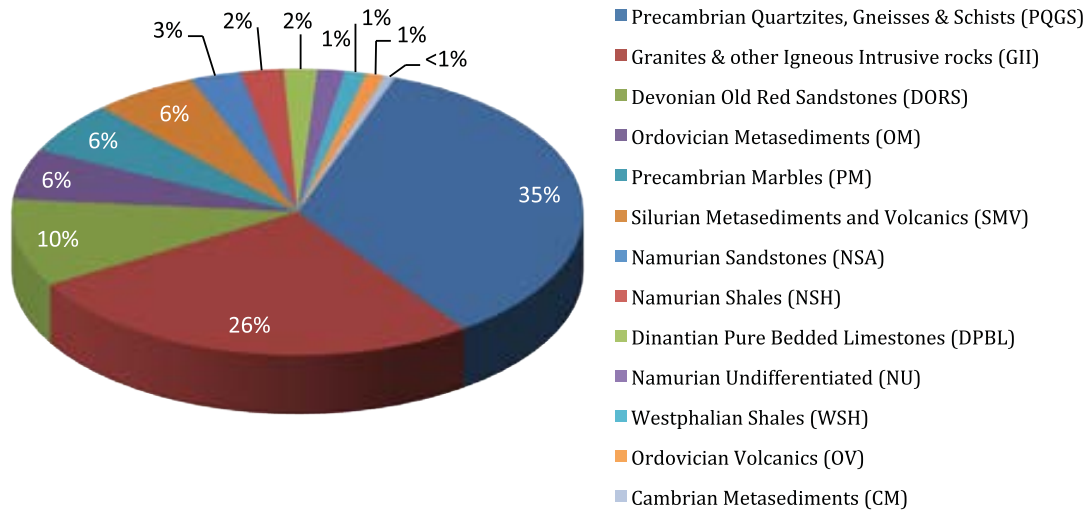
Figure 3.5. Predominant bedrock type in Ireland in areas of high LIP and shallow subsoils.

available permeability data were categorised according to the bedrock type and analysed for the proportions of hydraulic tests that resulted in bedrock permeabilities below or above a design-limiting value. Figure 3.6 shows the distribution curves for bedrock permeabilities observed in Ireland for different bedrock types.

The design guidelines for an imported media filter are outlined in the Code of Practice (EPA, 2009), which recommends a maximum hydraulic loading of 60 l/m^2 per day (including the rainfall onto the area) to ensure that appropriate treatment is provided before the effluent reaches the bedrock and the receiving water body. This equates to a minimum required filter surface area of roughly 2.7 m^2 per capita; however, for bedrock permeabilities of less than 0.06 m/d, this becomes the

factor that determines the size of the filter (increasing exponentially). The filter area can be increased up to a certain limit, above which the filter construction would not be economically feasible or the available space on a property would become a limiting factor. For example, for a filter area of up to 5, 10, 20 or 33 m^2 per person (equating to per capita construction costs of €1900, €2500, €3000 or €5000, respectively) this would result in minimum required bedrock permeabilities of 0.034, 0.0195, 0.0123 or 0.009 m/d, respectively, in order to ensure that the bedrock could accept the hydraulic load. Table 3.2 shows the proportion of bedrock permeability test results that were observed to be above these application limits for different bedrock types throughout Ireland. The Aquifer Parameter Database
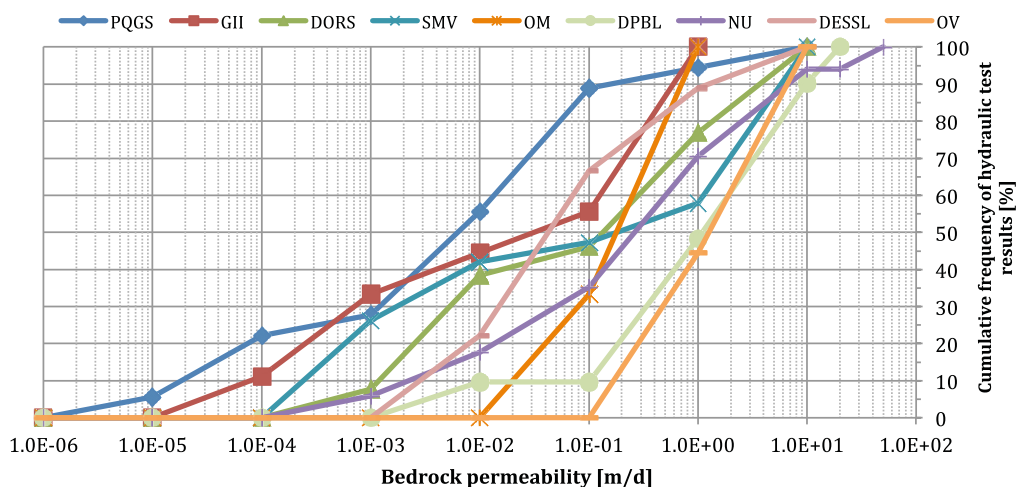
Figure 3.6. Distribution curves for bedrock permeabilities observed in Ireland for different bedrock types. DESSL, Dinantian (early) sandstones, shales and limestones.

