

Proceedings of the International Symposium on Domestic Waste Water Treatment and Disposal Systems



Trinity College, Dublin
Monday 10th and Tuesday 11th September 2012



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DUBLIN

INTERNATIONAL SYMPOSIUM ON DOMESTIC WASTE WATER TREATMENT AND DISPOSAL SYSTEMS

Trinity College, Dublin
Monday 10th and Tuesday 11th September 2012

Programme Day 1 - 11th September 2012

08:30 - 09:30 *Registration & coffee*

09:30 – 09:45 Introduction & Welcome Dr. Patrick Prendergast, Provost TCD

09:45 - 10:15 Setting the Scene Gerard O'Leary (Director EPA)

10:15 - 11:00 Keynote Speaker: Prof Chris Buckley (Uni. KwaZulu Natal, S. Africa)
On-site Wastewater: the hazards

11:00 - 11:30 *Tea & Coffee*

11:30 - 13:00 SESSION I CATCHMENT STUDIES

11:30 - 11:50 Linda May (CEH, Edinburgh)
The impact of on-site sewage treatment systems on river water quality in UK catchments.

11:50 - 12:05 Phil Jordan (Uni. of Ulster)
Rural point sources: small portions of pie and P in headwater catchments

12:05 - 12:20 Ray Flynn (Queens)
Quantifying contributions of on-site sewage disposal systems to nutrient fluxes in stream headwaters

12:20 - 12:35 Valerie McCarthy (Dundalk IT)
A field study assessing the impact of on-site wastewater treatment systems on surface water quality in a Co. Monaghan catchment

12:35 – 12:50 Discussion and Q&A

12:50 - 14:10 *Lunch*

14:10 - 15:20 SESSION II SUBSOIL ATTENUATION

14:10 - 14:30 Robert Siegrist (Colorado School of Mines, USA)
Using an in situ soil profile as a wastewater treatment unit: purification processes and renovation efficiencies.

14:30 - 14:45 Robbie Meehan
Subsoils across the Irish landscape; their textural and bulk density characteristics, and resultant variations in permeability

14:45 - 15:00 Laurence Gill (TCD)
The attenuation of on-site effluent contaminants through percolation areas of different Irish subsoils

15:00 - 15:15 Vincent O'Flaherty (NUI Galway)
Understanding microbial denitrification and pathogen transport in effluent and soils.

15:15 – 15:30 Discussion and Q&A

15:30 - 16:00 *Tea & Coffee*

16:00 - 17:20 SESSION III PACKAGED WASTEWATER SYSTEMS

16:00 - 16:20 Mark Gross (Orenco, Oregon, USA)
Packaged wastewater treatment systems for individual homes and small communities

16:20 - 16:35 Eoin Clifford (NUI Galway)
Research developments in the on-site treatment of wastewater

16:35 - 16:50 Eamonn Smyth (DECLG)
EN certification and National Annexes

16:50 - 17:05 Donata Dubber (TCD) -
Review of packaged systems in Ireland

17:05 – 17:20 Discussion and Q&A

Programme Day 2 – 12th September 2012

09:30 - 11:00 SESSION IV REGULATORY FRAMEWORK

09:30 - 11:00 Sarah West (EPA Victoria, Australia)
How Australia and New Zealand manage onsite wastewater 8 different ways.

09:50 – 10:05 Julia Black & Prof Robert Baldwin (London School of Economics)
The Challenges of Low Risks: Best Practice and the Development of a Strategy

10:05 - 10:20 Leo Sweeney & Margaret Keegan (EPA)
The National Inspection Plan for Domestic Waste Water Treatment Systems – A Proposed Approach

10:20 - 10:35 Seamus O'Brien (North Tipperary Co Co)
Local Authority Assessment of OSWWTS - Practice, Issues and Options

10:35 - 11:00 Discussion and Q&A

11:00 - 11:30 *Tea & Coffee*

11:30 - 12:50 SESSION V CONSTRUCTED PASSIVE AND INTERMITTENT SYSTEMS

11:20 - 11:40 Carlos Arias (Uni. of Aarhus, Denmark)
Current state of descentrilized waste water treatment technology in Denmark

11:40 - 11:55 Sean Curneen (TCD)
The Treatment of On-Site Wastewater using Willow Bed Evapotranspiration Systems in Ireland

11:55 - 12:10 Niall O'Lunaigh (La Trobe University, Australia)
The attenuation capacity of constructed wetlands to treat domestic wastewater in Ireland

12:10 - 12:25 Laurence Gill (TCD)
Constructed on-site sand filters as secondary and tertiary effluent treatment processes.

12:25 – 12:50 Discussion and Q&A

12:50 - 14:10 *Lunch*

14:10 – 15:40 SESSION VI RISK ASSESSMENT / DISPOSAL OPTIONS

14:10 - 14:30 Nancy Deal (Dept Env. & Natural Res, N. Carolina)
Theory and Practice of Soil and Site Investigation for Wastewater Dispersal Systems

14:30 – 14:45 Donal Daly (EPA)
A Risk Based Methodology to Assist in the Prioritisation of the Inspection of Domestic Waste Water Treatment Systems

14:45 - 15:00 Dave Smyth (NUI Maynooth)
The use of geospatial modelling in determining strategies for on-site wastewater treatment in areas of low permeability subsoil.

15:00 - 15:15 Mark Livingston, Head of Water Regulation Group NIEA
Regulation of treated sewage discharges from domestic properties in Northern Ireland.

15:15 – 15:40 Discussion and Q&A

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PREFACE

There have been significant developments as well as growing public and political attention on the subject of on-site wastewater treatment (“septic tanks”) over the past few years in Ireland and so it was felt an opportune moment to organize a meeting of researchers and policy makers in the area to share and debate the current national thinking as well as providing an international perspective. Hence, a two day International Symposium was convened in Trinity College Dublin on 10-11th September 2012, organized jointly by the Environmental Protection Agency (EPA) and the Environmental Engineering Group, Trinity College Dublin. The two day Symposium was attended by over 210 participants, a mixture of academics, representatives from Governmental Agencies and Local Authorities and those who work in the on-site industry (site-assessors, hydrogeologists, treatment plant manufacturers etc). The symposium was divided into six sequential themes as follows: *Catchment Studies, Subsoil Attenuation, Packaged Wastewater Systems, Regulatory Framework, Constructed Passive Systems and Risk Assessment / Disposal Options*, each one consisting of presentations by an international expert followed by three national presentations.

There have been significant advances in research in the area of on-site wastewater treatment and disposal over the last ten years, much of which has contributed towards the development of a new Code of Practice for on-site wastewater treatment and disposal. The *EPA Code of Practice (2009) Wastewater Treatment and Disposal Systems serving Single Houses (p.e. ≤ 10)* must now be used by all Local Authorities (LAs) *to assess planning applications for rural housing*. The Code sets out how a Site Assessment should be carried out and how all the collected information should then be integrated into a conclusion and an appropriate engineered design for the on-site wastewater treatment system. This is then submitted with the overall planning application to the appropriate LA.

The subject of on-site wastewater treatment and disposal is also high on the public's agenda at the moment in Ireland due to new legislation being drafted in response to the Irish state being prosecuted by the European court on failing to address adequately the issue of on-site systems. In 2009 the European Court of Justice (ECJ) made a judgment against Ireland (Case C-188/08) with respect to on-site wastewater treatment systems, their main issue being that there was no registration system in place in Ireland for such systems (with the exception of County Cavan which was exempt from the ruling). Hence, in order to address this ruling and prevent significant ongoing fines from the EU, the government published the *Water Services Amendment Act (2012)* which includes registration and inspection arrangements. All on-site systems will have to be registered. The local LAs will establish and maintain a registration system. The EPA will be responsible for the development of a national inspection plan which will use a risk-based approach to prioritize areas of higher risk to human health and water quality.

Many of the larger aquifers in Ireland are found in fractured / fissured bedrock where groundwater movement is potentially fairly rapid and so the conceptual model for groundwater protection from on-site wastewater disposal assumes that no further treatment occurs once the effluent has reached the water table. The subsoils in Ireland are generally fairly heterogeneous, largely a result of the recent glaciation, e.g. extensive areas of glacially derived tills. There are significant areas of low permeability clayey subsoils (e.g. in Counties Wexford, Leitrim, Monaghan) which, as discussed above, give rise to challenges with the design of percolation areas for on-site effluent. Equally, other areas present problems due to shallow subsoils, <2m (often above karstified aquifers) as well as high water tables (<2m). Hence, the challenge for on-site designs is often to maximize the depth of unsaturated subsoil into which the effluent is disposed.

The Symposium provided a fascinating two days covering the wide range of issues (technical, geological, political, sociological, regulatory etc) that all influence the subject of on-site wastewater treatment and disposal. The Symposium acted as an excellent forum for debate and discussion between national and international delegates alike to provide a rigorous analysis of where Ireland is at present as well as direction future developments.

DISCUSSION

The scene was set for the Symposium by Gerard O’Leary, Director of the EPA, who highlighted recent Irish research in the field of on-site wastewater treatment, the development of the EPA’s *Code of Practice: Wastewater Treatment and Disposal Systems Serving Single Houses*, the ongoing EU prosecution and the EPA’s response to this, as detailed in the Preface. This was followed by a thought-provoking keynote talk by Chris Buckley from the University of Kwa-Zulu Natal in South Africa who stressed that the most important driver for appropriate on-site sanitation should be public health. He explained that the lack of significant existing wastewater sanitation infrastructure in many developing countries and the magnitude of the demand is promoting more innovative thinking that would not be possible in a more developed or constrained environment. He then introduced the ideas of ecosanitation (composting toilets, urine separation etc) and the value of wastewater as a resource which should be recycled. He also discussed the socio-economic practicalities of introducing sanitation schemes and the importance of community buy-in.

The first session on **CATCHMENT STUDIES** had a strong focus on the environmental / ecological pollution from on-site systems, in particular the impacts of phosphorus. Linda May from the Centre of Ecology and Hydrology in the UK started the session and showed how recent studies had revealed that the number of consented on-site systems in the UK seemed to be grossly underestimating the real number of systems and therefore underestimating the phosphorus contribution into freshwater from these sources with respect to agricultural sources. Phil Jordan (University of Ulster) then followed, presenting results from intensive monitoring in small headwater catchments in Ireland which suggested that, whilst the total input of phosphorus into rural catchments from septic tanks was low on the whole compared to more diffuse inputs (agriculture etc) during rainfall-runoff events, it was more significant during the more critical low flow periods. Raymond Flynn (Queen’s University, Belfast) then presented an ongoing large scale catchment modelling project (the *Pathways* project) which is developing a Catchment Management Tool for policy makers. In particular he presented how the attenuation of on-site effluent along the different hydrological pathways contributing to river discharge are going to be modeled under a range of geological conditions. Finally, Valerie McCarthy (Dundalk Institute of Technology) presented the results from studies on DWWTSs in a low permeability subsoil catchment in County Monaghan which suggested that effluent run-off into surface water usually dominated over infiltration with resulting impacts on lake water quality.

The next session on **SUBSOIL ATTENUATION** focussed on the fundamental importance of the characteristics of the unsaturated zone for the treatment of on-site effluent. Bob Siegrist (Colorado School of Mines) ran through the latest advances in studies of the unsaturated zone to show that there is now a more quantitative understanding of key flow and transport

processes and the removal of pollutants and pathogens as affected by soil properties, system features, effluent quality and loading, and other design factors and environmental conditions. He also appealed for the use of accurate and consistent language within the field to prevent confusion. Laurence Gill (Trinity College Dublin) then summarised research carried out on several field sites across Ireland which have investigated the unsaturated zone on a range of subsoils receiving effluents of different quality (septic tank vs secondary treated). He showed how the extent of the biomat formation and therefore underlying hydraulic loading was determined by organic load and subsoil permeability. Robbie Meehan (Talamh Ireland) outlined the significant variations in subsoils across Ireland as a result of bedrock geology and recent glaciations and showed how the permeability of different subsoils are affected by grain size distribution and amount of clay and how this equates field saturated hydraulic conductivity (T-test) values. Finally, Vincent O’Flaherty (NUI Galway) highlighted recent research which has investigated the composition of the microbial denitrifier community in both soil and groundwater as well as the development of a Microbial Source Tracking methodology to determine the source (human and/or animal) of faecal contamination from on-site effluent in soils, groundwater and surface water.

Mark Gross (Orenco Systems) opened the session on **PACKAGED WASTEWATER SYSTEMS** giving a summary of the different types of generic secondary treatment processes from suspended growth systems to fixed film systems (including membrane bioreactors), with an assessment of their suitability for the on-site application. He especially highlighted maintenance as key criteria when choosing a system for use in a single house scenario. Edmond O’Reilly (NUI Galway) then covered some new advances in biofilm treatment processes for small scale wastewater treatment which particularly target nutrient and pathogen removal. He also described the Water Research Facility in Tuam which provides a research infrastructure to trial novel water and wastewater treatment technologies. Emanon Smyth (Department of Environment, Community and Local Government) covered issues surrounding certification with the new European EN12566 testing requirements for all systems in Ireland and also issues of the applicability of such tests carried out at one location given the different strength national effluents that are typical throughout different European countries. Donata Dubber (Trinity College Dublin) then finished the session with a review of package systems available in Ireland for on-site treatment which also compared energy costs and stressed the importance of ongoing maintenance. Her summary also revealed how most package systems would not meet required phosphorus consents for discharge directly into nutrient sensitive surface waters without some form of enhanced further treatment.

The session on **REGULATORY FRAMEWORKS** started with Sarah West (EPA Victoria, Australia) describing the development and current range of site and soil assessment regulations and procedures across Australian States and Territories as well as New Zealand’s Councils. This also revealed a much stronger focus on water recovery from on-site systems than the

respective regulations in Ireland. Julia Black and Robert Baldwin (London School of Economics) then introduced their GRID-GRAF concept for regulators involved in “low risk” activities (as deemed to apply to on-site wastewater treatment systems). This covered inspection and monitoring strategies and presented a methodology for risk assessment for a broad range of intervention tools (i.e. GRID) and then the use of a systematic method for assessing the success of the intervention (i.e. GRAF) and use of alternative engagement and incentive strategies. The EPA will use this methodology in order to encourage people to register their on-site systems as part of the response to the EU prosecution. Leo Sweeney and Margaret Keegan from the EPA then introduced the proposed approach for the National Inspection Plan for Domestic Waste Water Treatment Systems as a risk-based inspection regime to maximise the protection of human health and the environment from potential impacts of treatment systems. This outlined the regulatory strategy, incentive and engagement strategies and strategies for the site inspections. Seamus O’Brien (North Tipperary CoCo) then discussed some research that he had carried out by surveying all of Ireland’s Local Authorities to establish their current and proposed practices in relation to inspections of on-site systems. This revealed that many Local Authorities wanted guidance as to the relevant standards against which existing systems (particularly older sites) should be compared, as well as concerns over how such an inspection plan was to be resourced.

The next session focussed on **CONSTRUCTED PASSIVE AND INTERMITTENT SYSTEMS** opened up by Carlos Arias (University of Aarhus, Denmark) who covered 20 years of Scandinavian experiences with sand filters, willow beds and constructed wetlands. He observed that, although there had been a significant uptake in horizontal flow based constructed wetland during the 1980s and early 1990s, very few were now being built with most people currently installing vertical flow constructed wetlands as the preferred on-site system. Sean Curneen (Trinity College Dublin) reported on ongoing full-scale field research into the use of zero discharge willow treatment systems to treat on-site wastewater effluent in Wexford and Leitrim. Although the sites are in the early stages of development, the systems that had been operating for three or more years have shown the ability to provide zero discharge across the year. Niall O’Luanaigh (LaTrobe University, Australia) then reported on the results from his field research in Ireland on both subsurface flow reed beds and surface flow constructed wetlands used for on-site wastewater treatment. The research had found nutrient removal was generally poor through such systems but they did remove reasonable levels of organics and attenuate microorganisms. The subsurface flow reed beds did not significantly reduce on-site flows by evapotranspiration. Finally, Laurence Gill (Trinity College Dublin) discussed research in Ireland on stratified sand filters as well as filters using recycled glass. The results showed that sand filters can offer the same attenuation as percolation trenches over subsoil but with reduced plan areas. He also focussed on the phosphorus adsorption capacity of different naturally available media in Ireland which could be incorporated into such systems to improve phosphorus removal.

The final session of the Symposium covered **RISK ASSESSMENT / DISPOSAL OPTIONS**. Nancy Deal (Dept. Environment & Natural Resources, North Carolina) shared her experiences as a regulator and highlighted the variability of regulations across the United States. She revealed that in North Carolina they were wrestling with many similar issues to those of concern at present in Ireland (such as low permeability sites, legacy sites, maintenance issues etc). Dave Smyth (NUI Maynooth) then outlined a GIS modelling approach being developed as a strategic decision making tool for Local Authorities for sites in low permeability subsoils. The tool identifies potential problem areas and then will provide a ranked series of appropriate solutions from on-site systems to the clustering several properties together, options which can then be given more detailed consideration. Mark Livingstone from Water Regulation Group Northern Ireland discussed how the Northern Ireland Environment Agency has adopted a fairly pragmatic approach to the issue of on-site treatment and disposal (for example, allowing surface water discharges in some cases) and has been mainly reactive to any reports of unconsented domestic discharges as they are generally considered to be low risk. However, they are currently developing a risk based methodology using GIS tools in order to target a specific number of higher risks sites that will be incorporated into a wider inspection and monitoring plan. Finally, Donal Daly (EPA) finished off the Symposium explaining the EPA's proposed risk assessment approach in order to assess the relative risks to human health and the environment from DWWTs pollutants such as microbial pathogens, phosphorus and nitrogen. The approach uses hydrogeological information and GIS maps of soils, subsoils and bedrock to evaluate the movement and attenuation of pollutants arising from on-site effluent. This will allow the Local Authorities to identify areas of highest potential risk and thereby inform their strategy behind planning future inspection regimes.

Emerging issues

Several different issues emerged during the Symposium that were raised during questions to the speakers during the sessions as well as debated informally throughout the two day Symposium period, which are summarised here.

There was much discussion over the new ***inspection regime*** for DWWTs, especially concerns as to how and to what standard sites which fail the inspection will need to be upgraded. It was acknowledged that in many cases upgrades would have to be pragmatic as the sites may not be able to comply with the optimal Code of Practice standard, particularly for those sites in low permeability, high risk areas. The importance of getting the publicity right with positive messages in tandem to the introduction of the new regime was also highlighted - for example, describing the benefits to public health that the new regime is aiming to promote etc. It will be of particular importance to target dissemination at a local level via local newspapers, radio, leaflets in public health clinics etc, with clear and simple

messaging using accurate language. The messages should include how to install, use and maintain properly your on-site system and why this is important.

There was also much debate as to whether **direct discharges to surface water** (with associated consents and licensing) should start to be granted by Local Authorities, as it would appear to be the most practicable solution, particularly on low permeability sites. The consensus seemed to be that there should be allowance for such discharges depending on site specifics (size and quality of receiving water, permeability of subsoil etc). The option of **clustering** several single house systems together with a small bore sewer network down to a packaged treatment plant with a single licensed discharge to surface water also received much debate. It was acknowledged that a change in culture would be needed before this became a realistic option but consideration should be given in the future to such cluster systems as an opportunity as well as a challenge. One area would be to consider linking with the existing National Group Water Scheme Association as this operates on more or less the same principle only for drinking water. Local Authority housing schemes could also be considered to pilot the clustering concept.

There was also a lot of discussion around the issue of **phosphorus** contribution to surface water originating from DWWTS discharges. It was agreed that new approaches to reduce and remove of phosphorus from such effluent discharge need to be developed which would be appropriate for on-site scenarios (i.e. low maintenance, passive systems). There was also discussion around the issue of reducing the amount of phosphorus produced at source and how much effect the EU directives requiring the removal from washing powders have had. It seemed that more research was required on this area but again the issue should be addressed in education / publicity campaigns in order to promote changes in cultural use of such products. The issue of phosphorus however, was not seen as a major driver from the international speakers from Australia, United States and South Africa etc where **public health** seemed to be the main focus. There was a general consensus that more emphasis needed to be placed on the **maintenance** of existing systems.

Laurence Gill (January 2013)

KEYNOTE ADDRESS: ON-SITE WASTEWATER: THE HAZARDS

Chris Buckley¹

¹University of KwaZulu-Natal, Durban, South Africa.

ABSTRACT

The background to currently used water borne sanitation system is briefly described. Unless the excreta contaminated wastewater is adequately treated, it can lead to contamination of the drinking water. If the general population is healthy, this is not too serious. However if an infected person were to join the population then a serious outbreak could result. Any change in the HIV / AIDS prevalence would put increasing number of people at risk of opportunistic infection. The response of Durban, a city in South Africa, is described in order to illustrate that changing circumstances and a large unserved population lead to innovative approaches to sanitation. The link between public health, water supply, hygiene education and wastewater removal was emphasised. The opportunities presented by the Bill & Melinda Gates Foundation Water and Sanitation Reinvent The Toilet Challenge was emphasised. The objectives of the challenge were listed. Some of the innovative approaches were described. The paper concluded that change provides an opportunity for new approaches.

Keywords: on-site sanitation, pit latrines, free basic water, urine diversion toilets, community ablution blocks, reinvent the toilet challenge

INTRODUCTION

The provision of adequate sanitation is primarily one of public health. In developed countries with a healthy well educated population and functioning health services it is possible to forget the hazards associated with a breakdown in sanitation systems. It is thus instructive to reflect on poorly function systems or the situation in countries with large numbers of unserved households and to assess how close a functioning system is to collapse. While this may seem to be melodramatic, it provides an opportunity to examine extreme scenarios.

Isolated well contained decentralised systems can be intrinsically safer than large centralised systems if subject to infected users or a breakdown in the containment / treatment system.

It is in developing countries that we can expect the emergence of disruptive sanitation technologies. This is for two reasons; firstly there is a great need and potential market – 2.5 billion people; secondly there is no technology lock-in and there is no sunk-cost infrastructure that needs to be maintained..

The current western sanitation model is water based, and as has been well documented in recent years this technology predated the scientific discovery of bacteria and the germ understanding of disease. The prevalent theory of disease transmission at the time was the **miasmic theory** which held that diseases such as cholera, or the Black Death were caused by a miasma, a noxious form of *bad air* (Karamanou, et al., 2012). By discharging excreta into a water borne system, odours were reduced and the excreta was removed from the place of excretion. Without adequate wastewater treatment the water borne sanitation system reverts to institutional open defecation.

While there are many routes for infection from excreta, contaminated drinking water can be one of the most serious. This can either from the direct discharge of untreated sewage into drinking water sources, or from cross contamination of drinking vessels. Open defecation and subsequent contamination of food, fingers or eating utensils is another major contamination route.

The challenges faced by developing countries in meeting their sanitation challenges can result in innovation and non-conventional systems which may be rejected by users more accustomed to conventional systems.

As described by Jönsson et al. (2004) the annual per capita production of excreta (wet mass) is 550 kg/person,year of urine and 51 kg/person,year of faeces, while the dry mass is 21 kg/person,year of urine and 11 kg/ person,year of faeces. This amount is easily managed if it is not mixed with an average of 18 tons of water. Of even greater significance is that the nutrients are associated with the urine fraction while the pathogens are associated with the excreta. Microbiological degradation (while the excreta is moist) can further decrease the dry mass of the excreta.

Since nutrients are not retained by the body, the per capita intake of nutrients equals the per capita excretion. Hence as described by Jönsson et al. (2004), a person on a high grain diet, should be able to sustain themselves by utilising their excreta in agriculture.

The concept of *Peak Phosphorous* (Cordell and White, 2011) has provided a further interest in the separation and processing of urine.

This paper attempts to describe the provision of water and sanitation to a large backlog of households in a city in a developing country. The lack of an existing infrastructure and the magnitude of the demand allowed for innovative thinking that would not be possible in a more developed or constrained environment. The purpose of the paper is to illustrate that

every 50 or so years there is an opportunity to rethink a delivery system and to implement a technology leap.

SITUATION IN SOUTH AFRICA

At the start of the post-apartheid era in South Africa the sanitation backlog was estimated to be 21 million people (50%) did not have access to adequate sanitation facilities (DWAF, 1994). In 1996, the Department of Water Affairs and Forestry (DWAF) launched the National Sanitation Programme, which aimed to eradicate the sanitation backlog by 2010. By March 2009 the MDG Country Report for South Africa reported more than 10 million households (77%) had access to basic sanitation. A functioning basic sanitation system is defined as (Stats SA), ... *flush toilet connected to a public sewerage system or septic tank or a pit latrine with ventilation pipe* ... Nationally, as of 2010, 2.5 million households were using an unventilated pit latrine, 110 000 households were using the bucket system and 727 000 households had no toilet at all – none of these systems is considered as adequate.

The Bill of Rights, which is contained within the Constitution contains clauses that directly or indirectly refer to a right to basic sanitation. Section 24(a) of the Bill of Rights in the Constitution states that ... *everyone has a right*

to an environment that is not harmful to their health or well-being ... and (b)(i) ... to have the environment protected, for the benefit of present and future generations, through reasonable legislative and other measures that prevent pollution and ecological degradation ... This clause has often been interpreted as implying a right to basic sanitation for all (Mjoli et al., 2009).

Further the Bill of Rights states that ... *everyone has the right to have access to adequate housing...* and that ... *the state must take reasonable legislative and other measures, within its available resources, to achieve the progressive realisation of this right...*

There have been a number of court cases to interpret these rights for specific contexts, these have been reviewed by Tissington, 2011.

THE DURBAN SITUATION

Durban can be considered as a microcosm of the sanitation challenges in developing countries. It has a population of 3.6 million people (eThekweni Municipality, 2011). It was formed by the consolidation of over 20 smaller local entities. The consolidation brought challenges as the city moved from 100% coverage by sewer or septic tank to having to serve about 60 000 households with ventilated improved pit (VIP) latrines, over 90 000

households in rural areas with no water supply and sanitation, and roughly 900 000 people in informal settlements with no services.

The city is facing a potential water shortage, the lead time in providing additional water supplies are a strong driver for encouraging water saving measures (Department of Water Affairs, 2012).

The prevalence of HIV / AIDS in South Africa is very high at about 20% (WHO 2011). The provision of adequate sanitation and water supply is particularly important to these vulnerable population group.

New challenges and a large unserved population can provide an opportunity for innovative thinking.

WATER RETICULATION AND CHOLERA

An outbreak of cholera in 2000 (Mugero, and Hoque, 2001) resulted in a major campaign to supply reticulated potable water to all residents. It soon became apparent that the amount of water supplied to a site had to be related to the ability of the site to evapotranspire the water otherwise further health issues would arise. The concept of free basic water originated during this period. The current policy in this regard is that all indigent households receive free basic water which is controlled and limited to 300 l/household/d. **The universal provision of reticulated potable water inside all households is a major barrier to the transmission of disease.**

WATER RETICULATION AND DEMAND MANAGEMENT

Over time a number of different potable water reticulation systems have been developed to reduce the capital and operating cost and reduce the non-revenue water losses.

The amount of water supplied is coupled to the type of sanitation system provided and the ability of the site to adsorb the supplied water. Water losses are also a major concern, so much effort is being expended in reducing the network zonal pressure. The provision of water for fire fighting is no longer one of the criteria used for network design. Rural areas which receive free basic water are provided with an on-site storage container to which water is supplied through a solenoid valve coupled to a water meter. The system is programed to provide a maximum specific amount of water over a 24 h cycle. A similar system is being implemented for low cost housing, houses with on-site sanitation (VIPs) and for consumers that are concerned that their water bills are too high and need assistance in managing their water consumption.

Innovative systems can be installed to limit water consumption to reduce costs for all parties

WATER TARIFFS

This needs to be undertaken with a pricing policy that promotes multiple goals (equity, affordability, universal access, marginal cost recovery, long-run cost recovery, environmental sustainability). All households are supplied with a water meter which is read monthly and a monthly invoice is raised. The municipality has introduced a rising block tariff for potable water (Bailey 2003). The cost parameters are updated annually (http://www.durban.gov.za/City_Services/water_sanitation/Bylaws_Tariffs/Pages/default.aspx).

The tariff for wastewater treatment for households with the opportunity to connect to the reticulated sewerage system, are based on the volume of water supplied. This provides the consumer with a double incentive to practice water demand / water conservation measures.

Appropriately designed and updated water and sanitation tariffs can change consumer behaviour.

INTEGRATION OF WATER, SANITATION AND HYGIENE

After solving most of the water supply challenges it became apparent that for effective disease control, it was necessary for the three interventions (sanitation, water supply and hygiene education) to be applied simultaneously (Lutchminarayan, L., 2007).

There is no room for silo thinking.

URINE DIVERSION TOILETS

The challenge of emptying the estimated 60 000 VIP latrines that had filled resulted in the municipality investigating other sanitation systems for people in the rural areas. The dry on-site system of urine diversion was piloted for a few years prior to the full-scale rollout of 75 000 installations. In this system the urine is diverted to a soak away and the faeces are collected in a vault below the pedestal. The provision of two vaults enables the pedestal to be moved from vault to vault and provide a year for the faeces to dehydrate. The dehydrated faeces are manually removed by the householder using implements supplied by the municipality.

The segregation of urine and faeces has resulted in the municipality realising the advantages of urine separation in more conventional (sewer) situations. Trials are underway in installing

No-mix pedestals in public buildings. A research project is investigating the recovery of nutrients from urine (discussed later). If successful it will interface with city food garden projects for poor and unemployed residents.

Source separation leads to new opportunities.

VENTILATED IMPROVED PIT LATRINES

45 000 VIP toilets have been desludged after they had filled up over a period of about 14 y. After a few years of trials, a successful system has been devised to use local contractors to empty the pits and transport the material to a central processing site. The municipal engineers have developed a plant in which the material is pelletised, dehydrated and then pasteurised. Trials are underway to evaluate the agricultural potential of the pellets. Systems will be trialed whereby a continuous pit emptying service will be provided. An economic evaluation of the system is underway.

The development of pit emptying techniques and a sludge processing plant could be a game changer with regard to the rejection of VIP latrines.

A LEARNING INSTITUTION

The municipality has encouraged a culture of innovation, piloting and perseverance in attempting to solve new challenges. It has provided resources to enable concepts to be piloted and modified before limited full-scale roll-out. Further, resources have been provided to fix and improve latent design or construction faults. In taking this approach the city has partnered with many institutions (all the universities in the city) and organised business. It has opened up its operations and installations to researchers from around the world. It has created a formal learning institution within the municipal structure – Municipal Institute of Learning – MILE to partner with these external bodies (<http://www.mile.org.za>). MILE is also the vehicle by which the municipality shares its knowledge. Engineers from Durban are assisting a number of African cities improve their water and sanitation systems.

To be a learning organisation requires two-way communication.

CURRENT CHALLENGES

As of June 2010 the sanitation backlog was 138 569 households, which will be met in 15 y at current delivery rates. The city is cursed by its own success in that there is an increase in population due to an influx from surrounding areas of about 10% per year. The backlog is made up of 17 500 households which will receive urine diversion toilets. Approximately 900 000 people live in urban or peri-urban informal settlements. Sanitation in these

circumstances consists of community ablution blocks which serve up to 75 households or are within 200 m of a dwelling. These community ablution blocks could be within the *water borne edge* (ie within the sewered area) or outside the sewered area. The challenge faced for providing sanitation to the latter is to provide an appropriate treatment system.

A further challenge is to provide sanitation to new formal subsidised housing which is situated outside the *water borne edge*. In order to reduce the cost of land and services the erf size needs to be as low as possible. This will also reduce the construction of informal shacks on the sites. The area of the erf is too small to enable the wastewater to be disposed by evapotranspiration. At the planning stage is the construction of 250 houses on 300 m² sites connected by a gravity sewer to an anaerobic baffled reactor, a constructed wetland and an associated agricultural area. If successful, the next phase will be for about 2 500 houses.

Limited high-strength treated wastewater could be of major benefit to urban agriculture while solving wastewater disposal, unemployment and food security problems.

FUTURE DIRECTIONS

BILL & MELINDA GATES FOUNDATION SANITATION PROGRAMME

The Bill & Melinda Gates Foundation have initiated a Sanitation Programme (<http://www.gatesfoundation.org/watersanitationhygiene/Pages/home.aspx>) the budget for 2013 is in the region of USD 65 million. The target are the 2.5 billion people without improved sanitation. There are a number of sub-programmes the most exciting are the Grand Challenges (up to USD 100 000 per project) and the Reinvent the Toilet Challenge. to leverage advances in science and technology and create a new toilet that will transform waste into energy, clean water, and nutrients. The Reinvent the Toilet Challenge aims to achieve the following goals:

- Address the failures of the 18th-century toilet, which is not meeting the current needs of 2.5 billion people who lack access to sanitation
- Devote funding and attention to the need for a new toilet
- Generate innovation among a wider research and development community
- Support upstream research and development of a toilet that:
 - » Is hygienic and sustainable for the world's poorest populations
 - » Has an operational cost of \$0.05 per user, per day

» Does not discharge pollutants, but instead generates energy and recovers salt, water and other nutrients

» Is designed for use in a single family home

- Create a toilet that does not rely on water to flush waste or a septic system to process and store waste
- Create a toilet that is the basis for a sanitation business that can be easily adopted by local entrepreneurs living in poor urban settings
- Raise awareness about this research by publishing scientific papers in journals and articles in various media outlets

The above *stretch* targets had to be met in just over 12 months. A Toilet fair was held in Seattle in August 2012 where the 8 contestants (and a further 42 Grand Challenge grantees) demonstrated their prototypes. Although no entry satisfied all the requirements a range of innovative systems were demonstrated. Processes included fuel cells, plasma gasifiers, hydrothermal processes, solar powered systems, membrane processes and incineration. Associated developments included surface treatment to reduce (prevent) surface soiling, provide permanent disinfection using nano-particles of silver and coupled with ultra violet laser activated titanium oxide impregnated pedestals.

Many of these and other developments will receive a second round of funding and field evaluation. Negotiations are in progress to provide a field site in Durban to allow international researchers a platform to evaluate their processes using realistic raw materials.

Gates Foundation funded sanitation research projects in Durban include:

- investigating mechanisms of sludge breakdown in VIP toilets,
- evaluation of additives to accelerate the breakdown of contents in VIP toilets,
- the recovery of nutrients from urine,
- the links between menstrual management and sanitation,
- economic assessment of processes for the final disposal of VIP sludge,
- mechanical properties of faecal sludges,
- reinvent the toilet challenge

DISCUSSION AND CONCLUSIONS

By breaking the direct link between water supply and the transport of excreta a whole range of possibilities present themselves. One of the biggest vectors for the spread of disease is removed. Possibilities are presented to obtain energy and nutrients from the resulting excreta.

Source separation and the reduction or elimination of water from the sanitation system will have a major benefit in preventing institutional open defecation.

The resources (financial and intellectual) that are now being applied to the reinvention of the toilet are going to provide a major breakthrough in how we address sanitation issues in the future.

In conclusion the engineers at eThekweni Water and Sanitation did not know where they were going when they first took up the challenges associated with the cholera epidemic and large backlog of unserved households. Even at this stage it is not certain what the eventual system or systems will look like, but what is certain is that innovation and disruption have become a habit and that they look forward to the re-invented toilet. As a cholera epidemic in London led to the modern discipline of public health, the Durban cholera epidemic had led to local innovations in the provision of water and sanitation services.

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SESSION I CATCHMENT STUDIES

IMPACT OF ON SITE SEWAGE TREATMENT SYSTEMS ON RIVER WATER QUALITY IN UK CATCHMENTS

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ABSTRACT

Until recently, the impact of effluent discharges from on-site waste water treatment systems (STS) within the UK was believed to be negligible. This was mainly because the number of systems was unknown. When a range of different methods were used to estimate the number and location of STS more accurately, it was found that the number of consented systems was probably less than 10% of the total number of systems in many areas and registration schemes were introduced. At the site specific level, that discharges from these systems can have a high impact on downstream water quality can be high. In-stream phosphorus (P) concentrations can be up to four times higher downstream of STS than upstream. At the catchment scale, the likely impact of STS on downstream water quality is usually estimated from information on consented STS, only. When all STS are taken into account, estimated contributions from agricultural sources may be up to 20% lower than previously thought.

Keywords: on-site sewage treatment systems, septic tanks, phosphorus

INTRODUCTION

Until recently, discharges from on-site waste water treatment systems (STS) and other small rural sources of nutrient export to nearby water courses had rarely been documented within the UK. Indeed, there was a widely held belief that septic tanks were not a problem, especially at the landscape scale. However, mounting anecdotal evidence is emerging to

suggest that these sources may be causing water quality problems and warranted closer investigation.

In 2010, during a project funded by Natural England, May et al. (unpublished) compiled a wide range of case studies that supported this assertion. Many of these are summarised below. Some provide evidence of direct discharge whilst others show that exceptionally large discharges from such sources may be driven by high rainfall events. This is an important observation because this type of event is often missed by routine monitoring and may become increasingly common with climate change. Other case studies from across Europe, but not documented here, also provide evidence of the potentially large impacts that STS can have on water quality in rural headwater streams, especially during the ecologically-sensitive low flow conditions (Withers et al., 2012).

NUMBER AND LOCATION OF SEPTIC SYSTEMS WITHIN THE UK

There are an estimated 1.4 million STS across the UK, many of which are very close to watercourses. Most discharge to a soakaway, but some discharge directly to water *via* a discharge pipe or drainage ditch. In spite of their large numbers, the exact locations of STS across the UK are, for the most part, unknown.

In recent years many different methods have been used to derive their numbers and locations from existing datasets. Some of these are described below.

Postcode method: One method that has been used to estimate the number and location of STS is the 'postcode' method, which was originally developed and applied to the catchment of Bassenthwaite Lake (May et al., 1999). This method involves removing dwellings that are connected to mains sewerage systems from a 'master' list of all dwellings in an area, making the assumption that the remainder are served by STS. This method has potential for widespread use over large geographical areas if appropriate data on sewer connections are available. For example, it has been used to approximately locate septic tanks within the catchment of Loch Leven, Scotland, (Dudley *et al.*, 2007) and across the whole of Scotland (SNIFFER, 20068).

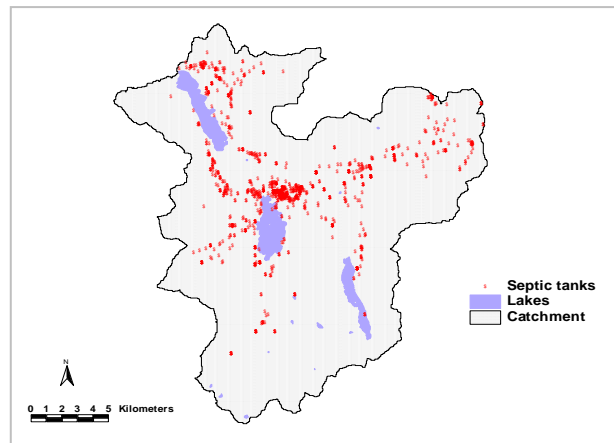


Fig. 1. Estimated location of septic tanks within in the catchment of Bassenthwaite Lake (*after May et al., 1999*).

Sewerage network method: Another method of estimating the number of septic tanks is described by Hilton *et al.*, unpublished. This involves using sewer system network diagrams to derive the area of a catchment that is served by the mains sewerage system. The method assumes that premises that are outside sewered areas are connected to STS. Although effective, this method is difficult to use in practice because utility companies may be unwilling to disclose the necessary information about their sewer networks because of its commercial value and security implications. Also, it cannot necessarily be assumed that all properties within an area served by a mains sewerage system are connected to that system.

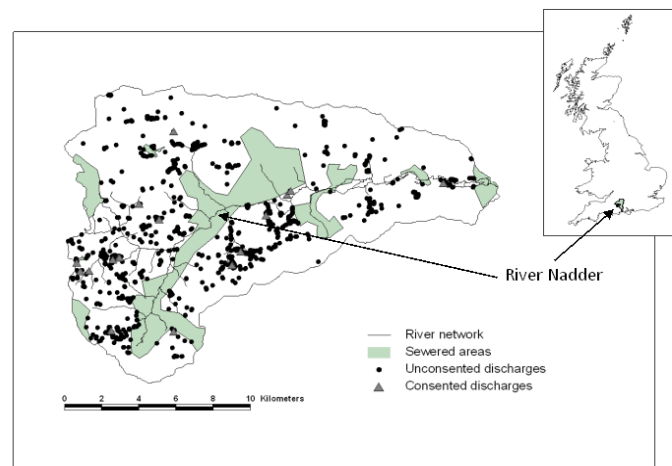


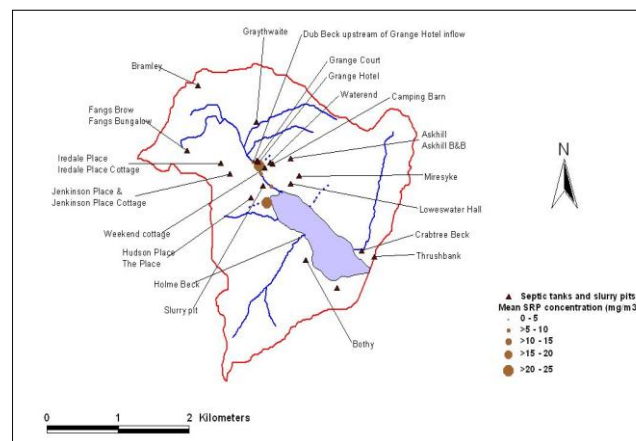
Fig. 2. The catchment of the River Nadder, Wiltshire, showing sewered areas, and consented and unconsented STS (*After Withers et al., 2012*).

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Aerial photography method: May et al. (unpublished) derived the number and position of STS within their study catchments from aerial photography. Each 'house' was digitised from the aerial photography using Erdas Imagine® and the authors made the assumption that residential properties ('houses') that were within these rural areas but beyond the area served by a mains sewerage network were had STS.

The location of these unsewered 'houses' in the River Nadder catchment is compared to the number consented STS discharges in Figure 2. Overall it was found that less than 1% of STS



in this area were consented. A similar pattern emerged from studies undertaken in three other rural catchments.

Fig. 3. Location of septic tanks within in the Loweswater catchment based on local knowledge (after Maberly *et al.*, 2006).

Local knowledge method: For small catchments, STS can be located by harnessing local knowledge. This method has been used to locate STS within the catchment of Loweswater, Cumbria (Maberly *et al.*, 2006; Figure 3) and in the River Clun catchment, Shropshire (Fildes, 2011). However, this method is not practical for application to large catchments that cover a wide area.

Large area statistics method: A recent re-analysis of data compiled by Faber Mausell (2003), Anthony et al. (2006) and Stapleton et al. (2006), has shown that, at the larger scale, the approximate number of STS in any given area can also be derived from nationally available datasets (Anthony, *pers. comm.*). The data used in this study comprised:

1. *For Northern Ireland:* information on septic tanks usage from 1991 population census returns
2. *For Scotland:* properties located within Postcode Sectors across Scotland, as derived from an OS Address Point database, and outside of a sewered area
3. *For North West England:* information on properties known to be using septic tanks from local water company data

These analyses were performed at district council and postcode sector level and the relationship between property density and percentage connection has not been validated for application elsewhere.

Although there are large uncertainties within these data, a clear relationship was found between the percentage of properties that are not connected to mains sewerage systems and the density of properties (Figure 4). Although this method does not provide details of the exact locations of individual properties and is probably too coarse for application at a site or catchment specific scale, it does provide a way of estimating the number of properties that are served by STS at the regional or national scale.