Table 3.2. Proportion (%) of bedrock permeability results above certain application limits. Values in parentheses exclude test results for deep bedrock

Bedrock types	No of considered test results	Bedrock permeability limit (m/d)			
		0.034	*0.0195*	*0.0123*	*0.0094*
PQGS	18 (12)	27.8 (33.3)	33.3 (41.7)	38.9 (50)	50 (58.3)
GII	9 (6)	44.4 (66.7)	44.4 (66.7)	55.6 (66.7)	55.6 (66.7)
DORS	13 (10)	53.8 (70)	61.5 (80)	61.5 (80)	76.9 (90)
SMV	19 (16)	57.9 (68.8)	57.9 (68.8)	57.9 (68.8)	57.9 (68.8)
OM	3 (0)	66.7 (n/a)	100 (n/a)	100 (n/a)	100 (n/a)
DPBL	31 (27)	90.3 (96.2)	90.3 (96.2)	90.3 (96.2)	93.5 (96.2)
NU	17 (15)	76.5 (73.3)	76.5 (73.3)	82.4 (80)	82.4 (80)
DESSL	9 (9)	44.4 (44.4)	77.8 (77.8)	77.8 (77.8)	77.8 (77.8)
OV	9 (9)	100 (100)	100 (100)	100 (100)	100 (100)

DESSL, Dinantian (early) sandstones, shales and limestones; DORS, Devonian old red sandstones; DPBL, Dinantian pure bedded limestones; GII, granites and other igneous intrusive rocks; NU, Namurian undifferentiated; OM, Ordovician metasediments; OV, Ordovician volcanics.

also provided information on the depth and the zones (subsoil, transition zones, shallow and/or deep bedrock) in which the hydraulic tests have been carried out, so that test values from the deep bedrock (that would be less relevant for the considered disposal option) could be excluded (see values in parentheses in Table 3.2). The type of assumption that can be made from this analysis are, for example, that with a construction cost limiting factor of €3000 per person, about 50% of sites underlain with Precambrian quarzites, gneisses and schists (PQGS) could be suitable for discharge through an imported media filter into this bedrock. Alternatively, if a site is underlain by PQGS, the chances are 50:50 that the site would be suitable for the proposed solution. However, there are considerable uncertainties with such a method in that there are no or too few data for several bedrock types in order to provide statistically robust and representative summary statistics. Hence, confirmation by an on-site assessment of the shallow bedrock's permeability would be required in most cases.

There are a number of different hydraulic tests, each addressing a different purpose, scale of measurement and potentially different pathways. Pumping tests (including slug and packer tests) are probably the most common tests carried out to estimate the hydraulic properties of an aquifer system. However, estimates of hydraulic conductivity, based on these measurements, reflect saturated, predominantly horizontal fluxes in deeper bedrock zones, whereas shallow, vertical, near-saturated fluxes would be of interest to the

proposed disposal solution. Furthermore, the expense of such a hydrogeological survey for a single-house site assessment, especially where no existing wells or boreholes can be used, would probably prove prohibitive. On-site falling-head permeability tests are often deployed for site investigations to determine hydraulic conductivity. They are typically carried out over short intervals and are usually restricted to a small area of relatively shallow depth. While this can be too restrictive in order to make estimates for the hydraulic parameters of an aquifer, it could be sufficient to determine whether or not the bedrock would take the required hydraulic effluent load for the proposed disposal scenario. For example, a simple falling-head infiltration test method developed recently by Mirus and Perkins (2012) using a bottomless bucket to estimate the field-saturated hydraulic conductivity (K_{fs}) of bedrock outcrops could be applied. Alternatively, a similar falling-head test (without the bucket) could be carried out at the bottom of the site assessment test hole which, in this case, is dug down to the bedrock. Because the subsoil, forming the walls of the test hole, will be of low permeability, it should be possible to ignore the lateral infiltration/water movement that could occur above the bedrock. However, because of the inherent heterogeneity in the predominantly fractured nature of bedrock in Ireland, the disadvantages of such a localised test should be acknowledged as these could result in the failure to detect major fractures, which might be quite extensive over the proposed area of percolation. As there can still be considerable costs and effort (time) associated

with such a site assessment, the strategic method described earlier could be used to give an indication of the probability of success prior to any site assessment. Finally, given that 90% of the relevant bedrock is poorly productive, it is likely that the transition zone will be the main pathway for flow via fairly short pathways into the nearest surface water feature (ditch or stream). Hence, it is essential that the effluent is treated to the required standard on reaching the base of the imported media filter.

3.4 Wastewater collection in storage tanks with regular disposal at centralised wastewater treatment plants

In areas with very low subsoil permeability, a storage tank (i.e. a cesspool) for wastewater collection and treatment at the local WWTP could be considered as a possible disposal option. To assess the economic feasibility of such a solution, emptying frequencies and associated costs were estimated for a private house/ holiday home in Wexford, as an example. Calculations were made assuming an occupancy rate of 3 PE with a standard per capita consumption (PCC) of 150 l/c per day and with PCCs based on the reduced wastewater

productions that can be achieved using different toilet systems, together with water-saving appliances (Dubber and Gill, 2013). A storage tank volume of 18,000 l was chosen given that the British Building Regulations (DoCLG, 2000) specify this as the minimum capacity for a single domestic dwelling with two residents. Quotes for emptying the tank were requested from local companies from which costs were estimated at €40/m^3 (excluding VAT).

The cost calculations in Table 3.3 show that annual net expenses are reduced by 44%–53% if water-saving devices are installed. Therefore, for a holiday house in areas with unsuitable subsoil permeability, this may be considered as a disposal option. However, it is not economically feasible for a full-time occupied house. The storage tank of a three-person household, without water-saving devices, would need to be emptied almost every month, resulting in annual costs of €6570. Using dual-flush toilets and other water-saving appliances extends the emptying interval to every 2–3 months and reduces the annual net cost to €3793 (€3400 with a urine-diverting dual-flush toilet); however, this is still too high to be considered an economically realistic option. It should be noted that, as companies charge per collected volume of wastewater, annual running costs cannot be decreased by using a larger storage tank.

Table 3.3. Emptying frequencies and costs (excluding VAT) associated with a wastewater storage tank solution for a holiday house used regularly (52 weekends, i.e. 119 days, per year) by three people

	Standard PCC	Reduced PCC with dual-flush toilet (3/4.5 l)	Reduced PCC with urine-diverting dual-flush toilet (0.6/4 l)
Wastewater production	150 l/c per day	86.6 l/c per day	77.6 l/c per day
Emptying frequency	Every 4 months	Every 7 months	Every 8 months
Annual costs for tankering	€2142	€1237	€1108
Annual water/energy cost savings	n/a		
Using electricity		€88	€90
Using gas		€48	€50
Annual net costs	€2142		
Using electricity		€1149	€1018
Using gas		€1189	€1058
Annual net costs per person	€714		
Using electricity		€383	€339
Using gas		€396	€353

n/a, not applicable.

3.5 Cost and sustainability assessments for domestic wastewater treatment solutions[1]

3.5.1 Economic feasibility

Tables 3.4 and 3.5 show estimated per capita construction and operational costs for the alternative effluent disposal options considered for application in areas with inadequate percolation. Costs are calculated for a single-house system serving three inhabitants, but it should be noted that per capita costs usually decrease if the house has a higher occupancy. Construction costs include material and labour, while annual operation costs include electricity, desludging and maintenance of the systems. Furthermore, how the construction and operational costs would reduce if water-saving devices were installed into a house is shown. Because of the reduced system size (42%–49%), material and construction costs for a willow system will be up to 43% lower than for a standard-sized system, which would cost around €5300 per person without the consideration of potential wastewater reduction. It should be noted that the proposed guidelines for these willow systems (see Section 2.3) are based on a design for effluent loading of only 100 l/c per day meaning that water-saving fixtures must

be included in any house where this technology will be installed. In contrast, for LPP and DD systems, the size does not significantly affect the construction time and, therefore, construction costs are only reduced by up to 5% and 9%, respectively (Table 3.4). The costs for the connection to an existing sewer were based on a sewer length of 100 m being needed to connect to the network. However, it should be noted that the economic feasibility of this solution increases with the number of houses that can be connected with the same sewer extension (to connect a total of two houses, it would cost €2280 per capita; to connect a total of three houses, it would cost €1540 per capita). The costs for the decentralised treatment solution were estimated based on the installation of an effluent sewer system. This type of sewer network collects the effluent from septic tanks either by gravity (septic tank effluent gravity) or using an effluent pump [septic tank effluent pump (STEP)] if the house is at a lower elevation than the main sewer line. The costs presented in Table 3.4 are average costs and, in practice, range from €1513 per capita for houses using gravity up to €2135 per capita for houses requiring a pump to feed their effluent into the collection network.

Water-saving actions will not only reduce the construction costs of on-site disposal systems, but also lower operational expenditures and a household's utility bills for water and energy (Table 3.5). Based on a volumetric water charge of €0.75/m³ for rural areas in Ireland (Brady and Gray, 2010), the estimated annual water cost savings are up to €17.36 per capita when using

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1 Costs have been calculated based on best available information in 2014 and may underestimate real costs for applications with respect to, for example, specific retrofitting requirements.

Table 3.4. Estimated per capita construction costs (excluding VAT) for alternative effluent disposal options. The numbers in brackets represent cost reductions compared with a system designed with standard per capita consumption (PCC)

Construction costs	Standard PCC	Reduced PCC with dual flush toilet	Reduced PCC with urine-diverting toilet
Connection to existing sewer[a]	€4420	n/a	n/a
Clustered decentralised system	€1820	n/a	n/a
Imported filter media onto bedrock[b]	€1634	€1499	€1480
LPP system[c]	€1510	€1450 (4%)	€1440 (5%)
DD system	€2180	€2000 (8%)	€1980 (9%)
Willow system	€5300	€3400 (36%)	€3030 (43%)
Storage tank (cesspool)	€1200	€1200	€1200

[a]Based on a road distance of 100 m to the existing sewer.

[b]For minimum required filter area according to a maximum hydraulic loading of 60 l/m² per day defined by the EPA (2009). Depending on bedrock permeability filter areas, costs will increase (see Section 3.4).

[c]Based on a T-value of 80.