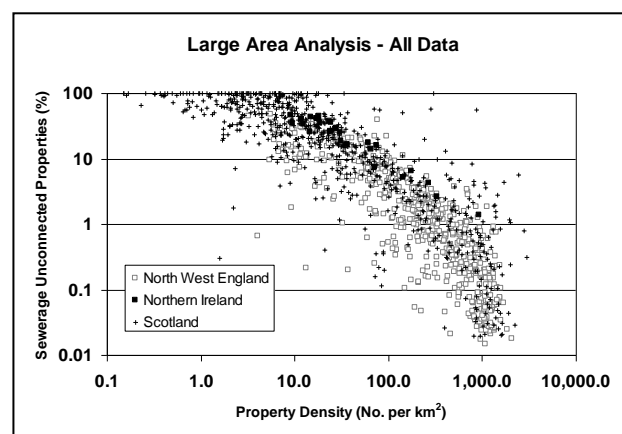


Fig. 4 Relationship between the percentage of properties using STS and the density of properties in a given area (*After Anthony, pers. comm., ADAS UK Ltd.*).

Although there are large uncertainties within these data, a clear relationship was found between the percentage of properties that are not connected to mains sewerage systems and the density of properties (Figure 4). Although this method does not provide details of the exact locations of individual properties and is probably too coarse for application at a site or catchment specific scale, it does provide a way of estimating the number of properties that are served by STS at the regional or national scale.

Improved registration: Under the Control of Pollution Act 1974 (CoPA) all sewage discharges to surface waters in Scotland required consent from the Scottish Environment Protection Agency (SEPA). However, in most cases, there was no requirement to obtain consent for STS discharges to soakaway. This led to incomplete records of septic tank

locations. Since 1 April 2006, there have been significant changes to the control of sewage discharges following the introduction of the Controlled Activity Regulations 2005 (or CAR). Under these new regulations, all new STS discharges from domestic properties serving less than or equal to a population equivalent (PE) of 15 are required register with SEPA. For population equivalents greater than 15, a discharge licence is required. Although SEPA discourages direct sewage discharges to waterbodies, where this is unavoidable these must also be registered.

A recent change to legislation within Scotland aims to address the problem of unconsented discharges retrospectively at the national scale. Since April 2006, all septic tanks must be registered with the Scottish Environment Protection Agency (SEPA) when properties change ownership. Over time, this will create a record of the size, location and discharge of all septic tanks in Scotland (SEPA, 2006).

A STS registration process was later extended to England and Wales as a result of new regulations introduced by the Department for Environment, Food and Rural Affairs (Defra) and the Welsh Government to meet the legal obligations imposed by the European Union Water Framework Directive. However, subsequently, compulsory registration has been suspended in England pending the outcome of a review of the need for such legislation. This is currently being undertaken by the Environment Agency and Government. In contrast to the situation in England, compulsory registration continues in both Scotland and Wales.

IMPACTS OF STS DISCHARGES ON WATER QUALITY

The impacts of STS discharges on water quality at the catchment scale can be demonstrated by the results from a wide range of studies. Such studies include upstream and downstream water quality monitoring and catchment 'hotspot' surveys. Examples of the type of evidence collected is summarised below.

Eye Brook: The first example is part of a study on the Eye Brook, Leicestershire (Withers et al., 2011). Samples were taken upstream and downstream of two STS locations within the catchment, Village East and Loddington North, October 2006 to October 2007. In general, median soluble reactive phosphorus (SRP) concentrations were found to be three to four times higher downstream of STS locations than upstream, while median total phosphorus (TP) concentrations were approximately double upstream of these locations where farming was the only source of P (Figure 5). The results of this study show that STS can affect water

quality in rural areas, especially if the effluent is being discharged directly into a stream. The results suggest that many STS are acting as direct point source inputs to headwater streams where there is very low dilution capacity. This causes degradation in water quality. The authors concluded that failure to take account of nutrient emissions from STS may undermine attempts to improve the ecological status of freshwaters by focusing control measures on major sewage treatment works and agriculture.

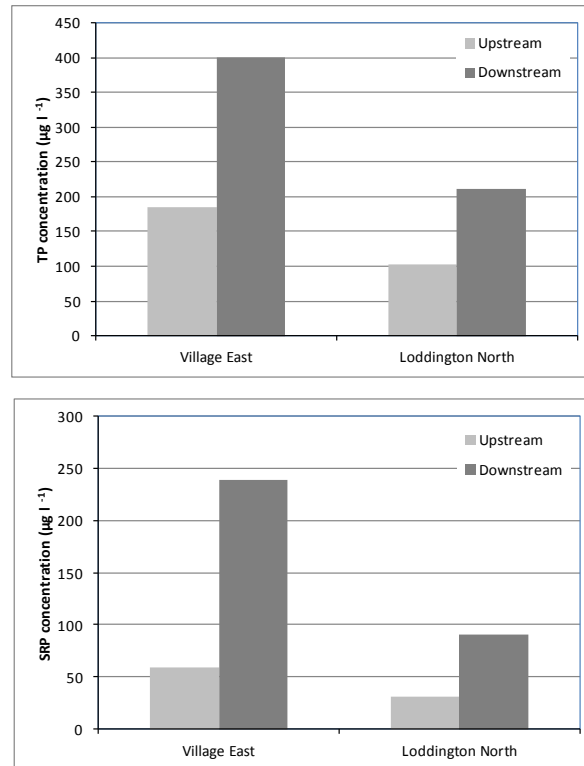


Figure 5: Median concentrations in total phosphorus (TP) and soluble reactive phosphorus (SRP) upstream and downstream of STS discharges at (a) Village East and (b) Loddington North.

River Wyre: The second example is from a study of the River Wyre catchment, Lancashire (Nicholson, 2007). This study aimed to determine whether septic tanks contributed to SRP concentrations and loads in a stream that ran close to a small cluster of houses served by a STS. By measuring flows and concentrations along this stretch of the stream, the author was able to detect a marked increase in P concentrations and loads downstream of the STS, which was between sampling sites 4 and 5 (Figure 6). In-stream concentrations rose from about $50\mu\text{g P l}^{-1}$ to about $400\mu\text{g P l}^{-1}$ over a distance of less than 100 m. As there were few other possible sources of P in this area, it was concluded that the sudden increase in P in the stream was attributable to STS discharges.

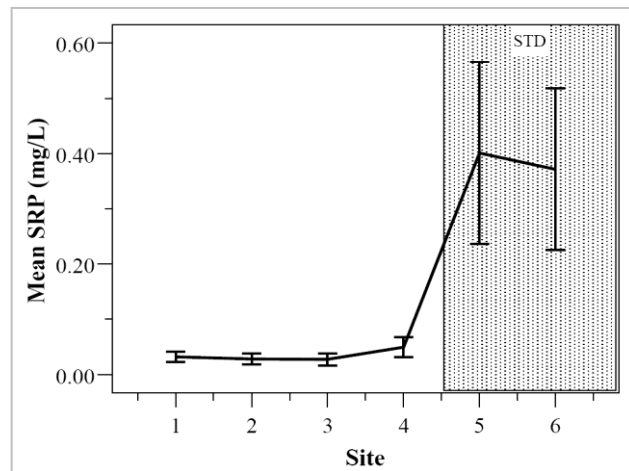


Figure 6: Mean SRP concentrations along a short stretch of river within the Wyre catchment showing a marked increase between sites 4 and 5 (After Nicholson, 2007).

Loweswater: The third example is from a small feeder stream that flows into Loweswater, Cumbria (Figure 7). Here, OP concentrations were monitored from October 2004 to September 2005 (Maberly *et al.*, 2006). Although annual mean OP concentrations in most of the inflows to the lake were low (i.e. $< 10 \mu\text{g P l}^{-1}$), this stream had a much higher OP concentration (i.e. $\sim 24 \mu\text{g P l}^{-1}$). Further investigation showed that the stream was receiving effluent from a faulty septic tank.

When combined with stream discharge rates, the mean daily OP load from this tank over the whole year was estimated to be approximately 8 g P d^{-1} or 2.9 kg P y^{-1} . However, during a storm event in December 2004, a single value of 122 g P d^{-1} (i.e. about 4% of the annual P load) was recorded. This highlights the importance of rainfall driven discharge events in delivering nutrients to watercourses from some STS. In this case, this was a STS of outdated design that received roof runoff as well as sewage and, therefore, tended to overflow during heavy rainfall.

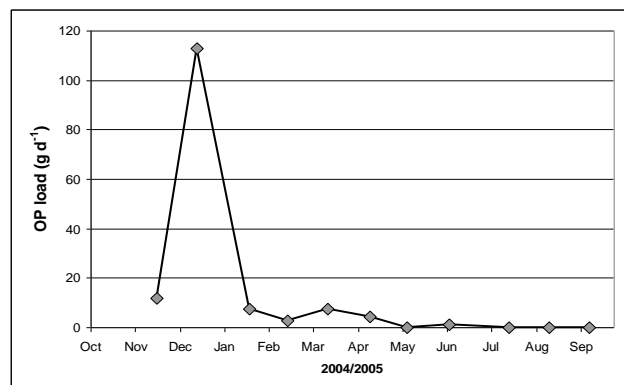


Fig. 7 Seasonal changes in orthophosphate (OP) loads in a small inflow to Loweswater in 2004/2005 (After Maberly *et al.*, 2006)

Hornsea Mere: Pollution impacts of STS at a wider spatial scale can be assessed by 'catchment walks', whereby samples are collected at a range of sites over a wide area to identify 'Hotspots' where phosphorus concentrations are high (e.g. Withers *et al.*, 2009).

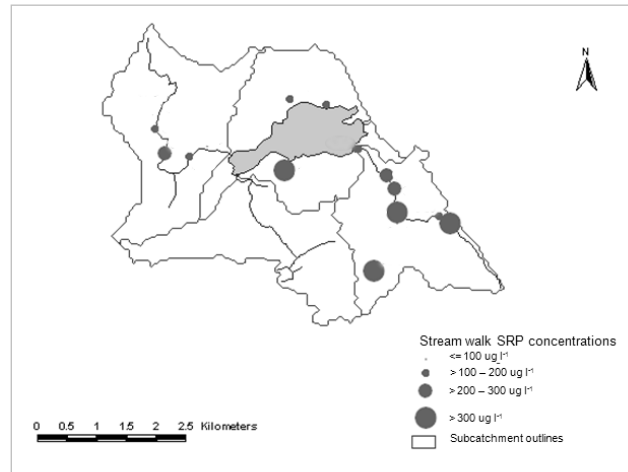


Figure 8 Orthophosphate concentrations ($\mu\text{g P l}^{-1}$) within the catchment of Hornsea Mere (After May *et al.*, 2010).

This type of survey was undertaken across the catchment of Hornsea Mere, Yorkshire by May *et al.* (2010). Orthophosphate (OP) concentrations were found to be very high in many locations, with values of $200 \mu\text{g P l}^{-1}$ to $300 \mu\text{g P l}^{-1}$ commonly recorded just downstream of known STS (Figure 8). An exceptionally high value of more than 2 mg P l^{-1} was recorded just downstream of one particular STS. This study demonstrates how widespread P pollution from STS may be in some rural areas.

Blackwater River: In addition to evidence at the site specific level, outlined above, Arnscheidt *et al.* (2007) showed that the impact of STS discharges on water quality can also be detected at the catchment scale. Their study involved a survey of the STS in three rural tributaries of the Blackwater River, Northern Ireland, together with high frequency (10 minute intervals) monitoring of TP concentrations at the catchment outlets.

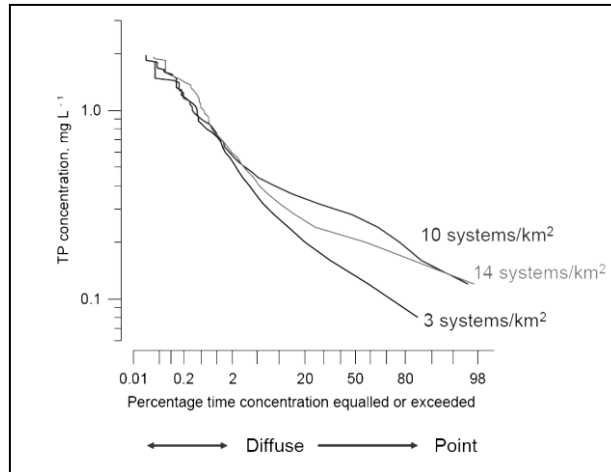


Figure 9: Percentage of time in which the total phosphorus (TP) concentration in three streams at each catchment outlet was greater than or equal to a given threshold concentration (y-axis) in comparison to the upstream density of STS (After Arnscheidt *et al.*, 2007).

Arnscheidt *et al.* (2007) showed that a range of threshold TP concentrations were exceeded more frequently in catchments with higher densities of STS than those with lower densities (Figure 9). They also found that more than 60% of these tanks were at high risk of causing water pollution because of their condition, management and location.

ESTIMATING P LOSSES FROM STS TO WATER AT THE CATCHMENT SCALE

It is relatively easy to estimate the amount of P that enters STS from domestic waste. However, it is much more difficult to determine the amount of P that is ultimately discharged into the environment after processing within the tank and retention within the drainage field. This is because there are few measured values and the level of discharge from a given system depends on a range of local environmental condition (May *et al.*, unpublished).

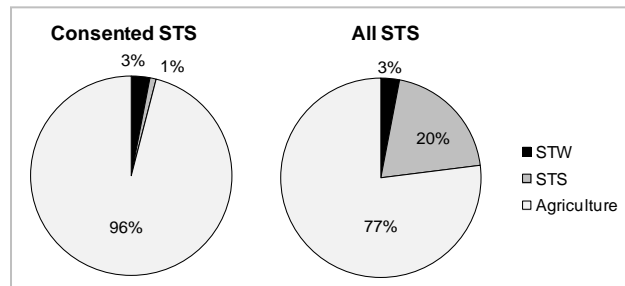


Figure 10: Source apportionment (%) of the total P load in the River Bure, without and with unconsented STS. P sources are: sewage treatment works (STW), on site waste water treatment systems (STS) and agriculture.

Nevertheless, one of the greatest uncertainties in the estimation of P losses to water at the catchment scale is the number of STS (Withers et al., 2012). Using case studies from the Norfolk Broads and Hampshire Avon, May et al. (unpublished) demonstrated that existing records of STS in many areas reflected less than 10% of the actual number of tanks. So, it was concluded that most estimates of the amount of TP emanating from these systems at the catchment scale was probably less than 10% of the actual value. Using this information and applying a simple *per capita* export coefficient approach, the authors compared estimated TP outputs from consented STS within several rural catchments with those from all STS within those catchments.

The results were used for the source apportionment of TP within the receiving rivers. If all STS were taken into account, the amount of TP in the receiving waters that was estimated to be coming from agricultural sources appeared to be up to 20% less than if consented STS, only, were taken into consideration (Figure 10).

DISCUSSION AND CONCLUSIONS

The number of STS across the UK has been significantly underestimated in the past. Recent studies have suggested that, if all STS are taken into account, P discharges from these sources may have a significant impact on downstream water quality. Also, when the total number of STS is used in source apportionment calculations, the apparent P input to the drainage system from agricultural sources may be reduced by up to 20%. This provides an important insight into where mitigation measures should be focused.

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RURAL POINT SOURCES: SMALL PORTIONS OF PIE AND P IN HEADWATER CATCHMENTS

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ABSTRACT

The phosphorus loads from rural point source effluents are commonly cited as minor proportions of annual inventories owing to their dispersed nature and low per capita losses. In some regions, however, the persistence of these sources is evident at lower river flows when dilution capacity is reduced and when the potential for loss from diffuse sources is minimal. Evidence from intensive monitoring in some Irish headwater catchments shows this to be a chronic feature; maintaining small streams in a highly eutrophic state during ecologically significant periods. If the monitoring is intensive enough, diurnal signals of phosphorus loss appear to indicate patterns of water use in catchments. Irish studies also indicate that these low flow patterns can be related to the density and condition of septic tank systems upstream of monitoring points. Particular challenges from these studies show that, when condition is mitigated, density can still pose a significant problem to water quality in catchments of low soil permeability. This is due to the risk of low soil attenuation of effluents and/or reduced assimilative capacity in receiving headwaters due to suppressed summer baseflows.

Keywords: low flows, catchments, phosphorus dilution

INTRODUCTION

Rural point sources of pollution in the landscape are differentiated from industrial and municipal point sources at larger river basin scales, and are generally considered to be smaller contributors of pollution in annual budgets. Two of the main rural point sources in Irish landscapes, and which can contribute to nutrient enrichment of receiving waters and pathogenic bacterial contamination from faecal matter, are farmyards and domestic waste water treatment systems (DWWTS).

Taking phosphorus (P) as an example of a nutrient pollution parameter, rural point sources are accounted for at large river basin and regional scales alongside diffuse sources such as

those transferred from agricultural soils and following slurry and other fertiliser applications. In a P budget study, for example, Smith et al. (2005) estimated that the point source contribution to P loads to inland waters in Northern Ireland was 30% of the total. Of this, just 7% was considered to be contributions from DWWTS (reported as rural septic tanks) and estimated from the product of the rural population and a coefficient based on total P production per person and a DWWTS retention factor (i.e. mean losses from DWWTS of $0.44 \text{ kg person}^{-1} \text{ yr}^{-1}$). Of the 70% reported as P loads from diffuse sources, 59% were estimated to be from diffuse agricultural sources, and largely based on a landuse export coefficient approach (Fig. 1).

Other studies have noted similar annual contributions from DWWTS in catchments of varying scale (e.g. Withers et al. – in press). It is postulated that this low percentage contribution is possibly a reason for the low importance attached to DWWTS mitigation and management in recent years as compared, for example, to municipal or agricultural sources of pollution. Here, controlling EU directives have been enacted resulting in considerable investment in sewerage and farmyard infrastructure.

However, with growing anecdotal and scientific research evidence of surface headwater stream impact (Arnscheidt et al., 2007; Greene et al., 2011; Withers et al., 2011; May et al., 2012 - these proceedings) and, in the Republic of Ireland at least, legal expedencies to properly manage DWWTS at a national level, mitigation of pollution from these systems has an increasing focus.

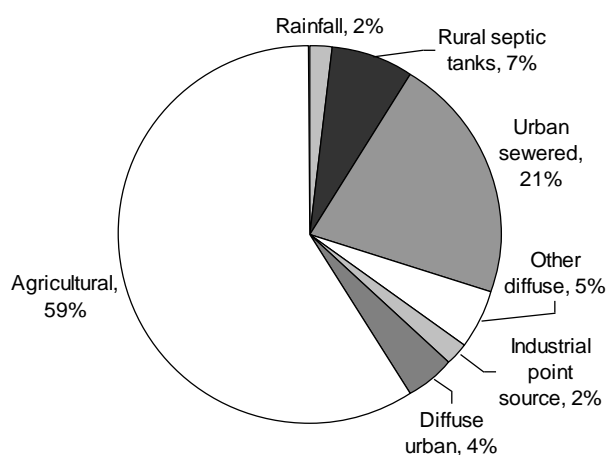


Figure 1: Pie chart of total P budget to inland waters of Northern Ireland taken from Smith et al. (2005) and showing a small 7% contribution from DWWTS (reported as rural septic tanks).

IDENTIFYING THE IMPACTS FROM SMALL P LOADS

Defining the impacts from DWWTS effluents can be difficult owing to variations in effluent treatment and retention (largely due to DWWTS type, age and condition) and soil type. Thus, the mean effluent export per person (for example cited above for P) is likely to have large margins of error. There will be similar uncertainties as catchment scale changes and where clustering of DWWTS has occurred in the landscape through housing developments. However, the small loads reported throughout the literature require effective dilution in receiving rivers to offset trophic or other pollution issues.

If it is assumed that water use in rural housing is equal to water waste and, during periods of low flow, risk of pollution (effluents reaching a surface water course) is manifested as changes in water chemistry several methods can be used to indirectly assess the potential trophic impact of DWWTS.

A simple method is a longitudinal survey where sampling of water along river reaches will highlight if a point source issue is present. Changes at specific points or reaches can then be compared with potential catchment sources. Examples of this technique are reported in Arnscheidt et al. (2007) in Northern Ireland (Fig. 2) and in Melland et al. (in press) in the Republic of Ireland.

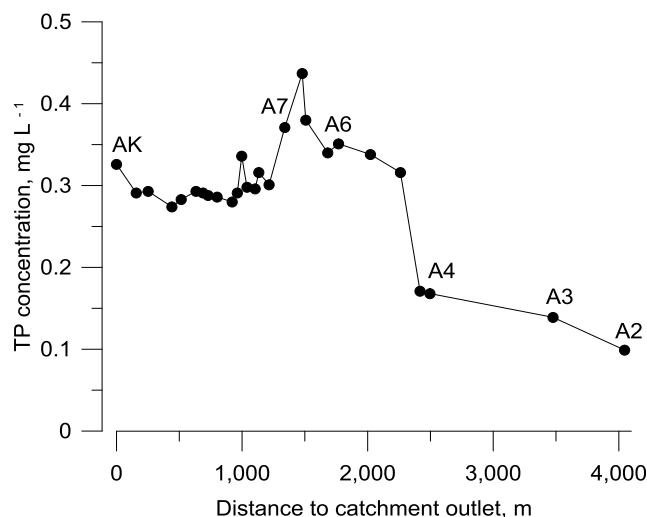


Figure 2: Longitudinal river survey in a 3km² Co. Armagh headwater catchment stream during summer low flow reported in Arnscheidt et al. (2007). Black circles are sample locations upstream from the outlet (AK) and the magnitude of total P concentration. Locations AK to A2 are locations where sampling was repeated throughout the summer low flow period.

One of the striking features in Figure 2 is the maintenance of very high total P concentrations following the rapid rise between locations A4 and A6 and which were coincident with clustering of rural housing very close to the stream. Nevertheless, using the coefficient cited in Smith et al. (2005), above, the annual P export from DWWTS in this catchment was estimated to be 39 kg compared with a total of 543 kg (Macintosh et al., 2011) – or 7%. This is the same (in this year) as the small proportion of the pie chart in Figure 1 for the whole of Northern Ireland but with high P concentrations at the small catchment scale due to the low baseflows typical of these headwater systems in the Irish border soils.

Another indirect method of assessing impact risk is to link catchment housing, where not served by central waste water treatment, with metrics of water quality. In the same Co. Armagh catchment highlighted in Figure 2, a catchment survey on the condition and number of DWWTS was undertaken and compared in each sub-catchment downstream of the monitoring points (AK to A2) where an average low flow total P concentration was measured during a summer season. The catchment survey was undertaken by independent engineers (Hyder Consulting Ltd) using a framework that scored a system up to 100 (worst case scenario) and using criteria that included type of system, method of discharge and maintenance/operation (Arnscheidt et al., 2007). The positive relationship noted in this catchment (Fig. 3) was offset in other catchments only by conditions of stream gradient which would most likely have attenuated effluent P during low flows (but likely to become mobile again at higher flows).

In further work, the pattern of low flow P concentrations as monitored by high-resolution equipment has revealed diurnal patterns when baseflows establish and this has also been interpreted as signals of water use and effluent transfer to receiving streams (Jordan et al., 2005; Jordan et al., 2007) (Fig. 4) but which only appears where dilution capacity is minimal (Jordan et al., 2012).

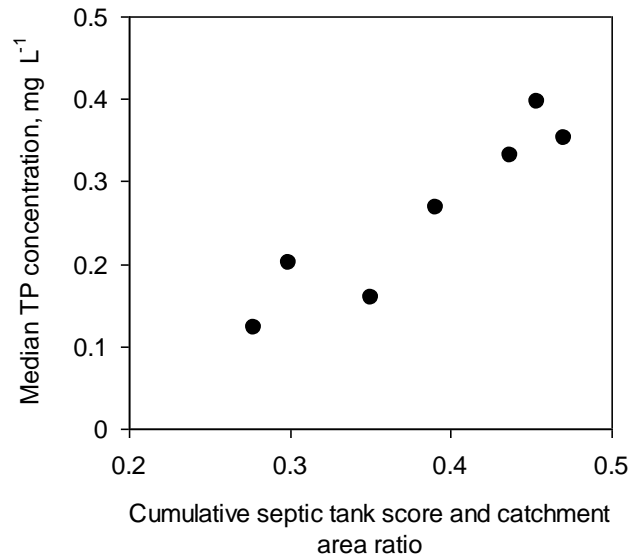


Figure 3: Relationship between sub-catchment cumulative DWWTS condition and average summer total P concentration during low flows in a 3 km² headwater catchment in Co. Armagh. Each datapoint corresponds to the locations (AK to A2) in Figure 2 (from Arnscheidt et al., 2007).

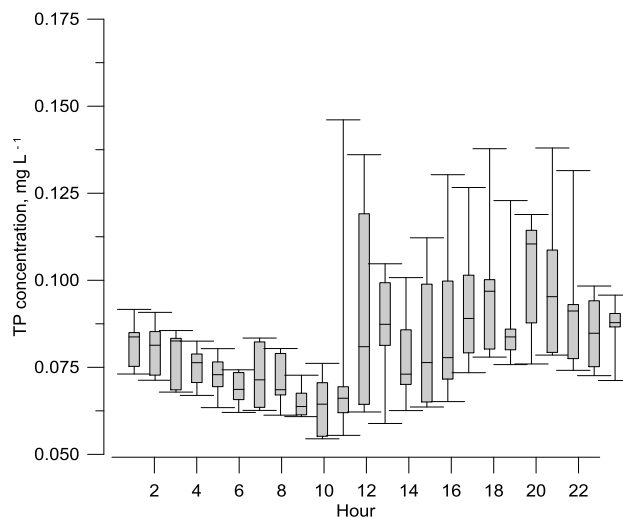


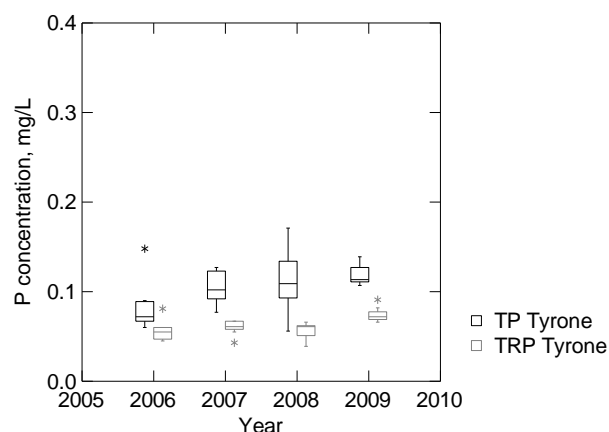
Figure 4: Several days of overlaid total P concentration from sub-hourly data in a 5 km² Co. Tyrone catchment stream during days with zero rainfall. The pattern appears consistent with domestic water use and effluent pollution with recoveries during the night (from Jordan et al., 2007).

The above are examples highlighting the magnitude of nutrient pollution and linking to the potential for DWWTS contamination of receiving catchment streams – through deduction. The premise is that the potentially constant transfer of effluent from multiple DWWTS rural point sources can elevate the background low flows of headwater streams and which is most potent during the summer period, when water temperatures are high, and so will

intensify a eutrophic impact. However, consistent with the budgets described in Figure 1, diffuse P transfers overwhelm the signal during rainfall-runoff events. While there is little doubt that these diffuse events have a large agricultural P contribution from residual soil and fertiliser losses, a particular challenge will be to untangle the contribution of DWWTS transfers during high flow events as it has been proposed that damaged or poorly operated systems may not be isolated from storm runoff (Bowes et al., 2010; May et al., 2012 – these proceedings).

MITIGATING POLLUTION FROM DWWTS

In a project described in Jordan et al. (2008), defective DWWTS in three Irish border sub-catchments of the Blackwater River (including the two catchments highlighted in Figs. 2, 3 and 4) were replaced subsequent to two years of high-resolution P monitoring. The new systems consisted largely of modern package plants with above ground polishing filters and chosen to replace the most defective according to voluntary uptake and budget constraints. The filter option was recommended following consultation as below-ground percolation was deemed unsuitable following standard tests (despite traditional DWWTS being installed previously). Following two further years of high-resolution P monitoring, it was assumed that the clearest signal of mitigation effect would be during comparative low flows between years. Data were extracted as hourly average concentrations at similar low flows and compared (Macintosh et al., 2011) (Fig. 5).



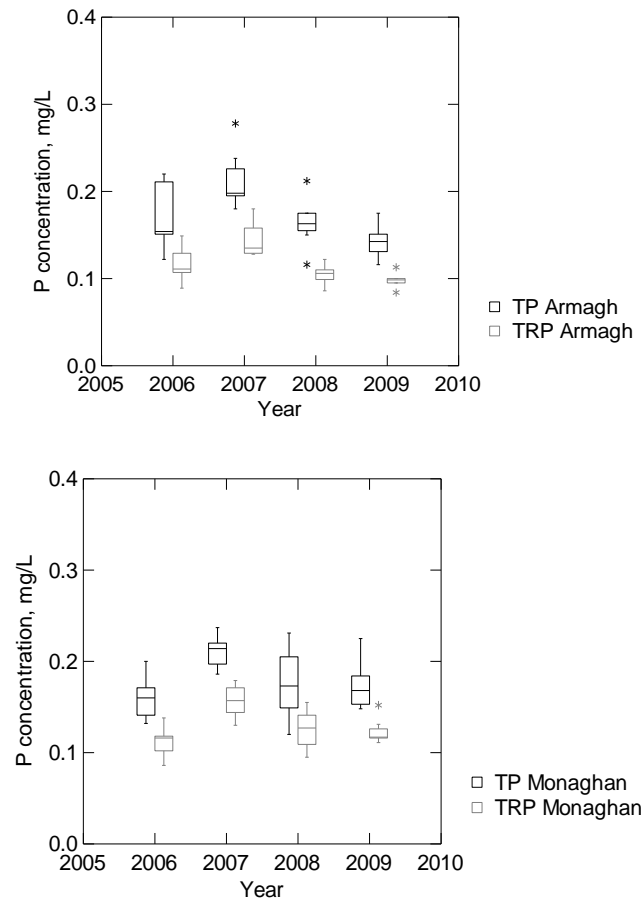


Figure 5: Comparisons of hourly average total P and total reactive P concentrations over four years in low flow discharges in three Irish border area sub-catchments. Statistically there appeared to be an increase in Co. Tyrone concentrations, a decrease in Co. Armagh concentrations and a small increase in Co. Monaghan concentrations (from Macintosh et al. 2011).

These low flow comparisons indicated variable mitigation success of replacement systems with a significant decrease in P concentration in the Co. Armagh catchment at low flows and an actual increase P concentration in the Co. Tyrone sub-catchment – also with a small increase in Co. Monaghan.

In a subsequent independent follow-up survey of the number of DWWTS in each sub-catchment to account for any new builds following the installation of new systems it was found that the Co. Armagh sub-catchment had the least number (one extra) of new builds over the period but that the other two catchments had increased significantly (Table 1).

Table 1: Survey of DWWTS ('systems') in three Irish border area sub-catchments in 2005-2006 (with replacement number of defective systems) and showing a large increase in numbers in two sub-catchments from a re-survey in 2010 (from Macintosh et al., 2011).

	Tyrone	Monaghan	Armagh
Catchment area, km ²	5	5	3
2005-2006 no. of systems surveyed	17	69	29
No. of systems chosen for replacement	7	11	10
2007-2008 no. of systems replaced	4	11	3
No. of additional systems since 2005-2006	6	17	1
2010 no. of systems surveyed	23	86	30

The overall density of DWWTS between each sub-catchment (as well as condition as accounted for by individual surveys within sub-catchments) appeared to be an over-riding influence in the control of low flow P concentration and this possibly indicates that either DWWTS catchment carrying capacity is not properly accounted for at the scale in question or *a priori* discharge and/or chemistry statistics are insufficient for planning at this scale (Fig. 6).

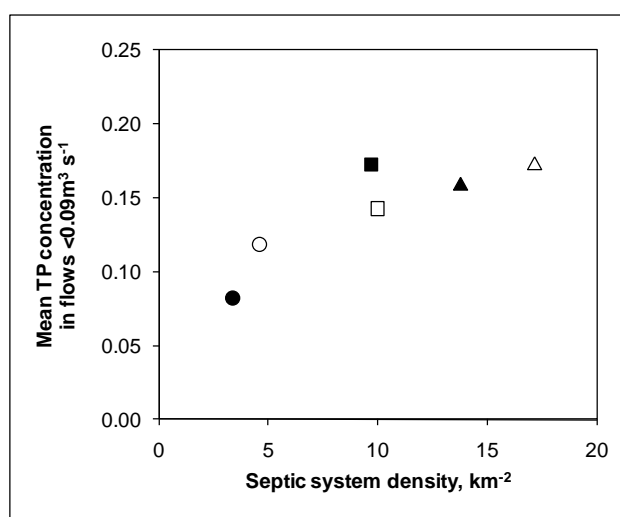


Figure 6: Scatterplots showing the trajectory and change of septic system density and average total P concentration in stream discharges $<0.09\text{m}^3\text{ s}^{-1}$. Circles, squares and triangles denote Cos. Tyrone, Armagh and Monaghan respectively. Closed symbols represent 2006-07 and open symbols represent 2009-10 (from Macintosh et al., 2011).

RISK SCALE AND FUTURE CATCHMENT CHALLENGES

Despite the ecological benefits of headwaters being recognised in terms of nursery habitats (Clarke et al., 2008) and also as catchment zones that can provide runoff for downstream dilution (Alexander et al., 2007), for the most part, they are not considered in Water Framework Directive (WFD) monitoring regimes. In the Republic of Ireland, the Small Stream Risk Score (Callanan et al., 2008) is likely to be the most suited to the spatial scale of identifying the impact of DWWTS effluent transfer but, lacking a chemical proxy or

fingerprint, may not fully disentangle the impacts of other, for example diffuse or morphological, effects.

The cumulative downstream impact of headwater DWWTS pollution is also largely unknown although questions have arisen on the relationships between the temporal scale of infrequent chemical monitoring, that is highly probably linked to low flows in any hydrological year (e.g. Cassidy and Jordan, 2011), and ecological metrics – the inference being that there is a high likelihood that these may be related to chronic point rather than acute diffuse sources of nutrients, where point sources are present (e.g. Hilton et al., 2006).

There are clearly challenges ahead relating to DWWTS technology, catchment density and the scale at which impact is determined.

In terms of technology, although indicating an improvement with changed technology where DWWTS density wasn't overly changed, the research highlighted above showed that the improvement in P concentration was insufficient to be considered at low risk with average low flow total reactive P concentrations still around 0.1mg l^{-1} following mitigation. While this may have been offset by a greater number of replacement systems, the potential for soluble P attenuation in above ground filtering media is likely to be limited – the operating system of package plants focusing as it does on suspended sediment and biochemical oxygen demand reductions through organic matter processing. New technology for secondary and tertiary attenuation of P, nitrogen (N) and bacteria in the liquor portion of septic effluents in new and existing systems (retro-fitting) is pressing.

Clustering of DWWTS through rural housing developments, or high densities dispersed in sub-catchments of high drainage density (and thus with high hydrological connection) is an issue that may not be solved through technological changes. Consolidation of complete systems or at least the liquor portion of septic effluents for attenuation and treatment could be a solution and warrants research through pilot projects.

While the recent EU ruling against the Republic of Ireland is mostly related to DWWTS pollution, human health issues and drinking water, the eutrophication of headwaters from these rural point sources is becoming manifestly clear with evidence accruing from Irish, UK and other EU research (Withers et al. – in press). How river systems are monitored and what the chemistry relates to, what importance is assigned to these headwaters and how the signals can detract from other improvements in pollution sources in sub-catchments at the same scale are also common questions.

In terms of priorities in headwaters, these challenges are likely to be combined in those zones of low soil permeability where the attenuation and dilution capacity of soils and sub-catchments, respectively, are the least.

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QUANTIFYING CONTRIBUTIONS OF DOMESTIC WASTEWATER TREATMENT SYSTEMS TO NUTRIENT FLUXES IN STREAM HEADWATERS

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ABSTRACT

Effective implementation of the Water Framework Directive requires a reappraisal of conventional approaches to water quality monitoring. Quantifying the impact of domestic wastewater treatment systems (DWWTS) in Irish catchments is further complicated by high levels of natural heterogeneity. This paper presents a numerical model that couples attenuation to flow along different hydrological pathways contributing to river discharge; this permits estimation of the impact of DWWTS to overall nutrient fluxes under a range of geological conditions. Preliminary results suggest high levels of attenuation experienced before DWWTS effluent reaches bedrock play a significant role in reducing its ecological impact on aquatic receptors. Conversely, low levels of attenuation in systems discharging directly to surface water may affect water quality more significantly, particularly during prolonged dry periods in areas underlain by low productivity aquifers (>60% of Ireland), where dilution capacity is limited.

Key Words: Water Framework Directive, wastewater, nutrients, modeling, baseflow.

BACKGROUND

Across the EU, identification of areas where human activity may impact water quality within a river basin district (RBD) poses a significant challenge to those seeking to determine the waterbody status. Implementation of Water Framework Directive (WFD) legislation requires RBD managers to re-evaluate conventional approaches to surface water quality monitoring. A focus on aquatic ecological health implies that “end of pipe” approaches, e.g. at a catchment outlet, may be insufficient to meet WFD requirements; such approaches may fail to detect impacts masked or removed by dilution/purification processes operating as contaminated water flows downstream. From a monitoring/enforcement perspective this implies that stretches of rivers and streams that had received limited attention under previous water quality legislation may require greater scrutiny under the WFD, e.g. low order streams and river headwaters.

In Ireland, understanding the impact of nitrogen and phosphorus on water quality, and attributing relative contributions to diverse point and diffuse sources, has proved

particularly difficult to characterise. This is due in part to the high levels of (physical and geochemical) geological heterogeneity encountered in soils, subsoils and bedrock; the composition and texture of geological materials influence the hydrological pathways by which contaminants are transported to aquatic receptors and the degree to which they may be attenuated en route. Approaches to predicting the impact of aquatic pollutants must take these variations into account if the effects of human activity on water quality are to be determined.

The Source-Pathway-Receptor (S-P-R) approach to investigating the fate and transport of contaminants transported by hydrological processes provides a simple but effective rubric for developing an understanding of the impact of water borne nutrients on the quality of water in rivers and lakes. The approach may be employed for both point source contamination (e.g. on site waste water treatment systems) and diffuse contamination (e.g. from land spreading of agricultural fertilisers). Nonetheless characterising the pathways followed by contaminants as they are transported to aquatic receptors remains challenging in Ireland. The relative importance of different hydrological pathways will vary between catchments (and subcatchments) as a result of geological, climatic and landuse factors. The EPA-funded STRIVE pathways project (2007-WQ-CD-1-S1) (Pathways Project) is investigating how these conditions influence the delivery of nutrients, sediment, and pathogenic micro organisms to surface water receptors (Archbold et al., 2010).

PATHWAYS PROJECT OVERVIEW

The Pathways Project builds upon recent advances made in mapping geological & geomorphological conditions across Ireland, along with the improved understanding of the properties of soils, subsoils, and bedrock arising from related research and focused database interrogation. Integrating these datasets with landuse data has permitted more confident assessment of the spatial variation of the risk of impacts to groundwater and surface water quality, arising from both point source and diffuse contamination, in Irish catchments. By highlighting areas of preferential pollutant loss and combining these with landuse data, critical source areas, where disproportionate quantities of pollutant discharge to water bodies, may be identified. This in turn permits limited resources to be focused toward improving water quality in a more scientific manner (Daly et al., 2012).

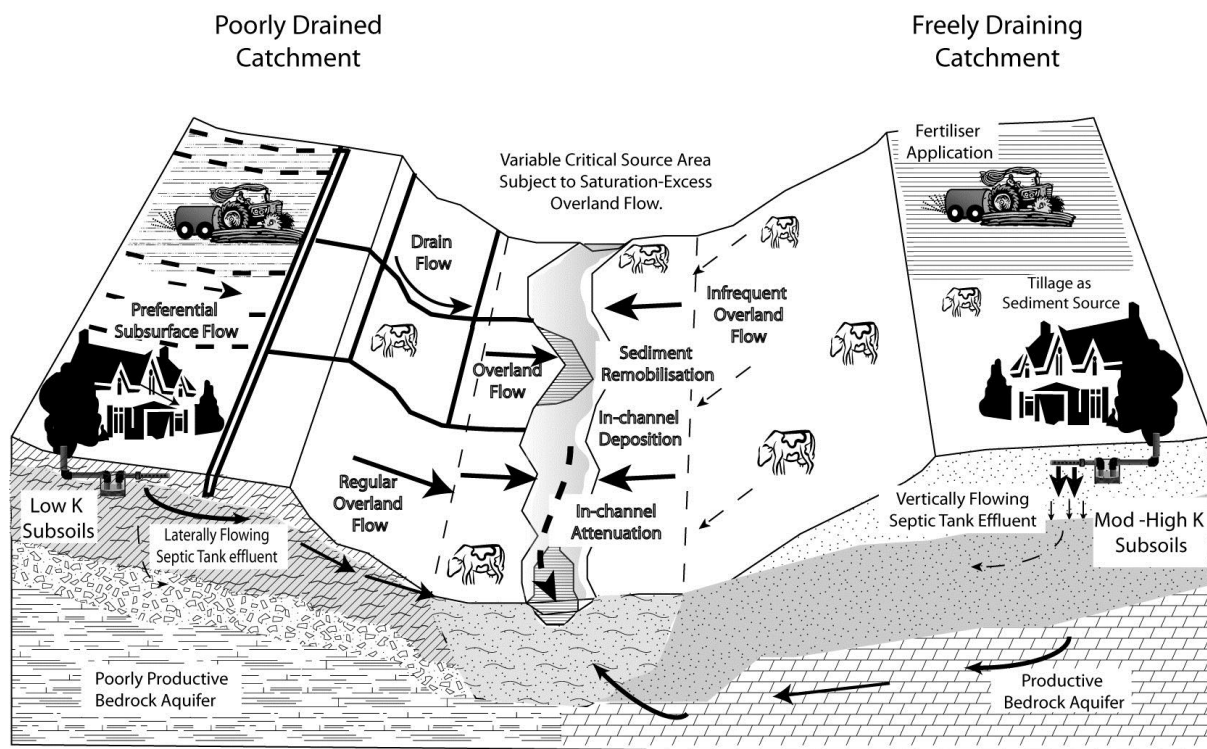


Figure 1: Generic conceptual model of nutrient transport and attenuation in poorly-drained (left) and freely-draining (right) rural Irish catchments.

The contribution of domestic wastewater treatment systems (DWWTS) to nutrient fluxes in a catchment depends upon both the density of systems and their location. Subsoil type and thickness will strongly influence the degree of attenuation experienced by contaminants discharging to ground. Attenuation modelling by the Pathways Project employs the values used by the Environmental Protection Agency for assessing risks of DWWTSs affecting water quality in order to determine the mass of a given contaminant discharging from a DWWTS to ground. Based on evidence collected across the country empirical attenuation factors are applied to simulate declines in contaminant concentration depending on subsoil type and thickness. Figure 2 summarises the approach currently employed for nutrients in a simplified hypothetical catchment.

Model outputs from this loading routine are combined with the output from the hydrological model, MIKE NAM. MIKE NAM permits flows along different flow pathways to be simulated by adjusting a range of empirical calibration parameters. Pathways considered in the current research include overland flow, groundwater flow and interflow; in the current model interflow is considered as all subsurface flow contributing to river discharge that is not groundwater flow through permeable subsoils (sand and gravel) or bedrock.