Table 3.5. Estimated per capita operational costs (excluding VAT) for alternative effluent disposal options. The numbers in brackets represent cost reductions compared with standard per capita consumption (PCC)

Annual operational costs[a]	Standard PCC	Reduced PCC with dual-flush toilet	Reduced PCC with urine-diverting toilet
Connection to existing sewer	€0[b]	n/a	n/a
Clustered decentralised system	€53–€63	n/a	n/a
Imported filter media onto bedrock	€139	€47 (66%) or €88 (36%)	€45 (68%) or €86 (38%)
LPP and DD systems	€139	€47 (66%) or €88 (36%)	€45 (68%) or €86 (38%)
Willow system	€50	€0 (100%) or €1 (98%)	€0 (100%) or €0 (100%)
Storage tank/cesspool[c]	€714	€383 (46%) or €396 (44%)	€339 (53%) or €353 (51%)

[a]**Net costs, incorporating water and energy savings for electricity or gas, related to water-saving devices.**

[b]**Currently no direct costs as covered by taxpayers' money (but will be subject to changes from Irish Water in the future).**

[c]**Only considered for holiday homes (see Section 3.4); costs are based on an average occupation time of 119 days per year.**

water-saving devices. At the same time, appliances such as flow-restricted shower heads and tap aerators that reduce the consumption of warm water will also promote a reduction in the energy used to heat water for domestic use. Based on a water-use reduction of up to 48%, it was estimated that up to €73 per capita per year or €32 per capita per year could be saved when water is heated using electricity or gas, respectively (Dubber and Gill, 2013). The generally low running costs for the ET-based willow system would be paid back completely by the annual savings made through water and energy savings. Per capita operational costs for the decentralised system are estimated at around €53, but increase to €63 if a STEP system is used. As discussed in Section 3.4, a cesspool is an option for holiday homes only if water-saving devices are installed. Although a cesspool requires the lowest

capital expenditure (Table 3.5), the high operational costs make it the least economic and sustainable solution, and cesspools should be used in only special cases if, for technical and/or environmental reasons, no other solution would be applicable.

3.5.2 Environmental sustainability

A comprehensive assessment of the environmental sustainability of DWWTSs in Ireland, including their impact on groundwater and surface waters as well as on greenhouse gas (GHG) emissions related to energy consumption and microbiological processes, has been carried out by Dubber and Gill (2014). Table 3.6 shows GHG emissions that are associated with the energy use (including diesel use for desludging services) of each considered alternative disposal solution. As for

Table 3.6. Estimated energy-related GHG emissions (as $kg\,CO_{2eq}$ per capita per year) for alternative disposal options

GHG emissions	Standard PCC	Reduced PCC with dual-flush toilet	Reduced PCC with urine-diverting toilet
Connection to existing sewer	22.5	13	11.6
Clustered decentralised system	36.6	n/a	n/a
Imported filter media onto bedrock	66.5	60.9	60.1
LPP and DD systems	66.5	60.9	60.1
Willow system	10.4	10.4	10.4
Cesspool[a]	49.8	28.8	25.8

[a]**Considered only for holiday homes (see Section 3.4); calculations are based on an average occupation time of 119 days per year and a distance of 20 km to the nearest WWTP for centralised treatment and disposal.**

construction and operation costs, a reduction in GHG emissions can be observed when wastewater production is reduced via water-saving devices. For DD and LPP systems, for instance, emissions related to the electricity used to distribute the effluent over the percolation area can be reduced by 5.6–6.4 kg CO_{2eq} per capita per year (Table 3.6). According to data from the UK, carbon emissions related to treatment at centralised WWTPs are estimated at 0.41 kg CO_{2eq}/m³ (Water UK, 2007), which yields 22.5 kg CO_{2eq} emissions per person per year when assuming a standard PCC of 150 l/c per day. These emissions, however, will decrease proportionally with the reduction in wastewater production. The carbon footprint from the operation of a cesspool depends largely on the distance from the house to the nearest central WWTP (assumed to be 20 km for the estimations presented in Table 3.6), but because of the reduced emptying frequencies (Table 3.6), similar reductions in CO_2 emissions can be expected.

In addition to savings related to wastewater treatment, GHG emissions are also reduced in relation to water treatment and warm water supply. Based on the average company performance values from Water UK (Water UK, 2007), 0.29 g CO_{2eq} is emitted for every litre of water supplied. Hence, the secondary emission rates for water supply for a person with a water consumption of 150 l/c per day is estimated to be 15.88 kg CO_{2eq} per capita per year, which could be reduced by 6.71 up to 7.67 kg CO_{2eq} per capita per year. However, the largest savings are obtained from energy savings associated with reduced warm water use. It has been shown that 305–421 kWh per capita per year in electricity can be saved if flow-restricted appliances are installed. Similarly, if gas is used as an energy source, 1491.8–2058.5 cf per capita per year (equal to 480–662 kWh per capita per year) can be saved by using flow-restricted appliances. By applying conversion factors of 0.562 or 0.206 kg CO_2/kWh (Hackett and Gray, 2009) to these values for electricity or natural gas savings, respectively, this equates to carbon emission reductions of 171–236.6 kg CO_{2eq} per capita per year or 99–136.4 kg CO_{2eq} per capita per year, depending on the energy source used to heat the domestic water. These reductions offset the higher GHG emissions that are expected in relation to the energy use for some of the disposal solutions.

3.6 Development of a GIS-based decision support tool

A decision support computer program has been developed that can be used by environmental planners and managers to assess the feasibility of alternative disposal options in rural areas with inadequate percolation. The program is based on a GIS and uses geospatial datasets of human settlements, the physical environment, comprising geology, land cover and hydrology, and infrastructure, such as transportation and utility networks (Box 3.1); these datasets were obtained from the GSI, EPA and LAs. A modelling architecture (Figure 3.7) has been developed that takes, as its initial input, an existing house located in an area of inadequate percolation. The proposed disposal solutions (see Box 3.1) and their suitability for the selected site are evaluated in parallel.