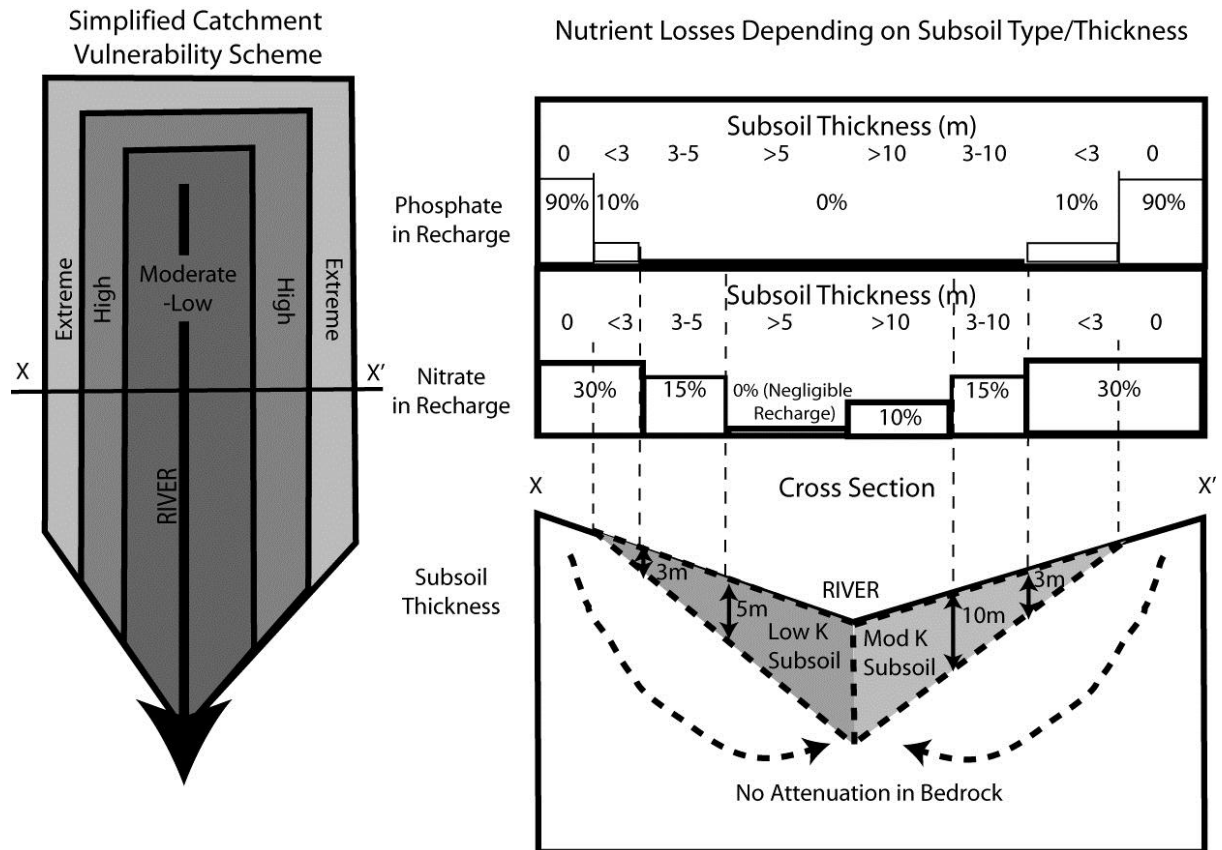


Figure 2: Schematic diagram summarising the levels of attenuation experienced by nitrate and phosphate according to subsoil permeability (K) and subsoil thickness. Percentages for “Phosphate in Recharge” and “Nitrate in Recharge” shown in the left half of the figure are proportions of the total load reaching groundwater in bedrock.

The hydrological activity and discharge rates along the different pathways vary depending on hydrological conditions. The Pathways attenuation model makes the initial assumption that a constant flux of DWWTS effluent discharges to the subsurface with equivalent attenuation operating along both pathways. Dividing the total load from DWWTS effluent by combined groundwater & interflow discharge, following application of relevant attenuation terms, permits DWWTS nutrient-derived fluxes, and resulting concentrations in a river, to be calculated.

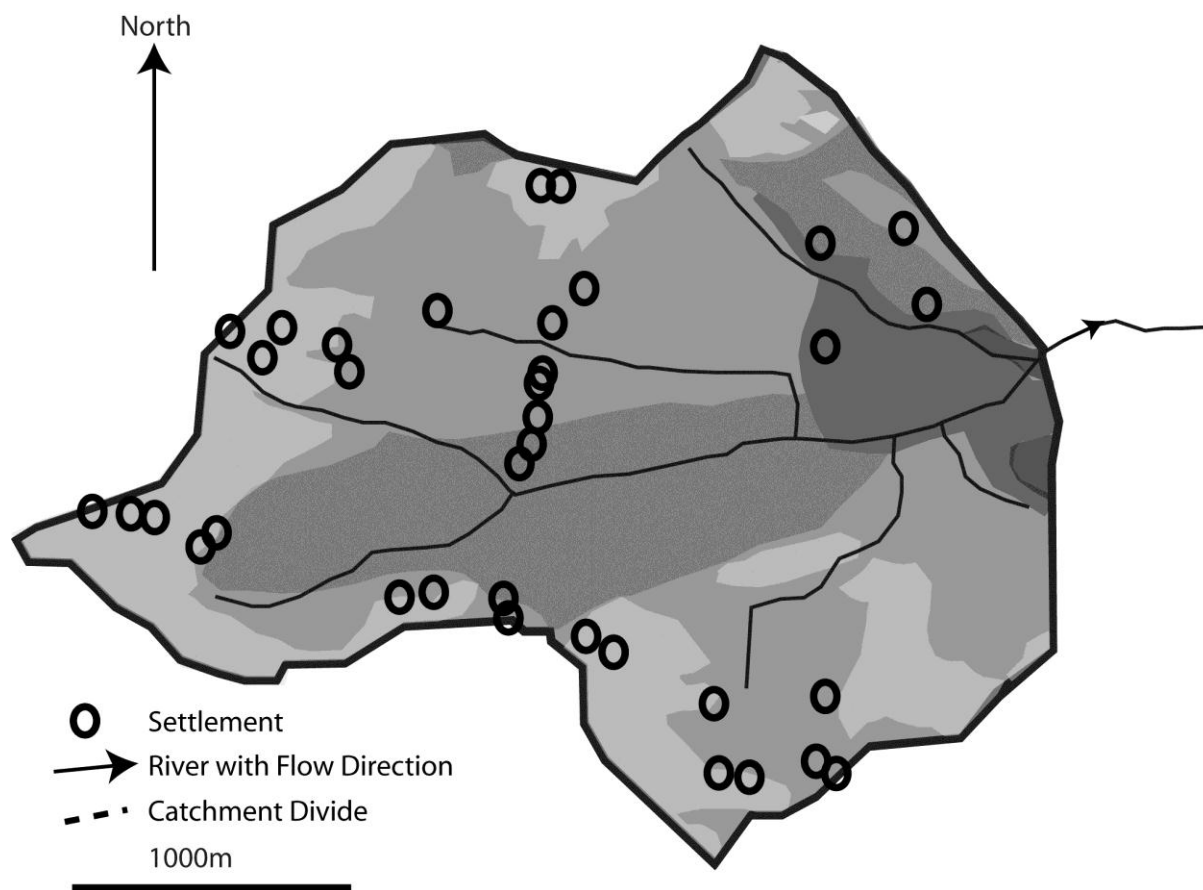


Figure 3: Map of Mattock Headwaters showing the location of housing and drainage. Darker shades of grey reflect thicker subsoil. (Data Source: Geological Survey of Ireland).

MODEL APPLICATION CASE STUDY – MATTOCK HEADWATERS

As part of the Pathways Project research programme, detailed characterisation of land use, hydrology and geological conditions has been carried out in four study catchments spanning a range of conditions encountered in Ireland. An important aspect of the characterisation process involves continuous monitoring of river water temperature, electrical conductivity, and discharge at two nested locations in each catchment. Similarly, monthly to bimonthly nutrient analysis of baseflow, where the influence of overland flow on discharge is absent, has been on-going over a 20 month period at between six and eight locations in each catchment.

To test the current model, calculated concentrations of nitrate (i.e. outputs of the combined loadings tool with MIKENAM) were compared with observed concentrations in water samples discharging from a 699ha subcatchment in the headwaters of the Mattock River, Co. Meath, between January 2011 and January 2012. Aerial photographs, coupled with field verification, showed the area to contain 39 active unsewered dwellings in an area otherwise used as pasture (Figure 3). Soil cover over greywacke bedrock consisted of low permeability soils ranging from less than 3m to greater than 10m in thickness.

Table 1: Preliminary results of a sensitivity analysis of the Pathways Attenuation Model evaluating the impact of subsoil attenuation factors on nitrogen concentrations in the summer base flow of the Mattock River Headwaters. With the exception of the “No Attenuation” scenario, all simulations assume a 40% removal rate by the DWWTS.

	No Attenuation		EPA Risk Assessment Guideline Value		EPA Guideline x 1.1		EPA Guideline x 0.9	
Soil Thickness (m)	Factor (%)*	Load (Kg)	Factor (%)*	Load (Kg)	Factor (%)*	Load (Kg)	Factor (%)*	Load (Kg)
<3m	0	76	0.7	14	0.77	11	0.6	18
3m-5m	0	102	0.85	9	0.93	4	0.77	14
5m-10m	0	93	1	0	1	0	0.8	6
>10m	0	59	1	0	1	0	0.8	4
Total Load (Kg)	N/A	331	N/A	23	N/A	15	N/A	40
Average Summer Baseflow Conc. In River (mg/l)	N/A	1.5	N/A	0.11	N/A	0.07	N/A	0.18

**Factor: Percentage of N load removed on passage to bedrock. This equates to (100%-Percentage Reaching bedrock) presented in Figure 2. Results are based on preliminary estimates and will be subject to revision as further data become available.*

MODEL OUTPUTS-INITIAL FINDINGS

Using the N-cycle routine to determine the mass of diffuse N infiltrating across the area, initial attenuation model outputs for the Mattock Headwaters suggested that infiltrating septic tank effluent made up approximately 6% of the catchment’s total N input.

At present the output from MIKE NAM simulations provide poor correspondence between observed and simulated discharges. This is suspected to be partially due to the very small catchment size coupled with drinking water abstraction from the river during the summer

period. Nonetheless model results suggest that the concentrations resulting from N-derived septic tank effluent outside the period from April to September are negligible; calculations using observed river discharge data for this period suggested that the contribution to the N flux in winter baseflow concentrations could be no more than 0.2mg/l. By contrast the model suggested this contribution could reach over 1 mg/l during the summer period. This is due to low levels of baseflow discharge, which significantly affect the capacity of the river to dilute pollutants, while at the same time pointing to the ability of pollutants to have disproportionate impacts on river ecosystems during prolonged dry periods.

Application of attenuation factors results in a dramatic reduction in N levels that may be derived from DWWTS effluent. Table 1 summarises the sensitivity of model concentrations to these attenuation factors (“Factor” in the table) for areas of differing soil thickness. When considered in conjunction with loads from diffuse sources of N in the area, levels compare favourably with baseflow levels of between 3.5mg/l and 6.5 mg/l.

PERSPECTIVES

Preliminary results from the attenuation modelling study suggest that DWWTS effluent may make quantifiable contributions to nitrogen levels in stream headwaters during low flow periods, especially in summer. These contributions have the potential to impact aquatic health. However, as model sensitivity analysis has demonstrated, attenuation factors can considerably reduce concentrations along pathways between DWWTS outlets and aquatic receptors. Further quantification of these factors for different soil types and thicknesses is necessary if the attenuation model is to be applied with greater confidence across the country. Application of the approach to other subcatchments within the pathways study areas aims to address this issue.

By contrast it is worth noting that at present the above model assumes that all DWWTS effluent infiltrates and that interflow and water entering the bedrock experience the same attenuation. On-going field studies in the Mattock Catchment and other Pathways study catchments suggest that a significant proportion of septic tanks discharge directly to drains. Similarly, treated effluent discharged to low permeability subsoils can reappear in nearby field drains.

Comparable observations have been made in other poorly-draining Irish catchments (V.McCarthy, 2012). Along these pathways rates of attenuation may be considerably lower, or attenuation may not occur. This would imply that DWWTS-derived nutrients would make up a considerably larger proportion of the total nutrient flux in rivers. In areas underlain by freer-draining subsoils and more productive groundwater systems, the impacts of these phenomena may not be significant. By contrast direct discharge of sewage effluent to streams in flashy catchments, where baseflow may provide very limited dilution, has the potential for much more significant ecological damage, and thus affect water body status more strongly.

The potential for the approach outlined above to assist in determining the impact of DWWTS on river and stream water quality should be viewed in conjunction with on-going advances in MIKE NAM modelling and continued data collection by the STRIVE Pathways Research Consortium. These will provide further data to allow the attenuation model more confidently inform the Pathways Project catchment management tool. This in turn will allow regulators and RBD managers to make more confident, scientifically sound decisions on how best to implement the WFD across Ireland.

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A FIELD STUDY ASSESSING THE IMPACT OF ON-SITE WASTEWATER TREATMENT SYSTEMS ON SURFACE WATER QUALITY IN A CO. MONAGHAN CATCHMENT

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ABSTRACT

Poorly functioning on-site wastewater treatment systems (OSWTS) can be among the many sources of pollution to groundwater and surface water, which are of critical concern owing to potential human and ecological health risks. An investigation into the effects of on-site wastewater treatment systems (OSWTS) on surface water quality has been undertaken at several sites within a catchment in Co. Monaghan. The study sites were located in areas of 'low' permeability, suggesting that run-off usually dominates over infiltration. Poor treatment performance of OSWTS within the catchment were found to be the result of several factors, including location in areas with unsuitable soil and site characteristics, incorrect installation, poor maintenance and inappropriate operation by the home owner.

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Keywords: On-site wastewater treatment systems, effluent, water quality, septic tank.

INTRODUCTION

In Ireland, it is estimated that there are approximately 400,000 on-site wastewater treatment systems (OSWTS), located principally in rural areas. Properly built and maintained OSWTS can treat effluent in an ecologically sound manner and return the water to the environment (Hill, 2004). However, poorly constructed, installed, maintained and operated OSWTS, have the potential to be among the many sources of pollution to both groundwater and surface water. In Ireland diffuse run-off from agricultural is suspected to be the principal source of nutrient input to water bodies (Foy *et al.*, 2002; Jennings *et al.*, 2003). However, there is mounting evidence to suggest that pollution arising from failing OSWTS, is also likely to be an important source of nutrients (Jordan *et al.*, 2005).

Poor treatment performance of OSWTS is generally the result of several factors including location in areas with unsuitable soil and site characteristics, poor operation and maintenance practices, and a general lack of knowledge by householders regarding appropriate use and general maintenance (Carroll, 2006).

Onsite wastewater treatment systems have been identified by the Water Framework Directive (WFD; 2000/60/EC) National Summary Characterisation Report as a significant diffuse pressure acting on water bodies. Given that the WFD requires the achievement of 'good' ecological and chemical status in all water bodies by 2015, it will be necessary to identify measures that are necessary to mitigate against existing and future pressures arising from OSWTS.

In October 2009 the Irish Government lost a case in the European Court of Justice over their approach to the regulation and inspection of OSWTS. In its judgment the court found that the Irish legislation did not conform with the provision of various EU Directives, in particular the 1975 Directive on Waste (75/442/EEC), which has since been replaced by Directive 2008/98/EC. The ECJ noted that regular inspection of OSWTS by Local Authorities were of fundamental importance and although there was sufficient scope for such inspections to be carried out in Irish legislation, these powers were not exercised sufficiently. In 2012, The Water Services (Amendment) Act was introduced, which requires water services authorities to carry out risk-based inspections of OSWTS. This legislation aims to enhance and protect public health and the environment. The evidence-based inspections are due to commence in 2013, in which a system will fail if evidence of endangerment of human health or the environment is shown.

The work presented here aimed to acquire an understanding of the likely risk posed by OSWTS to water quality within an Irish catchment. The project focused on systems which are currently in use, taking into account the possible effect of factors such as poor maintenance, incorrect installation and operation and location in areas with unsuitable site characteristics. The study focused on a poorly drained area where the impact of diffuse contamination on water quality has warranted more detailed investigation of wastewater delivery mechanisms to surface water bodies.

SITE DESCRIPTION

The Milltown Lake catchment in Co. Monaghan [Grid ref. 54° 8' 42", -6° 42' 41"] (Figure 1), has been the subject of an intensive monitoring programme since 2006. Milltown Lake is an abstraction point for the Churchill and Oram Group Water Scheme, servicing approximately 1,922 people with approximately 735 m³ of water being abstracted daily. The quality of water throughout the catchment was monitored and the various sources of point and diffuse pollution identified.

The main water quality pressures identified within the catchment include poor farmyard practices, cattle access to streams and runoff from slurry and fertiliser spreading. On-site wastewater treatment systems were also identified as a potential source of contamination within the catchment and as such they were characterised through a combination of household questionnaires, non-intrusive site inspections and an intrusive monitoring

programme. A total of 154 households were surveyed in 2006 and homeowners were asked questions relating to the age and type of their system, the frequency of desludging, the number of occupants and the number of sinks, showers baths and toilets in the house. A subset of 42 of these systems were then visually inspected to establish the materials used in the construction of the system, the type of effluent dispersal unit and the overall condition of the site.

Five sites (Sites; D, F, J, K and S) were selected for the intrusive survey. Sites were selected based on a number of factors including distance to the nearest watercourse, the presence of likely interfering or confounding features such as roads or pathways, or the presence of obvious sources of alternative pollution (e.g. nearby slatted house or slurry pit). Despite these criteria, the principal consideration was homeowner consent. As a consequence, two of the five sites are located outside the Milltown Lake study catchment. Nevertheless, all sites are located within the Lough Muckno catchment and have comparable geology and landuse.

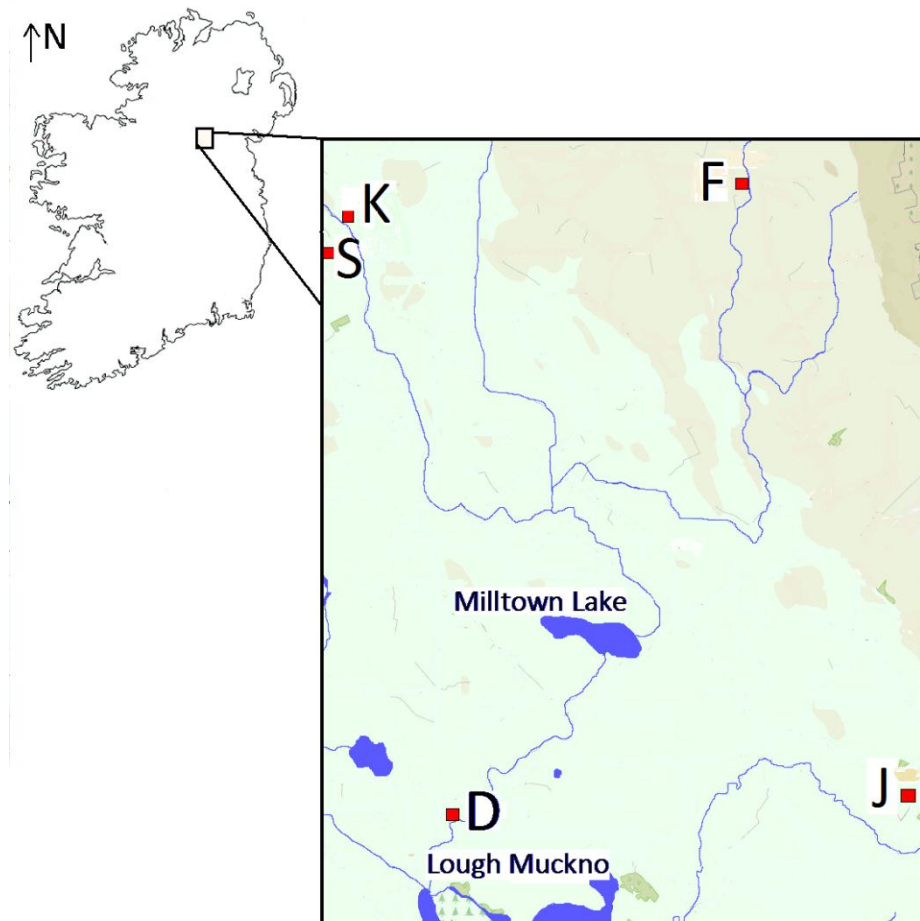


Figure 1: Sampling sites located within the Lough Muckno catchment, Co. Monaghan.

INVESTIGATION METHODOLOGY

All five intrusive survey sites were located in areas which were underlain by clayey glacial till resting on a poorly productive sequence of Lower Palaeozoic shales and greywackes (highly compacted/lithified sandstones). A Geological Survey of Ireland (GSI) subsoil permeability map records the subsoil in the general area as of 'low' permeability, suggesting that runoff usually dominates over infiltration. A site characterisation was carried out at each site in November 2008 in accordance with guidelines set out by the Irish EPA (EPA, 2000), which incorporated a percolation test to ascertain the assimilation capacity of the subsoil. This was determined using a modified version of the on-site standardised Irish falling head percolation test, the 'T-test'. The 'T-value' is the average time in minutes it takes for the water level to fall 25 mm in each of two percolation test holes dug at depths at least greater than 400 mm below ground level. A low 'T-value' (< 3) suggests a soil that is highly permeable in nature (generally sand or gravel) and is, therefore, not suitable for the installation of a septic tank, while a high 'T-value' (>50) is more indicative of a less permeable conditions, often clayey soil, which is again not suitable for a conventional septic tank (EPA, 2009). In addition to the 'T-value', the 'P-value' was also obtained. The P-test is generally carried out at ground level to establish a percolation value for soils with 'T-values' ≥ 50 and ≤ 90 that are being considered for an alternative treatment system, which would discharge effluent at ground surface or over ground through a soil polishing filter.

At each site a varying number of monitoring wells were installed down-gradient of each OSWTS depending on its proximity to the nearest water course. A control was also installed outside the predicted path of the effluent plume. Sampling began at these sites in August 2008, subsurface and surface water samples were taken on a bi-monthly basis for one year and analysed using standard methods for a range of parameters indicative of effluent pollution including, soluble reactive phosphorus (SRP), nitrate (NO_3^- -N), nitrite (NO_2^- -N), ammonia (NH_3^+ -N), total phosphorus (TP), total nitrogen (TN), chloride (Cl^-), sodium (Na^+), potassium (K^+), alkalinity, suspended solids, conductivity, pH, and dissolved Organic Carbon (DOC). Total coliforms and *E. coli* were enumerated using the QuantiTray® 2000 most-probable-number (MPN) format.

RESULTS AND DISCUSSION

The majority of the households surveyed via a questionnaire (> 90%) had a conventional septic tank. Over half of these systems were installed before 1991 (prior to SR6:1991 Wastewater, Treatment Systems for Single Houses), and consequently might not comply with existing standards, although they may have conformed with the standards current at the time of their installation. According to current EPA guidelines, it is important that the tank is de-sludged on a regular basis, with a recommended frequency of once every 12 months (EPA, 2009). Of those surveyed, 27% had never de-sludged their systems.

The questionnaire survey was followed up with visual inspections of a subset of 42 of these systems. Of these, 83% were conventional septic tanks and the remainder were packaged secondary systems of various types. A majority (64%) of the septic tanks inspected were

single chambered tanks and would, therefore, be considered sub-standard when compared to current standards. It was found that 3 of the 7 packaged secondary systems included in these visual inspections were not operating correctly, having never been turned on (confirmed through consultation with the homeowners). The type of effluent dispersal unit was also inspected for all systems, 45% of which were found to have percolation areas, whereas 38% relied on soakaways (i.e. pits filled with stone) and 17% were found to have direct discharge to a nearby ditch or water course, two of which had no settlement tank at all.

Details of the systems at the five sites selected for long term monitoring are summarised in Table 1. Only one site (Site D) had a secondary system, which was a mechanical aeration system. The remaining sites were reliant on two chambered septic tank systems, all with an estimated age greater than 30 years. In addition, three of the five sites had a soakaway, which did not conform to current guidelines (EPA, 2009).

Table 1: Summary of OSWTS at each of the five study sites, Co. Monaghan.

*DWT = Depth to water table, measured between Aug 2008 and Aug 2009.

Site	System	Dispersal Method	*DWT (m)	Age (yrs)	De-sludging
D	Secondary	Percolation	0.15 -2.65	~8	2/yr
F	Septic Tank	Soakaway	0.18– 2.9	~35	1/yr
S	Septic Tank	Percolation	0.12–1.67	~30	>5
K	Septic Tank	Soakaway	0.37–2.77	~30	>5
J	Septic Tank	Soakaway	0.77–1.37	~30	>5

Following site characterisation, Site D was found to have a system with percolation pipes located too close to the bedrock surface, additionally percolation pipes were concentrated together rather than forming a well-spaced network at Site S. A broken in-flow pipe was also suspected at Site F, of which the homeowner appeared unaware, and at this site a portion of the greywater (perhaps arising from the washing machine or relatively recently installed dishwasher) was piped separately to a nearby ditch running along the eastern side of the site, and flowing directly to the adjacent stream.

The average 'P- and T-value' recorded at the sites following site characterisation are summarised in Table 2. None of the sites were deemed suitable for a conventional septic tank system, and only two were considered suitable for the installation of an advanced wastewater sewage treatment system by the assessor, both with recommendations for a soil polishing filter. Water table levels at all sites ranged from 0.12 m to 2.77 m over the course of the one year monitoring period, indicating a reduced unsaturated zone

Finally, direct discharge points were observed at Sites K and J, with pipes observed by-passing the soakaway and discharging effluent directly to the nearby water course at both sites. It is worth noting that Site K was included in the visual inspection, but that the pipe was only identified after the intrusive study began. This suggests that the initial figure of 17 % may be an underestimation of the true number of direct discharge points within the catchment.

Table 2: Average percolation ‘T- and ‘P-values’ for each site

Site	T value min/25 mm	P value min/25 mm
Site D	38	23
Site F	98	25
Site S	>100	63
Site K	>82	60
Site J	>90	65

Results from monitoring wells collected between August 2008 and August 2009 indicate high concentrations of natural tracers such as electrical conductivity and chloride (Cl^-) in monitoring well water samples collected nearest to the tank. These subsequently decline with distance (Figure 2). A similar pattern was also observed for other measured parameters at all sites.

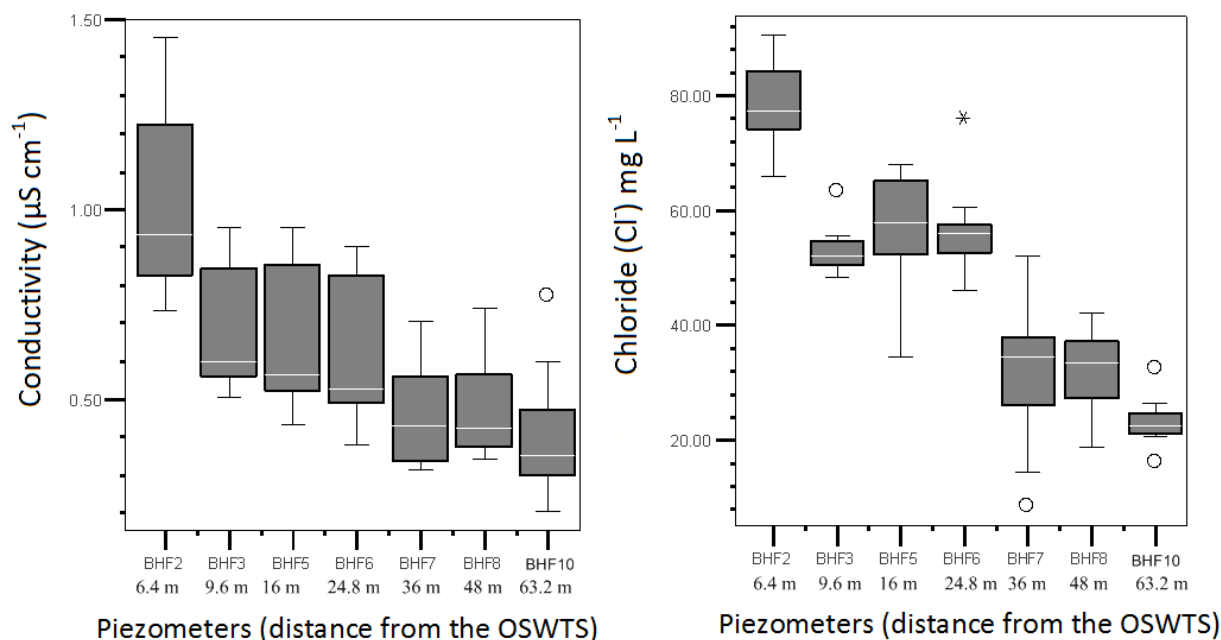


Figure 2: Distribution of conductivity and chloride (Cl^-) concentrations, Site F Aug 08 - Aug 09, showing the median, 75th and 25th percentile.

Highest concentrations of effluent contaminants were generally recorded at Site D, in which contaminants were observed to travel a distance of 22 m and perhaps more, having received little or no treatment whatsoever (mean $\text{NH}_3\text{-N}$ at 22 m of $128.7 \pm 5.7 \text{ mg L}^{-1}$), and up to 9.6 m at Site F (mean $\text{NH}_3\text{-N}$ of $40.5 \pm 2.0 \text{ mg L}^{-1}$). Values declined thereafter, but NO_3^- -N values increased suggesting that nitrification was occurring. However, NO_3^- -N experienced the lowest rate of attenuation of all contaminants other than the conservative tracer Cl^- . In contrast, phosphorus was generally largely removed during the soil treatment process at all sites. Nevertheless, numbers of both total coliforms and *E. coli* were relatively high in the monitoring wells located at the greatest distance from the OSWTS, and were generally higher than the background levels recorded at these sites.

CONCLUSION

The safe disposal of effluent arising from OSWTS is essential for the protection of both groundwater and surface water in Ireland. The chosen approach to this study aimed at gaining an understanding of the impact of operating OSWTS on water quality within a catchment.

For improved management of wastewater treatment systems it is important to be able to identify and quantify the effect that OSWTS may have on water quality. Information on the number, location and condition of OSWTS within a given management area is vital.

This study highlighted the need for correct installation of OSWTS. Of the sites intensively investigated, Site D and Site S had inappropriately designed percolation systems, and Sites J and K had pipes discharging directly to a watercourse. Therefore, all of these sites posed a potential risk to water quality owing to installation issues, even before the permeability and characteristics of the subsoil and soil were taken into account.

In addition, highest concentrations of effluent contaminants in subsurface samples were generally recorded at Site D, which was the only site with a secondary mechanical aeration system. However, this system appears to have malfunctioned, as it had never been switched on (Homeowner, *pers. comm.*). This highlights the need for effective education of homeowners regarding the correct care and maintenance of their OSWTS. This would improve understanding and awareness of their role and responsibilities regarding the correct function, operation and maintenance of their system as laid down in Section 70 of the Water Services Act, 2007.

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SESSION II SUBSOIL ATTENUATION

SOIL TREATMENT UNITS USED FOR EFFLUENT INFILTRATION AND PURIFICATION WITHIN ONSITE WASTEWATER SYSTEMS: SCIENCE AND TECHNOLOGY HIGHLIGHTS

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ABSTRACT

To develop a fundamental understanding of the principles and processes important to soil treatment units used within onsite and decentralized wastewater systems, a program of research has been ongoing for more than a decade within the Small Flows Program at the Colorado School of Mines in Golden, Colorado, USA. Recent and ongoing research concerning soil treatment and assimilation of effluents in the subsurface has been focused on two onsite system approaches: 1) effluent dispersal into a soil profile using shallow trenches outfitted with infiltration chambers and 2) effluent dispersal into the rhizosphere using drip tubing with pressure-compensating emitters. Research has included laboratory experiments, controlled field experiments with pilot-scale systems, field monitoring of full-scale systems, and mathematical modeling. This paper provides a summary of the research carried out. Due to space limitations this paper is focused on soil treatment using subsurface infiltration trenches. While many of the principles and processes are also applicable to soil treatment using drip dispersal of effluent into the rhizosphere, this soil treatment approach is not explicitly covered in this paper.

Keywords: Effluent dispersal in soil, pollutant and pathogen removal in soil.

INTRODUCTION

Today, the vast majority of onsite and decentralized systems in the U.S. include a unit operation involving soil to achieve tertiary treatment with natural disinfection (e.g., Siegrist *et al.* 2001, USEPA 2002). Similar to a confined treatment unit (e.g., septic tank, packed bed filter), an unconfined soil profile can be conceptualized as a *wastewater treatment unit operation* that is designed to: 1) hydraulically process and purify the effluent within the soil

profile to the extent needed to protect public health and water quality; 2) provide a long service life with low operation and maintenance requirements; 3) enable resource recovery and reuse; and 4) have an affordable cost (Siegrist 2006, 2007). Using the terminology, Soil Treatment Unit, reflects this conceptualization (Figure 1).

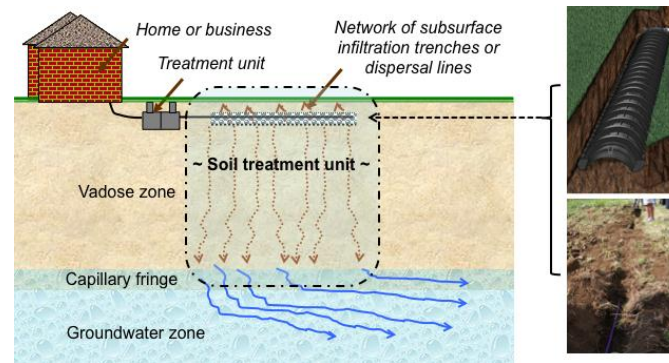


Figure 1: Illustration of a modern soil treatment unit, which uses infiltration chambers placed in trenches or drip dispersal tubing inserted in the rhizosphere within an *in situ* soil profile for infiltration and purification of primary or secondary treated effluents.

To develop a fundamental understanding of the principles and processes important to the design and performance of soil treatment units used within onsite and decentralized wastewater systems, research has been ongoing for more than a decade within the Small Flows Program at the Colorado School of Mines in Golden, Colorado, USA. Recent and ongoing research has been focused on soil treatment of different quality effluents using two onsite system approaches: 1) effluent dispersal into a soil profile using shallow trenches outfitted with infiltration chambers and 2) effluent dispersal into the rhizosphere using drip tubing with pressure-compensating emitters (Figure 2).

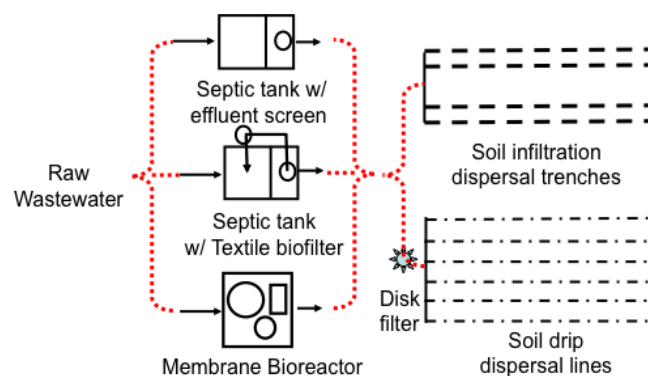


Figure 2: Schematic of the onsite wastewater system components involved in soil treatment unit research within the Small Flows Program at the Colorado School of Mines.

Within the Small Flows Program, soil treatment unit research has included laboratory experiments, controlled field experiments with pilot-scale units, field monitoring of full-scale systems, and analytical and numerical modeling. The program of research has been conceived to develop a quantitative understanding of soil treatment unit design and performance including flow and transport and the removal of pollutants and pathogens as affected by soil properties, system features, effluent quality and loading, and other design factors and environmental conditions. The research has also developed models and decision-support tools for soil treatment unit applications. This paper provides highlights of the research carried out. Additional details on a given topic may be found in the literature cited. Due to space limitations, this paper is focused on soil treatment using subsurface infiltration trenches. While many of the principles and processes are also applicable to soil treatment using drip dispersal of effluent into the rhizosphere, this soil treatment approach is not explicitly covered in this paper.

SOIL TREATMENT UNITS AND KEY PROCESSES

Flow and Transport Processes. When a partially treated effluent (e.g., STE) is applied to soil, effluent infiltration and percolation with eventual ground water recharge involves a complex set of hydraulic and purification processes. The major processes can be categorized to include:

- Effluent infiltration into soil pore networks
- Effluent water movement within a soil profile
 - Percolation – movement within the pore network
 - Groundwater recharge – transport into groundwater
 - Evapotranspiration – transport up and out of the soil
- Effluent pollutant and pathogen removal reactions
 - Kinetic reactions (e.g., biodegradation)
 - Capacity-based reactions (e.g., filtration, sorption)
 - Plant-based reactions (e.g., nutrient uptake)

These processes can interact in a dynamic manner, evolving as the soil treatment unit matures from startup through the first year(s) of operation.

Infiltration Rate Behavior. During normal continuous use of a soil treatment unit over months to years of operation, the application of effluent solids, as measured by total solids (TS) and total suspended solids (TSS), and the total biochemical oxygen demand (BOD) (ultimate carbonaceous BOD (cBOD) plus nitrogenous BOD (nBOD)) can contribute to pore filling at the infiltrative surface and loss of infiltration capacity (Bouma 1975, SSWMP 1978, Siegrist *et al.* 2001, Beach *et al.* 2005, Beal *et al.* 2005, Van Cuyk *et al.* 2005, Lowe and Siegrist 2008). With continued operation, the native soil's capacity to infiltrate wastewater effluent will decline substantially from the capacity for clean water infiltration that was

present prior to initiating effluent application to the soil infiltrative surface. The decline in infiltration capacity during operation is often characterized as a 3-phase process (Figure 3), which is caused by biomat formation and pore-filling at and near the location where effluent enters the soil pore network (effects II and III in Figure 4).

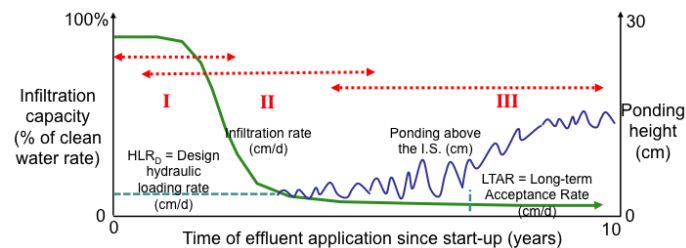


Figure 3: Illustration of the soil infiltration rate behavior during longer-term effluent application following a 3-phase process with an asymptotic approach to a long-term acceptance rate (LTAR).

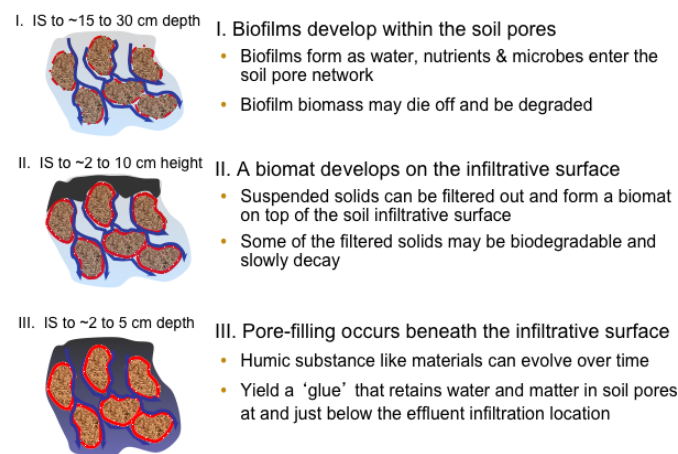


Figure 4: Key changes at and near the soil infiltrative surface in response to effluent application in a soil treatment unit (after McKinley and Siegrist 2010).

After a period of operation, a soil treatment unit can experience a sufficient decline in infiltration capacity such that intermittent or continuous ponding of the infiltrative surface ensues (Figure 3). However, the time to development or sustained occurrence of ponding does not necessarily correlate with long-term hydraulic or treatment performance. A soil treatment unit can operate effectively in an intermittent or continuously ponded condition for an indefinite period of time. The depth of ponding above the infiltrative surface can fluctuate dramatically on a daily or seasonal basis due to system operation and environmental conditions (Siegrist and Boyle 1987).

Under some conditions, such as when higher strength wastewater or higher daily loading rates occur compared to design assumptions, or after an extended period of continuous use (e.g., 20 years or more), excessive soil clogging can occur. This can lead to hydraulic dysfunction where the infiltrative surface becomes so impermeable that the daily wastewater loading can no longer be fully infiltrated (Siegrist and Boyle 1987, Siegrist *et al.* 2001).

Purification Behavior. Wastewaters treated by onsite systems can contain a variety of pollutants and pathogens at low to very high levels. The nature of the source and the water-use and waste-generation characteristics within it determine the composition of the wastewater stream that must be handled by an onsite system (Lowe *et al.* 2009). Traditional constituents of potential concern include oxygen consuming compounds, particulate solids, nitrogen, phosphorus, heavy metals, bacteria and virus (Table 1). Emerging constituents of concern include an array of organic compounds (e.g., caffeine, nonylphenols, Tricosan) which can be referred to as trace organics (due to their relatively low concentrations). Consumer product chemicals can routinely occur at varied levels depending on the source (e.g., residential dwellings vs. commercial establishments) (Conn *et al.* 2006, Conn *et al.* 2010a). Pharmaceuticals, pesticides and flame retardants can also occur, but much less pervasively and typically at much lower levels (Conn *et al.* 2010a).

Soil treatment units are often expected to achieve tertiary treatment and natural disinfection. For this to occur, highly unsaturated flow under aerobic conditions is normally critical. This flow regime facilitates contact between wastewater constituents and the soil grain surfaces and their associated biofilms and provides for a relatively long period for treatment processes to occur (Emerick *et al.* 1997, Schwager and Boller 1997, Van Cuyk *et al.* 2001, Siegrist *et al.* 2001, Van Cuyk *et al.* 2004, Van Cuyk and Siegrist 2007). Unsaturated flow conditions can be achieved by hydraulic design if the design hydraulic loading rate (HLR_D) is limited to a small fraction of the soil's saturated hydraulic conductivity (K_{sat}) (e.g., $HLR_D = 1$ to 5 cm/d vs. $K_{sat} = 100$ to 1000 cm/d) and application is achieved by intermittent dosing through pressurized piping networks. Also, over time, effluent infiltration can lead to soil clogging and unsaturated flow conditions irrespective of hydraulic design attributes (Siegrist 1986, Siegrist and Boyle 1987).

Pollutants and pathogens can be removed in a soil treatment unit by many physical-chemical and biological processes. BOD removal can occur by biodegradation in biofilms that grow on soil grains and within soil organic matter. Suspended solids can be removed by physical filtration and absorption followed by biodegradation. Reduced forms of nitrogen (e.g., NH_4^+) can be biologically oxidized completely and some total N can be removed by nitrification. Phosphorus removal varies widely depending on soil mineralogy and its P-sorption properties. Pathogens such as parasites and bacteria can be filtered out and die-off while virus can attach to grain surfaces and be inactivated. Effluent infiltration can also lead to the establishment of a biozone (Figure 4) that can provide more rapid and extensive

treatment of the constituents in the applied effluent (e.g., by enhanced sorption, nitrification, and biological decay at and near the soil infiltrative surface) (Siegrist 1987, Van Cuyk *et al.* 2001, Siegrist *et al.* 2005, Van Cuyk *et al.* 2005, Van Cuyk and Siegrist 2007, Tomaras *et al.* 2009).

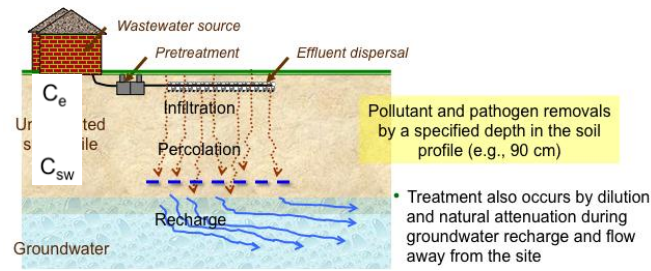
Purification of trace organic compounds (e.g., caffeine, nonylphenols, Tricosan) in a soil treatment unit principally occurs by sorption and biodegradation. Achieving high removal efficiency for a particular organic compound depends on the properties of the compound as well as the conditions present in the soil treatment unit (Conn *et al.* 2010b).

SOIL TREATMENT UNIT PERFORMANCE

A soil treatment unit can be designed and implemented to reliably achieve tertiary treatment with natural disinfection over a service life of 20 years or more. Key conditions that are required to achieve this performance level include: 1) the hydraulic conductivity of the infiltrative surface zone is not dramatically reduced by compaction, smearing, or solids deposition during installation and startup; 2) the HLR_D and/or concentrations of pollutants that cause soil clogging are not excessive compared to design assumptions; 3) there is an adequate soil profile depth for treatment - depending on effluent loading rate and quality, a certain depth of unsaturated aerobic soil is needed for treatment to occur; 4) there is unsaturated flow in the soil profile with long travel times so kinetic processes can achieve pollutant removals (e.g., removal of BOD, NH₄⁺, Fecal coliforms); 5) there is an adequate volume of soil profile to provide soil grain surface area for sorption processes (e.g., P removal); and 6) subsurface conditions are conducive to treatment (e.g., circumneutral pH, high Eh, moderate temperatures, no biotoxins).

The treatment efficiencies normally expected of a well-designed and properly operated soil treatment unit are given in Table 1. The effects of several key design factors and environmental conditions and their relative contributions to overall performance are highlighted in Table 2.

The inherent nature of a soil treatment unit can complicate the use of quantitative treatment expectations (e.g., Table 1) and the ability to verify their achievement through monitoring. This is because, unlike a tank-based unit such as a sand filter, there is no outlet pipe and “effluent” per se from a soil treatment unit. Rather, the “end-of-pipe equivalent” is the soil solution at some depth (e.g., 0.6 m below the infiltrative surface which may be where shallow groundwater exists) (Figure 5).



$$\% \text{Removal} = \left[\frac{C_e - C_{sw}}{C_e} \right] \times 100\%$$

Figure 5: Assessing treatment efficiency by comparing concentrations in soil pore water at a specified depth (C_{sw}) to the effluent applied (C_e).

Depending on the environmental setting, further purification of this “end-of-pipe” effluent equivalent can occur as reclaimed water moves through the subsurface (deeper vadose zone or ground water zone) and exits the ground water into surface water. This assimilation of effluent from a soil treatment unit and attenuation of residual constituents of potential concern can be critically important to achieving public health and water quality protection goals (e.g., attenuation of nitrate-nitrogen, virus, trace organics).

Use of a mass discharge approach for evaluating treatment effectiveness and impacts incorporates this attenuation within the vadose zone and groundwater system. Figure 6 illustrates the concept of mass discharge as applied to a soil treatment unit.

MODELING OPERATION AND PERFORMANCE

Analytical and numerical models of varying scope and complexity are available to aid design of an isolated system or clusters of soil treatment units as well as for assessment of onsite system impacts at the local, development, and watershed scale (e.g., Beach and McCray 2003, McCray *et al.* 2005, Poeter *et al.* 2005, Siegrist *et al.* 2005, Heatwole and McCray 2007, McCray *et al.* 2009, 2010, Geza *et al.* 2009, 2010, 2012). Modeling tools are also available to evaluate the relative environmental effects of onsite and decentralized systems vs. centralized wastewater facilities within a particular planning area (e.g., Kellogg *et al.* 1997, Siegrist *et al.* 2005, Lemonds and McCray 2007, Geza *et al.* 2010).

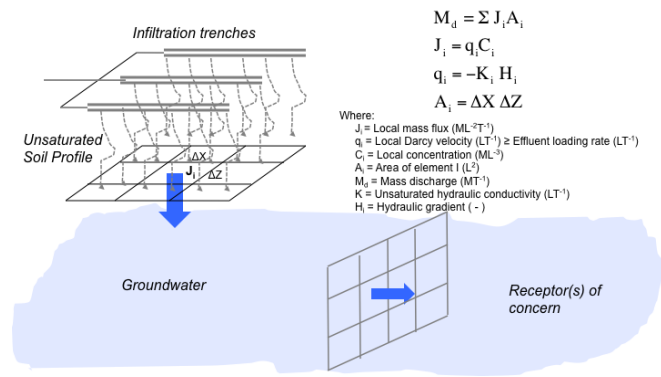


Figure 6: Assessing treatment effectiveness of a soil treatment unit using a pollutant mass discharge (M_d) approach.

An example of a model which can be used to predict the hydraulic performance of a soil treatment unit is the infiltration rate (IR) model developed by Siegrist and Boyle (1987). This model predicts the IR response during long-term application of a particular effluent quality at a certain hydraulic loading rate (see Figure 7). The model consists of a regression equation that was developed based on long-term field experiments in a silt loam soil. Subsequent to its development, the model has been applied to soil treatment units in other soil systems and it has been found to be reasonably predictive (e.g., sandy loam soils as shown in Figure 7).

A prime example of a model which can be used to predict purification performance is STUMOD (Soil Treatment Unit Model) (Geza *et al.* 2009). STUMOD was developed to predict the fate and transport of nitrogen in a soil treatment unit. STUMOD calculates nitrogen species concentrations with depth in the soil profile (Figure 8a) and the fraction of nitrogen remaining with depth (Figure 8b). By repeatedly running STUMOD using randomly selected values from ranges of potential values, the probability that a certain fraction of the total nitrogen in the effluent infiltrated will reach a specified soil depth can be estimated (Figure 8c).

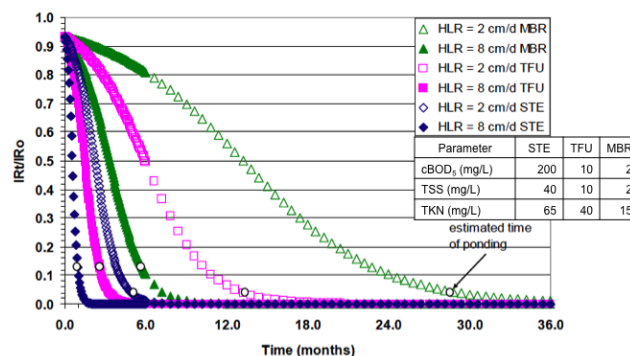


Figure 7: Model simulation of the infiltration rate (IRt) decline as a ratio of the initial infiltration rate (IRo) for a sandy loam soil as affected by three effluent qualities and two loading rate (Siegrist and Boyle (1987) model as reported in Van Cuyk *et al.* 2005).

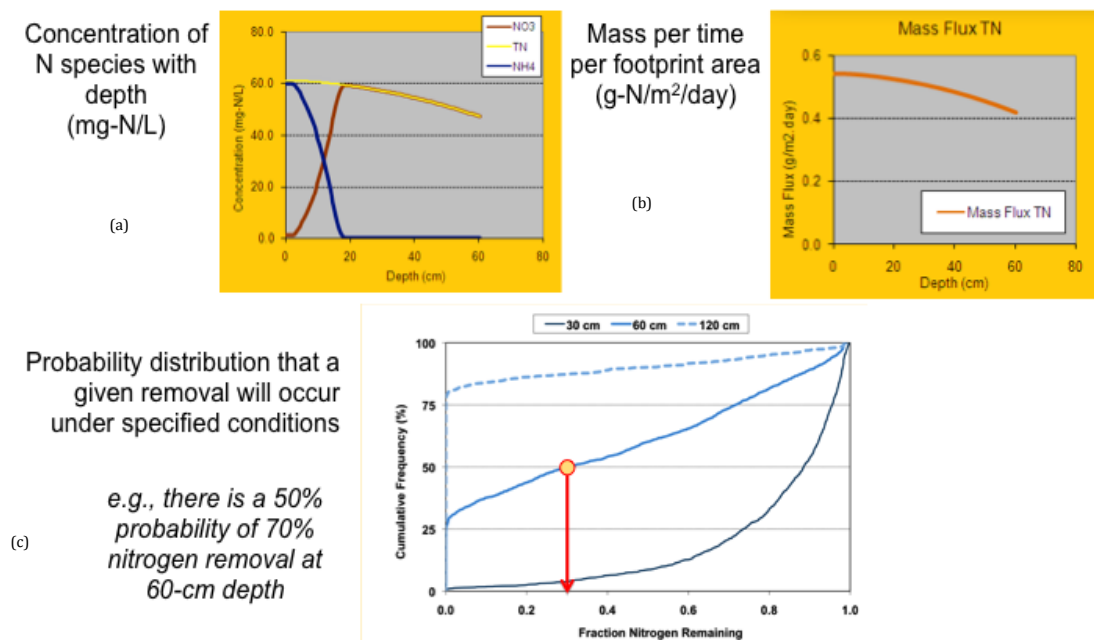


Figure 8: STUMOD simulation of nitrogen transformation and fate when 2 cm/d of STE is applied to a soil treatment unit installed in a sandy loam soil (Geza *et al.* (2009, 2010, 2012) as reported in McCray *et al.* 2010).

STUMOD involves a number of complex equations, which are implemented in a spreadsheet. It is relatively simple to use but can account for important processes such as ammonium sorption, nitrification and denitrification. STUMOD is being modified to account for: 1) evapotranspiration and plant uptake on nitrogen removal; 2) fate and transport during lateral movement in an aquifer (by linkage with a groundwater model); and 3) transport of organic nitrogen and trace organics in the vadose zone.

While these types of relatively simple models provide insight into the operation of soil treatment units and quantitative estimates of performance as affected by a range of conditions, many complex processes and less-common operating conditions can be better addressed by numerical models such as HYDRUS (Šimůnek *et al.* 1999). HYDRUS has been used for a variety of modeling studies exploring soil treatment unit design and performance (e.g., Beach and McCray 2003; Radcliffe *et al.* 2005; Pang *et al.* 2006; Bumgarner and McCray 2007; Heatwole and McCray 2007; Radcliffe and West 2007; Beal *et al.* 2008; Finch *et al.* 2008).

SUMMARY

Within the Small Flows Program, one of several research thrusts has focused on soil treatment units. Over the past decade, research has encompassed laboratory experiments, controlled field experiments with pilot-scale systems, field monitoring of full-scale systems, and analytical and numerical modelling. The results of this research have enhanced the understanding of fundamental principles and processes important to system design and performance. There is now a more quantitative understanding of key flow and transport processes and the removal of pollutants and pathogens as affected by soil properties, system features, effluent quality and loading, and other design factors and environmental conditions.

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Table 1. Wastewater constituents of concern and treatment expectations from a well-designed and properly operated soil treatment unit treating 1 to 5 cm/d of domestic septic tank effluent.

Constituents of potential concern	Basis for concern over wastewater constituent	Example unit of measure (units)	Domestic septic tank effluent ¹	Treatment efficiency after 90 cm of unsaturated aerobic soil
Oxygen demanding substances	Can create anoxic or anaerobic conditions and can contribute to soil clogging	BOD ₅ (mg/L)	140 to 200	>90%
Particulate solids	Contributes to soil pore filling and accelerated soil clogging	TSS (mg/L)	50 to 100	>90%
Nitrogen	Can contribute to oxygen demand, can be toxic via drinking water ingestion, can upset ecosystems	Total N (mg-N/L)	40 to 100	10 to 20%
Phosphorus	Can cause increased productivity in sensitive surface waters	Total P (mg-P/L)	5 to 15	100 to 0% ²
Bacteria	Infectious disease transmission via drinking water, contact with seepage, or recreational water activities	Fecal coliforms (org./100 mL)	10 ⁶ to 10 ⁸	>99.99%
Virus	Infectious disease transmission via drinking water, contact with seepage, or recreational water activities	Specific virus (pfu/mL)	0 to 10 ⁵ <i>(episodically high levels)</i>	>99.9%
Heavy metals	Potential toxicants to humans by ingestion in drinking water or to ecosystem biota	Individual metals (ug/L)	0 to low levels	>99%
Trace organic compounds	Potential health effects to humans by ingestion of drinking water or vapor inhalation during showering or environmental effects to ecosystem biota	Specific organics associated with consumer products, pharmaceuticals, pesticides, and flame retardants (ng/L or ug/L)	0 to trace levels	Low to >99% ³

¹ Note: STE concentrations given are representative of those for residential dwelling units. However, commercial sources such as restaurants can produce STE that is markedly higher in some pollutants (e.g., BOD₅, COD, TSS, trace organics) while other sources can produce STE that is markedly lower in some pollutants (e.g., laundry can have lower total nitrogen and pathogen levels).

² P-removal is highly dependent on media sorption capacity and P loading rates and time of operation.

³ Removal of trace organic compounds (e.g., nonylphenol, Triclosan, EDTA, caffeine,...) is highly dependent on the properties of the organic compound and conditions within the soil treatment unit (e.g., conditions conducive to sorption and biotransformation during adequately long hydraulic retention times).

Table 2. Key design factors and environmental conditions and their relative importance in determining the performance of a soil treatment unit.

Design factors and environmental conditions	Effects on performance of a soil treatment unit (<i>relative importance</i>) ¹
Initial Ksat of the soil profile	For well-drained soil profiles with Ksats of ~5 to 2500 cm/d, the long-term acceptance rates (LTAR) for infiltration of domestic STE under continuous routine use will normally approach ~2 cm/d (<i>minor to moderate</i>) ²
Soil profile conditions during operation	Higher temperatures, lower soil water contents, and higher aeration levels tend to enable relatively higher LTAR's and treatment efficiencies (<i>moderate</i>)
Geometry and infiltrative surface architecture	Horizontal infiltrative surfaces that are aggregate-free in low-height, narrow trenches characterized by sidewall-to-bottom area ratios of 0.5 to 1.0 and placed shallow in the subsurface enhance infiltrability and higher LTAR's and improved treatment (<i>moderate</i>)
Actual operating HLR (HLR _A)	For a given effluent quality, the actual HLR _A exerts a major effect by 1) determining the mass loadings of total BOD and TSS, which control wastewater-induced soil clogging behavior, and 2) the loadings of other pollutants of concern that need to be treated to acceptable levels (<i>major</i>)
Quality of effluent being treated in the soil treatment unit	At a given HLR _A , effluent quality exerts a major effect on the treatment requirements for pollutants and pathogens of concern (<i>major</i>)
Method of effluent application to the soil treatment unit	For systems in continuous daily use, major effects may be exerted on the volume processed per unit area and time but the LTAR and pollutant removal may be unaffected or enhanced (<i>minor to major</i>)
Continuity of use	Infrequent or intermittent use with long periods of resting can sustain higher soil infiltration capacity and LTAR's as well as enhanced treatment capability (<i>major</i>)

¹ This table is presented for illustrative purposes and the attributes and effects given are not intended to provide comprehensive coverage of this subject.

² The descriptors used have the following meanings: "minor" indicates a relative effect of ~+/-20% or less, "moderate" indicates an effect on the order of +/-50%, and "major" indicates an effect on the order of +/- 100% or more.

SUBSOILS ACROSS THE IRISH LANDSCAPE; THEIR TEXTURAL AND BULK DENSITY CHARACTERISTICS, AND RESULTANT VARIATIONS IN PERMEABILITY

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ABSTRACT

Subsoils across the Irish landscape generally comprise glacial and deglacial materials (i.e. those laid down during the last glaciations to affect Ireland), or post-glacial sediments. The majority of Irish subsoils are glacial tills, which are unsorted and unbedded 'boulder clays', which are remarkably heterogeneous on a broad scale. Other subsoil types include sands and gravels, glaciolacustrine deposits, alluvium and peat. The permeability of subsoil is largely a function of (a) the grain size distribution, (b) the amount (and sometimes type) of clay size particles present, and (c) how the grains are packed together. It can also be influenced by other factors such as discontinuities (fissures/cracks, plant roots, pores formed by soil fauna, isolated higher permeability beds or lenses, voids created by weathering of limestone clasts) and density/compactness of the deposit. Subsoil permeability across Ireland varies markedly, but broad patterns emerge and permeability has been mapped on a regional scale by the Geological Survey of Ireland (GSI). Results of site specific percolation tests ('T' tests) show that as the dominant subsoil grain size decreases the permeability also decreases. Though an increase is seen in general in the 'T' test results (corresponding to a decrease in permeability) from GRAVEL through SAND through SILT and SILT/CLAY, a wide range of 'T' test results are seen across the SAND and SILT classes. Where percolation tests were completed in CLAY, the values are exceptionally high and more a function of the time spent monitoring the test holes rather than the actual percolation rate. The CLAY subsoils therefore have a very low permeability and are not suitable for ground discharge from a domestic wastewater treatment system.

Keywords: subsoils; permeability; on-site wastewater treatment.

INTRODUCTION

Irish subsoils were deposited during the Quaternary period of glacial history, which encompasses the last 1.6 million years and is sub-divided into the Pleistocene (1,600,000-10,000 years ago); and the more recent Holocene (10,000 years ago to the present day). The Pleistocene, more commonly known as the 'Ice Age', was a period of intense glaciation separated by warmer interglacial periods. The Holocene or post-glacial period, saw the onset of a warmer and wetter climate approaching that which we have today.

During the Pleistocene the glaciers and ice sheets laid down a wide range of deposits, which differ in thickness, extent and lithology. Material for the deposits originated from bedrock and was subjected to different processes within, beneath and around the ice. Some were deposited irregularly by ice and so are unsorted and have varying grain sizes, while others were deposited by water in and around the ice sheets and are relatively well sorted and coarse grained. Mapping of subsoils in Ireland was undertaken by Teagasc in a project funded by the EPA during the period 1998-2006. This mapping formed the foundation of subsequent subsoil permeability assessments. Subsoil distribution is presented in Figure 1.

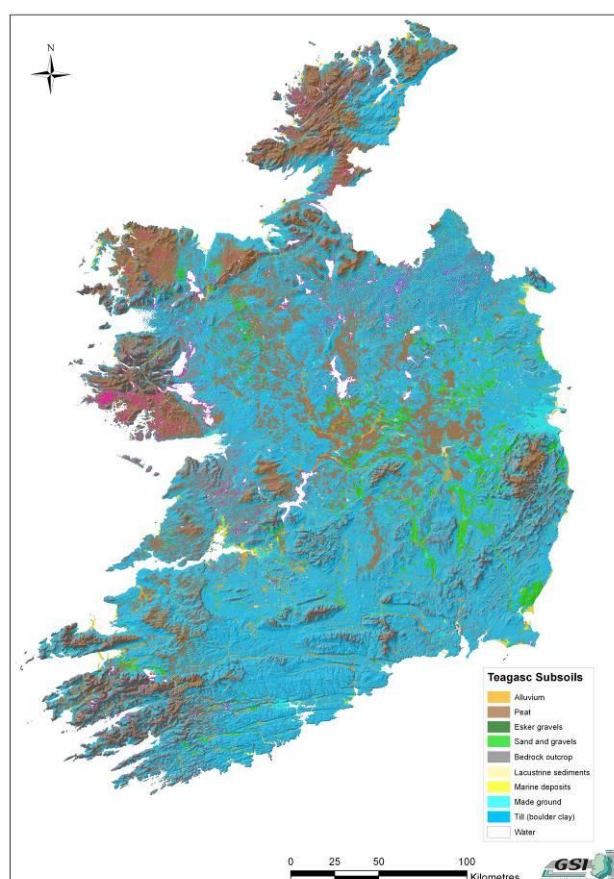


Figure 1: Subsoils map showing different subsoil types across the Irish landmass (Geological Survey of Ireland/Teagasc).

SUBSOIL CLASSES

There are five main subsoil types identified across Ireland and shown on Figure 1:

- ◆ till
- ◆ sands and gravels
- ◆ lake deposits
- ◆ alluvium
- ◆ peat

Areas where bedrock outcrops or comes within 1 m of the surface are recognised as “bedrock outcrop and subcrop”.

Diamictons (mostly tills)

Diamictons are unsorted deposits with a wide variety of particle sizes. They include tills and head deposits, but in the Irish context most diamictons are tills. Till (often referred to as boulder clay by engineers) is the most widespread subsoil across Ireland as can be seen on Figure 2.3, covering over 43% of the country area at surface, and probably another 25% beneath peat or floodplain sediments.

Till is sediment deposited by or from glacier ice. Glacial ice is the principal depositional agent, but gravity and, in some cases, water, also play a part. Tills are often over-consolidated, or tightly packed, unsorted, unbedded, possessing many different particle and clast (stone) sizes, and commonly have sharp, angular clasts. Tills may be categorised according to their dominant petrographical component, or by the grain size of the matrix, or the texture of the till. This determines permeability, which is important for wastewater treatment processes. Thus, tills may be described as gravelly till, sandy till, silty till or clayey till.

Tills include glaciomarine sediments, which are deep-sea sediments that have originated in glaciated land areas and have been transported to the sea by glaciers or icebergs. These sediments are present along certain portions of the Irish coast and have the same geotechnical (and hence permeability) characteristics as tills.

In describing the textures of tills, the British Standard Soil Description Classification System is used. This system places subsoils into groups defined by the grading of their coarsest particles and the plasticity of their finer particles, characteristics that play a major role in determining subsoil engineering and permeability properties. The main attributes of the

subsoil described include the mass characteristics, material characteristics, geological deposit type and age and classification.

Sands and gravels

Glaciofluvial sands and gravels are different from tills in that they are deposited by running water only, when the glaciers are melting. The gravels usually possess stratification (layering) and usually have rounded edges.

Sand and gravel deposits are usually loosely packed. When the huge amounts of meltwater produced by the melting of the ice sheets that covered Ireland at the end of the last glacial period are considered it is not surprising that these deposits are very common in Ireland. They represent the stagnation and decay of the ice sheets. They give rise to a variety of different landforms, including kames, kame terraces, sandar, moraines and, in some cases, drumlins.

Esker sands and gravels are laid down by glacial meltwaters in tunnels and crevasses in stationary or retreating ice sheets, and are seen on land as long, narrow, sinuous ridges. They commonly possess rounded boulders and cobbles. Clasts are usually much larger overall than in other glaciofluvial deposits. Sand may or may not be present.

Esker gravels, as with all glaciofluvial sands and gravels, have very high permeabilities.

Alluvial deposits

In the Holocene Epoch, the warmer climate effected a large change on the environment. The modern fluvial systems were superimposed on, although largely controlled by, the pre-existing glacial landscape. The floors of these modern valleys take the form of alluvial floodplains.

Alluvium is a post-glacial deposit and generally consists of gravel and sand with a minor fraction of silt and clay. However, this deposit may consist of gravel, sand, silt or clay in a variety of mixes and usually consists of a fairly high percentage of organic carbon (10%-30%).

The alluvial deposits are usually moderately to well sorted and are bedded, consisting of many complex strata of water-lain material left both by the flooding of rivers over their floodplains and the meandering of rivers across their valleys. Alluvial fans and modern deltas are included as part of this deposit type

Glaciolacustrine deposits

Glaciolacustrine deposits were deposited into a large number of meltwater-fed lakes during and shortly after deglaciation. Deposits consist of sorted gravel, sand, silt and clay.

They are found normally in wide flat plains, or in small depressions in the landscape. The deposits have different permeabilities depending on the dominant grain size. Deltas (or other near lake shore or beach deposits), which are formed as sediment is deposited at a river mouth on entry into a glacial lake, usually contain interbedded sands and gravels which dip lake-ward. These are left as sand and gravel hills when the ice disappears and the lake drains away. Lacustrine basins, which are distal parts of the lake system, usually contain finer sediments, such as clays and silts which have settled from suspension from the water body. The differentiation of the dominant grain sizes within lacustrine sediments is imperative as such a wide variety of grain size combinations is possible, each resulting in a different texture.

Peat

The change in climatic conditions in the Holocene also resulted in the growth of peat (bog). Peat is also a postglacial deposit, consisting mostly of vegetation which has only partially decomposed in an ombrotrophic (nutrient poor) environment. Bog peats are formed in acidic waters and vary according to the main plants involved in their growth. They cover just under 20% of the country.

Blanket bog is associated with highland areas where poor drainage enabled the build-up of oxygen-starved, partially decomposed biomass. This is thought to have begun approximately 4,000 years ago.

Raised bogs developed in many small lake basins, spreading over time to the surrounding land. Vegetation fills and compacts in marshes, ponds and other lakes carved out and left by Quaternary ice sheets. Thus, on the lowlands in Ireland, peat usually overlies badly drained

glaciolacustrine silts and clays. In the last few centuries, much of Ireland's raised peat has been cut away for burning as solid fuel.

Fen peats consist of unspecified organic materials formed in a minerotrophic environment due to the close association of the material with mineral rich waters. In fen peats the presence of calcium in the groundwater neutralises acidity, giving a black, structure-less peat. The material is normally moderately to well decomposed, with decomposition greater at later depths.

SUBSOIL PERMEABILITY

The permeability of subsoil is largely a function of (a) the grain size distribution, (b) the amount (and sometimes type) of clay size particles present, and (c) how the grains are packed together. It can also be influenced by other factors such as discontinuities (fissures/cracks, plant roots, pores formed by soil fauna, isolated higher permeability beds or lenses, voids created by weathering of limestone clasts) and density/compactness of the deposit.

Subsoil permeability has been mapped across Ireland by the Geological Survey of Ireland as part of the National Groundwater Protection Scheme. The permeabilities mapped are qualitative regional assessments of the subsoil based on how much potential recharge is infiltrating and how quickly potential contaminants can reach groundwater. A map of subsoil permeability across Ireland is shown in Figure 2.

In poorly sorted sediments such as glacial tills, the characteristics described above detail the engineering behaviour of the materials as detailed in the subsoil description and classification method derived from BS 5930:1999 (Swartz, 1999). This method is used to assess the subsoil permeability at any locality, and is combined with recharge and drainage observations in the surrounding area for a regional, three-dimensional classification. Each approach used in assessing the permeability is discussed briefly here. Some are described in more detail in the research theses of Lee (1999) and Swartz (1999):

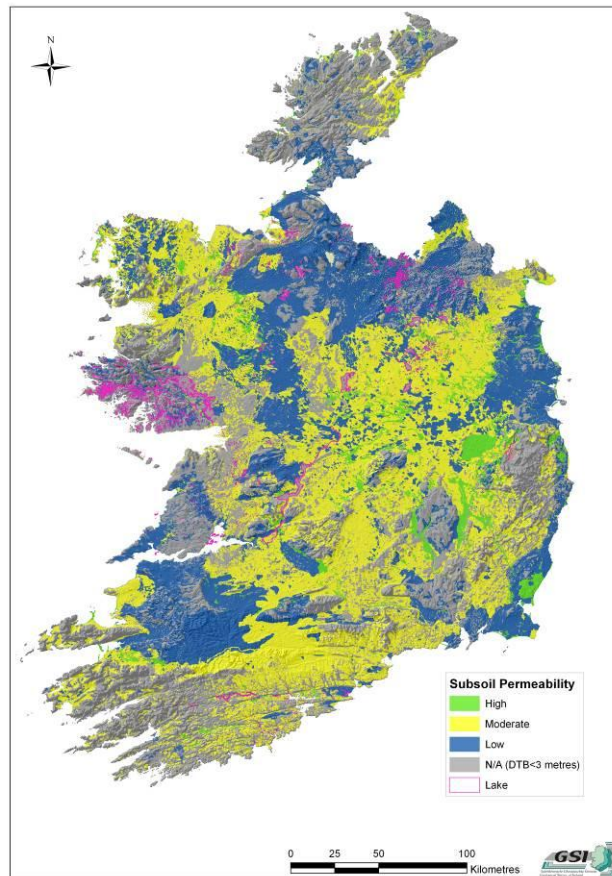


Figure 2: Subsoil permeability map showing different permeability classes across the Irish landmass.

Subsoil Description and Classification Method (derived from BS 5930:1999). Using this method, subsoils described as sandy CLAY or CLAY has been shown to behave as low permeability materials. Subsoils classed as silty SAND¹ and sandy SILT, on the other hand, are found to have a moderate permeability (Swartz, 1999). In general, sands and gravels that are well sorted have a high permeability. Permeability mapping focuses on areas where soil and subsoil are thicker than 3m, since areas with thinner soil/subsoil than this are automatically considered ‘Extremely Vulnerable’ with respect to groundwater pollution.

Particle Size Analyses. The particle size distribution of sediments describes the relationships between the different grain sizes present. Well-sorted sediments such as water-lain gravels (high permeability) or lacustrine clays (low permeability) will, on analysis, show a predominance of grain sizes at just one end of the scale. Glacial tills, on the other hand, are more variable and tend to have similar proportions of all grain sizes.

¹ silty SAND refers to a textural classification of unsorted tills, rather than sorted glaciofluvial or fluvial material.

Despite their complexity, evaluations of the grain size analyses for a range of tills in Ireland have established the following relationships (Swartz, 1999):

- i. Samples described as 'moderate permeability', based on observations of recharge indicators (vegetation, drainage density); typically have less than 35% fines (silt plus clay).
- ii. These 'moderate permeability' samples also tend to have less than 12% clay.
- iii. Samples described as 'low permeability' frequently have more than 50% fines.
- iv. These 'low permeability' samples also tend to have more than 14% clay.
- v. 'High permeability' sand/gravel deposits tend to be sorted and have less than 7.5% fines (O Suilleabhain, 2000).

Once the general characteristics and variations have been identified, these can be extrapolated to other similar areas where permeability observations may be lacking.

Subsoil Parent Material. The subsoil parent material, which is often the bedrock, plays a critical role in providing the particles that give rise to different subsoil permeabilities. Sandstone, for example, gives rise to a high proportion of sand size grains in the deposit matrix; pure limestone provides a relatively high proportion of silt, whilst shale, shaly limestone and mudstone break down to the finer clay size particles. A good knowledge of the nature of the bedrock geology is therefore critical. It is also useful to know the direction of movement of the glaciers and the modes of deposition of the sediments as these will dictate where the particles have moved to, how finely they have been broken down, and what the relative grain size make-up and compaction are. Understanding these processes enables more informed extrapolations to be made where observations are lacking.

Recharge Characteristics. Examining the drainage and recharge characteristics in an area gives an overall representative assessment of the permeability. Poor drainage and certain vegetation species can indicate low permeability subsoil providing iron pans, underlying low permeability bedrock, high water-tables, and excessively high rainfall can be ruled out. Well-drained land suggests a moderate or high permeability once artificial drainage is taken into consideration (Lee, 1999). Table 1 indicates the broad permeability categories for natural drainage density. Rigorous analysis of drainage density was not undertaken in this project, but general abundance or absence of drainage ditches was recorded.

Table 1: Broad Permeability Categories for Natural Drainage Density (Lee, 1999)

Permeability	Natural Density	Drainage	Units
High	<0.6		km/km ²
Moderate	0.6-1		km/km ²
Low	>1		km/km ²

Table 2: Broad Permeability Categories for Artificial Drainage Density (Lee, 1999)

Permeability	Artificial Density	Drainage	Units
High	<2		km/km ²
Moderate	2-8		km/km ²
Low	>8		km/km ²

Topsoil Map. The soils map can be used to indicate broad drainage characteristics, especially where specific site recharge observations are not available. Poorly drained soils such as surface water gleys are usually related to underlying low permeability subsoil; the more free draining soils such as grey brown podzolics are more typical of the sandy and silty moderate permeability subsoil.

Quantitative Analysis. From a limited number of national field permeability measurements, the boundary between moderate and low permeability is estimated as approximately 10^{-8} - 10^{-9} m/s. While the moderate to high boundary has not yet been examined in detail, one study suggests this boundary may be in the region of 10^{-4} m/s (O'Suilleabhain, 2000). However, permeability measurements are highly scale dependent: laboratory values are often up to two orders of magnitude lower than field measurements, which in turn tend to be lower than regional assessments based on large scale pumping tests. Thus, for regional permeability mapping, qualitative assessments of the recharge characteristics and engineering behaviour of the subsoils are more appropriate than specific permeability measurements.

None of these methods can be used in isolation: a holistic approach is necessary to gain an overall assessment of each site and thereby build up a three dimensional picture of the regional permeability. In a given area, as many factors as possible are considered together in order to obtain a balanced, defensible permeability decision. In order to extrapolate from

point data to area assessments, the country is divided into geomorphic regions, usually on the basis of similar subsoil and/or bedrock characteristics. It is intended that the assessments will allow a broad overview of relative permeability across the country, in order to help focus field investigations for future developments in areas of interest. In mapping Ireland, the process is aimed at providing regional guidance is not intended to be applied to a site-specific level. Consequently, it has always been stressed that the permeability assessments are not a substitute for site investigations for specific projects. Hence the requirement for site specific trial hole assessments and percolation tests for the purposes of on-site wastewater treatment.

SITE SPECIFIC PERMEABILITY – ‘T’ TESTS

There are several different approaches to the on-site assessment set out in the CEN standards but the overall principle is to unify all the different methods of soil investigations to a single comparative numerical parameter termed LTAR (Long Term Acceptance Rate). The LTAR is defined as, “the amount of pre-treated effluent which the system can infiltrate during its lifetime without water logging or clogging” in units, l/m²/d.

The process of the site assessment in Ireland begins with a desk study to ascertain regional bedrock, subsoil permeability (see above) and aquifer characteristics, followed by a visual site inspection. Following this a trial hole is excavated, from which samples of subsoil are taken and tested on-site to determine their characteristics (gravel, sand, silt, clay). The BS5950 methodology for subsoil classification is used.

Following from this, if the trial hole shows that the site may be suitable for ground discharge, a secondary investigation is completed, which aims to determine the percolation rate of the subsoil and ultimately the LTAR. In Ireland the ‘T’ test used is an on-site falling head test, which measures the time taken for water to fall 25mm in a 0.3m x 0.3m hole. The result is expressed in minutes. The hole itself is 0.4m deep, and three test holes are dug on each site.

‘T’ TEST RESULTS

Comparison was made between 198 individual ‘T’ test results and the corresponding primary BS:5930 subsoil class.

As expected, the high permeability subsoil classes had lower 'T' test results, and hence were more permeable. GRAVEL subsoil had an average value of 3.6, whereas SAND, silty SAND and clayey SAND had an average result of 14.2.

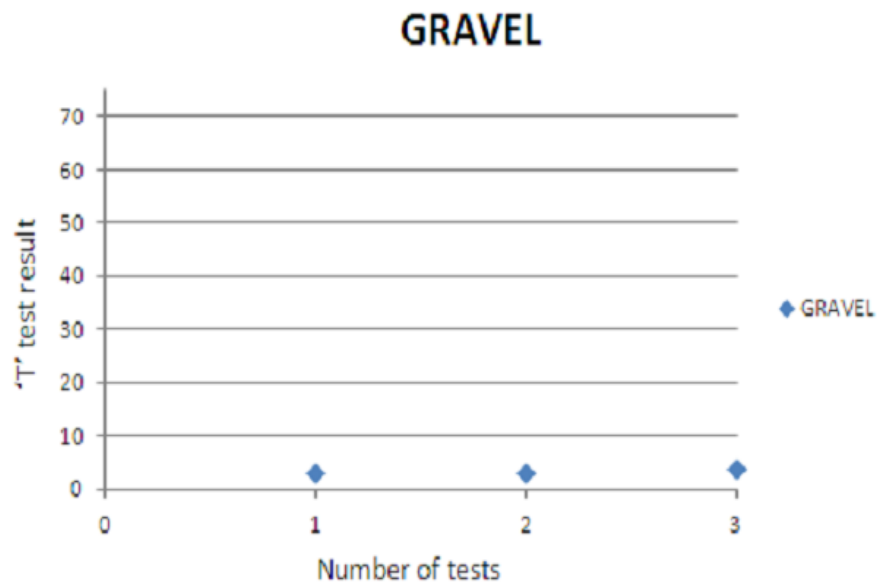


Figure 3: 'T' test results (3) for GRAVEL-dominated subsoils².

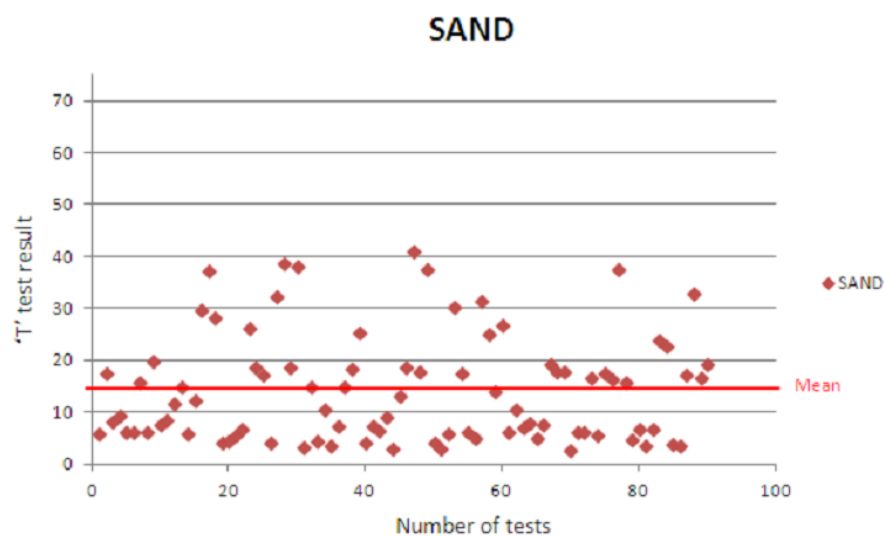


Figure 4: 'T' test results (90) for SAND-dominated subsoils.

² Few GRAVEL-dominated samples are recorded as GRAVEL accounts for a minute proportion of the subsoils across Ireland.

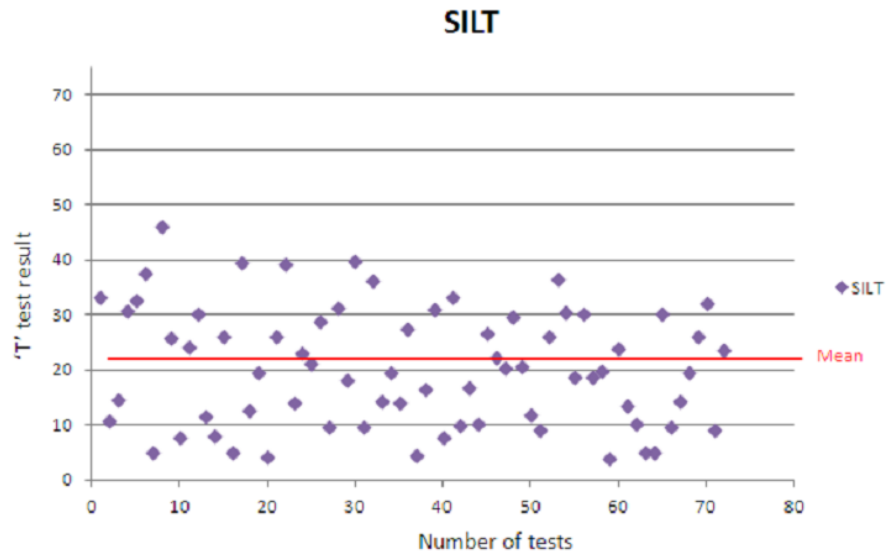


Figure 5: 'T' test results (72) for SILT-dominated subsoils.

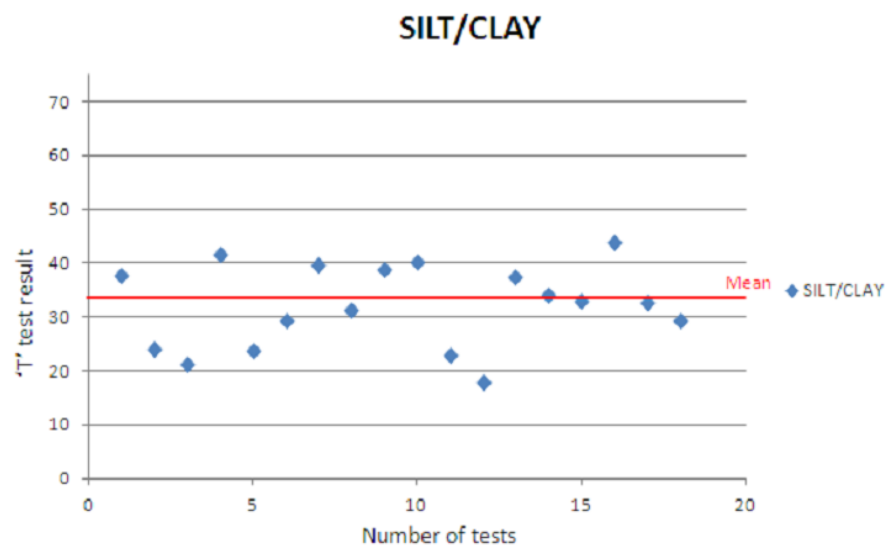


Figure 6: 'T' test results (18) for SILT/CLAY-dominated subsoils.

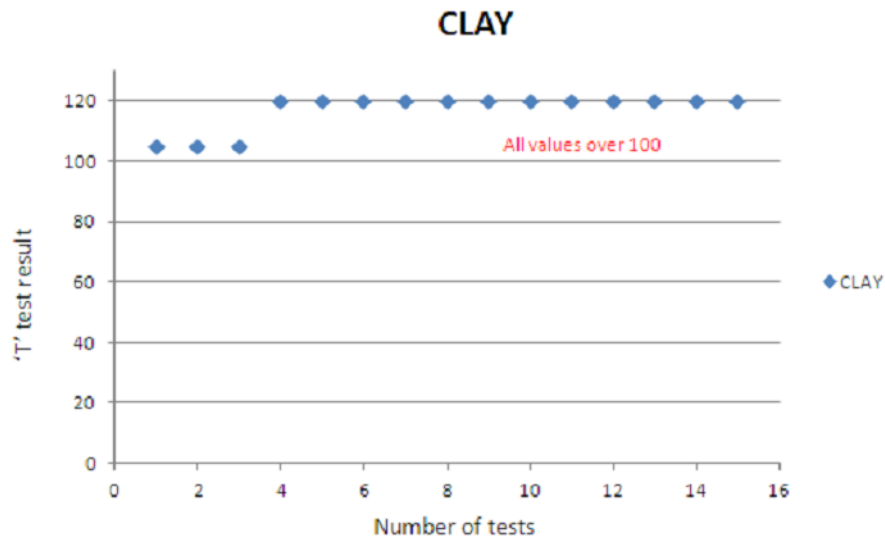


Figure 7: 'T' test results (15) for CLAY-dominated subsoils³.

As expected, as the permeability class moved through moderate to low, the 'T' test result slowed. The average 'T' result for SILT-dominated subsoil was 14.2 and that for SILT/CLAY was 32.1. However, the results for CLAY subsoils were all over 100, and measured at either >105 (*i.e.* water had not fallen 10mm in 7 hours) or >120 (*i.e.* water had not fallen 10mm in 8 hours).

A number of observations can be made from the remainder of the data and results.

Firstly, SAND- and SILT-dominated subsoils have a broad, and overlapping, range of 'T' values. The quickest 'T' result for SAND was 2.8 while the slowest was 41.1, whereas the quickest SILT result was 3.9 and the slowest was 46.3. The average result for SAND was much quicker than that for SILT, however.

The range for SILT/CLAY is less broad, between 17.7 and 43.7. The average value for this subsoil class is 32.1.

³ Few CLAY-dominated samples are recorded as sites are usually rejected at the trial hole stage for ground discharge if CLAY is present, and the site assessment rarely proceeds to the empirical percolation tests stage,

All of the CLAY results were over 100, in fact there was generally no movement in water level in CLAY subsoils at all. When measured out the 't' test result in a CLAY generally depends on the amount of time the water level is monitored, rather than relating to any permeability class in the subsoil. However, it is recognized that some CLAY subsoils will have some element of permeability, so this generalization does not hold across the board.

DISCUSSION AND CONCLUSIONS

It can be seen from the results that, as expected, the average 'T' test result, which measures the time in minutes for the water level to fall 25mm in a 0.3m x 0.3m hole, increases from GRAVEL through SAND through SILT and SILT/CLAY.