Box 3.1. Considered treatment and disposal strategies to be evaluated by the GIS-based decision support tool and the integrated GIS datasets

Considered treatment and disposal options

- Connection to existing sewer network for centralised treatment
- Clustering of houses with decentralised treatment and surface water discharge
- Discharge onto bedrock through imported media filter
- On-site pressurised dispersal systems (LPP and DD)
- On-site ET (Willow) systems
- Wastewater collection in on-site cesspool with regular emptying and centralised treatment

GIS datasets used

- LIP
- *x*- and *y*-coordinates for septic tank systems
- Road networks
- Urban and rural sewered areas
- Rivers
- WWTPs
- Depth to bedrock
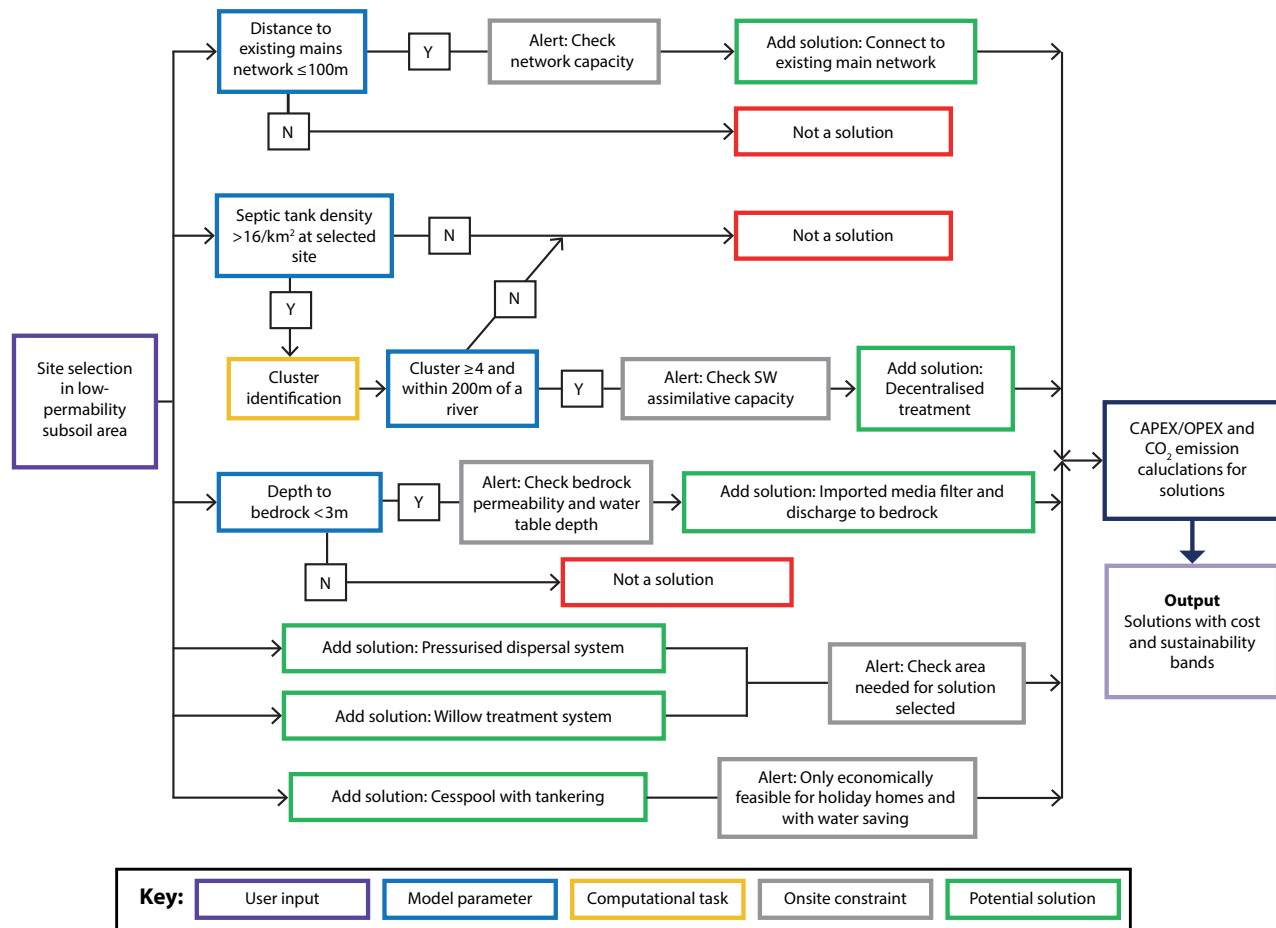
Figure 3.7. Modelling architecture for the GIS-based decision support tool.

While the on-site solutions are always included as a suggested option, the selection of other appropriate solutions depends principally on four major model parameters: (1) distance from an existing sewerage network; (2) existing septic tank density (for reasons of economies of scale); (3) the distance to surface water; and (4) the depth to bedrock (Figure 3.7). Through a series of interconnected GIS geoprocesses, the model provides appropriate solutions for a site and gives information on sustainability and cost. However, it should be noted that while the model can assess the suitability of treatment options to a certain extent, the final decisions are still dependent on on-site-specific constraints. Therefore, each solution is accompanied by an alert message that provides additional information for the user to refine the output list according to the available local site-specific information. ArcGIS (Esri, Redlands, CA) tools and functions have been applied to carry out the computational tasks for the evaluation of each disposal strategy, as explained below.