The average result in CLAY dominated subsoils is very high, as the water level in the hole generally does not move following filling.

The relationship between 'T' test results and texture of the subsoil is therefore an exponential one, where, once a critical % of CLAY is exceeded, the subsoil becomes effectively impermeable.

This has implications for the installation of domestic wastewater treatment systems in Ireland, as discharge to ground will and does not work effectively in CLAY-dominated subsoils.

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THE ATTENUATION OF ON-SITE EFFLUENT CONTAMINANTS THROUGH PERCOLATION AREAS OF DIFFERENT IRISH SUBSOILS

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ABSTRACT

*Extensive field studies on six percolation areas receiving both septic tank and secondary treated on-site effluents from single houses in Ireland was carried out to ascertain the attenuation effects of unsaturated subsoils with respect to on-site wastewater effluent. The development of a biomat across the percolation areas receiving secondary treated effluent was restricted on these sites compared to those sites receiving septic tank effluent. This created significant differences in terms of the potential nitrogen loading to groundwater. The average nitrogen loading per capita at 1.0 m depth of unsaturated subsoil equated to 3.9 g Total-N/d for the sites receiving secondary treated effluent, compared to 2.1 g Total-N/d for the sites receiving septic tank effluent. Relatively high nitrogen loading was, however, found on the septic tank sites discharging effluent into highly permeable subsoil that counteracted any significant denitrification. Phosphorus removal was generally very good on all of the sites although a clear relationship to the soil mineralogy was determined. The research also indicates that the septic tank effluent was of an equivalent quality to the secondary treated effluent in terms of indicator bacteria (*E. coli*) after percolating through 0.6 m depth of unsaturated subsoil, with slightly more evidence of faecal contamination occurred at 1.0 m depth at the sites with more highly permeable subsoils. Three bacteriophages were also spiked into two systems and mainly reduced to their minimum detection limit at a depth of 1.0 m below the percolation trenches. However, slightly higher breakthroughs of MS2 and PR772 contamination were detected at the same depth under the trenches receiving secondary treated effluent.*

Keywords: on-site wastewater; septic tank; percolation; subsoil; nitrogen; phosphorus.

INTRODUCTION

The domestic wastewater of approximately one third of the population in Ireland, ~500 000 dwellings, is treated by on-site domestic wastewater treatment systems (CSO, 2011). The potential impacts of such on-site effluent are the pollution of either groundwater and / or surface water. If the effluent loading on the subsoil is too high, the permeability of the subsoil excessive or there is an insufficient depth of subsoil then the groundwater beneath a percolation area is at risk of pollution, in particular from microbiological pathogens and / or nutrients. Alternatively, if there is insufficient permeability in the subsoil to take the effluent load, surface ponding may occur with associated health risks, and there will be a risk of effluent discharge / runoff of pollutants to surface water and also wells. The nutrient load in the effluent (either as direct discharge or from groundwater baseflow) can contribute to eutrophication in sensitive water bodies, whilst contamination of water sources by human enteric pathogens can promote the outbreak of disease. The unsaturated subsoil above the water table or bedrock into which on-site effluent is discharged (i.e. the percolation area) is therefore an integral part of the overall on-site treatment system, particularly since the main aquifers in Ireland occur in fissured or fractured bedrock formations overlain by subsoils of variable thickness and permeability.

The soil treatment system, comprised of a series of subsurface percolation trenches, is a crucial component of the gravity flow treatment system with much research concentrating on the flow of effluent and the mechanisms of pollutant attenuation within the subsoil (Jenssen & Siegrist, 1990; Beal *et al.*, 2005 etc). The biogeochemical mechanisms for purification and hydraulic performance are complex and have been shown to be highly influenced by the biomat zone which forms at the soil-gravel interface along the base and wetted sides of the percolation trenches. Reduced percolation rates through the biomat due to clogging as a result of anaerobic activity can cause the effluent to pond above the biomat but leaves unsaturated conditions below, for aerobic degradation processes to operate on percolating effluent (Bouma, 1975; Siegrist & Boyle, 1987). The development of a biomat takes several months but will eventually reach a steady state equilibrium which the Long Term Acceptance Rate - the basis of design codes in Europe (CEN, 2006) and elsewhere – attempts to define.

Two sequential projects funded by the Irish EPA have been undertaken to test out the efficacy of on-site wastewater treatment systems on a range of sites with subsoils of different percolation characteristics receiving domestic wastewater effluent to assess the implications particularly with respect to groundwater pollution due to housing development in unsewered, rural areas.

METHODS

Site selection and construction

Six separate households in Ireland were used for the research projects on sites of varying subsoil permeability. In all sites new on-site treatment systems were constructed at the beginning of each study. Two-chamber septic tanks were installed on Sites 1, 2 and 3 while secondary treatment systems were installed on the other sites: a rotating biological contactor, RBC (Biodisc®, Klargester) on Site 4 and peat filters (Puraflo®, Bord na Mona) on Sites 5 and 6. The six sites were chosen to cover a range of subsoil conditions as shown on Table 1. The percolation T-values of the different subsoils span the range 3.7 to 52 as determined by the onsite standardised Irish falling head percolation test, the T-test, equivalent to field saturated hydraulic conductivities in the range 0.08 to 1.05 m/d (Elick & Reynolds, 1986). The effluent from all six sites (three sites discharging septic tank effluent (STE) and three sites discharging secondary treated effluent (SE) from packaged plants) entered percolation trenches at 2.45 m centres built to EPA specifications (EPA, 2000) consisting, in each case, of 110 mm diameter perforated PVC pipe sitting on a bed of 250 mm of washed gravel (20 to 30 mm diameter) in a 450 mm wide trench (see Fig. 1) at a slope of 1:200. The design specification was generally one 20 m long trench per person per household. The achievement of an equal loading rate on each trench was achieved by a distribution box specifically designed for the projects.

Table 1: Summary of site characteristics

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Effluent type	STE	STE	STE	SE	SE	SE
No. residents	6	4	4	3	5	4
T-value	3.7	15	33	4.5	29	52
Subsoil classification [†]	gravely SILT	sandy CLAY	gravely, clayey SAND	sandy GRAVEL	sandy SILT / CLAY	gravely, clayey SAND
k_{fs} , m/d [#]	1.05	0.28	0.13	0.84	0.15	0.08
LTAR, L/m ² d [§]	40	20	10	40	15	10

[†] see BS5930 (British Standards Institution, 1999)

[#] Field saturated hydraulic conductivity

[§] Long Term Acceptance Rate - see CEN, 2006

INSTRUMENTATION AND SAMPLE ANALYSIS

The flow profile to the percolation areas from the septic tanks and secondary treatment systems was measured using tipping bucket flow-gauges (Unidata, Australia) placed underneath each of the four distribution box outlets at each site. Automatic samplers (Bühler Montec) collected 24 hour composite samples of STE and SE. Suction lysimeters (Soilmoisture Equipment Corporation) were installed along the length of each trench to nominal depths of 0.3, 0.6 and 1.0 m below the invert of the percolation trenches respectively (Fig. 1). At each site nine tensiometers (Soil Measurement Systems) were installed at the same three sampling depths along separate trenches in order to obtain a profile of soil moisture tension across the percolation area. Meteorological variables (rainfall, temperature, wind speed, relative humidity, solar radiation and sunshine hours) on each site were recorded by a weather station (Campbell Scientific) and tipping-bucket type rain gauges (Casella).

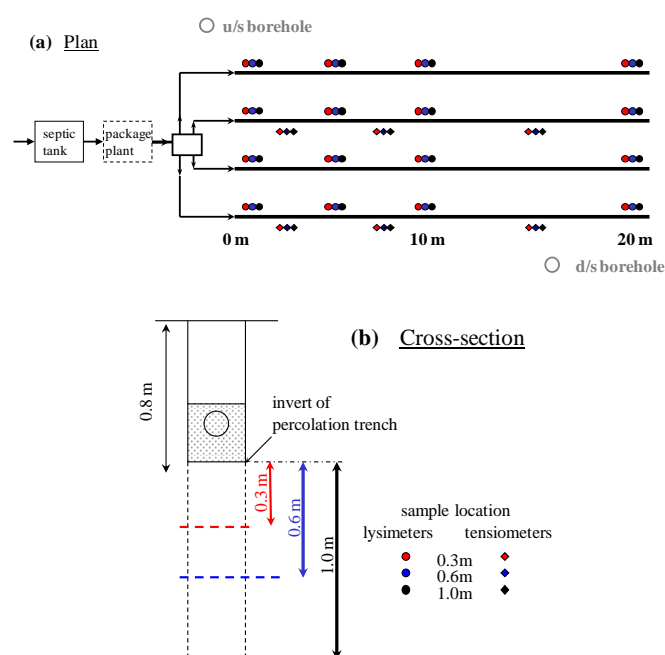


Figure 1: (a) Plan and (b) cross-section view of instrumentation layout of study sites.

All STE, SE and soil moisture samples were analysed for ammonium, nitrite, nitrate, chemical oxygen demand, orthophosphate and chloride using a Merck Spectroquant Nova 60[®] spectrophotometer and associated reagent kits. Samples were also tested for total nitrogen using a Hach Lange DR2800 to ascertain the fraction of organic and inorganic nitrogen present. The presence of *E.coli* in samples was detected using the Idexx Colilert[®]-18 test (IDEXX Laboratories Inc. Westbrook, Maine).

To investigate the potential fate and transport of enteric viral pathogens in the subsurface environment of the percolation area, a multi-phage injection experiment was conducted using a selection of bacteriophages (MS2, ΦX174 and PR772) on each of the percolation trenches of highly permeable subsoil Sites 1 and 4. They were obtained from the American Type Culture Collection (Manassas, VA, USA) and grown on their host *E. coli* lawns by the agar-overlay method while enumeration of the phages was performed by the plaque forming unit (PFU) method. The minimum detection limit for all three phages was < 10 PFU/mL.

Sites 2, 3, 5 and 6 were each studied for a period of 12 months, compared to Sites 1 and 4 which were studied over 32 months in order to monitor any further biomat spread and trends in removal processes with respect to time.

RESULTS

The extent of the active percolation area (i.e. spread of the biomat) first needed to be established in order, (i) to decide which of the lysimeters were sampling percolating effluent and, (ii) to factor in the effect of any rainfall dilution on the samples due to recharge through the subsoil.

Extent of active percolation area and effect of dilution by rainfall. Chloride was used initially as a tracer to identify the extent of the percolation area which was receiving wastewater effluent. The results of the Cl analysis at the different sample positions at the three different depth planes and along each trench were analysed with respect to time. From this a conceptual model (assuming isotropic and homogeneous soil properties) was derived for the analysis of the attenuation of the percolating effluent according to how far across the percolation area the biomat was spreading. The concentrations of all the other measured parameters were then averaged across all trenches and across all sampling positions where the effluent was known to have reached at the 0.3, 0.6 and 1.0 m depth planes.

An example of the method of calculating loads is shown in Fig. 2(a) for Site 2, where similar chloride loading rates for all planes indicate that the effluent had spread across the whole length of the percolation area. Similar analysis of Cl levels carried out on Site 4 (see Fig. 2(b)) showed decreased concentrations at the 5, 10 or 15 or 20 m sample distances along the same depth planes, suggesting that the average results across the trenches and depth planes recorded at the 0 m sampling positions only would be the more representative way to assess effluent attenuation on those sites.

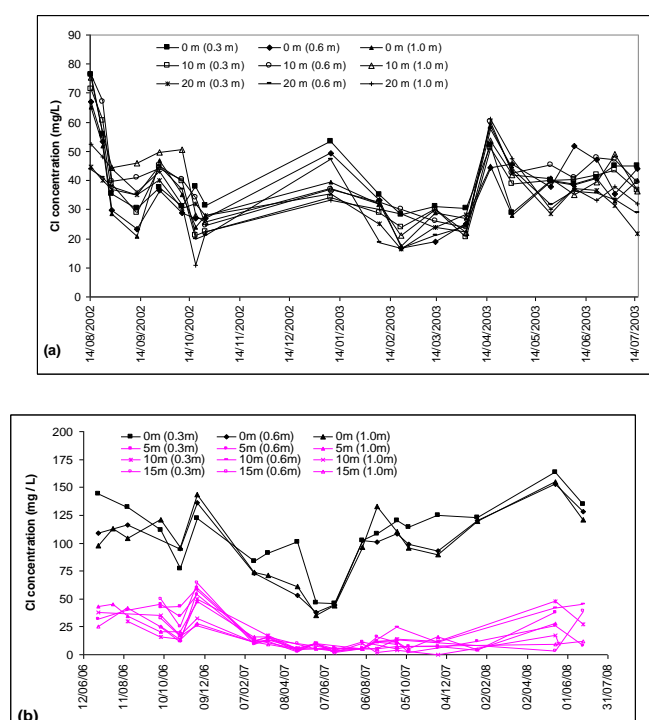


Figure 2: Average Cl concentrations across all depth planes across the percolation area: (a) Site 2 and (b) Site 4.

Loading rates of pollutants at the different depths were then calculated on a daily basis according to a mass balance of effluent flow plus any rainfall recharge. The extent of effluent dilution in the subsoil from effective rainfall was calculated using rainfall and evapotranspiration values obtained on site and calculations based on the Penman-Monteith method. Effective rainfall was calculated by subtracting the daily actual evapotranspiration and accumulated soil moisture deficit figures from the daily rainfall measurement.

The effect of dilution on the attenuation of the percolate was then calculated by determining the zone of contribution around each trench, using the difference between data showing the reduction in Cl concentration between the trench influent (i.e. STE, or SE) and at the three depth planes during periods of effective rainfall compared to periods of zero effective rainfall. Having quantified the dilution factors at each depth plane, a simple mass balance approach was adopted to estimate the zone of contribution (A_c) of effective rainfall at each depth plane. These calculations were carried out on all sites to yield accurate lengths of the biomats along the trenches, as shown on Table 2. It is evident from the data that the reduced organic loading brought about by the additional secondary treatment of wastewater prior to discharge to the percolation areas had inhibited the formation of a biomat preventing distribution of the effluent along the entire base of the trenches. The installation of the secondary treatment systems greatly reduced the organic load on the percolation areas as expected with an average 77% lower COD loads being discharged to the

trenches on Sites 4, 5 and 6 compared to the sites with septic tanks. The effluent only reached up to a maximum of 4 m along the length of the SE percolation trenches with a considerably shorter biomat development for the higher permeability subsoil sites.

Examination of the soil moisture tension values from the tensiometers installed at the different sample positions (front, middle and back) along the trenches were also used to corroborate the conceptual models for each site. For example, tensiometers at the three depth planes sample locations where no effluent was recorded responded directly to variations in effective rainfall over the total sampling period. By comparison, the tensiometers at sample locations within the plume remained more stable across the trial period as the subsoil soil moisture conditions were influenced more by the percolating effluent than by the contribution of effective rainfall.

Nitrogen loadings and removal. The mean daily on-site wastewater flow rates on the six sites as measured continuously over the sampling periods by the instrumentation are shown on Table 2. The average total-nitrogen loads of STE and SE discharged into the percolation trenches on each site over the research periods are also shown. When these were compared to the respective loading rates after percolating through 1 m depth in the subsoil (taking into account the dilution by effective rainfall) significant differences were found between the sites receiving STE and SE, particularly when comparing the sites of moderate permeability (Sites 2 and 3 (STE) and Sites 5 and 6 (SE)). The nitrogen in the STE, discharging mainly in ammoniacal form, was seen to nitrify in the gravel and in the upper 0.3 m of subsoil and then denitrify in the sporadically saturated conditions below the biomat, often termed anoxic microsites. In comparison, the secondary treatment systems on Sites 4, 5 and 6 were clearly discharging at least partially nitrified effluent but while the SE underwent further slight nitrification within the subsoil, the nitrate remained largely unchanged as it percolated down through the subsoil, leaving potentially higher total-N loads to the groundwater. The equivalent reduction in total-N load between the SE and after 1.0 m of unsaturated subsoil on Sites 4, 5 and 6 was only 25%, 9% and 24% respectively compared to 59%, 76% and 89% on the STE sites. Fig. 3 compares this difference between the percolating STE on Site 2 to the SE on Site 5 where total-N loads of 16.7 g/d were found at the nominal point of discharge to groundwater on Site 5 compared to 6.8 g/d at the same depth on Site 2. The inhibition to biomat formation along the percolation trench on the sites receiving SE would result in a reduction in microbial denitrification (promoted in the biomat by the reducing conditions). In addition, even if saturated conditions existed locally within the subsoils on Sites 4, 5 and 6, the organic load of the percolating effluent may not have been sufficient to support the facultative heterotrophs required for denitrification.

Table 2: Site characteristics and average total nitrogen loads across research periods

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Mean flow	119	105	82	90	60	123
per capita (L/d)						
Biomat length	11	20	8	0.4	3	4
along trenches (m)						
COD loading	72.0	40.2	108.4	12.8	14.1	24.5
per trench (g/d)						
Effective hyd.	24.0	11.7	22.8	373.3	41.7	54.7
load [†] (L/m ² .d)						
Influent total-N	60.4	28.9	19.6	20.2	18.4	17.8
to subsoil (g/d)						
Total-N after 1m depth (g/d)	25.0	6.8	2.1	15.2	16.7	13.6
Total-N per capita after 1m depth (g/d)	4.17	1.70	0.53	5.07	3.34	3.40

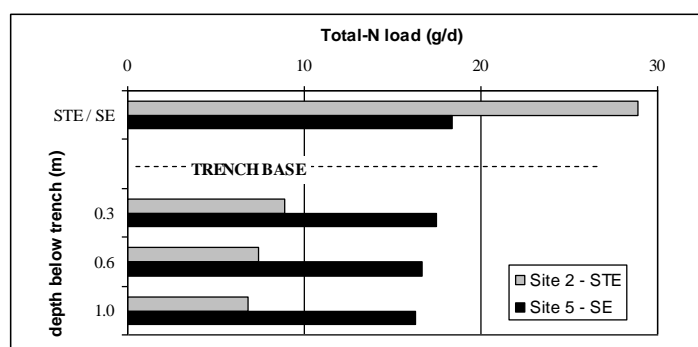


Figure. 3: Total N-loading with depth (Site 2 compared to Site 5).

An interesting correlation can be seen in Table 2 between higher total-N loads per capita at 1 m depth below the base of trench and subsoil permeability (Table 1), for both STE and SE.

This is particularly evident on the highly permeable subsoil Sites 1 and 4. The SE again remained largely untouched in its nitrate form en route to the groundwater as expected, but the STE on Site 1 did not appear to be denitrify as thoroughly as on Sites 2 and 3 whilst nitrification was still occurring between the 0.3 and 1.0 m depth planes. The high permeability subsoil had muted the spread of the biomat even under a high organic loading promoting a relatively high hydraulic loading on that localized area but maintaining unsaturated aerobic conditions in the subsoil beneath.

The average results from the sites show that the potential total-N load to groundwater beneath the percolation areas from the secondary treatment systems was approximately twice that compared to the sites receiving just septic tank effluent, or up to three times greater if the high permeability sites (which are rare in Ireland) are ignored.

Phosphorus loadings and removal. The secondary treatment systems on Sites 4, 5 and 6 had little effect on the ortho-P loadings of the STE, acting to reduce their respective influent loads by an average 12%. These package treatment systems are not usually designed to remove P, apart from the reduction that occurs as a result of biological assimilation.

The attenuation of ortho-P within the subsoil is controlled by soil adsorption and mineral precipitation, reactions that are closely related to soil pH. The results in Table 3 show that the majority of the phosphate removal on Sites 1, 2, 3 and 4 were achieved within the first 0.3 m of subsoil. X-ray diffraction analysis of the subsoil in Site 1 showed that it contained quartz, feldspars, chlorite and calcite with a small amount of swelling illite-smectite clay (soil particles of a larger specific surface area which are more conducive to phosphorus adsorption) in addition to presence of calcite under the prevalent alkaline conditions. Particle size distribution (PSD) analysis of the subsoils on Sites 2 and 3 showed that they contained a higher clay content which may account for the greater percentage removal of ortho-P above the 0.3 m depth plane. The high phosphate removal in Site 4 is more surprising perhaps, as the mineralogy of the subsoil did not suggest significant potential for phosphate load removal: X-ray diffraction analysis of the sandy silt revealed no calcite or oxides of Al, Fe and Mn. The PSD analysis however, did reveal 11% clay at 0.2 m depth plane, which may be partially responsible for sorption of the phosphate ions and a reduction in clay content with subsoil depth, mirroring the reduction in phosphorus fixation with depth. Interestingly, the high hydraulic loading on this site resulting from the limited spread of the biomat (see Table 2) also did not appear to affect the phosphate removal.

Table 3. Site characteristics and average ortho-phosphate loads across research periods.

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Effluent	STE	STE	STE	SE	SE	SE
pH of influent	7.4	7.7	7.7	7.2	6.4	6.4
pH at after 1 m depth	8.0	8.0	7.1	7.6	8.2	6.6
Influent ortho-P (g/d)	13.2	5.9	1.2	7.0	9.5	2.0
Ortho-P at 0.3m depth (g/d)	1.2	1.2	0.1	0.8	6.9	1.7
Ortho-P at 0.6m depth (g/d)	0.8	0.7	0.0	0.2	5.8	1.3
Ortho-P at 1m depth (g/d)	0.6	0.6	0.0	0.2	2.2	0.2
Ortho-P at 1m per capita (g/d)	0.10	0.15	0.00	0.07	0.44	0.05

The phosphate removal through the subsoil at Site 6 was relatively low through the first 0.6 m of subsoil, particularly as the influent load was comparatively small on that site. PSD analysis of the subsoil showed it to contain a relatively low clay content, despite its low permeability (which was attributed to the high density of the matrix) and X-ray diffraction analysis revealed neither calcite nor oxides of Al, Fe and Mn. The comparatively low pH of the percolating effluent on Site 6, along with the high hydraulic loading, may explain the less efficient phosphate removal with depth. The results at Site 5 stand out from the other sites as significant ortho-P concentrations (average 6.9 mg/L) were picked up in the effluent after percolating through 1.0 m of subsoil. There was a noticeable increase in ortho-P fixation between the 0.6 m and 1.0 m depth planes. X-ray diffraction analysis showed that while the subsoil at this depth contained calcite, it was devoid of Al, Fe and Mn oxides, hydrous oxides or dissolved ions and therefore, fixation would be confined to the high pH range – consistent with the alkaline conditions that did exist at Site 5. This site also had a relatively high hydraulic loading due to the limited spread of the biomat and this may have contributed to the less effective phosphorus removal.

Finally, the potential for a reduction in phosphate removal over time - due to the available sites for PO₄-P adsorption sites becoming filled - was examined on all sites but, showed no apparent reduction in removal performance during the trial periods.

E. coli removal. The average concentration of total coliforms in the STE from the sites was 7×10^7 MPN/100mL with average *E. coli* concentrations of 8×10^5 MPN/100mL. The installation of secondary treatment systems greatly reduced the bacterial loads on the percolation areas but still left relatively high concentration of bacteria in the SE (average *E. coli* concentrations of 3×10^4 MPN/100mL) which would be unsuitable for discharge to groundwater prior to further attenuation in the subsoil.

The results of the bacteriological analysis demonstrated the ability of the subsoil and associated biomat to remove enteric bacteria from the percolating effluent. The presence of *E. coli* in samples obtained from the 1.0 m depth plane on Site 2 showed that, except for the one incidence where 10 MPN/100mL was detected, complete removal of enteric bacteria was achieved by the 1.0 m depth plane. While the installation of the secondary treatment systems greatly reduced the bacterial load, there was some evidence of *E. coli* contamination at the 1.0 m depth plane on Site 5. The subsoil at this site had a high sand content which may have facilitated the movement of bacteria through the subsoil. The reduced biomat development, as discussed earlier, would also have had the effect of increasing the hydraulic load per unit area. The results from Sites 3 and 6 at the 0 m sampling positions indicated no evidence of *E. coli* contamination past the 0.3 m depth plane.

Analysis of the *E. coli* data measured at the two sites with highly permeable subsoil (Sites 1 and 4) showed almost complete removal of enteric bacteria from the STE at Site 1 was achieved over the 2 year monitoring period by the 1.0 m plane with the exception of isolated incidences of low faecal contamination on two separate dates (in May and December 2007) coinciding with high levels of rainfall. Equally, the results at Site 4 showed almost complete removal of enteric bacteria from the SE was achieved over the 2 year monitoring period by the 1.0 m depth plane, although there were a slightly higher number of isolated breakthroughs of low concentrations of enteric bacteria (10 to 20 MPN/100mL) and one more significant breakthrough at 260 MPN/100mL. Such incidences again appeared to coincide with periods of increased hydraulic load to the percolation area owing to significant effective rainfall events which could have promoted temporarily saturated conditions thereby facilitating microbial transport and breakthrough at depth.

Bacteriophage spiking trials. Fig. 4 illustrates the minimum planar log removal of each bacteriophage detected at all sample positions over the duration of the trial at Site 1. Phage values considered were only those seen to coincide with incidences of bromide evidence (spiked at the same time) at that point. Greatest removal in the initial 0.3 m below the infiltrative surface was achieved by phage MS2 (1.1 log-unit), followed by PR772 (0.6 log-unit ~ 78%) and Φ X174 (0.6 log-unit ~ 75%). With the majority of the MS2 now retained by the 0.3 m depth, the most significant removal of Φ X174 and PR772 phages occurred between the 0.3 and 0.6 m depth planes with a further 1.6 and 2.9 log units attenuated,

respectively. By the time the percolate had reached the 1.0 m depth plane on day 3 (as evidenced by bromide detection) MS2 had been reduced to their minimum detection limit of < 10 PFU/mL, whilst ΦX174 and PR772 recorded 20 PFU/mL on one trench and < 10 PFU/mL at the expected peak time of arrival. This equates to an overall percentage removal of > 99.82% (MS2), > 99.77% (ΦX174) and > 99.99% (PR772) between the infiltrative surface and 1.0 m depth plane. Hence, in summary, almost complete removal of the three phages was achieved over an unsaturated subsoil depth of 1 m on Site 1.

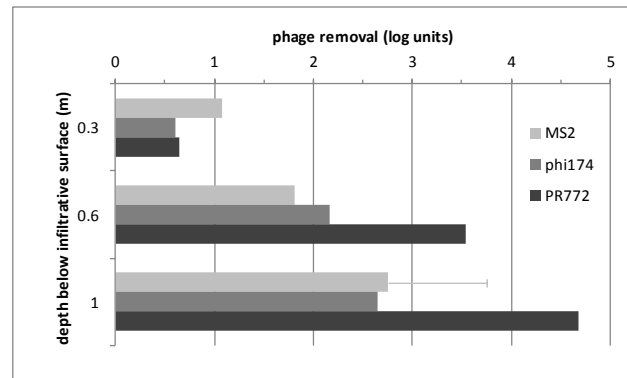


Figure. 4: Phage log-unit removal (Site 1) recorded at each depth plane at all sample positions over the duration of trial.

The phage tracer experiment carried out on the percolation trenches receiving SE at Site 4 showed similar initial removal of all three phages over the first 0.3 m to Site 1 at rates of 1.3 log-unit, 1.6 log-unit and 0.9 log-unit for MS2, ΦX174 and PR772, respectively. Subsequent attenuation of the phages over the remaining 0.7 m resulted in all three achieving a further 1.2, 1.9 and 3.9 log-unit removal for MS2, ΦX174 and PR772, respectively. Averaging the results from the three phages produces a mean phage removal of at least 3.6 log-units between the infiltrative surface and the 1.0 m plane. While the majority of phage concentrations recorded at the critical 1.0 m depth plane were below the limit of detection (< 10 PFU/mL), MS2 was detected at a concentration of 50 PFU/mL on day 1 while PR772, at 20 and 30 PFU/mL, was picked up at two samples point a day later. Phage detection at this depth over such a short-term trial highlights a risk of viral contamination through such highly permeable subsoil in the underlying aquifer. However, as these were single spiking trials, more comprehensive research is needed to investigate the exact physical and chemical properties governing viral fate and transport in the vadose zone which will help to understand the hydrogeological formations affecting groundwater vulnerability.

CONCLUSIONS

The results from the field studies on six sites have shown that the septic tank and percolation area provided a comparable treatment performance with respect to groundwater protection to the packaged secondary treatment system with percolation area, without the need for ongoing maintenance or energy consumption.

The potential nitrogen loading per person to the groundwater beneath percolation areas receiving on-site secondary treated wastewater effluent was approximately two to three times that from the equivalent septic tank percolation areas. The reduced biomat formation along the percolation trenches receiving secondary treated effluent resulted in more concentrated hydraulic loading of the effluent, with a lower organic content that has limited denitrification. There was also a correlation between higher resultant nitrogen loadings beneath the percolation areas with the more highly permeable sites for both secondary treated and septic tank effluent, which again could be linked to spread of the biomat.

Phosphorus removal was linked to subsoil mineralogy and appeared to be independent of the hydraulic loading rate. However, the relatively low potential P loadings beneath the on-site wastewater percolation areas may still be significant with respect to groundwater baseflow discharging into rivers and other surface water bodies.

The analysis of indicator bacteria (*E. coli*) in the percolation areas showed that the percolating septic tank effluent had reached an equivalent quality to the percolating secondary treated effluent by the 0.6 m depth of unsaturated subsoil and almost complete removal was achieved by the 1.0 m depth plane. However, higher breakthroughs were picked up on the two sites with more highly permeable subsoil (T-values ~4) linked to significant rainfall events. Equally, the bacteriophage spiking trials at the sites with highly permeable subsoils showed that all three bacteriophages were significantly reduced to close to the minimum detection limit after passing through a depth of 1.0 m subsoil below the percolation trenches receiving both septic tank effluent and secondary treated effluent, although isolated incidences of breakthrough were measured on both sites with slightly higher breakthroughs of MS2 and PR772 contamination detected under the trenches receiving secondary effluent.

In general this research has highlighted the crucial role that the unsaturated subsoil and associated biomat formation at the trench-subsoil interface plays with respect to the attenuation of pollutants of on-site effluent. The development of a biomat across the percolation areas receiving secondary treated effluent by gravity flow was restricted on these sites compared to those sites receiving septic tank effluent which created significantly higher localised loadings on the subsoil within those areas.

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UNDERSTANDING MICROBIAL DENITRIFICATION AND PATHOGEN TRANSPORT IN EFFLUENT AND SOILS

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ABSTRACT

Inadequate percolation areas, associated with single house wastewater treatment systems, can result in the inefficient treatment of domestic wastewater. The release of large volumes of effluent onto low permeability sub-soil, in particular, can lead to surface ponding and the subsequent release of pollutants to surface water bodies. Furthermore, the intensification of agriculture has led to an increase in organic and inorganic fertiliser inputs to agro-ecosystems, which can result in negative effects on water quality. Key issues include: eutrophication, depletion of dissolved oxygen, excessive algal growth in surface water bodies and the outbreak of infectious disease caused by human enteric pathogens. A better understanding of the factors influencing microbial denitrification and the behaviour of pathogens in sensitive environments may lead to the identification of management options for reducing nitrous oxide (N₂O) emissions and nitrate (NO₃⁻) leaching; and for enhanced protection of public health. This paper summarises investigations on the potential for microbial denitrification in soils, at six sites under varying land management regimes; and in groundwater sampled from four aquifers. Nitrogenous emissions (N₂O; N₂) were successfully correlated with the size and composition of the microbial denitrifier community in both soil and groundwater, while studies on denitrification in septic tank effluent percolation areas are ongoing. We also report on the development of a method, which allows the determination of the source (human and/or animal) of faecal contamination in soils, groundwater and septic tank, or other wastewater treatment plant, effluents. This approach should facilitate future studies of pathogen transport in soils impacted by human or animal faecal inputs.

Keywords: wastewater treatment; nutrient removal, denitrification, enteric pathogens, Q-PCR.

INTRODUCTION

Nutrient inputs (especially phosphorus and nitrogen; Spon & Spon, 1999) can cause eutrophication in natural waterbodies. Eutrophication refers to enhanced biological production associated with a reduction in the available dissolved oxygen in the water column. Worldwide, eutrophication has been observed as a consequence of human population increases, especially in relatively small geographical areas. Eutrophication occurs not only in lakes or rivers near cities, however, but also increasingly in rural areas, where natural nutrient recycling processes may be disrupted (Örnólfsson *et al.*, 2004). In rural areas, where “main-line” sewer systems are not available, nutrient levels can increase in local water bodies as a result, for example, of malfunctioning septic tank soakaways and the excessive use of agricultural fertilizers (Conley *et al.*, 2009; Figure 1). Areas with low permeability subsoil are particularly vulnerable, as surface ponding may result in loss of nutrients to near-by surface water bodies.

Nitrogen (N) release to soils, surface water and groundwater systems, originating mainly from excessive application of fertiliser, manure and sewage sludge, has increased over recent decades (Stark & Richards, 2008). Legislation, such as the Nitrates Directive (91/676 EEC, European Commission) aimed at controlling agricultural management practices, has been introduced to limit nitrate (NO_3^-) leaching from soils to rivers, lakes and underground aquifers. There can be serious health effects associated with consumption of water containing elevated nitrate levels. For example, links have been established between elevated nitrate levels and methaemoglobinemia (blue baby syndrome), increased incidence of gastric and intestinal cancer and adverse reproductive effects (Schlesinger, 2009). Microbial denitrification removes nitrate in various environments as it proceeds along a pathway comprising four distinct sequential enzymatic steps involving: (i) nitrate reductase (*nar*), (ii) nitrite reductase (*nir*), (iii) nitric-oxide reductase (*nor*; producing N_2O) and (iv) nitrous oxide reductase (*nos*; producing N_2) to complete denitrification (Zumfit, 1997; Braker *et al.*, 1998; Figure 2). Many organisms, such as *E. coli*, can carry out partial denitrification, but lack the *nos* gene and thus produce N_2O as the end-product. Assays, based on the presence of the *nos* gene or gene product, which catalyses the further reduction of N_2O to N_2 , can thus distinguish partial (N_2O producers), from complete denitrifiers (N_2 producers; Zumfit, 1997).

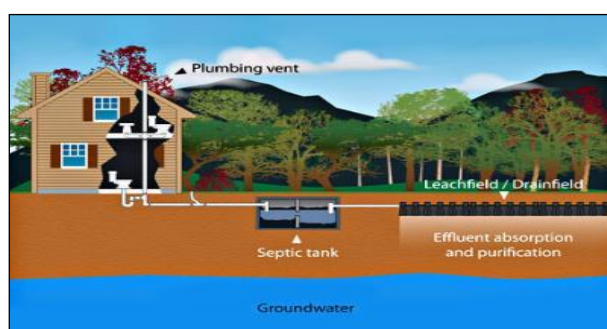


Figure 1: A typical rural domestic wastewater system (www.epa.ie).

In addition to the problems of eutrophication and nitrate leaching, there are significant public health concerns associated with water borne infectious disease outbreaks arising from inadequate wastewater treatment and agricultural discharges. For example, there are in excess of 200,000 private drinking water supplies in Ireland, with >25% of these regularly contaminated by human and animal faeces (EPA, 2010). Ireland consists of a total land area of 70,282 km², with 50% of this area comprised of karstic regions (Daly *et al.*, 2005). The rapid infiltration of microbial contaminants through overlying soil and bedrock is a concerning feature of karst aquifers, with infiltration further amplified following heavy rainfall, inadequate wastewater treatment and application of organic fertilisers. Most regions of Ireland rely heavily on these aquifers for potable supplies (86-90% in certain areas; EPA 2010).

Monitoring and establishing the source of faecal pollution is imperative for the protection of water quality and human health. Conventional culture methods to detect such pollution, via faecal indicator bacteria, are well-established but do not determine the source of pollution. To address this issue, microbial source tracking (MST), an emerging molecular tool, can be applied to detect the ubiquitous, but highly host-specific, faecal bacterial group the *Bacteroidales* (Kildare *et al.*, 2007). 16S rRNA gene-based assays for host-specific *Bacteroides* markers have been developed to distinguish between human and animal sources in faecally contaminated water, but these have not been routinely applied to complex matrices, such as soil or wastewater. For example, Kildare *et al.* (2007) developed universal (BacUni-UCD), ruminant-specific (BacCow-UCD) and human-specific *Bacteroidales* assays (BacHum-UCD) for water quality monitoring.

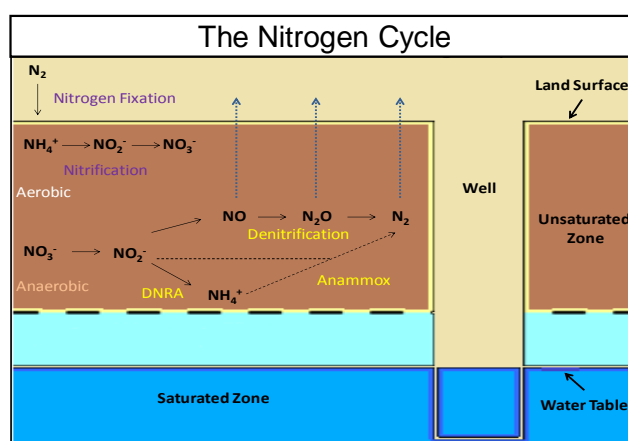


Figure 2: The biological Nitrogen cycle.

The aim of this study was to quantify denitrifying microbial populations in soil, groundwater and the percolation areas of household wastewater treatment systems, using real-time polymerase chain reaction (PCR) assays targeting the nitrite reductase (*nirS* or *nirK*) and nitrous oxide reductase (*nosZ*) functional genes. A further aim was to establish a methodology for MST of faecal contamination detected in soil, groundwater, surface water

and effluent samples, and to monitor the transport of pathogens specific to on-site wastewater outflows.

MATERIALS AND METHODS

Soil samples. Soil samples were collected from six sites under four different land management strategies (Barrett *et al.*, 2012b): Johnstown Castle (grazing grassland); Solohead (clover and ryegrass grassland); Oak Park forest (conifer forest); Oak Park tillage (barley/wheat); Oak Park (Grassland) and Dairygold (grassland). Soil variability at each site was taken into account by sampling three separate soil profiles.

Groundwater. Groundwater was sourced from multi-level piezometers, which were installed to target different groundwater zones as described by Jahangir *et al.* (2011).

Wastewater treatment systems. At two low permeability subsoil sites in Co. Monaghan and Co. Kilkenny (Table 1), suction lysimeters were installed in the soakaway area in two zones at a range of depths (1.2 m, 1.45 m and 1.8 m; Figure 3). The lysimeters were pressurised 24 hours prior to sample retrieval under suction of approximately 50.7 kPa (Gill *et al.*, 2009).

Table 1: Sample site characteristics.

Site	Type Dwelling (no. of residents)	Subsoil Type	T-value (Percolation test)	Groundwater Vulnerability
Co. Monaghan	Detached Bungalow (4)	Clay	73	Moderate
Co. Kilkenny	Detached Bungalow (3)	Shales and sandstone till	75	High

Septic tank, soil moisture/effluent, groundwater (upstream and downstream boreholes) and surface water (where possible) samples were retrieved at each site on a near-monthly basis throughout a seven month period from November 2011-July 2012. Samples were collected using a vacuum-pressure hand pump, collection flask and extraction tube. Samples were transported in a cool box to the laboratory and stored at 4°C until filtration. Total coliform and *E. coli* counts were performed using the IDEXX Quanti-Tray/2000 on each of the groundwater and surface water samples upon arrival in the laboratory in which the Most Probable Number (MPN) of bacterial cells per 100 ml of sample was calculated.

Soil samples were retrieved from the core of both upstream and downstream boreholes at both sites. Samples were taken at 0.25 m points on a depth gradient, transported to the laboratory on ice, and stored at -20°C until further analysis.

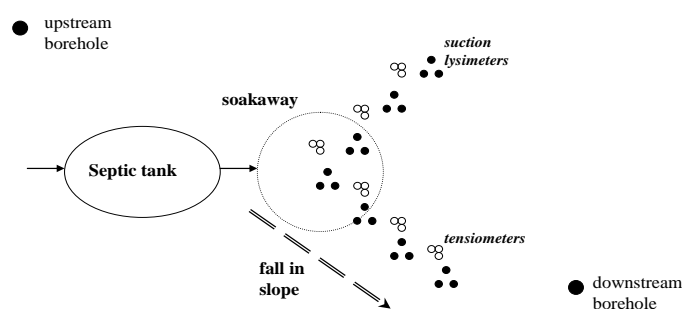


Figure 3: Outline of installed lysimeters and groundwater boreholes.

DNA extraction. Total genomic DNA was extracted from soil and groundwater and samples as described in Barrett *et al.*, (2012a) and (2012b), respectively. Nucleic acid DNA was extracted from borehole samples, taken from the two wastewater treatment sites, using the “Ultraclean® Soil DNA Isolation Kit” (MO BIO Laboratories) using the optimal conditions outlined in the manufacturer’s instructions.

Real-time quantitative PCR analysis. Standard curves and assay conditions for absolute quantifications of archaeal and bacterial 16S rRNA genes; and the three denitrification genes (*nirS*, *nirK* and *nosZ*) were established as detailed by Barrett *et al.* (2012a). The assay conditions for MST qPCR assays, BacUni, BacHum, BacCow and BacCan were identical to those described by Kildare *et al.*, (2007). Samples of human and animal faecal DNA, soil, groundwater and effluent were used to assess the feasibility of the use of the MST approach in the context of wastewater discharges to soils.

Statistical analysis. Non-metric Multi-dimensional Scaling (NMS) ordination was applied to jointly plot the N₂O and N₂ fluxes with the absolute abundance of the target genes, using PC-ORD ver. 5 (MjM Software Design). Coupled to NMS analysis, 2-way analysis of variance (ANOVA; Kruskal-Wallis test) was also carried out using the same data in order to establish relationships between bacterial 16S rRNA and denitrifier gene concentrations, soil and water chemical properties, and the concentrations of N₂O and N₂ (Garten *et al.*, 2007).

RESULTS & DISCUSSION

This study firstly examined the relationship between the abundance and activity of bacterial denitrifiers in groundwater and soils, differing with respect to overlaying land management. The most abundant denitrifying functional genes in all sites were *nirS* and *nosZ*. The distribution and abundance of bacteria harbouring denitrifying functional genes varied between sites and with respect to soil depth ($P < 0.01$), with mean levels of *nir* genes being generally 2-fold higher than *nosZ* (e.g. Table 2). This reflects the widespread capability of soil bacteria to carry out partial denitrification.

The typical *nirS* concentrations in groundwater ranged from 1.4×10^2 gene copies l^{-1} in water to 2.0×10^3 gene copies l^{-1} at the soil-bedrock interface; while *nosZ* ranged from 0.2×10^2 gene copies l^{-1} in groundwater to 1.9×10^2 gene copies l^{-1} at the soil-bedrock interface (Barrett *et al.*, 2012a).

Data relating to denitrifier abundance were successfully related to gaseous nitrogen emissions and to the physicochemical properties of both soil and groundwater. For example, *nir* abundance was positively correlated to N_2O production in groundwater ($P < 0.01$), while *nosZ* concentrations were also positively correlated with excess N_2 ($P < 0.01$). The concentration of dissolved organic carbon (including that associated with septic tank discharges) and the depth of the water table were also positively correlated with increasing *nir* and *nosZ* denitrifier abundance, while *nosZ* gene copy number was also positively correlated with increasing pH ($P < 0.01$; Barrett *et al.*, 2012b).

Table 2: Concentrations of target genes (copies per gram) in relation to soil depth at the Johnstown Castle site.

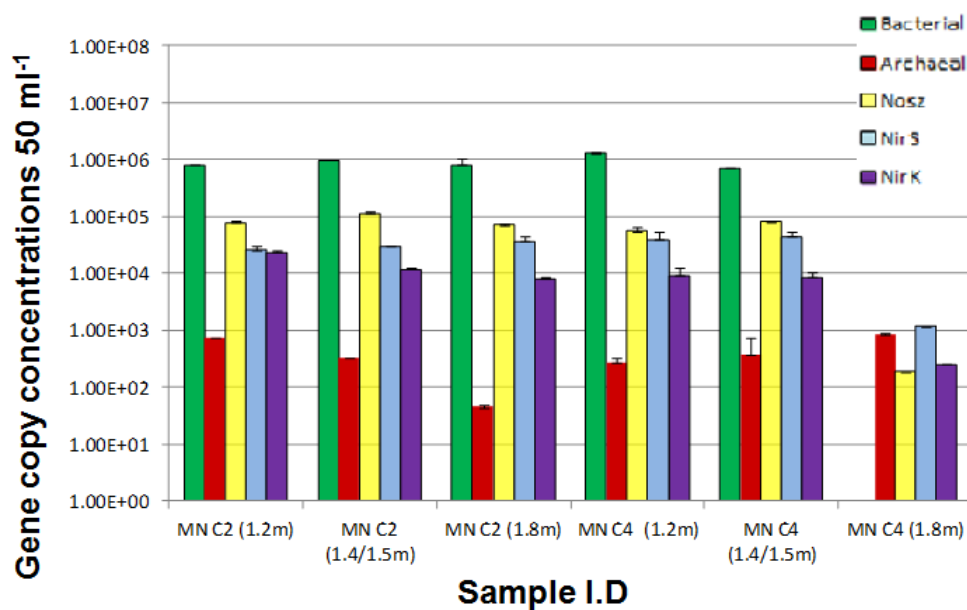
Soil depth (cm)	<i>nirS</i> + <i>nirK</i> (g^{-1})	<i>nosZ</i> (g^{-1})	Bacterial 16S rRNA: <i>nir</i> ^{total}	Bacterial rRNA: <i>nosZ</i>	16S
10	1.3×10^7	7.7×10^5	2.9	57.5	
45-55	1.5×10^7	4.41×10^5	9.4	288	
120-130	4.9×10^6	4.6×10^4	1.3	48.5	

These results indicate that it may be feasible to establish relationships between the abundance of bacterial denitrifiers and denitrification potential, but further study is required to fully evaluate the robustness of the methodology. With respect to soils impacted by septic tank effluents, the release of N_2 through the metabolic activity of anammox bacteria needs to be considered, as well as the N_2O , which may be produced during nitrification (Dong *et al.*, 2009; Di *et al.*, 2010). The integration of microbial community composition and functional diversity information will be important in future, to allow us to elucidate the role and impact specific microbes have under varying conditions.

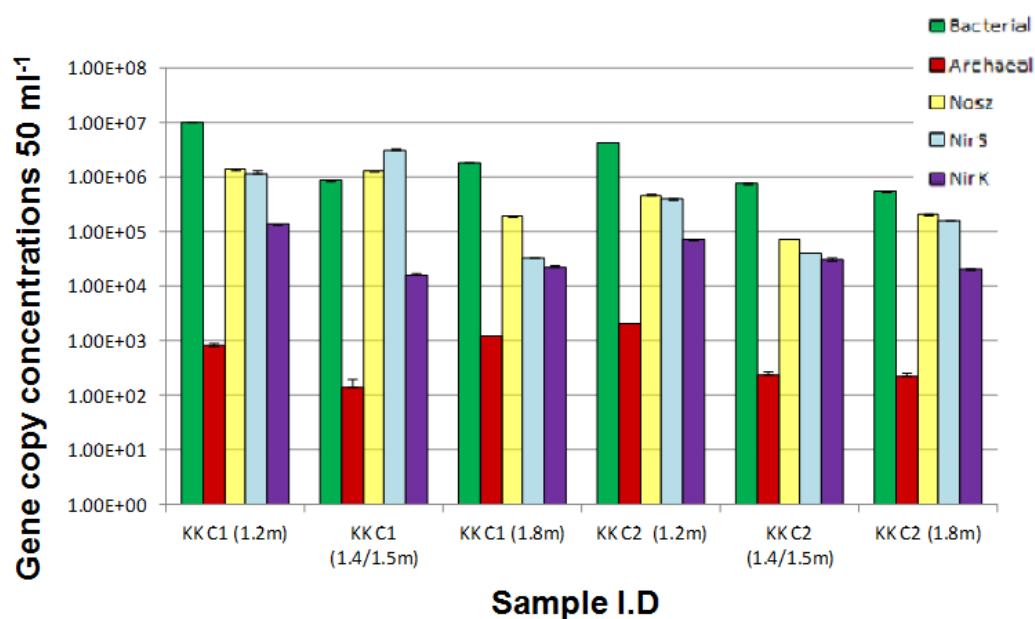
The potential for partial and complete denitrification is currently being assessed by targeting the functional genes *nirS*, *nirK* and *nosZ* at the two wastewater treatment sites. To this end, the molecular methods were successfully adapted to allow the detection and quantification of the target organisms in septic tank effluent. Results to date provide an indication of the existing denitrification potential in the soakaway areas, while future work will assess the impact of remediation measures at the two study sites. Two shallow, pressure-dosed systems consisting of a drip distribution and a low-pressure pipe system have been installed, in parallel at both sites. They are composed of narrow diameter piping placed at a shallow depth in the soil. The wastewater is then dosed at controlled volumes into the growth zone of the soil. The rest period between doses is important as it allows air to re-enter the soil preventing ponding and allowing sufficient treatment of the effluent.

Due to the shallow installation depth these systems also provide a solution to on-site wastewater disposal in an area with a high water table.

With respect to the potential for pathogen transfer, total coliform and *E. coli* levels give a strong indication of the level of faecal contamination present in local water bodies (e.g., Figure 5), but do not inform as to the source of this contamination. We successfully adapted the MST methodologies of Kildare et al. (2007) to allow the detection of human and animal faecal contamination in soil, groundwater and septic tank effluent samples. DNA extracts from cow, horse, and human faecal specimens, together with soil, water and septic tank effluent samples were tested against five quantitative PCR assays designed to detect universal, human-specific, ruminant-specific and cow-specific faecal *Bacteroidales* genetic markers. The universal primer (BacUni-UCD) detected *Bacteroidales* in all faecal and environmental samples studied, while ruminant (BacCow-UCD and BoBac) and human (BacHum) primers detected and quantified *Bacteroidales* and *Bacteroides*, only in non-human and human faecal samples and in spiked environmental samples, respectively.



A



B

Figure 4: Real-time PCR (qPCR) quantification of archaeal, bacterial 16s rRNA and functional denitrification genes (*nirS*, *nirK* and *nosZ*) from Co. Monaghan (A) and Co. Kilkenny (B) lysimeter samples.

The ability to predict pathogen occurrence in relation to standard culture-based indicator threshold levels was evaluated using a weighted measure that showed the universal *Bacteroidales* genetic marker to have a comparable or higher mean predictive potential than standard coliform based tests.

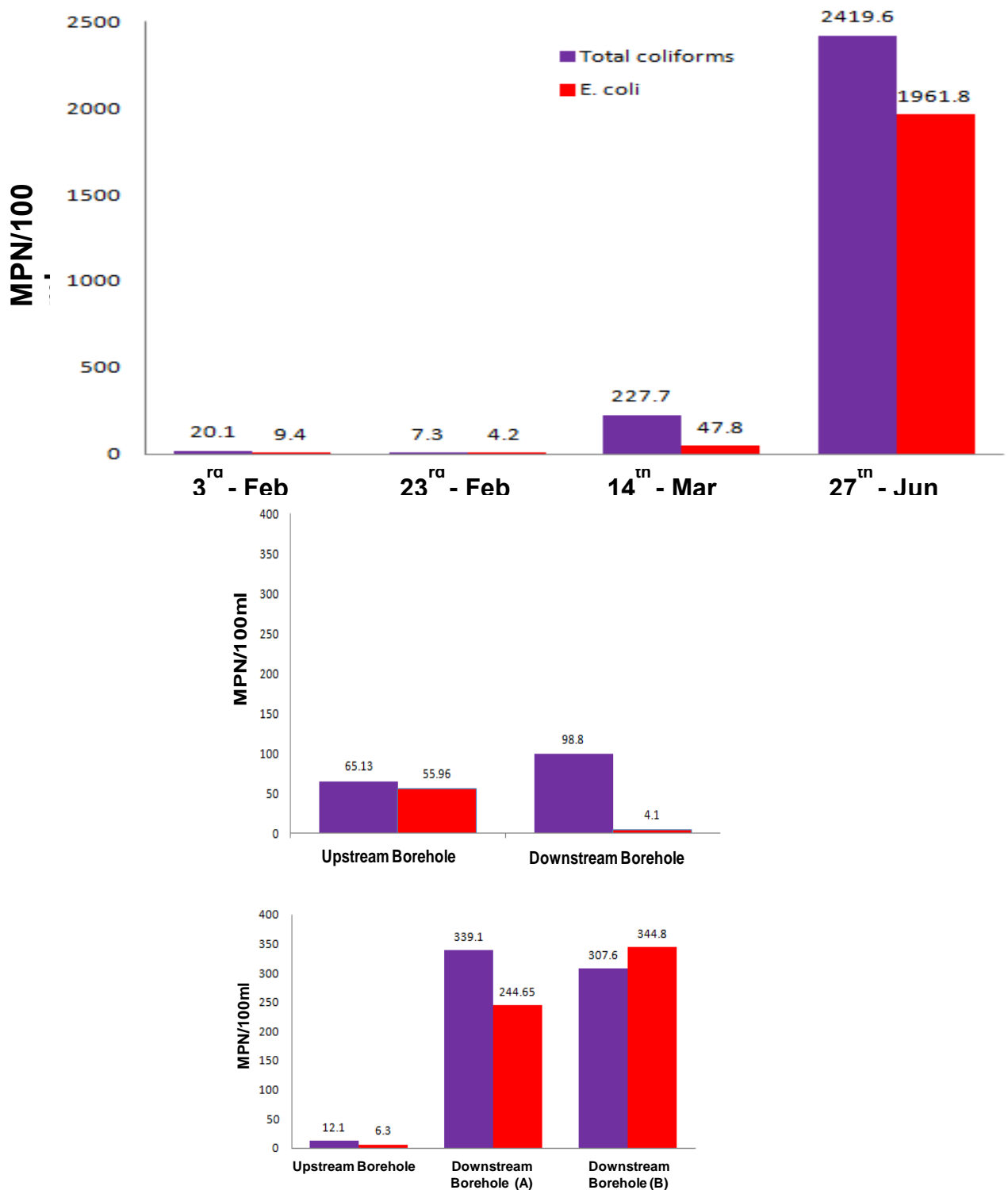


Figure 5: Detection of faecal indicators in waterbodies impacted by two septic tank systems. (A) Most probable number (MPN) of total coliforms (purple) and *E. coli* (red) per 100ml of surface water samples taken between February and June 2012; (B) Groundwater sampled at the Co. Kilkenny site; and (C) groundwater sampled at the Co. Monaghan site.

This predictive ability, in addition to the *Bacteroidales* assays providing information on contributing host fecal sources, supports further research towards the use of *Bacteroidales* assays for water quality monitoring (Schriewer *et al.*, 2010). The successful establishment of

an MST methodology applicable to soil, wastewater effluent and groundwater, in an Irish context, will underpin future investigations into the fate and transfer of pathogens, in a variety of settings, and could also inform and support management and remediation decisions. When many diverse point and non-point sources are present in a watershed or catchment, meaningful information about the most important sources, potential pathogens, and health risks may be important to evaluate the trade-offs between management costs, expected health benefits, and economic and environmental impacts, in choosing mitigation actions.

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SESSION III PACKAGED WASTEWATER SYSTEMS

PACKAGED WASTEWATER TREATMENT SYSTEMS FOR INDIVIDUAL HOMES AND SMALL COMMUNITIES

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ABSTRACT

Packaged or pre-engineered wastewater treatment systems are required for some sites where traditional septic tanks and soil dispersal systems do not provide acceptable effluent quality for dispersal into the receiving environment. In other cases, providing higher-quality effluent may allow a particular site to accept the cleaner water. In some cases, sensitive environments may require treatment beyond the septic tank and dispersal system to meet a particular quality such as a low nutrient concentration. Pre-engineered systems may be configured around any of several different process modifications. One of the earliest pre-engineered configurations used extended aeration activated sludge, targeting an effluent quality of 30 mg/L 5-day Biochemical Oxygen Demand (BOD₅) and 30 mg/L Total Suspended Solids (TSS). Attached-growth systems have been introduced in the past 30 years, having natural or synthetic media for the microbial growth to populate. The media generally has a light dry weight so it can be packaged and delivered in modules. As the pre-engineered system industry has grown, additional technology has been modified from large municipal processes such as integrated fixed activated sludge (IFAS), a process that utilizes both suspended growth and fixed film technology. Similarly, moving bed bioreactors (MBBRs) incorporate both fixed microbial growth and suspended growth, with the media being lifted and mobilized by the mixing action imposed by aerators. Membrane bioreactors (MBRs) have come into the market and are manufactured for small systems. All of these processes may produce low concentrations of BOD₅ and TSS, and are capable of nitrification to convert the ammonium-nitrogen to nitrate. Some of them are configured to recirculate the nitrified effluent to an anaerobic process for denitrification and reduction of total nitrogen concentration to low levels. Generally, individual home package plants do not address phosphorus removal, although precipitation and exchange systems have been tried, and commercial products may be available. In all cases, the pre-engineered systems require oversight including maintenance and management. Most studies examining packaged treatment systems for individual homes conclude that a high percentage of the systems are out of regulatory compliance because they need more maintenance. Depending upon the process, the product, and the technology, the packaged treatment systems vary in their amenability to being maintained and properly operated in a manner that make the systems sustainable and affordable.

COMMON TYPES OF SYSTEMS

Package wastewater treatment systems, also known as pre-engineered systems are commonly used to treat wastewater prior to discharge. The systems are typically packaged to treat a particular maximum daily flow rate, and models are manufactured in several sizes ranging from 900 L/day up to 3800 L./day. The sizing is typically based upon a household

population equivalent ranging from 4 to 18 persons. Generally, the package wastewater treatment systems are designed for residential strength wastewater for the particular maximum flow rate. However, some of the systems also include maximum mass loading rates for particular parameters such as Biochemical Oxygen Demand (BOD), 5-day Biochemical Oxygen Demand (BOD₅), or perhaps 5-day Carbonaceous Biochemical Oxygen Demand (CBOD₅). In those cases where a nitrogen limit is addressed, a maximum allowable mass loading for nitrogen may also be indicated. Many, but not all, of the package wastewater treatment systems require primary treatment upstream of the unit.

Pre-engineered wastewater treatment systems may use any of several technologies and processes. Extended aeration activated sludge is common. Attached-growth media filter systems are often used. The attached growth media filters may rely upon either natural media materials or synthetic material. Integrated fixed activated sludge (IFAS) units have become popular systems for package plants. The media in the IFAS systems may be suspended as in a moving bed bioreactor (MBBR) or the media may be static with the wastewater moving around the media. One example of the latter is the FAST® (Fixed Activated Sludge Treatment) unit by BioMicrobics of Kansas, USA. The Bison™ NASF is a static media IFAS system manufactured in Ireland. Anua's PuraMax® Membrane bioreactor is another example of a product made in Ireland. MBBRs are being used for facilities ranging in size from individual home applications to thousands of cubic meters per day. Sequencing batch reactors, SBRs, utilize all of the unit processes of suspended growth activated sludge while minimizing the volume of the reactors by using a single tank and performing each process in sequence. Two parallel treatment trains must be provided if continuous flow is required. Improvements to membrane technology has resulted in the development of small membrane bioreactors, MBRs, that allow activated sludge to be operated at a very high microbial concentration yet eliminate the need for a final clarifier.

TECHNOLOGY DESCRIPTIONS

Extended aeration activated sludge systems may include a primary tank, often a small tank about half the volume of a typical septic tank. The small primary tanks are sometimes called "trash tanks" and only remove the large solids from the sewage flow. Not all individual residence package activated sludge plants include the trash tank, and in those cases, the sewage flows directly into the aeration chamber. The main tank in the treatment system normally contains the aeration chamber as well as the final clarifier.

The sewage enters the aeration chamber and is mixed and aerated. The treated wastewater and biosolids enter the clarifier where the biosolids settle out of the mixture and the clarified effluent is discharged.



Figure 1: Package Activated Sludge System

The air to the small package activated sludge units is supplied by a blower or compressor, depending upon the manufacturer.

Attached growth media filters may utilize natural or synthetic media. Natural media has included clean, washed sand of a particular gradation and uniformity and peat. Synthetic media includes textile fiber, open cell foam, polystyrene beads, and bottom ash from coal-fired power plants. Experimentation with crushed glass, crushed plastic (automobile tail lights), basic oxygen furnace slag, and expanded clay minerals has been conducted with varying success. Some of the media filters are single-pass or intermittent filters receiving clarified wastewater as a single pass through the media prior to discharge. Other media filters are multiple pass or recirculating systems where the clarified wastewater passes over the media and a portion of the treated wastewater is recirculated to a recirculation and processing tank to pass through the media filter repeatedly prior to discharge. Some of the attached growth systems are operated to maintain microbial populations in the endogenous phase, essentially steady-state processes so there is no excess biomass and therefore no sloughing from the media. Other attached growth systems have larger media and are designed as trickling filters. Those systems slough biomass and the treatment media is followed by a final clarifier. Biomass from the clarifier is typically returned to an

equalization tank or to a primary tank upstream of the media filter. The recirculated effluent serves to dilute the primary wastewater and the recirculating systems are loaded at a higher hydraulic loading rate than the single-pass or intermittent systems.

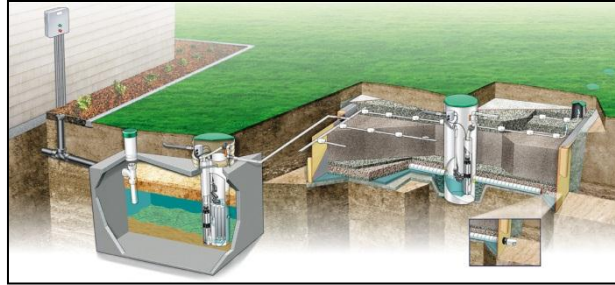


Figure 2: Single-Pass Sand Filter



Figure 3: Textile Media Recirculating Attached Growth System

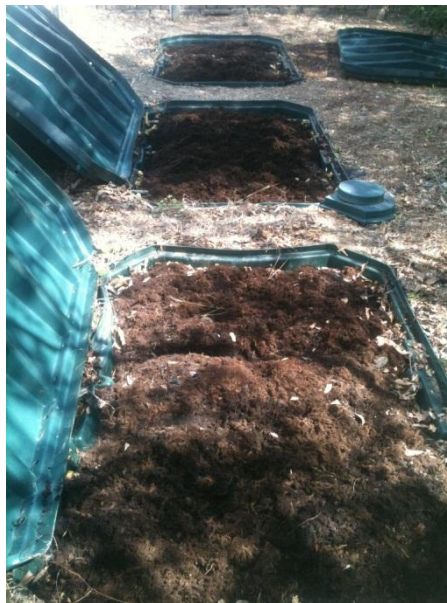


Figure 4: Peat Biofilter

Integrated Fixed Activated Sludge (IFAS) systems include primary treatment of the raw wastewater upstream of the aerobic reactor. The primary treatment is typically a septic tank or a trash tank with approximately half the detention time of a septic tank. The clarified effluent from primary treatment flows into the aerobic reactor where both suspended growth and attached growth processes are used to treat the wastewater. The mixed liquor flows to a final clarifier for settling and returning the biomass to the aeration basin.



Figure 5: Package Trickling Filter System



Figure 6: Integrated Fixed Activated Sludge System

Sequencing Batch Reactors, or SBRs are used on a limited basis for individual home systems, but more often for treatment for installations where regular professional operation and maintenance are provided. All of the processes of an activated sludge system are provided in a single vessel. The goal is to reduce tankage while providing unit processes. The same tank is used for sedimentation, aeration, final clarification, and decanting the clarified supernatant. In cases where continuous flow is required two parallel treatment trains must be used.

Membrane Bioreactors, MBRs are suspended growth activated sludge systems with the reactor basins followed by membrane filters rather than final clarifiers. The membrane filtration can be microfiltration (MF), ultrafiltration (UF), nanofiltration (NF) or reverse osmosis (RO) membranes. The membranes may be flat sheets (plate and frame) arrangements, pleated membranes, or hollow fibers. Pressure-driven membranes are typically located outside the reactor basin as a side stream. Vacuum-driven membranes are generally located inside the aerated reactor basin.

TECHNOLOGY PERFORMANCE, OPERATION AND MAINTENANCE

Small Activated sludge extended aeration basins generally target an effluent BOD₅ concentration of approximately 30 mg/L and a TSS concentration of approximately 25 to 30 mg/L. Extended aeration activated sludge operational parameters include a sludge age (mean cell residence time, or solids retention time) of approximately 40 to 60 days and a mixed liquor volatile suspended solids (MLVSS) concentration ranging from 2500 mg/L to 3500 mg/L. The MLVSS is based upon residential strength sewage and the normal reactor basin size of around 1900 liters and maintaining a food to microorganism ration (F:M) around 0.04 to 0.05. The aerator horsepower required is enough to provide the oxygen transfer for the metabolism as well as complete mixing within the aerated reactor. Normal annual operation and maintenance includes checking the pressure on the downstream side of the blower or compressor to determine if the aeration piping or distribution requires cleaning, cleaning the air filter, ensuring that adequate mixing and aeration is occurring by checking the dissolved oxygen concentration in the aeration basin, and performing a field settleability test of the mixed liquor. Other common field evaluations include dissolved oxygen, pH, and turbidity in the effluent. The most common repair and replacement consideration is replacing the blower. Desludging the final clarifier is recommended every year as needed to prevent sludge compaction and anaerobic conditions that could result in gas bubbles that can cause sludge to carry over into the discharge. Also, the top of the sludge blanket should be kept below the outlet tee.

Media filter systems are loaded differently depending upon the type of media and whether they are recirculating or single-pass (intermittent) systems. Single-pass intermittent sand filters use clean natural sand media with an effective diameter of approximately 0.3 mm

and a uniformity coefficient of less than 3.5, depending upon the local jurisdiction. The daily design loading rate for residential strength wastewater is 5 cm per day. The expected effluent quality from an intermittent sand filter loaded with residential septic tank effluent is approximately 10 mg/L BOD₅, and less than 10 mg/L TSS. Properly designed and loaded intermittent sand filters have historically reduced total nitrogen by approximately 30 % with an effluent ammonium in the range of 5 mg/L. In the past, intermittent sand filters were loaded with septic tank effluent by flooding the surface of the filter, but research shows that small, frequent doses of approximately 1 liter per orifice per dose and approximately 18 to 24 doses per day, evenly distributed over the surface of the filter extends the life of the media while providing much more effective treatment. Peat filter systems are normally loaded by either a pump and pressurized distribution system if flow equalization is used, by gravity flow through a distribution system of pipes and relatively large (10 mm) holes, or by a tipping bucket onto a corrugated distribution plate over the peat.

Intermittent sand filters are quite often buried, and the media cannot easily be examined or evaluated. The normal operation and maintenance procedure includes annually removing and cleaning the pump, depending upon the type of pump (multiple stage or single-stage). Float switches and controls and alarms should be evaluated to ensure they operate properly. The distribution network should be flushed annually, and following flushing, the distal residual head should be measured using a sight glass and the head compared to the startup conditions. If the intermittent sand filter has a downstream discharge basin, the pump and controls in that basin should be examined and if there is biomass accumulation in the discharge basin, it should be cleaned and desludged.

Peat filter maintenance should follow similar procedures to intermittent sand filters. Commercially-manufactured peat filters are constructed in modules that can be opened to examine the condition of the media and to service the distribution network. The experience in the eastern US is that the peat decomposes after approximately 10 years, and must be removed and replaced with new peat media. The decomposed peat must be properly disposed of, and some states allow disposing of the peat in municipal landfills.

Recirculating sand filters use clean media that is at least 2 mm effective diameter with a uniformity coefficient less than 2.0. Some designers prefer larger media, even up to 10 mm effective diameter. However, the larger media produces poorer quality effluent and does not nitrify as effectively.

With 2.0 mm effective diameter media, recirculating sand filters can be expected to produce effluent concentrations of less than 15 mg/L, BOD₅ less than 10 mg/L TSS, and achieve approximately 50% net nitrogen removal. Recirculating sand filters are dosed at approximately 20 cm/day design flow with each dose being approximately 8 liters per orifice per dose distributed evenly over the media surface. Recirculating sand filters are not covered with soil, but the media surface may have a porous textile cover for blocking

vegetation growth over the surface of the filter. Clean media is essential for both the single-pass and recirculating media filter systems.

Textile and other synthetic media attached growth systems are an outgrowth of research into a lightweight synthetic media that can be a replacement for sand, but light enough to be installed in the filter container, and hauled to the site. The media properties can be closely controlled using the synthetic media, unlike some of the natural media such as sand, where the sand is often not clean or the transportation cost can be high to obtain good-quality media. The container containing the lightweight synthetic media is set into the excavation using normal construction equipment on site for the system installation. The synthetic media has a high water holding capacity to retard the water movement, allowing time for the wastewater to be in contact with the microorganisms. The media also has a large specific surface area for microbial attachment. Yet, the media needs adequate hydraulic conductivity and porosity for water and air movement. Textile and other synthetic media recirculating systems are expected to produce effluent BOD₅ less than 10 mg/L, TSS less than 10 mg/L and achieve a total nitrogen reduction of over 60%. These results are achieved when the systems are loaded with typical domestic strength wastewater. The systems have recirculation ratios ranging from 4:1 to as high as 50:1 for the bottom ash media. Typically, the recirculation ratio is in the range of 4:1 to 6:1 for natural and synthetic media systems.

Operation of the recirculating media filter systems includes annually cleaning the distribution network, examining the media to ensure excess growth is not present, checking for proper air flow, either passive or induced, to the media system, checking the residual head in the distribution network, cleaning any effluent screens or filters and making the normal measurements of turbidity, dissolved oxygen and pH in the effluent. Floats are checked for proper functioning and the control panel is cycled to ensure proper operation. In most systems, since the service provider is on site, checking the septic tank sludge and scum levels is also a routine task.

The IFAS systems were developed to take advantage of the properties of completely mixed activated sludge systems as well as attached growth systems. The MLVSS for operating an IFAS system is typically maintained in the range of 1500 mg/L. The IFAS systems can achieve effluent concentrations similar to attached growth media filter systems, but the IFAS systems incorporate a smaller footprint. The tradeoff between IFAS and media filter systems is that the media filters require a larger footprint area, but the IFAS systems require much more energy to function. Typical O&M for the IFAS system is checking the blower operation, cleaning the screens and filters for the blowers, checking the air distribution system for proper air delivery, clogged air diffusers, and pressure and flow through the air system. Excess biomass may need to be removed by pumping. Any upstream primary treatment device may need to be desludged. Typical effluent operational parameters that are checked include turbidity, pH, dissolved oxygen, and solids settleability in the final

clarifier, if there is one. The media portion of the system is examined to ensure that excess growth and sloughing is not occurring that would cause the media in a moving bed bioreactor to stick together or not properly circulate as designed.

MBR systems operate very similarly to activated sludge systems, however, there is no final clarifier, and the membrane produces an effluent that is essentially free of suspended solids, with a very low BOD and virtually free of coliform. The MLVSS in a membrane bioreactor is maintained in the range of 12,000 mg/L. Typical service activities include cleaning the screen on the blower or compressor, checking the vacuum pump for proper function if the membrane is vacuum driven, checking the discharge pump for proper function if the membrane is pressure driven, analyzing for turbidity, dissolved oxygen and pH, and checking all float and control functions. Periodically, the system will require removing excess biomass and inert material. The membranes must be chemically cleaned periodically depending upon the wastewater characteristics. Reportedly, membranes must be replaced approximately every 5 years.

OPERATOR AND SYSTEM CERTIFICATION

The treatment system approval varies from state to state in the US. Some states accept the NSF International certification as the only requirement for acceptance in that particular state. Other states require field testing of the particular product and process in that state prior to acceptance. The field testing protocol varies, but most states require at a minimum analyzing for BOD₅, TSS, and Coliform. More often in recent years, the states have required analyzing for nitrogen species – usually resulting in determining the effluent Total Nitrogen concentration. The protocol also varies from state to state in terms of the rigorousness of the testing program. Some states require third-party testing from sampling to analysis to reporting being conducted by an independent agency or certified laboratory. Third-party field testing is expensive and can take years - typically 2 to 3 years. Many manufacturers have vehemently fought against third-party testing and some of the states where it was required in the past have dropped the requirement. Unfortunately, that makes for an uneven playing field for the sales and marketing of products and systems when one manufacturer is required to undergo rigorous and expensive testing while another manufacturer does not.

Certifying and licensing service providers also varies across the states, with some states having a certification program and others not having any program. The states with certification programs typically have either a required training course, an examination, or both. Some states are certified through the Department of Environmental Quality, Department of Health, or their equivalent. Other states have licensed service providers, designers, and installers through a state licensing board.

Some states also require installers and designers to carry a bond.

THE FUTURE

Developments in wastewater treatment for individual residences and small communities must address both capital cost and long-term costs while producing wastewater that meets increasingly more stringent discharge limits. Possibly the most important factor in small wastewater systems of the future will be energy consumption. With more efficient electrical motors that consume less electricity, variable frequency drives to reduce amperage surges during motor starts, and innovations into more effective treatment, the possibility of meeting the discharge limits effectively is achievable. However, capital costs may in fact increase as long-term costs and energy efficiency decrease. Regardless of the geometry, process, or configuration of the particular product or treatment system, the same biochemical reactions will apply as long as wastewater treatment is through the use of biological systems. The challenge is to optimize the processes through better technology.

RESEARCH DEVELOPMENTS IN THE ON-SITE TREATMENT OF WASTEWATER

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ABSTRACT

The removal of nitrogen and phosphorus from wastewaters arising from single houses and small communities, as well as the need for effluent disinfection, is receiving more attention in recent years. Some research developments in these areas in an Irish context (with some international examples) are presented. Key to the development of novel approaches to on-site treatment and disposal of wastewater are the availability and access to suitable research infrastructure allowing for the replication of 'real-life' conditions. To this end, the Water Research Facility is introduced and current technology development work presented.

Keywords: on-site wastewater treatment; nutrient removal, disinfection, water research facility, technology development

INTRODUCTION

On-site wastewater treatment systems can generally be divided into two categories; (i) small scale treatment systems for single houses (1 – 10 population equivalent, PE) and (ii) treatment systems for small communities (10 – 500 PE). In recent years significant improvements have been reported in wastewater infrastructure for communities with PE above 500 and 2000 in Ireland and the EU respectively (EPA, 2012; EC, 2011). Implementation of the Urban Waste Water and Water Framework Directives (WFD) has been a key driver in achieving these improvements. While on-site systems can offer economic, societal and environmental benefits there is an increasing awareness of the particular challenges offered by on-site wastewater treatment systems (Muga & Mihelcic, 2008). These challenges include a requirement for robust, cost-effective, low maintenance technologies that can achieve the required levels of treatment (Clifford *et al.*, 2010). Decentralised systems do not have permanent operators and indeed, may be infrequently

maintained and monitored. Thus compliance with increasingly stringent regulation can be difficult to achieve.

The WFD requires, by 2015, that anthropogenic pollution on water bodies must be reduced and waters restored to “good status”. The latest report on groundwater quality in Ireland classified 14.4% of monitored groundwater as of “poor status”. The main contaminants of concern were phosphate, nitrate and faecal coliforms. Ammonium was of lesser concern though increased ammonium concentrations were associated with the detection of faecal coliforms and can indicate groundwater vulnerability (EPA, 2010). The quality of drinking water from private wells is also of concern. Recent HSE data indicated 76% non-compliance with Drinking Water Regulations. Coliforms, E-coli and enterococci were among the parameters most commonly exceeded in 41 private wells surveyed in Sligo and Leitrim (DoECLG, 2010). On-site treatment systems were likely contributors to these breaches. Where on-site systems are poorly designed, inappropriately installed and/or maintained, contamination of surface waters is also a potential risk (McCarthy *et al.*, 2009).

Given the necessity to meet (i) the provisions of the WFD (or, for example CFR 40 502(14) in the USA), (ii) recent provision for registering and inspection of on-site wastewater treatment systems in Ireland, and (iii) concern over water supply integrity and contamination of waters used for recreation or food production; there is growing interest in Ireland and internationally in the development and study of new technologies and processes for nutrient and pathogen removal from on-site wastewater discharges (Cucarella & Renman, 2009; Gill *et al.*, 2009; Conn *et al.*, 2011; Rietveld *et al.*, 2011; Tanner *et al.*, 2012). Such research should be underpinned by accessible on-site test facilities where new technologies and processes can be tested, demonstrated and optimised in a real-world environment (Clifford *et al.*, 2011).

This paper presents some current research and technology developments pertaining to nutrient and pathogen removal by on-site wastewater treatment systems. Available research and technology development facilities are outlined and future technical challenges for on-site wastewater treatment facilities are proposed.

NUTRIENT REMOVAL USING ON-SITE WASTEWATER TREATMENT SYSTEMS

Nitrogen removal

Nitrogen removal is typically achieved via nitrification-denitrification processes; particularly in temperate climates. Typically on-site wastewater treatment systems are not designed to achieve nitrogen removal. Increased focus should be paid to low cost, self sufficient technologies that do not require regular maintenance.

Denitrification in soils used for percolation, where possible is a simple and attractive, though effectiveness can vary depending on a number of issues including the presence of organic carbon, the level of soil saturation and the soil texture. Gill *et al.* (2009) observed that where nitrates in groundwater are a problem an upper limit on subsoil permeability may need to be defined as denitrification is less likely occur in soils with higher permeability.

Biofilm systems can offer a low cost solution and are adept at treating wastewaters with high substrate variations (Rodgers *et al.*, 2010)

The horizontal flow biofilm reactor (HFBR), recently developed and commercialised at NUI Galway was designed as a low cost solution that enables nitrogen removal. Nitrogen removal is achieved by using a simple step-feed mechanism (Figure 1).

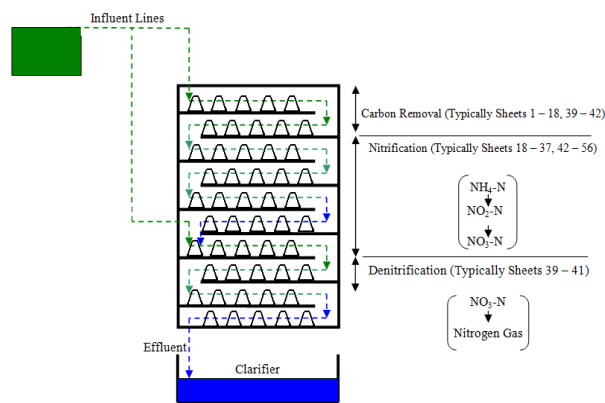


Figure 1: Schematic of a HFBR with a step feed (Regan, 2011).

The technology has been trialled both in the laboratory and on-site and has proven effective in the treatment of wastewaters ranging from domestics-strength (Table 1) to on-site dairy washwaters.

As a passive aeration technology, the system can operate without energy if the site conditions are favourable. Otherwise the system requires only a single pump to move wastewater onto the reactor. Running costs, in this scenario, for a single house are as low as € 10 – 20 per annum. Table 2 summarises the performance of various on-site wastewater treatment systems treating agricultural wastewater. In this study HFBR running costs were 0.03 kWhr/m³ treated.

Table 1 Comparison of nitrogen removal from various on-site wastewater systems treating municipal strength wastewater (adapted from Clifford *et al.*, 2010)

System	BOD ₅ (COD)		TN	
	Loading rate (g/m ² /d)	% removal	Loading rate (g/m ² /d)	% removal
HFBR ^{1, 2}	141.5	97.4	15.7	61.7
RF with forced aeration ^{2, 3}	(5.6)	73	-	-
St SF ²	22	99	2.4	27
SF with added carbon layer ²	8.4	83 - 93	1.8	67
Constructed wetlands ²	-	-	8.1 – 14.1	55 - 80

RF = recirculating filter, St SF = stratified sand filter, SF = soil filter, ¹At 11°C; ² Synthetic wastewater used in these studies; ³ Filtration media included gravel, clay loam, zeolites, iron particles and jute

Table 2 Comparison of the HFBR to other on-site agricultural wastewater treatment systems (adapted from Rodgers *et al.*, 2008)

System	Wastewater	NH ₄ -N	
		Plan loading rate (g/m ² /d)	Removal rate (g/m ² /d)
HFBR	Agricultural/ dairy	6.7	3.9
Reed bed	Cheese dairy	2.1	1.4
Wetland	Agricultural/ dairy	1.3	0.26
Wetland	Agricultural/ dairy	~ 0.04	0.04
Wetland	Livestock wastewater	5.3	0.78
Wetland	Agricultural/ dairy	0.7 (TKN)	0.2

Recent work by Tanner *et al.*, (2012) compared the use of wetlands (in conjunction with carbonaceous bioreactors) with attached growth bioreactors (followed by vertical flow media wetlands and carbonaceous bioreactors). In both cases excellent removal of ammonium-nitrogen (NH₄-N), total nitrogen (TN) and e-coli (system (i) only) were achieved.

The combination in series of specific technologies to achieve removal of a variety of contaminants proved successful with system (ii) generally achieving higher removals. Both systems were low maintenance; however running costs were higher for system (ii) due to the need for recirculation. In both cases increased land use may be of concern; where this is not an issue such hybrid systems can provide solutions where high performance is required.

Other examples of “passive” nitrogen removal systems (passive systems should use no more than 1 pump) include the use of media to promote denitrification. In many cases organic carbon may be limiting and thus the use of additional media can promote nitrogen removal. Typically, organic carbon is used as an electron donor in the denitrification process. For on-site treatment plants, media used for nitrogen removal should be low cost, easily transported and require infrequent replacement. The Florida Onsite Sewage Nitrogen Reduction Strategies project focuses on sulphur-based autotrophic denitrification processes.

In this process elemental sulphur is used as an electron donor for denitrification in anoxic conditions (replacing the role of organic carbon). Removals of up to 97% nitrogen have been achieved using zeolite and sulphur media and also separately using an expanded clay and sulphur media. Further research is ongoing to optimise reactor design and loading rates, the use of an alkalinity source where necessary (autotrophic denitrification consumes alkalinity) and long term performance and costs (Smith *et al.*, 2008; BOSP, 2010).

Alternative media currently being studied at NUI Galway are biodegradable polymers that include thermoplastic starches and polyhydroxyalkanoates. These materials can be used to promote heterotrophic denitrification where carbon is limiting. The process design would be similar to that for the autotrophic denitrification reactors previously mentioned and is outlined in Figure 3.

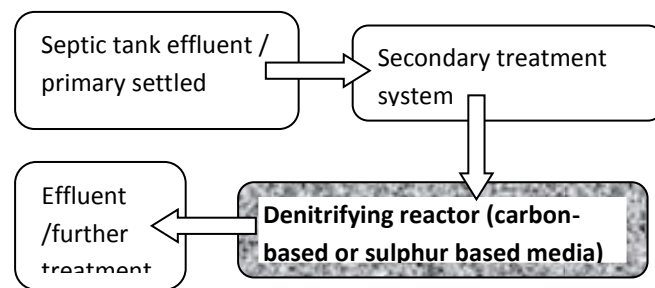


Figure 2: Schematic of a denitrifying reactor integrated into an on-site wastewater treatment system.

Early stage studies indicate that this media, possibly used in reactors operating in recirculation mode, could provide a low maintenance nitrogen (and potentially phosphorous) removal alternative. High loading rates could lead to smaller reactor sizes and infrequent media replacement could be possible. However, these materials are currently expensive and further research on material structure, reactor performance, nitrogen loading rates and recirculation rates is required.

While recent research work has mainly focused on the use of passive systems, sequencing batch (both activated sludge and biofilm) and moving bed reactors can also offer effective nitrogen removal performance. Operating and maintenance costs can however be higher than for passive systems.

Phosphorus removal

Phosphorus (P) removal from effluents arising from on-site wastewater treatment systems can be particularly necessary when discharge is into, or close to, a water course. In these situations, dedicated P removal techniques need to be applied to ensure limits are achieved.

Constructed wetlands are often used to for nutrient removal from wastewater, but can be less efficient at removing P than removing N (Cooper *et al.*, 1996) as the main P removal mechanism is through adsorption and can usually be directly related to the adsorptive capacity of the gravel/soil substratum (filter media) of the wetland system. Where the absorptive capacity of the gravel/soil substratum is low, the main removal mechanisms are plant uptake and subsequent harvesting. In a nitrification/denitrification study on a number of variations of horizontal and vertical flow constructed wetlands treating domestic wastewater, Tanner *et al.* (2012) found average P removal across all variation to be similar and attributable to the gravel media. Vohla *et al.* (2011) recently carried out an extensive review of wetland filter materials for the removal of P from wastewater, and also found that P removal is closely associated with the P sorption capacity of the filler material. A broad range of filler materials were reviewed, from natural materials (40 g P/kg for heated opoka), man-made materials (12 g P/kg for Filtralite), to industrial byproducts (up to 420 g P/kg for some furnace slags).

Dedicated adsorption-type systems can offer effective P removal on sites in sensitive areas where discharge is into or close to a water course. A recent study carried out by Rodgers *et al.* (2010) demonstrated an ion exchange adsorption system capable of consistently achieving effluent orthophosphate-phosphorus ($\text{PO}_4\text{-P}$) concentrations of less than 0.03 mg $\text{PO}_4\text{-P/l}$. The ion exchange material had a high P adsorption capacity, measured at 1,800 g $\text{PO}_4\text{-P/g}$ ion exchange material before break-through occurred (i.e., before the effluent concentrations of 0.03 mg $\text{PO}_4\text{-P/l}$ was exceeded). Automatic regeneration of the material is then possible resulting in (i) the adsorptive capacity of the material being restored, and (ii) a highly concentrated P solution suitable for P recovery. Although these types of adsorption systems can have a relatively high initial capital cost, they can offer solutions for larger populations (20 – 100 PE) in highly stressed areas.

PATHOGEN REMOVAL

Pathogen removal from wastewater treated with on-site treatment systems may not be necessary at many sites. However, there may be circumstances where disinfection is required and these can include (i) where discharge can potentially result in pollution of drinking water sources, aquaculture farms and bathing waters, (ii) where surface water discharge is being considered, (iii) where treated water is being re-used, and (iv) prior to discharge to highly permeable soils.

Where pathogen removal technologies are installed it can be a challenge to ensure adequate monitoring takes place. Analysis of samples is expensive for many pathogens and time consuming and can limit monitoring frequency (Conn *et al.*, 2011). For example, norovirus detection by real-time PCR detects both infectious and non-infectious particles and underestimates the reduction of infectious virus. Research into the use of surrogates as

accurate indicators of pathogen removal is ongoing at NUI Galway and may provide more cost effective and accurate solutions to current measurement techniques.

Membrane bioreactors (MBR), can offer a solution where pathogen removal is required. When using MBRs for separated grey and brown water treatment, Atasoy *et al.* (2007) observed 100% removal efficiency for coliforms. Dorgeloh & Kaiser (2007) used a MBR system to treat wastewater from a small community and achieved higher faecal removal than a UV system (< 10/100 ml achieved using the MBR and < 100/100 ml using the UV). While having a low footprint, MBRs require skilled maintenance and operation. Energy costs are relatively high with costs ranging between about 1.7 kWhr/m³ for greywater and up to 3.5 kWhr/m³ for non-segregated wastewaters.