With regard to geoprocessing services for disposal option 1, two ArcGIS models have been constructed in order to identify the nearest sewered area, as well as the closest road junction points, to the selected site. Intersections between roads and the sewered areas are used as reference points in order to identify the closest possible connection to an existing sewer network. ArcGIS Network Analyst software was applied in a third model to determine the road distance from the selected site (using the road junction) to the edge of the closest sewered area. This tool was selected over a standard buffering or distance function in order to estimate the length of the required sewer connection, which would be installed alongside the road, and hence estimate the installation costs as accurately as possible.

As regards geoprocessing services for disposal option 2, the ArcGIS kernel density function calculates the density of point features around each output raster cell and was employed in this study to calculate house densities in rural areas. If the septic tank density in the area of a selected site is greater than 16 systems/km^2,

an iterative buffering and clipping sequence is used to identify houses that are close enough to each other to be connected up to a small decentralised sewer system. This geoprocessing service was implemented using Python scripting through ArcGIS geoprocessing tasks. It further assesses whether or not a river is close enough to the cluster to facilitate a possible discharge.

For geoprocessing services for disposal option 3, GIS maps of the depth to bedrock were consulted. A maximum depth of 3 m was considered economically realistic for the excavation of the existing subsoil. Hence, the depth to bedrock for the selected site location was obtained from the maps to determine whether or not this would be a possible treatment and disposal option.

The remaining disposal options (4, 5 and 6) that would be suitable at a single-house level are always considered a solution by the program. This is because the suitability of these systems is mainly dependent on site-specific constraints that are not available from GIS maps. Hence, the user will be given additional information together with the suggested solution that enables the output list to be refined according to the available local site-specific information.

In order to evaluate the alternative disposal options suggested by the program on the basis of both cost–benefit and environmental sustainability principles, calculations to estimate capital and operational expenditures (CAPEX and OPEX), as well as GHG emissions associated with each solution (see Section 3.5), are included within the model. All cost and emission calculations integrate and use results from GIS-based computational tasks as input parameters for their estimations, making them as site-specific as possible. Examples of these parameters are the road distance of a house to the nearest sewer network, the size of a suggested house cluster and the distance from a house to the next WWTP for sludge or wastewater disposal. The calculations are implemented using Python scripting through ArcGIS geoprocessing tasks, which retrieves any required parameters from a Microsoft Excel® (Microsoft Corporation, Redmond, WA) file.

The subsequent coding was carried out to initially test the program on four counties: Wexford, Leitrim, Sligo and Limerick. The model was developed with Esri's ArcGIS Server 10.1 and ArcMAP 10.1 and is configured as a thin client/server application. Adobe Flash Builder 4.6 (Adobe Systems, Mountain View, CA) was used to deploy the program on a website (presentation layer) from where it interacts with the geoprocessing services, which are placed together with GIS data layers and maps as a web service on the ArcGIS server (data/service layer) (Figure 3.8).

The tool is designed to provide as great a flexibility to the user as possible. Therefore, the user interface
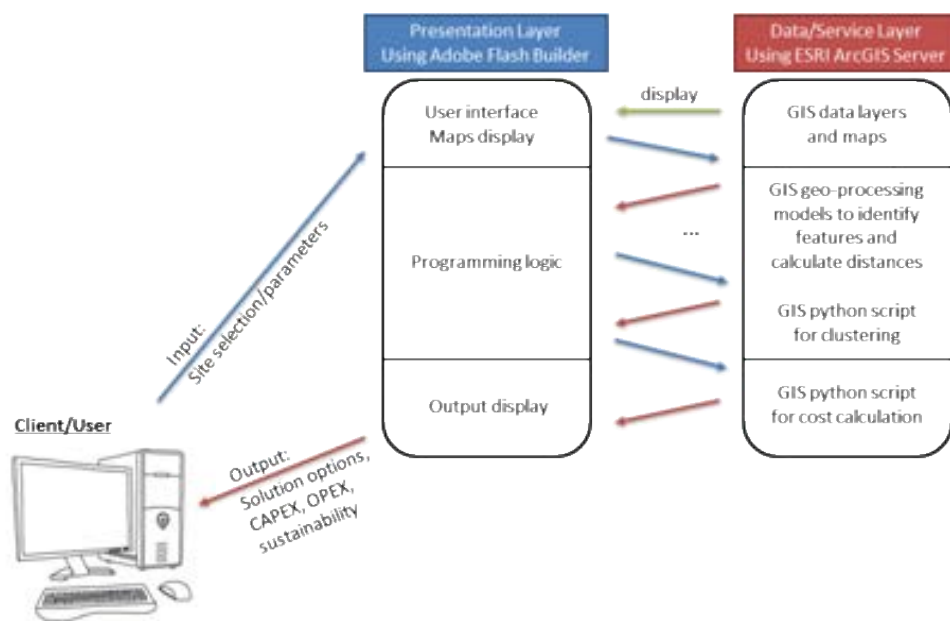
Figure 3.8. System structure of the web-based GIS decision support tool.

allows the possibility of changing some of the parameters, including the PE, the maximum considered road distance to an existing sewer network, the maximum house and water body distance used to identify the clusters, as well as the minimum cluster size considered for a decentralised solution. After a house has been selected for analysis, the tool runs through the different processes and presents the considered solutions, as well as the calculated parameters. The identified cluster of houses as well as the shortest routes to the sewer network and closest WWTP are also graphically highlighted in the map. As costs and sustainability are only estimates and are subject to change (prices might vary within the country), values are presented only in categories. Nevertheless, this will still allow a direct comparison between suggested solutions.