UV disinfection is commonly applied in the water and wastewater treatment sector; however its efficiency in the disinfection of domestic wastewater has not received significant attention. UV is relatively inexpensive, does not produce disinfection by-products and is relatively simple to install and operate. It is very effective against cryptosporidium and coliforms. Friedler and Gilboa (2010) demonstrated the excellent performance of UV disinfection of greater effluent for small communities. The disinfected wastewater was reused for toilet flushing. 100% removal of E. Coli and a viral indicator (surrogate – FRNA bacteriophage) was observed. However, UV resistant bacteria such as heterotrophic counts and *Pseudomonas aeruginosa* were not removed completely. For water reuse in toilet flushing this was not found to be an added health risk. Pre-treatment of the wastewater comprised primary settlement, secondary treatment (via a rotating biological contactor and MBR in parallel) and pre-disinfection balancing.

As viral resistance to UV disinfection varies between species thus further work is required on its effect on other species or the relationship between F-RNA phage removal and specific viral removal (as is ongoing at NUI Galway).

Conventional disinfection methods such as UV, ozone and chlorine are not effective against resistant pathogens such as nematode. Gamma radiation could provide an alternative solution; however, given costs further work is necessary before it can be applied to small scale wastewater treatment plants. Its future may initially lie in areas where water shortages are common. De Souza *et al.* (2011) found that gamma radiation is effective in the inactivation of *Ascaris lumbricoides* eggs from human faeces.

Sand and soil filtration and wetlands technologies have received increased recent attention; this in turn has led to an improvement in understanding of removal mechanisms and the design criteria necessary for pathogen removal (Torrens *et al.*, 2009). Key parameters influencing pathogen removal include sand filter depth and hydraulic residence time. It has been noted that the presence of plants may not affect the removal of pathogens (Torrens *et al.*, 2009). Gill *et al.* (2011) further noted the effect of hydraulic loading rate and the presence of a biomat on e-coli and bacteriophage removal in percolation trenches. The

study further highlights to necessity for careful design of the overall on-site system and the need for further research on attenuation mechanisms and the effect of environmental conditions on pathogen removal.

Slow sand filters have historically offered a very effective method of water treatment. While initial capital costs and land requirements can be high, low running costs and excellent pathogen removals can be achieved (1 – 3 log units coliforms; 2 – 4 log units enteric viruses; 2 – 4 log units *Giardia* cysts) (NESC, 2000). Recent innovations include developments to improve the efficiencies of these filters and significantly reduce filter downtime and recovery after backwashing (Manz, D., 2007). Further understanding of the mechanisms underpinning pathogen removal or augmentation of the filter could lead to improved design (Bradley *et al.*, 2011; Gibbs *et al.*, 2012).

WATER RESEARCH FACILITY

It is increasingly recognised that research infrastructure is vital for (i) promoting innovation, technology and policy development, (ii) supporting the conditions for leading-edge research, (iii) creating sustainable employment, and (iii) educating new generations of researchers and innovators (Research EU, 2011). Such research infrastructure can also play a larger societal role by increasing knowledge among the general population on issues relevant to their daily lives. In the water and wastewater sectors, there is limited large scale infrastructure available for the education, research and industrial communities. Examples include the Wastewater Technology Centre in Ontario, Canada and the Institute for Urban Water Management (ISA), Aachen University, Germany that provide full scale test facilities and/or certification for water and wastewater technologies.

The development of the NUI Galway/EPA Water Research Facility (WRF) (Figure 3) in conjunction with Galway County Council provides a research infrastructure that allows for the examination and trialling of proprietary and novel water and wastewater treatment technologies suitable for on-site treatment and disposal, and for technologies requiring a discharge licence. The WRF is located on the Tuam municipal wastewater treatment plant (WWTP) site with core funding provided by the Irish EPA with support from Galway County Council and NUI Galway. Access is provided to wastewater at each stage of the treatment process (i.e., raw influent, settled influent, secondary effluent, and tertiary effluent) allowing for a wide spectrum of technologies to be trailed and accessed.

An influent side-stream is taken directly from the wastewater entering the Tuam WWTP (prior to the storm over-flow location and primary settlement tanks) and pumped at user defined intervals and volumes to the WRF's primary settlement tanks. The WRF can currently process a maximum of about 50 m³ /day, though this volume can be varied depending on the work being carried out at any given time and the level of treatment required.



Figure 3 Photograph of the WRF in Tuam, Co. Galway

From there, wastewater is available for technologies being tested at the WRF that require primary-settled influent wastewater (in this case, package wastewater treatment systems for on-site wastewater treatment and disposal (Figure 4). The main secondary treatment technology employed to treat the bulk of the wastewater at the WRF is the pumped flow biofilm reactor (PFBR) – a novel attached growth wastewater treatment technology, developed and patented in NUI Galway (Rodgers *et al.*, 2004).

The installation of alternative secondary treatment technologies (other than the PFBR) is also possible. Secondary effluent can be passed through a number of tertiary treatment systems installed post secondary treatment. Currently, automatic sand filtration and UV disinfection are installed. The tertiary treatment system has been designed in a “plug and play” manner allowing technologies undergoing trials to be easily plugged into the system such as chlorine contact tanks, activated carbon filters, zeolite adsorption systems, etc.

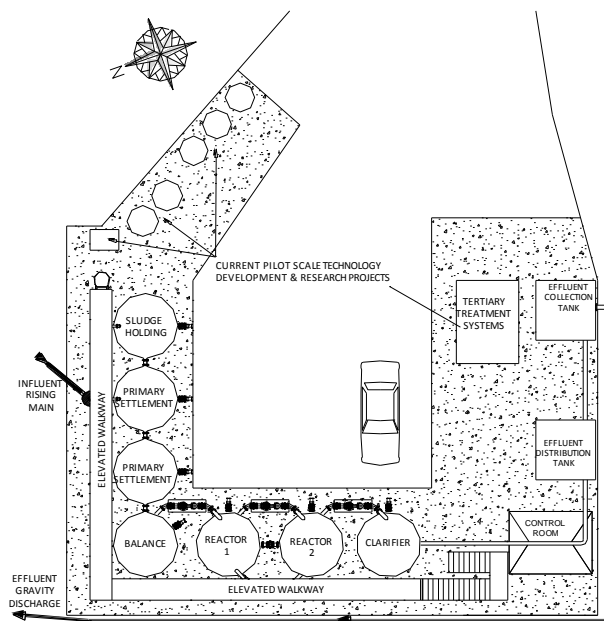


Figure 4 Layout of the WRF

A bespoke control system was developed for the WRF whereby a programmable logic controller (PLC) manages the operation of all the mechanical and electrical equipment for the main secondary wastewater treatment system. A touch screen human machine interface (HMI) allows the operator on-site access to all aspects to the system. All system controls and data are viewable and recorded by the PLC, displayed on a HMI and accessed remotely using an internet connection. Real time measurements of dissolved oxygen (DO), pH, oxidation reduction potential (ORP) and flow are combined with operational parameters such as pump run time and energy usage to optimise the treatment cycle; these data can be viewed remotely and in real time.

Technology development

A number of wastewater treatment technologies have already been trialled and have gone through various stages of developmental work at the WRF. NUI Galway researchers carry out contract research work (with Enterprise Ireland funding sourced through industry) at the WRF where individual companies require limited developmental work to be carried out on their own technologies – the WRF provides an ideal facility for such work. Work in this area included (i) the evaluation of a new phosphorus adsorption technology, (ii) novel sand filtration systems with lower backwashing requirements, and (iii) novel adsorption material for nutrient removal from wastewater.

One example of technology development work that has led to commercialisation was given above with a description of an on-site wastewater treatment unit known as the *horizontal flow biofilm reactor* (HFBR). A second example is that of the *pump flow biofilm reactor* (PFBR). In this technology (suitable for 100 PE – 5000 PE), a biofilm process is employed whereby aeration is achieved by alternately exposing the biofilm attached to stationary plastic media in each of two reactor tanks, to atmospheric air and wastewater; eliminating the need for forced aeration. The PFBR had been operated and tested in the Environmental Engineering Laboratories, NUI Galway for a number of years for a range of operation conditions. Extensive trials of the PFBR were also carried out at the WRF using the background research work from bench-scale testing in the laboratory. While operating over an period of 4-5 months, removals of 94% biochemical oxygen demand (BOD₅) (effluent concentration of < 10 mg BOD₅/l), 90% suspended solids (SS) (< 10 mg SS/l) and 97% NH₄-N (< 1 mg NH₄-N/l) were achieved at an energy requirement of 24 – 35 kWh/PE·yr for small populations of about 165 – 200 PE. The sludge yield for these trials was estimated at 0.13 g SS/g COD_{t removed} (O'Reilly *et al.*, 2011).

Subsequently, and in conjunction with an industrial partner (Molloy Precast Products) and Offaly County Council, the PFBR has been installed as a stand-alone wastewater treatment plant in Moneygall, Co. Offaly and has been designed for 750 PE (Figure 5).



Figure 5: PFBR installed in Moneygall, Co. Offaly

In this installation, the PFBR was designed as a dual stream system with each stream capable of catering for 375 PE, and has been in operation for about two months treating about 136 m³/d (summary of initial results in Table 3). The estimated biological load entering the plant is currently 578 PE (40 g BOD₅/PE-day as well settled influent wastewater) with an estimated energy usage of 0.25 kWh/m³_{treated}, or 21 kWh/PE·yr.

Table 3: Summary of influent and effluent results from Moneygall (all data in mg/l, number in parenthesis are standard deviations and number of samples)

Parameter	Influent		Effluent		% removal
	Balance Tank		Clarifier		
BOD ₅	170	(44.3, 19)	5	(4.9, 22)	97%
SS	166	(96.1, 31)	5	(4.1, 31)	97%
NH ₄ -N	10.3	(2.5, 38)	3.0	(1.2, 38)	71%
NO ₃ -N	-		5.1	(1.5, 36)	-
TN _t	13.6	(2.5, 12)	10.6	(1.3, 27)	22%

In general, BOD₅ and SS removal has been excellent and broadly in-line with expectations. Good nitrification has already been achieved – the inclusion of an anoxic phase in the next stage of operation will aim to reduce the effluent TN. The recent installation of an online ammonium/nitrate probe will allow for further refinement in the nitrification/energy requirement nexus.

DISCUSSION AND CONCLUSIONS

The removal of nutrients and pathogens from on-site wastewaters is a particular challenge. While technologies exist to achieve this, combining suitable technologies with the necessity

for low running and maintenance costs can prove a challenge. The use of hybrid systems (i.e. specific combinations of technologies), with minimal mechanical parts and electrical requirements could offer a solution. A focus on technology simplicity should be encouraged and indeed, where on-site conditions are appropriate the combination of a septic tank system and percolation area can offer effective treatment.

Where nutrient removal and/or disinfection are required, suitable monitoring mechanisms should be put in place. For small communities the use of robust sensors could result in increased plant performance and reduced energy costs. Future work in this area could mean individual domestic systems could be monitored in real-time in a cost-effective manner.

There exists significant opportunities in Ireland to lead research developments in the area of on-site wastewater treatment. A particular feature has been the development of on-site facilities for testing new technologies at the previously mentioned WRF, Moneygall and an on-site domestic wastewater treatment facility at Esker Hills, Co. Offaly which is currently facilitating work by NUI Galway and TCD. Other on-site work in Wexford, Limerick and Leitrim (see TCD) on willow systems could offer the advantage of zero discharge to ground and surface waters.

Some areas where future work and development could focus include:

Nitrogen removal:

- The use of passive technologies, with step-feed mechanisms to achieve nitrogen removal
- The potential of primary settled solids as a carbon source for denitrification
- The use of biodegradable polymers as a carbon source for denitrification
- Further development of autotrophic denitrification processes
- Development of standardised criteria to determine when nitrogen removal is required

Phosphorous removal:

- Significant work is required on the use of adsorption materials for on-site phosphorous removal. The logistics and cost of media replacement are currently cost-prohibitive in most cases
- Biological phosphorous removal processes require further development before widespread use in single house wastewater treatment systems

Pathogen removal:

- Technology combinations that could enable discharge of on-site effluents to surface waters
- Improved pre-treatment of wastewaters to ensure effective pathogen removal
- Development of cost-effective monitoring of systems where disinfection is required
- Further work on the development of filtration technologies as effective means of pathogen removal

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EN CERTIFICATION AND THE NATIONAL ANNEX

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BACKGROUND

The development of biological systems as small onsite wastewater treatment systems in Ireland occurred in the mid-nineties. At that time there was no applicable national standard. The only standard pertaining to on-site wastewater treatment systems was the SR 6 1991 which dealt with Septic Tanks and their associated percolation systems.

Biological treatment systems were considered “Innovative” products and compliance with the regulations and proof of suitability for use in Ireland was, largely, done by Irish Agreement Board certification (IAB). In this system of certification the “Royal Standard” for effluent previously applied to municipal treatment systems was used. This required that the effluent emerging from the treatment system had a Bio-chemical Oxygen Demand (BOD) concentration < 20 mg/l and Suspended Solids (SS) concentration < 30 mg/l.

CONSTRUCTION PRODUCTS DIRECTIVE (CPD) AND HARMONISED STANDARDS

At that time, and indeed currently for many products, local authorities relied on *national standards for showing compliance with* Building Regulations or writing Specifications for Public Works/ Supplies contracts. These standards are generally Irish Standards (I.S.) or British Standards (BS) or standards of a Member State of the EEA which provide in use an “equivalent” level of safety and suitability.

While these national standards have helped to achieve quality in building, they vary widely from one Member State to the next and have acted as a technical barrier to international trade in construction products. Such barriers must be removed if there is to be more effective competition in the construction supply chain and on construction prices, within the EU Internal Market. Ireland has one of the most open construction markets in the EU-it is estimated that about 40% of construction materials and products are imported.

To address the problem of technical barriers to international trade caused by varying national standards, the EU adopted the **Construction Products Directive (CPD)- 89/106/EEC**- for the harmonisation of construction product standards. The CPD was legally implemented in Ireland by the **European Communities (Construction Products) Regulations 1992 (SI No. 198 of 1992)**.

The EU adopted a second **Directive (93/68/EEC)** amending the prescribed format of **CE marking** to be used on, inter alia, construction products complying with the CPD. This Directive was legally implemented in Ireland by the **European Communities (Construction Products) (Amendment) Regulations 1994 (S.I. 210 of 1994)**.

The CPD aims to remove technical barriers to trade in construction products between Member States in the European Economic Area (EEA). To achieve this, the CPD provides for the following four main elements:

- a system of harmonised technical specifications (products standards and technical approvals);
- an agreed system of Attestation of Conformity (AOC) for each product family (with the product specifications);
- a framework of Notified Bodies; and
- the CE marking of construction products.;

The CPD does not aim to harmonise Building Regulations across Europe. Member States are free to set their own requirements on the performance of building works and, therefore, construction products. What the CPD harmonises are the methods of test, the methods of declaration of product performance values, and the method of conformity assessment. The choice of required values for the chosen intended uses is left to the national regulators in each Member State.

The national Building Regulations of Ireland and/or the related Technical Guidance Documents are amended, on a phased basis, to take account of new European classifications and standards.

The purpose of the harmonised technical specification for a product is to cover all the performance characteristics required by Regulations in any Member State. In this way, manufacturers can be sure that the methods of test and methods of declaration of results will be the same for any Member State, (although the *values* chosen by regulators may be different from one Member State to another). Currently, there are over 400 hENs covering a broad range of construction products. hENs are progressively becoming the norm as conflicting national standards (e.g. Irish and British Standards commonly used here) are being withdrawn.

EN 12566 STANDARDS

The standard EN 12566 is a suite of standards relating to wastewater treatment products. The seven sections of this standard are listed below:

IS EN12566-1 Prefabricated Septic tanks

ISEN12566-2 Soil infiltration systems - Technical Report (TR)

ISEN12566-3 Packaged and/or site assembled domestic wastewater treatment plants

ISEN12566-4 Septic tanks built in situ from prefabricated kits

ISEN12566-5 Pre-treated effluent filtration systems - TR

ISEN12566-6 Prefabricated treatment units used for septic tanks effluent

ISEN12566-7 Prefabricated tertiary treatment unit.

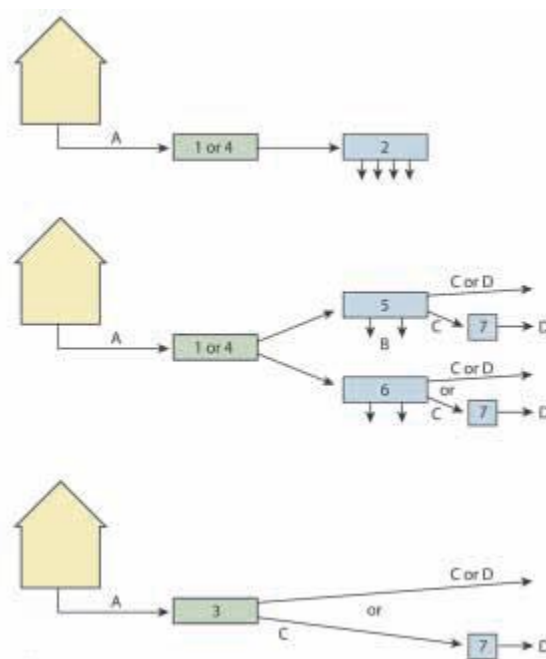


Figure 1:. Methods of wastewater treatment in line with EN12566.

Parts 1, 3 & 4 are Harmonised standards while parts 2 and 5 are technical reports. Parts 6 and 7 are currently nearing publication and will be harmonised standards.

With the adoption of the EN 12566 standards and the withdrawal of existing conflicting standards the national Building Regulation Part H / Technical Guidance Document (TGD) H *Drainage and Waste Water disposal* was revised to reflect this change and call-up the EN standard. IS EN12566-3 *Packaged and/or site assembled domestic wastewater treatment plants* is the standard applicable to biological treatment systems.

This Standard specifies requirements, test methods, the marking and evaluation of conformity for packaged and/or site assembled domestic wastewater treatment plants (including guest houses and businesses) used for populations up to 50 inhabitants. Small

wastewater treatment plants according to this European Standard are used for the treatment of raw domestic wastewater.

It covers plants with tanks made of concrete, steel, PVC-U, Polyethylene (PE), Polypropylene (PP) and Glass Reinforced Polyester (GRP-UP).

The unit is tested for the essential characteristics:

- Treatment efficiency,
- Treatment Capacity,
- Watertightness
- Crushing resistance and
- Durability.

The standard specifies various methods of testing depending on the tank material. For example the watertightness test can be done by a Water Test, Vacuum test or Pneumatic pressure test and the crushing resistance test has 6 different methods of testing with the pit test being the only one applicable to all tank materials.

The treatment efficiency test is a complex and lengthy testing procedure taking over 38 weeks with the exact test sequence specified within the standard. Raw domestic wastewater is used but must have the following quality:

a) BOD₅ or BOD₇(ATU): 150 mg O₂/l to 500 mg O₂/l

or COD 300 mg O₂/l to 1000 mg O₂/l

b) SS: 200 mg/l to 700 mg/l

c) KN: 25 mg/l to 100 mg/l

or NH₄ - N: 22 mg/l to 80 mg/l

The following core parameters are then monitored during the test for both the influent and the effluent:

- a) total chemical oxygen demand (COD)¹ and total biochemical oxygen demand (BOD)²; after a certain period, BOD of the influent only can be calculated from COD value;
- b) suspended solids (SS);
- c) temperature (liquid phase);
- d) total power consumption of the product if applicable;
- e) daily hydraulic flow.

The following parameters may also be measured if required:

f) pH;

g) conductivity;

- h) nitrogen parameters;
- i) total phosphorus;
- j) hourly hydraulic flow;
- k) dissolved oxygen concentration;
- l) sludge production;
- m) ambient air temperature.

NATIONAL ANNEXES

With the exception of watertightness or durability, these standards (en12566 series) only required that a value for the product be declared and thus there was no minimum value. The standard harmonized how the product should be tested and specifies parameters for which performance should be measured and declared by the manufacturer inline with the CPD but sets no performance criteria unlike traditional standards. This is recognized in the text where it states: *“NOTE: The ratios obtained do not automatically mean that the regulatory requirements on effluent qualities in a given country are met. A calculation should be made to indicate the final effluent qualities which should be compared to the requirements valid in the place of use.”*

Ireland, in order to give the same protection to the public and the environment that it had with its original requirements developed national annexes to the harmonised parts of the standard. These set down the minimum criteria to be achieved by the product for use in Ireland.

National Annexes to the Harmonised parts of EN 12566, namely Part 1: Prefabricated Septic Tanks, Part 3: Packaged and/or site assembled domestic wastewater treatment plants and Part 4: Septic tanks assembled in situ from prefabricated kits have now been published by the National Standards Authority of Ireland (NSAI) in conjunction with the Dept. of the Environment Community and Local Government and other organisations.

The following Table 1 gives the performance levels to be achieved by packaged Wastewater treatment plants for use in Ireland.

Table 1: EN 12566-3: Packaged and/or site assembled domestic wastewater treatment plants.

Essential Characteristics		Requirement Value
Effectiveness of Treatment (Treatment efficiency ratios)		$BOD_5 < 20\text{mg/l}^A$ $SS < 30\text{mg/l}$ $NH_4-N < 20\text{mg/l}^B$
Treatment Capacity (Nominal designation)	Nominal hydraulic daily flow	To be Declared (Expressed in Cubic Metres per day)
	Nominal organic daily load	To be Declared (Expressed in Kg of BODx per day)
Watertightness		Pass
Structural behaviour		Pass
Durability		Pass
Electric Consumption		To be Declared (in kWh per day)
A For an average BOD influent of 300mg/l or greater. B In nutrient sensitive areas the local authority may require a lower limit.		

The treatment values chosen by Ireland for the Part 3 systems are the original values used by local authorities for the municipal plants, and the Irish Agreement Board in assessing small wastewater treatment plants i.e. 20BOD, 30SS, 20NH4N.

In an amendment to EN12566 part 3 (Amendment A2 2012 currently being finalized by the CEN) Ireland requested that nitrogen be included as a parameter to be declared.

It should be noted that the treatment efficiency expressed as a percentage reduction, in the standard, does not give the information necessary to confirm the suitability of the final effluent unless there is acceptable national influent values. As Ireland does not have agreed influent characteristics for one-off dwellings the method of showing suitability is to declare the average final effluent values under the treatment efficiency tests within the standard.

The Certificates now produced by the Notified bodies have been revised to show the average effluent after treatment based on the 20 normal loadings and also the average value of the

influent with regard to BOD. (see excerpt below) These must be less than those given in the National Annex if the plant is to be suitable for use in Ireland.

Nominal organic daily load	0.26	kg BOD ₅ /d	
Nominal hydraulic daily load	0.90	m ³ /d	
Material	glass reinforced plastic (GRP)		
Watertightness	pass		
Crushing resistance	pass		
Treatment efficiency (nominal sequences)		Efficiency	Effluent
	COD	91.6 %	52 mg/l
	BOD ₅	95.9 %	11 mg/l
	SS	95.3 %	16 mg/l
	NH ₄ -N	56.7 %	15 mg/l
Electrical consumption	1.1	kWh/d	

Ireland, in the current review of EN 12566 part 3 standard, will be recommending that the average resultant effluent be declared.

Examples of the certificates can be seen on PIA notified Body website:

http://www.pia-gmbh.com/index.php?option=com_content&view=article&id=56&Itemid=44&lang=ga

It should be noted that the **Construction Products Directive Council Directive 89/106/EEC** has now been replaced by the **Construction Products Regulation (EU) No 305/2011**. One of the main impacts of the CPR is that from **July 2013, CE MARKING** of construction products covered by harmonised European Standards is mandatory. As such waste water treatment plants conforming with EN12566 series will be required to have a Declaration of performance in line with what has been discussed above and bear a CE mark.

THE SUITABILITY OF PACKAGED WASTEWATER TREATMENT SYSTEMS FOR DIRECT SURFACE WATER DISCHARGE IN RURAL IRELAND - A REVIEW OF PERFORMANCE AND COST EFFICIENCIES

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ABSTRACT

Areas with low permeability subsoils are often unsuitable for effluent discharge to ground. Therefore the consented surface water discharge of treated on-site wastewater effluent might need to be reconsidered as a disposal option in such areas.

This review shows that packaged wastewater treatment plants are able to achieve high organics, solids, and ammonia removal. Average effluent concentrations are usually lower than the required surface water discharge limits. However, nutrient removal is still limited so that surface water discharge in sensitive areas cannot be considered without further treatment. Enhancing denitrification results in higher process complexity using anoxic zones and effluent recirculation. Chemical phosphorus removal is possible in packaged OSWWT systems but involves expensive, hazardous chemical usage and storage as well as more frequent desludging. Biological TP removal on the other hand is not controllable in automated systems and cannot be considered as an option for packaged systems. The use of phosphorus adsorbing filter material is a promising solution to meet TP discharge limits in sensitive areas. Cost analyses have shown that decentralised systems for clusters of single houses can reduce the per capita cost for operational and capital expenditures compared to single house systems.

For all technologies available on the market it should be noted that their reliable treatment efficiency is highly dependent on regular maintenance and protection of user abuses. Thus the on-site and decentralised wastewater treatment with consented surface water discharge can only be environmentally sustainable where appropriate regulations and a suitable management plan are in place.

Keywords: on-site wastewater treatment; low permeability subsoils; effluent quality; nutrient removal; surface water discharge; cost analyses.

ABBREVIATIONS

CAS - Conventional activated sludge

MBBR - Moving Bed Bioreactor

MBR - Membrane Bioreactor

OSWWT - On-site wastewater treatment

RBC - Rotating Biological Contactor

SAF - Submerged Aerated Filter

SBR - Sequencing Batch Reactor

WWTP - Wastewater treatment plant

INTRODUCTION

The domestic wastewater of over one third of the population in Ireland (approx. 500,000 dwellings) is treated by on-site systems (CSO, 2011) of which the percolation area (soil attenuation system) is an integral part of the overall treatment system. However, where there is insufficient permeability in the subsoil to take the effluent load, surface ponding and runoff of pollutants to surface waters may occur. This represents a serious health risk and can also contribute to eutrophication in sensitive water bodies. The recent specification (EPA, 2009) of a lower limit to subsoil permeability (set at $T=90$) for effluent discharge to ground in conjunction with surface water discharges generally not being licensed for one-off housing, means that many areas will be unsuitable for single house development in the future. To allow further development in areas of low permeability subsoils and to achieve the desired river water quality according to European objectives (S.I. No. 272, 2009), alternative wastewater treatment and disposal options will need to be considered for new as well as for existing sites. While other appropriate on-site technologies, such as zero-discharge and alternative soil infiltration systems, are being investigated, Local Authorities might also need to reconsider the consented discharge to suitable water courses as a disposal options in areas with low permeability subsoils. For example in areas of relatively dense settlement it could be economically feasible to connect single houses via a small bore sewer system and treat the wastewater at a decentralised plant before consented discharge to a nearby watercourse.

There is a large variety of products available on the Irish market so that an overview of all available technologies is needed to be able to select a suitable plant for individual cases. This review will focus on the performance and suitability of available small scale treatment

technologies for a direct surface water discharge. Furthermore operational and capital costs of the different systems will be compared.

DATA COLLECTION

An extensive online search was undertaken to collect basic information about the available package treatment plants in Ireland. Detailed data on maintenance requirements, treatment performance and costs was then obtained through contacts made with manufacturers and suppliers. According to the Irish Buildings Regulations (DoEHLG, 2010) as well as the Code of Practice (EPA, 2009) packaged wastewater treatment plants must conform to the International/ European Standard I.S. EN 12566-3 including the relevant national annexes. Therefore the treatment performance of most available systems has been tested at European test facilities to obtain the relevant certificate. These test results have been collected in order to assess the system's ability to meet surface water discharge limits.

RESULTS AND DISCUSSION

As the data collection is still in process the presented results should be seen as preliminary findings.

Treatment processes used in packaged WWTPs. About 40 packaged wastewater treatment systems were found to be available in Ireland. While 6 plants are only available for larger applications (up to 5000 PE), 34 package plants are suitable for the treatment of domestic wastewater from single houses. Figure 1 shows the proportion of treatment processes that are used in packaged wastewater treatment plants in Ireland. With 15 plants the majority of systems use the fixed film process to treat domestic wastewater, while 13 systems are using the activated sludge process and only few media filter and MBRs are available in Ireland (Fig. 1). The 6 systems only available for large scale applications comprise two MBBRs, two MBRs and one SAF and RBC each.

A new fixed film system, the Horizontal Flow Bioreactor (HFBR), has recently been developed (Rodgers *et al.*, 2006) and will soon be launched into the market as well.

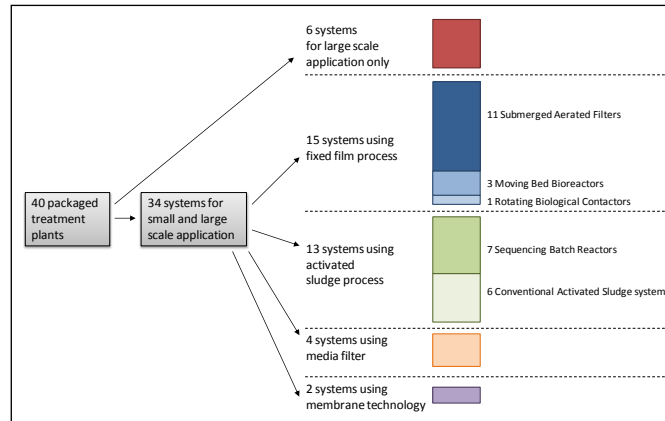


Figure 1: Proportions of treatment processes used in packaged wastewater treatment plants in Ireland

Treatment performance and effluent qualities. Over 70% of the available package treatment systems achieved average effluent BOD₅ concentrations of ≤ 12 mg/L during testing with SS concentrations usually being below 20 mg/L. Nitrifying bacteria, that convert ammonia to nitrate, are very temperature sensitive so that ammonia removal is dependent on water temperatures within the bioreactor. With decreasing temperatures nitrification can be inhibited. For better comparison of the nitrification potential of packaged treatment plants average ammonia removal rates were determined for temperatures $\geq 12^{\circ}\text{C}$. Most plants (about 80%) obtained average effluent concentration between 0.2 and 8 mg NH₄-N/L. However, consulting detailed performance data from the entire test period showed that even at winter temperatures (reached at the test locations in Germany) ammonia effluent concentrations usually stay within discharge limits.

Figure 2 shows the average effluent concentrations for BOD₅, SS and NH₄-N that are achieved by package plants using different treatment processes. As the data collection has not been completed yet the graph represents performance data from 3 MBBR, 8 SAF, 5 CAS, 5 SBR, 3 filter media, and 2 MBR systems. From the figure it can be seen that the average effluent concentration obtained from most package treatment plants are well below the current surface water discharge limits of 20:30:20 mg/L for BOD₅, SS and NH₄-N (EPA, 2009). In direct comparison it is apparent that the best treatment performance is achieved by plants that use filter media or membrane technologies (Fig. 2). All filter media plants produced effluent with BOD₅ and SS concentrations of ≤ 5 mg/L while MBRs reached concentrations of ≤ 2 mg/L. No major differences in performance could be observed between the other treatment processes. While 75% of CAS systems achieve generally good BOD₅ removal with effluent concentrations of ≤ 10 mg/L, SS concentrations are rather high with a maximum of 23 mg/L being observed as the average effluent concentration in one plant.

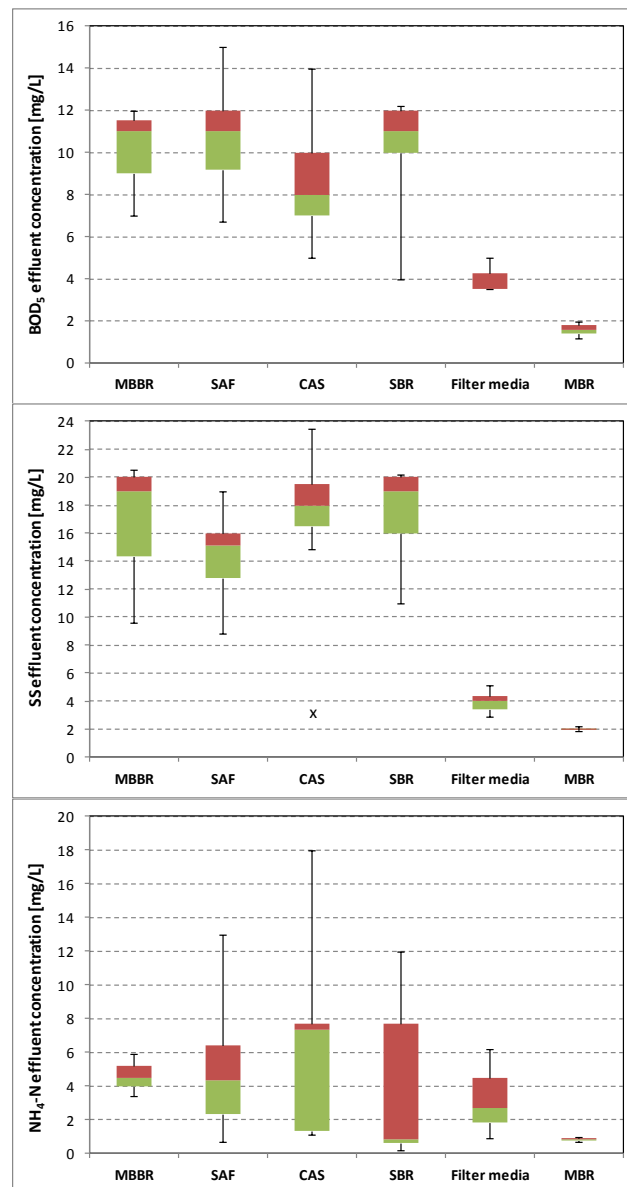


Figure 2: Distribution of average effluent concentrations (BOD₅, SS and NH₄-N) obtained from package plants using different treatment processes. The boxplot shows the lower quartile (green), the upper quartile (red) as well as minimum and maximum concentrations obtained.

However, one CAS plant has the option of an additional filter being incorporated into the system that decreases SS concentrations down to 3 mg/L (indicated as an outlier (i.e. x) in Fig. 2).

The MBR systems also achieved the best nitrification rates with NH₄-N effluent concentrations below 1 mg/L (Fig. 2). Due to the membrane filtration solids are contained efficiently within the biological process so that usually long sludge ages and high mixed liquor suspended solids (MLSS) concentrations are reached. These conditions support the slow growing nitrifying bacteria resulting in high nitrification. Filter media systems and

MBBRs also achieved good ammonia removal. Most plants produce effluents with ≤ 5 mg NH_4/L . Effluent concentrations from the other treatment processes vary largely between the different plants. Average ammonia effluent concentrations in CAS systems for instance vary from 1 up to 18 mg/L between the different plants.

Table 1 summarises the treatment performance and effluent qualities that are obtained from the best performing system of each process type.

Table 1: Treatment performance and effluent qualities (according to EN 12566-3 test results) obtained from the best performing system of each process type

System		BOD ₅	SS	NH ₄ -N [†]	TN [†]	TP
RBC[‡]	[mg/L]	<15	<25	<15	n/a	n/a
	[% removal]	89.4	94.8	88.6	45.7	47.6
MBBR	[mg/L]	7	9.6	5.9	20.3	5.1
	[% removal]	97.3	96.5	80.3	59.6	35.4
SAF	[mg/L]	6.7	8.8	2.5	17.9	3.8
	[% removal]	97.1	96.9	92.2	64.6	50
CAS	[mg/L]	7	17	1.1	n/a	n/a
	[% removal]	97.2	94	96.7	61.7	47.4
SBR	[mg/L]	4	11	0.2	n/a	n/a
	[% removal]	98.6	96.5	99.2		
Filter media	[mg/L]	5	4	0.9	12	n/a
	[% removal]	97	99	96	64	
MBR	[mg/L]	1.2	1.9	0.7	20.1	3.6 / 0.17*
	[% removal]	99.6	99.5	97.9	61.6	53.2 / 97.9*

[†] for temperatures $\geq 12^\circ\text{C}$

[‡] effluent concentrations according to manufacturers information

* with P-removal by adsorption to granulate

n/a = data not accessible

Nutrient removal. Several package 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 (Table 1). Only one of the reviewed package treatment plants (Filter media system) has shown that it can meet the discharge limit of 15 mg/L given

by the EU Council Directive (91/271/EEC) for the treatment of urban wastewater. No secondary treatment system was able to reach the required limit of 5 mg TN/L (EPA, 2009) to allow discharge to surface waters in nutrient sensitive areas.

To enhance denitrification effluent recirculation into the anoxic zones is needed. This would allow nitrates, produced in the aerobic stage via nitrification, to be removed from the water. However, this further adds to the systems complexity, increasing costs and maintenance requirements.

Phosphate removal is traditionally achieved by chemical precipitation using salts of iron, aluminium or lime as coagulants. 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 OSWWT systems. Furthermore additional costs for chemicals and more frequent desludging brought about by increased sludge production have to be considered. In order to achieve biological P-removal from wastewater microorganisms are subjected to alternating anaerobic and aerobic conditions. However, this involves a high process control and difficulties in assuring stable and reliable operation have been reported for municipal WWTPs (Blackall *et al.*, 2002). Thus, biological P-removal is not a realistic option for automated OSWWT systems. As a consequence most of the package treatment plants do not incorporate any P-removal in their standard models but some suppliers offer chemical P-removal for customised systems. Some package treatment plants achieve up to 50% P-removal even without any chemical dosing. However, discharge concentrations of effluent are still above 3 mg/L (Table 1) so that additional removal is needed if effluent discharge to surface waters in sensitive areas (2 mg/L required) is to be considered (EPA, 2009).

In past years different phosphorus-adsorbing materials have been extensively examined for their P-sorption capacity and their potential use in constructed wetlands or in other small-scale filter systems for TP-removal from domestic wastewater (Westholm, 2006). So far there are only few data for performances under field conditions and no empirical data on the longevity of these P-adsorbing substrates, but it is certain that for application purposes it will have to be replaced at some point after saturation. This needs to be considered when these substrates are incorporated into wastewater treatment systems so that it is accessible for the replacement. Furthermore the handling of the P-saturated filter media has to be considered. It can either be disposed at the landfill or find some beneficial use i.e. as fertilizer or soil conditioner in agriculture. Blast furnace slag for instance has been shown to release absorbed phosphorus in a form that is readily available to plants for assimilation (Westholm, 2006). However, the use in agriculture might be restricted as toxic pollutant such as metals might be adsorbed to the substrate as well.

Based on these concepts a new P-removal filter material will be soon available on the Irish market. The material is reported to not only remove TP down to levels below 2 mg/L but

also to reduce the microbial loading of the effluent by 95.5% (as total coliforms removal). However, the disinfection is achieved by an increase of the pH to about 10 so that a pH adjustment will be needed before effluent can be discharged to surface waters. One of the available MBRs can also be supplied with an additional P-adsorption filter bed. However, the manufacturer does not supply the filter separately as a trouble-free operation can only be assured for the treatment of effluent with low BOD₅ and SS concentrations.

Impact of higher influent concentrations on treatment performance. With the view to save water and energy costs, or to reduce the hydraulic load to either the subsoil or to water courses, water minimisation devices might become more widespread in rural households. Reducing the wastewater production will proportionally increase concentrations of organics, nutrients and other pollutants which may have an impact on the wastewaters treatability. Table 2 shows concentrations typically expected for domestic wastewater (Metcalf and Eddy, 1991) and how these concentrations will increase with lower water consumptions by using common water saving devices such as dual flush toilets, low flow shower heads and tap aerators. How this will affect package treatment plant performance has not been quantified yet and tests using higher wastewater concentrations may be needed in the future.

Table 2. Change of wastewater concentrations with the use of water saving devices

Contaminants	Typical domestic wastewater concentrations [mg/L]			Wastewater concentrations [mg/L] for volume reduced by 42.2%		
	weak	medium	strong	weak	medium	strong
BOD	110	220	400	190.2	380.4	691.7
COD	250	500	1000	432.3	864.6	1729.2
TS	350	720	1200	605.2	1245.0	2075.1
TSS	100	220	350	172.9	380.4	605.2
TN	20	40	85	34.6	69.2	147.0
org. N	8	15	35	13.8	25.9	60.5
Ammonia	12	25	50	20.8	43.2	86.5
TP	4	8	15	6.9	13.8	25.9

Cost comparison. Figure 3 shows the plant costs per person when serving a range of population equivalents. For a single house with 5 inhabitants the MBRs (€1600-1800 pp) and

MBBRs (€1200 pp) are the most expensive while SAF (€400-700 pp) and CAS systems (€450-550 pp) have the lowest capital costs. While some filter media are in the upper price range (€1000 pp) there are systems that are cost competitive with SAF and CAS plants (€700 pp). 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 (Fig. 3). For an SAF plant for instance the plant cost per person is reduced by 50% down to €200 for a plant serving 40 compared to 5 PE. Although Figure 3 only shows costs of plants serving up to 50 PE there are several plants available for small communities of up to 400 PE. Some manufacturers also give the option to design a larger customer specific system outside of their standard range.

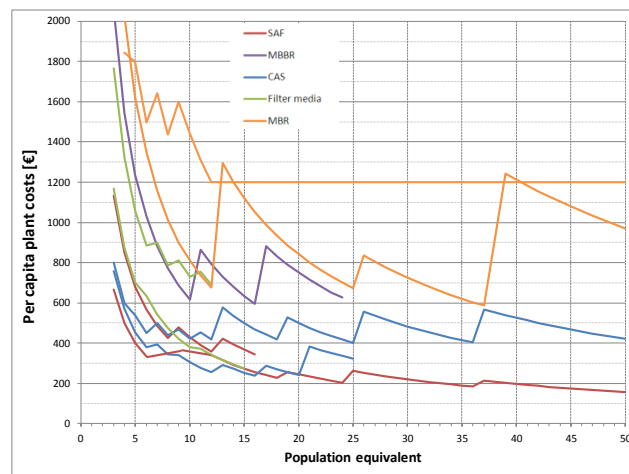


Figure 3: Per capita plant cost development for increasing population equivalents served by different package treatment plants

SAF, MBBR and CAS systems require an air blower running continuously for 24 hour to supply oxygen to the aeration chamber. For a single household with 5 inhabitants the annual electricity costs for such a system is estimated to range between €20 and €30 per person. An MBR needs an additional vacuum pump for the filtration process so that running costs can be about twice as high. The rotor of an RBC system uses less energy than an air blower so that annual electricity costs are estimated to be around €13 per person in a single household. Figure 4 shows that 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 below €10 per person per year. In some filter media systems the effluent is distributed by gravity so that no electricity is needed. Other systems however use a pump with float switch to intermittently apply the effluent over the media. The arising electricity costs to run these pumps are below €5 per person per year. As the pump operates depending on the wastewater production per capita costs are expected to stay similar for larger systems (Fig. 4). The aeration in an SBR system does also

not run on a continuously basis and treatment cycles are only started when enough wastewater has been collected in the primary chamber. Observed electricity consumptions during the EN 12566-3 testing period has shown that the annual running costs for small SBR plants (up to 8 PE) will range between €4 and €6.60 per person.