The program and its structure represent a first version of the software that could be extended and applied to the rest of Ireland. There is potential for enhancing the precision and user interaction of the software, and for including further automated feasibility assessments. For example, it would be possible to take into account topography to identify areas where sewage could be collected by gravity and those where a pump solution would be needed. Furthermore, the assessment of assimilative capacities for receiving waters could be automated and integrated into the software. This would be based on existing water flow and quality information and could employ watershed delineation to make estimations for ungauged subcatchments.

Eventually, the program could become an essential tool for decision making and environmental planning for LAs and in connection with the development of a suitable programme of measures to minimise the impact of DWWTSs on water bodies and to help reach Water Framework Directive (WFD) objectives. However, it should be noted that, while the tool presents feasible options from technical and environmental perspectives, management and legal issues will need to be considered during the final decision-making process.

4   Conclusions and recommendations

4.1   Conclusions

- Existing soakaway systems in low-permeability soils are likely to be causing shallow lateral flow of effluent into the nearest surface depression, promoting a risk to surface water pollution. However, the natural attenuation of bacteria and P in such lateral flow pathways does seem to be significant.

- In higher permeability soils, the effluent from soakaways is more likely to be percolating downwards and may give rise to a risk to localised groundwater pollution if the water table is shallow. However, no negative influence was picked up at any of the on-site wells in the deeper groundwater at those sites.

- The remediation of the existing soakaway systems to alternative pressure-dosed distribution systems resulted in a decrease in faecal contamination of groundwater, as well as the prevention of surface ponding of effluent at both of the low-permeability sites studied.

- The field results and calibrated models of the unsaturated zone have shown that the LPP system could be a solution for sites with T-values of less than 90 and the DD system for sites with T-values of less than 120 after secondary treatment. However, the method of assessing subsoil percolation, using the falling-head percolation test, is very insensitive at such low-permeability values and should be changed to a more rigorous test for such site assessments.

- Evapotranspiration systems using willow trees are unlikely to be able to act as zero-discharge systems if the existing low-permeability subsoil is used as a backfill (which would appear to be the most realistic option, particularly from an overall sustainability perspective). However, they do promote excellent pollutant attenuation and they significantly reduce net effluent discharge to the environment and so should be considered as a viable passive treatment for both existing (legacy) and new developments. It should be noted that this would require a change in current consent procedures to allow for such a controlled discharge to surface water.

- Microbial source tracking was successfully employed to confirm the source of faecal contamination in groundwater, and is a suitable new monitoring tool to support and add value to detection methods based on traditional faecal indicators, such as *E. coli* and other bacteria. Future work should develop these methods (and others, such as human-specific chemical and viral indicators) and tiered strategies to provide a practical, cost-effective and robust approach to water quality assessment and protection in Ireland.

- Molecular biological techniques can help to provide a better understanding of the factors influencing microbial denitrification and the transport of pathogens.

- Surface water discharge will need to be reconsidered in areas where discharge to ground is simply not possible. Most packaged WWTPs, if correctly installed and regularly maintained, are able to achieve effluent concentrations lower than the usual minimum required discharge limits. This could provide an environmentally sustainable solution; however, for discharge to nutrient-sensitive areas, improvements in the removal of TN and TP will be needed. New technologies in this area should be as passive and low maintenance as possible.

- The concept of clustered decentralised systems shows promise for targeting a significant proportion of potentially poor sites in low-permeability areas, and could lower the burden of monitoring associated with discharge consents. Furthermore, analyses indicate that they can be economically favourable compared with single-house systems. Hence, this option should be investigated and developed further from a technical, social, economic and legal (e.g. ownership, liability, etc.) perspective.

- The extension of existing sewer networks should be considered where WWTPs still have available treatment capacity. In order to reduce excavation work and costs, alternative effluent collection systems could be used to feed into the main sewer network.

- The excavation of impermeable subsoil and discharge to bedrock will be a viable solution only for a minority of cases, and a site investigation, including some sort of infiltration test, will be necessary to determine a site's suitability.
- Because of high operational costs, cesspools will not be a sustainable long-term solution and might be considered only in extreme cases and for holiday houses with water-saving devices installed.

- The developed decision support tool has demonstrated how geospatial information can help environmental consultants to identify possible solutions by, for example, highlighting where clustered systems could be considered, and comparing options based on cost and sustainability.
- The use of water-saving devices can increase feasibility and sustainability of most effluent disposal options and should be considered where possible.

4.2   Recommendations

Table 4.1. Recommendations for implementation and uptake of research findings

Issue	Recommendation	Target	Time frame
On-site solutions for areas with subsoils with T-values of >75	Introduce DD, LPP and ET system design recommendations into EPA Code of Practice (2009)	EPA, DECLG	<1 y
Retrofit options for existing sites failing under National Inspection Plan	Disseminate research findings from soakaway work, both with respect to pollutant transport in higher permeability subsoil as well as lateral effluent transport in lower permeability subsoils. Also, disseminate research findings from LPP, DD and willow systems to LAs	EPA, site assessors, LAs	<1 y
Strategic assessment of DWWTS by LAs	Workshops for LAs to demonstrate the use of the GIS-based decision support tool	EPA, LAs, DECLG, IW	1–2 y
Decentralised cluster systems for wastewater treatment	One or two pilot schemes to be funded by DECLG to determine the socio-economic challenges associated with such systems	DECLG, LAs, IW	2–5 y
Nutrient input from DWWTS into surface waters (re. WFD)	Use of GIS decision support tool to quantify inputs and also incorporation of findings from the project into the EPA catchment management tool	EPA	1–2 y
Risk of well contamination from DWWTS	Further research into use of microbial source tracking techniques and other fingerprinting methods	EPA, DoH, HSE	1–3 y
Introduction of surface water discharge licences for single houses in low-permeability areas	Consideration for such licences to be granted in limited, specific situations. Also, decision as to the status of whether or not rainfall runoff from a willow system needs to have a discharge consent	EPA, DECLG, DAFM	1 y
Discharge of treated effluent to bedrock	Further research to be undertaken to assess different techniques to measure the hydraulic conductivity of bedrock	EPA, DECLG	2–4 y
Reduction of wastewater production (and energy saving)	Awareness campaign for public to retrofit water-efficient devices	DECLG, EPA, IW	1–2 y

DAFM, Department of Agriculture, Food and the Marine; DoH, Department of Health; HSE, Health Service Executive; IW, Irish Water.

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Abbreviations

BOD$_5$	5-day biochemical oxygen demand	**MBBR**	Moving bed bioreactor
CAS	Conventional activated sludge	**MBR**	Membrane bioreactor
CO$_2$Eq	Equivalent CO$_2$	**MPN**	Most probable number
DD	Drip dispersal	**N**	Nitrogen
DECLG	Department of the Environment, Community and Local Government	**NO$_3$-N**	Nitrogen present as NO$_3$
		P	Phosphorus
DWWTS	Domestic wastewater treatment system	**PCC**	Per capita consumption
		PE	Population equivalent
EPA	Environmental Protection Agency	**PFBR**	Pumped flow biofilm reactor
ET	Evapotranspiration	**PO$_4$-P**	Phosphorus present as PO$_4$
GC/g	Gene copy concentration in gene copies per gram	**PQGS**	Precambrian quarzites, gneisses and schists
GC/l	Gene copy concentration in gene copies per litre	**RBC**	Rotating biological contractor
		SAF	Submerged aerated filter
GHG	Greenhouse gas	**SBR**	Sequencing batch reactor
GIS	Geographic Information System	**SS**	Suspended solids
GSI	Geological Survey of Ireland	**STE**	Septic tank effluent
HFBR	Horizontal flow bioreactor	**STEP**	Septic tank effluent pump
K_s	Hydraulic conductivity	**TCD**	Trinity College Dublin
LA	Local authority	**TN**	Total nitrogen
l/c	Litre per capita	**TP**	Total phosphorus
LIP	Likelihood of inadequate percolation	**WFD**	Water Framework Directive
LPP	Low-pressure pipe	**WWTP**	Wastewater treatment plant

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 Aeráide, Ceadúnaithe agus Úsáide Acmhainní
- An Oifig Forfheidhmithe i leith cúrsaí Comhshaoil
- An Oifig um Measúnú Comhshaoil
- An Oifig um Cosaint Raideolaíoch
- 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.

Assessment of disposal options for treated waste water from single houses in permeability subsoils

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

The potential impacts of incorrectly situated or poorly constructed domestic wastewater treatment systems (DWWTSs) include the pollution of either groundwater and/or surface water and places a risk on human health particularly via private wells. It is estimated that the more than one third of the country may have inadequate site conditions for the installation of such on-site systems discharging to ground. This research has assessed the performance of existing soakaway systems in a range of different soil permeability settings and then evaluated a range of potential solutions for the treatment and discharge of on-site domestic wastewater into low- permeability soils.

Identifying Pressures

This research has evaluated the pollutant transport to groundwater and surface water from old existing soakaway systems in a range of different soil permeability settings. This showed that such soakaway systems in low permeability soils are likely to be causing shallow lateral flow of effluent into the nearest surface depression promoting a risk to surface water pollution. However, the natural attenuation of bacteria and phosphorus in such lateral flow pathways does seem to be significant. This research has also shown the potential for the use of microbial source tracking techniques using human- specific Bacteriodales bacteria as well as other fingerprinting methods.
A decision support tool has also been developed based upon geospatial modelling to identify existing DWWTSs on potentially problematic sites with low permeability soils, information which can feed into wider strategic planning debates.

Informing Policy

This research has demonstrated with field trials a range of systems that may be a solution for on-site wastewater treatment and disposal in low permeability soils, although changes in current policy and legislation would be required to facilitate their use. EPA staff will review the report and liaise with DECLG in relation to the relevant recommendations/findings highlighted by the authors and incorporate the findings into national guidelines as appropriate
A decision support tool, also produced by this study, has been developed for strategic assessment at a Local Authority level to inform policy. This has highlighted that surface water discharge may need to be reconsidered in areas where the discharge to ground is very problematic. It has also shown that the concept of clustered decentralised systems could target a significant proportion of potentially poor sites in low permeability areas and therefore lower the burden of monitoring associated with individual discharge consents.

Developing Solutions

The decision support tool also recommends a suite of suggested strategic solutions for systems in low permeability areas based on economic and sustainability criteria such as: extension of existing sewer networks; clustered decentralised systems with surface water discharge; and excavation of impermeable subsoil with discharge to bedrock. The use of water-saving devices in homes has also been identified as a method to increase the feasibility and sustainability of most effluent disposal options and should be considered where possible.

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