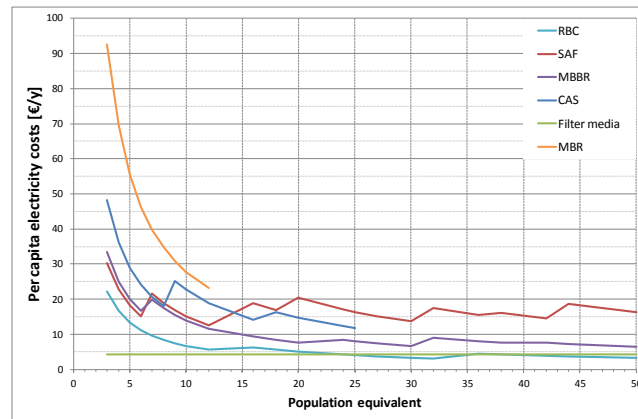


Figure 4: Per capita electricity cost development for increasing population equivalents served by different package treatment plants

Although MBRs are generally more expensive for single houses, one of the systems becomes cost competitive with other systems when used as a decentralised system (Fig. 3). For example, a decentralised plant serving about 225 PE the capital costs are estimated to decrease down to €454 per person. Researchers from the Centre for Water Science at Cranfield University in the UK investigated and calculated the capital and operational expenses of small MBR package plants for domestic use (Fletcher *et al.*, 2007). Their results indicated that it is possible to produce a single household MBRs at similar capital costs as one for the more expensive traditional packaged secondary treatment systems although the power requirements would still be 4 times higher for MBRs. However, when using systems designed for more than 20 PE they showed that the cost difference per head becomes negligible.

Tertiary treatment units. To improve effluent quality, tertiary treatment units such as constructed wetlands, sand filters or other packaged units (e.g. for disinfection or specifically designed nutrient removal systems) can be added after secondary treatment and before the discharge to rivers. These systems can reduce the number of micro-organisms present in the treated wastewater or further decrease organic, solids and nutrient concentrations to achieve standards depending on the sensitivity of the receiving waters. Part 7 of the European Standard deals with "Prefabricated tertiary treatment units", but is

only available as a preliminary draft (prEN 12566-7:2009). However, it is conceivable that in future these systems can be tested in a similar way as secondary treatment package plants.

Maintenance and operational issues. Often the success of a package treatment plant and effective wastewater treatment is dependent on correct installation and regular maintenance of the system. All available plants require at least one service visit per year and need to be desludged regularly. Many suppliers offer service and maintenance contracts that help keeping the system in good working conditions in order to ensure continued treatment efficiency. Significant operational problems could also appear due to user abuse (e.g. disposal of wipes, nappies and greases, or overuse of antibacterial cleaners) resulting in insufficient treatment performance with effluent concentrations exceeding discharge limits.

To ensure a successful implementation of advanced secondary and tertiary treatment units for single houses and small communities in rural areas, the correct installation and regular maintenance needs to be enforced by the regulating authorities. Furthermore the risk of user abuse needs to be reduced by appropriate education about wastewater treatment.

CONCLUSIONS

- Appropriate treatment technologies are available to meet even strict discharge limits for BOD₅, SS and NH₄-N.
- Further developments in TN removal technologies are needed to meet discharge limits for sensitive areas.
- Additional TP removal is needed to meet discharge limits for sensitive areas. Adsorption filters could be a sustainable solution.
- The possible increase of influent concentrations due to water saving actions and its effect on the plants treatment performance needs to be considered.
- Decentralised systems can lower the burden of monitoring associated with discharge consents
- Decentralised plants can represent an environmentally and economically sustainable solution providing that an appropriate management system is in place to operate and maintain plants.

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SESSION IV REGULATORY FRAMEWORK

SITE ASSESSMENT FOR SUSTAINABLE ON-SITE WASTEWATER MANAGEMENT SYSTEMS IN AUSTRALIA AND NEW ZEALAND: REGULATORY MODELS TO GATHER, MEASURE AND EVALUATE INFORMATION

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ABSTRACT

On-site wastewater management is governed by State, Territory and Local Government (Council) legislation and regulations in Australia and by National and Local Governments in New Zealand. The only guidelines for site and soil assessment procedures for on-site wastewater treatment and land application systems that is common to both countries, is the Australian and New Zealand Standard AS/NZS 1547: 2012 On-site domestic wastewater management. This Standard only has legal standing when it is mandated by a jurisdiction. This paper outlines the regulatory framework and procedures for evaluating the capability of single residential allotments to safely and sustainably treat and disperse wastewater to land in jurisdictions across Australia and New Zealand. Several Councils and a water authority that have developed scientifically-based, comprehensive site and soil risk assessment models, hazard categories and maps, and sophisticated decision support tools to provide the highest protection to public health and the environment are showcased.

Keywords: on-site wastewater, site and soil assessment, hazard rating, regulations.

INTRODUCTION

Australia and New Zealand face many of the same issues as Ireland when dealing with the risks and robustness of site and soil assessments for on-site wastewater treatment and dispersal systems. However, as wastewater is under the jurisdiction of the States and Territories in Australia there is no single national legislative or regulatory framework. Each State and Territory has its own legislation, policies, regulations, codes of practice, guidance documents, practice notes and information bulletins governing the treatment, recycling and disposal of domestic wastewater on a single household allotment (see Table 1). This includes regulations for land capability assessment, the design, installation, operation, maintenance and monitoring of on-site wastewater management systems, compliance and enforcement requirements and fee setting. Where a local government Council or drinking water supply authority considers that the State regulations for site assessment, installation, maintenance or management are not sufficiently robust or locally relevant, they may

publish their own regulatory documents and technical manuals under the State planning provisions or potable water supply legislation.

Each Australian State and Territory has a two-tier accreditation system in which a government department issues conditional Certificates of Approval for specific models of wastewater treatment systems to be sold in that State or Territory. Local Councils issue permits for wastewater treatment and land application systems to be installed and operated on specific allotments. The responsibility for issuing Approvals in each jurisdiction is:

- Australian Capital Territory (ACT) – Department of Health www.health.act.gov.au
- New South Wales (NSW) - Department of Health www.health.nsw.gov.au
- Northern Territory (NT) - Department of Health www.health.nt.gov.au
- Queensland (QLD) – Department of Housing and Public Works www.hpw.qld.gov.au
- South Australia (SA) - Department of Health www.health.sa.gov.au
- Tasmania (TAS) – Department of Justice, Workplace Standards www.justice.tas.gov.au
- Victoria (VIC) – Environment Protection Authority www.epa.vic.gov.au
- Western Australia (WA) - Department of Health www.health.wa.gov.au .

(see Table 1 for the legislation and regulations for each jurisdiction).

New Zealand (NZ) also has three levels of government, but instead of Australia's Federal, State and Local government (Council) divisions, New Zealand has the National Government and Regional and District Councils. The national legislation (see Table 1) empowers Regional and District Councils to publish policies, regulations, plans, technical manuals and guidelines "to promote the sustainable management of natural and physical resources" (RMA, 1991). Councils are also empowered under the Health Act 1956 to protect their citizens from any sanitary convenience or premises that may be injurious to health or offensive.

As well as the generally accepted requirements to protect public and environmental health, the concepts, protocols and rules developed in New Zealand for on-site wastewater management have been moulded by the Maori culture and beliefs. New Zealand legislation requires that Maori concepts of stewardship and sustainable management of the environment are included in regulations and policies; for example: "Maori cultural and traditional values shall be recognised and provided for in the management of water quality" (ARC Regional Policy Statement, Policy 8.4.24, 1999). Maori culture requires that contaminants are not discharged to water or air (RMA, s. 15, 1991). Consequently, treated sewage effluent must be dispersed to land via sub-surface irrigation or trench systems, not

to the air (via spray irrigation) or discharged to receiving waters. These cultural rules practised by all New Zealanders are intended to not only protect public health, livestock, native animals and aquatic life, but also sacred sites and the life force of water itself.

The only guidance document that encompasses all levels of government and jurisdictions in Australia and New Zealand is the Australian Standard / New Zealand Standard AS/NZS1547 *On-site domestic wastewater management* (2012). Although it is a national Standard for the two countries, it only has regulatory power when a State or Territory government department or New Zealand Council mandates its use in a regulatory instrument, such as a policy document or a Code of Practice.

DISCUSSION

The site assessment requirements for on-site domestic wastewater systems are variable across the state, territory and local government jurisdictions of Australia and New Zealand because, apart from the fundamental objectives of protecting human health and drinking water supplies, the diverse governing institutions may also have other objectives to fulfil. Local Councils, water authorities and state/territory government departments have legislative and regulatory requirements to protect the health and amenity of the environment and the beneficial uses of groundwater and surface waters. The State departments charged with overseeing the safe and sustainable management of on-site wastewater management systems in Queensland and Tasmania do this through plumbing and drainage legislation and regulations. Victoria is the only State where an Environment Protection Authority (EPA) regulates on-site wastewater management. In all other States and Territories, and at the national government level in New Zealand, the Departments (or Ministries) of Health fulfil that role.

Queensland and Tasmania have mandated that Australian / New Zealand Standard 1547 (AS/NZS 1547) be used for site and soil assessments and the design, installation, management and monitoring of on-site wastewater treatment and land application systems. Although, other States, Territories and Councils refer to AS/NZS 1547 and may mandate its use, they have incorporated other site and soil assessment requirements in their Codes of Practice or regulatory guidelines for on-site wastewater management systems.

Australian / New Zealand Standard 1547 *On-site domestic wastewater management* This Standard takes a risk management based approach to assessing, designing, installing,

operating and monitoring on-site wastewater management systems. This includes the identification, assessment, reduction and monitoring of risks to public health, the environment and local amenity. Risks need to be well managed to avoid:

- contamination of drinking water supplies,
- contamination of recreational waters,
- skin contact with effluent,
- inhalation of effluent aerosols,
- ingestion of effluent,
- contamination of vegetables, fruits and herbs,
- negative impacts on aquatic habitats and organisms,
- negative impacts on terrestrial ecosystems,
- reduction in the amenity value of land, water and air through odours, bogginess, ponding, scums and algae overgrowth,
- contamination of shellfish,
- negative impact on the health of livestock and native animals.

Risk identification and reduction is inherent in site assessment and on-site wastewater system design process. After clarifying the property owner's objectives, the site assessment process encompasses a desk top study, an on-site and surrounding area field check and land capability testing and evaluation. The on-site wastewater design process aims to determine whether the property is large enough to accommodate an appropriately-sized treatment system, land application (irrigation or disposal) system and duplicate reserve area for the size and location of the house and infrastructure that the property owner wishes to build. Where there is insufficient land to sustainably manage the proposed volume of effluent, the number of bedrooms in the proposed house will need to be reduced. In some jurisdictions a "water balance" may be required to determine the size of the land application area needed to sustainably absorb the effluent in the wettest month of the year. A nutrient balance may need to be carried out for environmentally sensitive areas such as sandy soils with high watertables near sensitive lake and river ecosystems or in drinking water supply catchments.

A desk top study should identify features on and adjacent to the property, such as:

- underlying geology and soil types,
- potentially poorly drained areas, drainage lines and flood frequency,
- legal and planning information including boundaries and existing and proposed infrastructure,
- landuses,
- location, depth, nature and value of aquifers and bores,

- potable water supply catchments, dams and waterways,
- rainfall and pan evaporation readings for at least 30 years,
- topographical features including slope and aspect,
- vegetation type and density, especially water loving plants (indicating boggy land).

A soil assessment is carried out through inspection and categorisation of the soil from boreholes or in test pits. A soil permeability (constant head) test maybe mandated by a jurisdiction or may only be undertaken to determine the hydraulic conductivity of the soil where there is a dispute or substantial doubt about the field assessment of the soil categories. The permeability values are used to determine the soil category and relevant Design Loading Rate (DLR) or Design Irrigation Rate (DIR) from the tables in AS/NZS 1547. A site and soil assessment must report on (as a minimum):

- the degree of previous soil disturbance, contamination, compaction and imported fill,
- risk of erosion and land slippage,
- ephemeral drainage lines, seepage, watercourses,
- risks from stormwater flows and flooding,
- slope and shape of land,
- distance to surface waters, road cuttings, embankments, retaining walls, fences and buildings,
- presence of restrictive soil horizons and bedrock and shallow soils,
- soil surface conditions – stoniness, dampness, hardness, soil cracks,
- geology and rocky outcrops,
- holes in the soil profile created by decayed plant roots or worms,
- sodic and dispersive soils,
- salinity
- exposure to sun and wind (aspect),
- soil permeability test results,
- vegetation cover,
- soil horizons and characteristics including mottling or gleying,
- depth to dry-weather watertable, shallow perched or seasonally high watertable.

The 'Site and Soil Evaluation Form' in the Standard is used as a template to develop more comprehensive and regionally specific pro-formas for displaying, categorising and assessing the information gathered. Several jurisdictions have developed hazard maps and hazard rating tools and /or land capability rating scales to help reduce some of the subjectivity inherent in the site and soil assessment process.

Victoria

It is estimated that there are around 270,000 on-site wastewater systems in Victoria (servicing 20% of the population). The Model Land Capability Assessment (LCA) Report (MAV & DSE, 2006) was developed to provide best practice protocols for undertaking site and soil assessment and evaluation of proposed new subdivisions, new allotments and existing premises with failing systems. The LCA Model includes a land capability assessment and site evaluation matrix with 23 site and soil characteristics. A rating of '1' for very good to '5' for very poor is allocated to each relevant characteristic. The rating for the site is determined by the lowest capability (i.e. highest rating) for any characteristic, not the average of the ratings. Any characteristic that is allocated a value of '5', is an indication that the site is not suitable for an on-site wastewater management system. Depending on the category, remedial work may ameliorate the problem i.e. where the soil depth is <0.5m to a watertable or an impervious layer, a mound system could be installed to increase the depth of soil. The poorer the land capability rating for the site, the higher the level of treatment that is required e.g. a site with a rating of 4 may require a secondary treatment system, whereas a site with a rating of 1 or 2 may be suitable for a primary treatment system.

Correctly sizing the land application (irrigation, trench, bed or mound) area is crucial to sustainably dispersing the household wastewater without causing a negative environmental or public health impact. The highest risk of negative impacts from on-site wastewater systems is frequently encountered in the wettest and coldest months of the year. The LCA Model provides formulas and worked examples to assist assessors in creating electronic spreadsheets to calculate the 'water balance' and 'nutrient balance' for any site. The water balance for an effluent dispersal area is expressed as 'precipitation + effluent applied = evapo-transpiration + percolation' for the wettest month of the year. A nutrient balance is calculated to determine the area of land that is required for the nutrients to be assimilated by the soil and vegetation cover. A nutrient balance is generally only required by the local Council or water authority in environmentally sensitive areas, such as sandy soils in the vicinity of a lake, or where the property is in proximity to a drinking water supply. Although the LCA Model is currently a technical manual with no regulatory power, when the new EPA Victoria Code of Practice for On-site Wastewater Management is published it will be a mandated as the LCA procedure and evaluation tool to be used in Victoria.

New South Wales (NSW)

It is estimated that NSW has over 300,000 on-site wastewater management systems (DLG, 2005). In 1997, an outbreak of food poisoning caused by eating contaminated oysters from Wallis Lakes in NSW resulted in the death of one person and over 440 people contracting

Hepatitis A. This became known as ‘the Wallis Lake Incident’. Investigations pointed to septic tank effluent as being the most likely source of pathogens in the oysters that all the victims had eaten. This incident led the State government to change a number of Acts of Parliament and regulations to better protect public health through better managing the flow of on-site wastewater to groundwater and surface waters.

A state-wide ‘Septic Safe’ program was set up to coordinate, support and fund each of then 175 (now 152) Councils to locate and audit their on-site wastewater systems and to develop a plan for their ongoing management. Each on-site wastewater system was categorised as high, moderate or low risk depending on their current and potential impact on the environment and public health. The three categories of risk were correspondingly colour coded as red, yellow or green (equivalent to the traffic light colour code). The Septic Safe program provided comprehensive support for Councils to improve their administrative systems and establish a management plan. Council on-site wastewater management plans are based on investing more resources on inspections and issuing notices to property owners to upgrade high risk treatment systems and land application areas. The intention of the Septic Safe system was that on-site wastewater systems which have been assessed and categorised as ‘red’ would be audited every 2 years, ‘yellow’ every 4 years and ‘green’ every 6 years. Some Councils with limited staff and thousands of on-site systems struggle to fulfil the goals within the timeframes.

The Environment & Health Protection Guidelines: *On-Site Sewage Management for Single Households* was jointly published by five State government departments in 1998 to provide guidance on site evaluation and treatment system options for new developments. Due to repeated failure of the NSW government to update and revise the Guideline, many Councils and other regulatory authorities (i.e. water authorities) have developed their own area specific regulations and guidelines, especially for site, soil and risk assessment. Local jurisdictions with landscapes and soil types that pose a high risk to public health and the environment are at the forefront of developing best practice protocols and guidance to map and rate the risks. Four contiguous Councils on the mid-North Coast of NSW (Port Stephens, Greater Taree, Great Lakes and Lake Macquarie) have had a large increase in population over the last three decades. Sandy soils, a warm climate and estuarine landscapes make them popular tourist and residential destinations and ideal for recreational water sports and growing oysters.

Port Stephens Council

With the Wallis Lake Incident still a prominent reminder of the risks of sewage pollution to public health, when Port Stephens Council found human DNA in oysters from an oyster lease in Tilligerry Creek, resources were swiftly mobilised to improve monitoring of groundwater, waterways and oysters and to develop a locally-specific hazard rating tool to assess unsewered blocks of land and the cumulative impact of on-site wastewater management systems. To ensure that the evaluation and rating tool was a mandatory instrument it forms the basis of the 'On-site Sewage Development Assessment Framework' (DAF) (2011) manual which has regulatory power through the Port Stephens Council Planning Scheme Development Control Plan (DCP). The DAF manual states:

"The Framework sets out Council's levels of investigation, acceptable solutions (deemed to satisfy) and minimum standards for sewage management in unsewered areas. All unsewered allotments in Port Stephens have been assigned an On-site Sewage Management Hazard Class. This Hazard Class (Low to Very High) determines the level of detail required for supporting information submitted with development applications and applications to install or alter sewage management systems."

To reduce the subjective nature of LCA evaluations, the cost of LCA duplication and the burden of delays caused by inadequate land capability assessments, Port Stephens Council commissioned a consultancy company to assess, evaluate and map all unsewered blocks in the shire and produce a policy document and a technical manual. Assessment of slope, soils, climate, flooding potential and proximity to waterways and aquifers formed the basis of the landscape evaluation and allocation of Hazard Classes to all areas. This was carried out through desk-top studies and ground-truthing. The data was entered into a GIS system for ease of manipulation, comparison, display and sharing with the public. The resulting Hazard Maps display the Hazard Class (Low, Medium, High or Very High) as coloured overlays on aerial maps with the boundaries of unsewered lots outlined on the landscape. The Port Stephens Council 'On-site Sewage Management Technical Manual' (2011) sets out the technical basis for the land capability and development assessment framework and the scientific and engineering principles and techniques that can be used to demonstrate compliance with the Development Assessment Framework.

Prospective and current property owners can view the Hazard Classes and Maps to ascertain the level of documentation and scientific justification required and the likelihood of a short or long timeframe for approval. Where a block of land is coloured yellow for low risk or light brown for medium risk a plumber or consultant can complete the simple Low or Medium Hazard Class Site Assessment table (with twelve tick boxes) that forms part of the PSC 'Site,

Soil, Systems and Environmental Assessment report for an on-site sewage management system'. When Council has verified the accuracy of the data, the approval to install is issued with little delay.

The DAF sets out the minimum requirements and proposes solutions for on-site wastewater management systems for constrained sites. A checklist is provided for each Hazard Class and used to confirm whether a proposed on-site treatment and land application system is an acceptable solution based on Council planning, development and sewage management policies. Council has developed minimum standards that apply to all site and soil assessment, system selection and sizing, constructability and cumulative impacts. This reduces the uncertainty associated with how much information is required for approval and expedites the approval process. Soil characteristics, additional to the requirements of AS/NZS 1547, that must be investigated and reported on in accordance with the DAF are:

- electrical conductivity,
- pH
- Emerson Aggregate Class (soil dispersivity)
- Cation Exchange Capacity
- Exchangeable Sodium Percentage
- Phosphorous sorption.

Land that is colour coded mid or dark brown (signifying High or Very High risk) requires a high degree of expertise to assess and evaluate the soil and site characteristics and may require a constant head permeability test to be undertaken. Applications to install or alter an on-site wastewater system on a High or Very High Hazard Class site must be accompanied by a Wastewater Management Report prepared by a suitably qualified scientific or engineering professional. The consultant must describe the site and soil characteristics in sufficient detail and demonstrate how the proposed on-site wastewater management system can overcome the site constraints. There is a high probability that Council will not issue an approval to install unless significant remedial work, such as a mound system, is proposed. Council's preferred solution for sites with a Very High Hazard Class is connection to sewer.

Sydney Catchment Authority

The Sydney Catchment Authority (SCA) manages and regulates the catchments and water storages for Sydney's drinking water supply. Its area of operation covers 16,000 square kilometres of land, 11 major dams and 15 Council areas. The *Sydney Water Catchment*

Management Act 1998 required a planning instrument be created to ensure that Councils only grant development consent within the drinking water catchment where it can be shown that the development would have a neutral or beneficial effect on water quality (Greene *et. al.*, 2012).

The SCA has produced the 'Neutral or Beneficial Effect (NorBE) on Water Quality Assessment Guideline' (2011) and a web-based decision support system to enable Councils to carry out comprehensive and consistent NorBE assessments. The NorBE module for assessing proposed on-site wastewater management systems consists of 67 questions and a series of computer generated spatial views, but it does not replace the need for site inspections and ground-truthing by Council staff. Embedded in the NorBE tool is a GIS-based effluent plume generation modelling tool [the Wastewater Effluent Model (WEM)]. The WEM uses spatial information for data input and the proposed wastewater system design calculations to predict the extent of an effluent plume downslope of the land application area. It models the direction and distance that phosphorous, nitrogen and faecal coliforms travel below-ground. If the effluent is predicted to leave the site or reach a waterway the system will not have demonstrated a neutral or beneficial effect and Council must refuse the development application (Greene, *et. al.*, 2012; Noonan & Greene, 2012).

New Zealand

Many Councils in New Zealand have developed their own on-site wastewater management regulation, site and soil assessment guidelines and technical manuals (see Table 1). The Auckland Regional Council has been a leader in the production of comprehensive assessment, design and management manuals. The current third edition of Technical Publication No. 58 (TP 58) (ARC, 2004) includes a nine page 'On-site Wastewater Disposal Site Evaluation Investigation Checklist'. Councils such as Horizons Regional Council, Gisborne District Council and Marlborough District Council have used TP 58 as a template or starting point for creating their own area specific manuals and guidelines. The 'Manual for On-site Wastewater Systems Design and Management' published by Horizons Regional Council (HRC) in 2007 sets out a site assessment methodology for evaluating surface features and sub-surface characteristics. In addition to AS/NZS 1547 soil and soil requirements, two other items that must be reported on are the performance of neighbouring on-site wastewater systems and the 'sum of' all the site constraints.

Although the manuals and guidelines are comprehensive and scientifically based, the decision-making process is based on professional expertise and does not have the support of a decision rating tool.

CONCLUSION

Australia and New Zealand have a range of site and soil assessment regulations, procedures, evaluation checklists and decision-making tools that are used in the many diverse landscapes across a multitude of jurisdictions. The Australian / New Zealand Standard AS/NZS/NZS 1547 *On-site domestic wastewater management* provides a valuable and important foundation for site and soil assessment in both countries. However, where on-site wastewater management systems have posed a high risk to public health through contamination of drinking water supplies, recreational waters and shellfish, Councils and other regulators have developed comprehensive risk assessment procedures, maps and decision-making rating tools. These scientifically based methods and tools provide a transparent technical rationale to guide the land capability assessment process and more rigour to the decision-making process.

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Table 1: Australian and New Zealand On-site Wastewater Management Legislation, Regulations, Policy and Guidance Documents

ACT	Act	Policy	Regulations	Code of Practice	Other Guidance
	Public Health Act 1997				
NSW	Local Govt Act 1993 s. 68, 109, 124 Public Health Act 2010 Protection of Environment & Operations Act 1997	State Environment Protection Policies SEPPs 14, 62, 71	Local Government (General) Regulation 2005 Cl. 40 & 41 Local Govt (Approvals) Regs 1999 Part 4, Cl. 43(1) Hunter Water (Special Areas) Regs 2003		-S/T & C/W Accreditation Guideline Dec 2001 -SMF Accreditation Guideline May 2005 -WCT Accreditation Guideline May 2005 -DGTS Accreditation Guideline Feb 2005 - Environment & Health Protection Guidelines, Onsite Sewage Management for Single Households Feb 1998 -Sydney Catchment Authority, Designing & Installing On-site Wastewater Systems 2012 - Port Stephens Council, Onsite sewage development assessment framework 2011 - Port Stephens Council, Onsite sewage management technical manual 2011 - Australian / New Zealand Standard AS/NZS 1547
NT	Public & Environmental Health Act 2011		Public Health (General Sanitation, Mosquito Prevention, Rat Exclusion and Prevention) Regulations	Code of Practice for Small On-site Sewage and Sullage Treatment Systems and the Disposal or Reuse of Sewage Effluent 1996	EHO – Guidance Note: Processing Applications for Product Approval of an Onsite Wastewater System
NZ	Resource Management Act 1991 S.15, 104, 105, 107, 108, 314-325, 334, 338, 339 Building Act 2004 S. 3 Health Act 1956 S. 23, 29, 33, 34, 39, 41, 42,44,54, 60 Local Government Act 2002 S. 127, 128	National Policy Statement Council Regional Policy Statements	Building Regulations 1992	Building Code 2004 Cl. B1, B2, G1, G13, G 14 Council Regional Plans: Air, Land, Water (for discharges to air, land, water & coastal areas)	- Auckland Regional Council Technical Publication No. 58 (TP58) 3 rd Edition, 2004 - Gisborne District Council – Guidelines for On-site wastewater management, 2011 - Horizons Regional Council – Manual for onsite wastewater systems design and management, 2007 - Marlborough District Council – Guidelines for new onsite wastewater management systems, 2005 Australian / New Zealand Standard AS/NZS 1547
QLD	Plumbing & Drainage Act 2002		Standard Plumbing & Drainage Reg 2003	Plumbing & Wastewater Code Feb 2009	Australian / New Zealand Standard AS/NZS 1547
SA	Public Health & Env Act Local Govt Act 1999 s.177		Public Health (Wastewater) Regs 2012	Onsite Wastewater Systems Code Oct 2009	Australian / New Zealand Standard AS/NZS 1547
TAS	Building Act 2000 s. 3 (1) Interpretation s. 59 Authorisation and accreditation	State Policy on Water Quality Management 1997	Plumbing Regulations 2004 s. 3 (1)	Tasmanian Plumbing Code Authorisation & Accreditation 2006	- Tasmanian Plumbing Code Authorisation & Accreditation – Plumbing and Drainage Products Authorisation Process & On-site Waste Water Management Systems Accreditation Process (GB351) - Application for NEW accreditation of an on-site waste water management system (GF078) Australian / New Zealand Standard AS/NZS 1547
VIC	Environment Protection Act 1970 s. 53 J-O, Schedule A Local Government Act 1989 Local Laws Public Health & Wellbeing Act 2008 s. 58	State Environment Protection Policy – Waters of Victoria 2003 Cl. 32 - 34	Schedule of Fees Regs Plumbing Industry Commission Regs 2008	Code of Practice, Onsite Wastewater Management Pub. 891.2, 2008	Pub 935 – applying for a Certificate of Approval Pub 746.1 – LCA for onsite domestic wastewater management Australian / New Zealand Standard AS/NZS 1547 MAV-DSE LCA Model Report, 2006
WA	Health Act 2011	Government Sewage Policy – Perth Metropolitan Region, Draft Country Sewage Policy.	Health (Treatment of sewage & disposal of effluent and liquid waste) Regs 1974.	- Code of Practice for Product Approval of Onsite Wastewater Systems 2010. - Code of Practice for the Design, Manufacture, Installation and Operation of Aerobic Treatment Units (ATUs) 2001. - Code of Practice for the Reuse of Greywater in WA 2010.	Wastewater Treatment Onsite Domestic Systems WQPN 70 Sept 2006. Guidelines for the Non-Potable Uses of Recycled Water in WA 2011 (based on AGWR Phase I).

Reference: Australian Standard / New Zealand Standard AS/NZS 1547: 2012 *Onsite domestic wastewater management*

ASSESSING THE EFFECTIVENESS OF REGULATORY ACTIVITIES AT 'LOW RISK' SITES AND PROPOSING GOOD PRACTICE GUIDELINES

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This presentation was an amalgamation of two papers previously produced by these authors

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WHEN RISK-BASED REGULATION AIMS LOW: APPROACHES AND CHALLENGES

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THE NATIONAL INSPECTION PLAN FOR DOMESTIC WASTE WATER TREATMENT SYSTEMS - A PROPOSED APPROACH

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ABSTRACT

Domestic waste water treatment systems are generally considered to be a low-risk activity. The central principle of the National Inspection Plan will be the protection of human health and the environment. The National Inspection Plan for Domestic Waste Water Treatment Systems will incorporate a multi-strategy approach utilising engagement and incentive strategies as well as the traditional site inspections. The initial focus will be on ensuring compliance with the registration process and the implementation of engagement strategies to encourage best practice for the operation and maintenance of domestic waste water treatment systems. Site inspection strategies will be predominantly risk based, taking account of sensitive receptors and will be integrated with other regulatory inspections, as appropriate.

Keywords: domestic waste water treatment systems, national inspection plan, risk assessment.

INTRODUCTION

The Environmental Protection Agency (EPA) considers that the overall risk from domestic waste water treatment systems (DWWTSs) to the environment at a national scale is significantly lower than agricultural activities and urban wastewater treatment systems and consequently considered to be 'low risk' activities. There are, however, areas of the country where the potential risk from DWWTSs may be important at a local level due to the density of systems and the prevailing ground conditions.

Domestic waste water disposal accounts for approximately one-third of residences in Ireland. There are an estimated 497,000 (CSO, 2012) DWWTSs in Ireland treating waste water from single houses not connected to a public sewer system.

All DWWTSs are designed to:

- Protect humans from contact with waste water;
- Treat wastewater to minimise contamination of soils, lakes and rivers;
- Prevent direct discharge of untreated waste water to groundwater or surface water;

- Keep animals, insects, and vermin from contact with waste water; and
- Minimise the generation of foul odours.

In most cases the system utilised is a conventional septic tank system. The conventional septic tank system consists of two main parts:

- The septic tank itself, and
- The percolation area, which comprises the effluent distribution system and the adsorption and treatment beneath it, which occurs in the soil and subsoil layers and where the main treatment of the effluent takes place.

Secondary treatment systems (often called 'Advanced' systems) are also employed to treat waste water from domestic houses. These systems offer secondary treatment of discharged effluent and include those constructed on-site and packaged treatment systems. Septic tank systems require greater depths of subsoil and a larger area for distribution of discharged effluent than secondary treatment systems.

The National Inspection Plan (called the Inspection Plan) will assist Ireland in meeting the objectives of the *Water Framework Directive* (2000/60/EC). The objectives of the *Water Framework Directive* (WFD) are to protect all high status waters, prevent further deterioration of all waters and to restore degraded surface and ground waters to good status by 2015. The Inspection Plan specifically addresses one of the measures in the River Basin Management Plans, which deals with inspection and remediation of DWWTSs.

RISK POSED BY WASTEWATER TO HUMAN HEALTH AND THE ENVIRONMENT

Domestic systems pose much less of a risk to watercourses than public sewerage discharges and agricultural effluents - nevertheless they can cause localised pollution. Where systems are not properly located, designed, installed, operated or managed they pose a risk to the health of the individual homeowners (and their children and pets (Figure 1)) through possible contamination of water in private wells or because of effluent ponding in gardens. Such effluent also poses a risk to watercourses and can result in problems for bathing waters and other amenities.

There are a number of different pollutants in waste water, each of which can cause problems for health and the environment. Microbial pathogens are one form of pollutant and these can cause illnesses such as gastro-enteritis, eye infections, polio, hepatitis and meningitis.



Figure 1: Photo of an exposed waste water distribution pipe providing a drinking source for the family pet. (source: A.Goggin)

The presence of *E.coli* bacteria provides evidence of recent faecal contamination from human or animal wastes; drinking water for example is tested for the presence of *E.coli* as it is a clear indicator that pathogens are present. Typically there are approximately 1 million *E.coli* bacteria in one litre of effluent from a septic tank serving a normal household while the drinking water standard is zero. Phosphorus is another pollutant in domestic waste water and it encourages the growth of algae, depletes oxygen, and may cause algal blooms and fish kills.

INSPECTION FRAMEWORK

The proposal for the EPA's Inspection Plan has been informed by international best practice in regulation such as *Recommendation of minimum criteria for environmental inspections* (RMCEI) (2001/331/EC), *Better Regulation* (Hampton, P. (2005)), and the '*Good Practice Framework for Lower-Risk sites*' (SNIFFER, 2011).

Experience in other countries when dealing with low-risk activities suggests that adopting a single strategy may be unnecessarily constraining and in some instance may lead to ineffectiveness (Simon 2010). Black and Baldwin suggest that regulators should develop a mix of strategies to achieve a regulatory outcome. In the case of low-risk sites a variety of non-routine inspection strategies and in particular proxy (e.g. monitoring of water quality), third party, engagement and incentive strategies are available to regulators (Baldwin & Black, 2008; Black & Baldwin, 2012) to protect human health and the environment. These strategies will involve working closely with householders and other stakeholders to ensure that those who are responsible for DWWTSs know how to comply and are encouraged to do so.

The Good Regulatory Intervention Design (GRID) framework operates on the basis that potential intervention strategies are selected and applied on the basis of two factors:

- the potential risk of the domestic waste water treatment system, which includes their proximity to sensitive receptors, and
- the attitude of the homeowner to compliance (as indicated by observance of the registration process).

Figure 2 illustrates the regulatory framework for DWWTS and different types of strategies will be used to achieve a regulatory outcome.

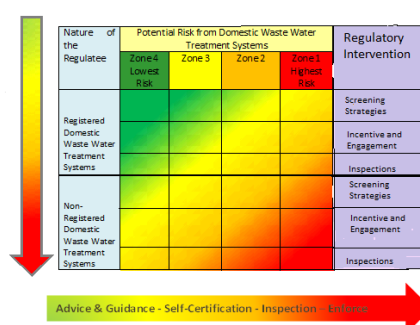


Figure 2: Illustration of the proposed Good Regulatory Intervention Design (GRID) for Domestic Wastewater Treatment Systems

(1) Risk Assessment

The EPA in conjunction with staff from the Geological Survey of Ireland (GSI), Groundwater Section and external expert consultants developed a risk assessment methodology, which has identified the potential risk posed by domestic waste water to human health and the environment. The basis for the risk assessment methodology is the source-pathway-receptor (S-P-R) model where the *source* is the origin of the potential pollutant (the DWWTS), the *pathway* is the link between source and receptor (e.g. the subsoil), and the *receptor* is the medium of possible impact to human health or the environment (e.g. the river) - see Figure 3. In its simplest form the S-P-R model combines the sources (DWWTS) with the pathway (hydrological and hydrogeological characteristics) to determine the risk to the receptor (human health, groundwater and surface water).

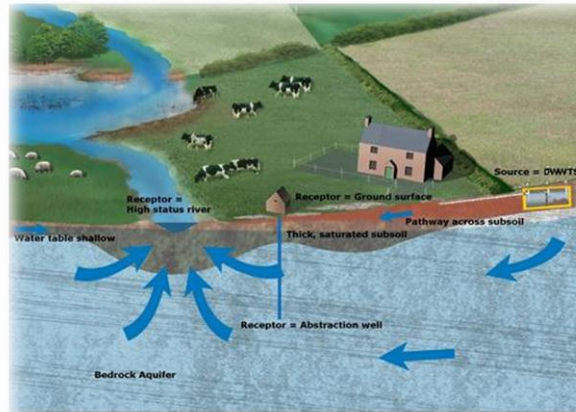


Figure 3: S-P-R Model for domestic wastewater treatment systems with impermeable subsoil (graphic sources from the WFD Visual website, SNIFFER, 2007)

The 'Potential Risk' from DWWTS to human health and the environment is represented by the different zones, Zone 1 (highest risk) to Zone 4 (lowest risk). The different potential risk categories are derived from the Risk Assessment Methodology and the areas can be depicted on GIS maps. The potential risk is a combination of the likelihood of a problem occurring and its impact and is determined by combining data from a number of GIS layers following the S-P-R model. Thus they would include density of houses (source), subsoil permeability (pathway) and aquifer classification (receptor). The impact on sensitive receptors will be taken into account in applying the site selection criteria at a local level. The actual risk derived from the inspections will be used to verify and amend where necessary the risk assessment methodology.

(2) Registration Compliance

Registration is the first step in complying with the regulatory requirement. Awareness raising campaigns will mean that the population will be informed and can choose to comply. Different intervention strategies are proposed for those that have registered as compared with those that have not. The basis for doing so is the assumption that those who register are more likely to operate and maintain a DWWTS in a compliant manner than those who fail or refuse to register.

By combining the potential risk of a DWWTS and the registration compliance level, inspections can initially be targeted at the highest risk areas where the registration compliance is low.

Aim, Objectives and Principles of the Inspection Plan

In developing the Inspection Plan the EPA will adopt a pragmatic, risk-based inspection regime to maximise the protection of human health and the environment from potential impacts of treatment systems. It is proposed that the Inspection Plan will incorporate a twin track approach using citizen engagement and incentive strategies as well as the more traditional site inspection strategies.

The Inspection Plan will be designed to ensure that:

- Owners of domestic waste water treatment systems are informed of their responsibilities and of the appropriate actions required to operate and maintain their systems;
- Adequate treatment of domestic waste water is in place;
- Risks to human health and the environment are known and managed;
- Where risks to human health and the environment are identified, they are remediated; and,
- Waste water sludge from domestic waste water treatment systems are managed appropriately.

Regulatory Intervention aspects of the Inspection Plan

The 'Regulatory Intervention' aspects will form the building blocks of the Inspection Plan. These management based strategies are divided into three categories:

1. Regulatory strategy
2. Engagement and incentive strategies
3. Inspection / monitoring programmes

The regulatory intervention intensity which is depicted by the arrows in Figure 2 is an indication of two factors. First, the intrinsic intensity of the intervention tool chosen (is it high cost, high intervention, such as an on-site inspection, or low cost, low intervention as with a general advice and guidance). Second, it looks to the severity with which the chosen tool is applied on the ground (e.g. its frequency of use and the severity of the associated sanctions or actions demanded).

The higher the risk, the more justifiable it is to intervene with a tool of higher cost and intervention level (e.g. an individual site inspection as opposed to a general advice and guidance) and to intervene with greater frequency and intensity to encourage compliance with the legislation. The proposed strategies are described in more detail below:

1. Regulatory Strategy

Owners of properties connected to a domestic waste water treatment system are required by law to register their systems by 1 February 2013. This is the first step in compliance with the *Water Services (Amendment) Act 2012*. By registering on or before 28th Sept 2012

owners can avail of the reduced fee of €5.00, after that date the registration fee rises to €50.00. In the case of a new build, the domestic waste water treatment system should be registered upon occupancy of the house.

Owners of properties that have a domestic waste water treatment system are also obliged by law to ensure that their systems are operating and managed properly and are not creating a risk to human health or the environment. The Inspection Plan will assist owners to achieve this second step.

2. Incentive and Engagement Strategies

A key priority within the Inspection Plan will be incentive and engagement strategies to promote best practice relating to the operation and maintenance of DWWTSs and encourage registration by homeowners. It is proposed that the early part of the 2013 should focus on these strategies thus raising awareness and changing behaviour prior to the initiation of targeted site-inspections.

i. Incentive Strategies

A reduced registration fee for homeowners (for a three month period) has been introduced to incentivise the early registration of DWWTSs. The Inspection Plan may also consider good performance incentive strategies such as self-certification outlined below.

ii. Information Campaigns, General Advice and Recommendations

It is proposed that the Department of Environment, Community and Local Government (DoECLG) in conjunction with the EPA and Water Service Authorities (WSAs), embark on a series of activities to raise awareness and compliance related to the operation and maintenance of domestic waste water treatment systems. Information campaigns may include the communication of international best practice to WSAs and Inspectors and specific local guidance to NGOs and homeowners.

Campaigns should target priority areas within WSA functional regions and information related to inspection and best practice should be distributed in the targeted area thus informing and driving compliance. e.g., Pearl Mussel Area Plans or Zone of Contribution (ZoC) of Public Water Supply (PWS), catchment area for bathing water, etc.

iii. Dialogue with Stakeholders, NGOs and Interested Parties

The Inspection Plan will be used as a basis for discussion with interested parties e.g. National Federation of Group Water Schemes (NFGWS) as well as NGOs such as SWAN, An Taisce and others. Ideas for bespoke engagement strategies will be sought from the individual groups to encourage awareness raising and behavioural change amongst their members and the wider community.

The International Symposium for Domestic Waste Water Treatment presents an opportunity to communicate and exchange best international practice and guidance between regulatory authorities and NGOs.

The EPA, WSAs and other stakeholders may participate in public speaking events wherever possible to communicate best practice in relation to the operation and maintenance of domestic waste water treatment systems.

iv. Advise and Assist Inspections

WSA and the EPA may engage in providing advice and training to NGOs and other stakeholders as appropriate on the inspection, operation and maintenance of DWWTS to ensure that they are not giving rise to an impact on human health or the environment. Examples include the NFGWSs.

v. Self-Certification & Reporting

Self-certification and reporting to WSAs by home owners operating DWWTS are to be considered as part of the overall regulatory effort. Examples of areas that might be covered include: homeowners submitting certification of system installation; evidence of a system maintenance contract in place; and evidence of tank de-sludging. Where such an approach is adopted in the Inspection Plan a statistical sample of random inspections of such systems will be undertaken.

3. Site Inspection Strategies

Targeted site inspections together with incentive and engagement strategies should be specified to ensure effective implementation of the Inspection Plan. It is proposed that these site inspections will focus specifically on registration, operation and maintenance of DWWTSs. Selection criteria and risk maps will be provided to individual WSAs or River Basin Districts or combinations of both, for implementation. Inspections will vary in both type and frequency as the outcome of the incentive and engagement strategies is measured.

It is envisaged that WSAs themselves will select the individual sites depending on their priority areas taking into account the selection criteria that will be set out in the Inspection Plan and that risk based inspections will begin in 2013.

An Information Technology system is currently being developed by the EPA for use by the Water Service Authorities to enable the site inspection information to be captured and reported in a consistent manner.

The toolbox of inspections that are available within the proposed Inspection Plan include the following:

i. Registration Inspections

It is proposed that inspection of non-registered DWWTS will take precedence over registered systems initially. It is envisaged that each WSA will undertake inspections of non-

registered properties within their functional area. Appropriate enforcement action shall be pursued by WSAs in respect of homeowners who fail to comply with the registration process.

ii. Non-routine Inspections

WSAs may undertake a reactive inspection in certain circumstances including complaints, incidents or accidents, or where the operation of DWWTS is considered to give rise to an impact on human health or the environment. It is anticipated that such inspections will be recorded and responded to in accordance with the requirements of the legislation.

iii. Routine Risk Based Inspections

The Inspection Plan proposes that risk based inspections be undertaken with an initial focus on unregistered systems. The inspections will focus on the operation and maintenance of DWWTS. It is anticipated that adoption of the GRID approach will mean that inspections are likely to be targeted at the most sensitive environments (Zone 1 and 2 risks).

The Inspection Plan envisages that the WSA will prioritise the location of individual inspections based on local knowledge and WFD priorities in their functional area. Thus, not only will the most sensitive areas e.g. designated Pearl Mussel areas, areas discharging to bathing water areas, water bodies with poor water quality status etc. be inspected most intensively, they will be regulated as a matter of highest priority.

A statistical number of random inspections in low risk areas will also be carried out to validate the risk based approach.

iv. Other Routine Inspections

During other routine environmental inspections by WSAs it may be appropriate to incorporate and include inspections of domestic waste water treatment systems, as part of the regime. Examples include routine farm inspections, and implementation of Pearl Mussel Area plans.

v. Proxy Inspections

Proxy inspections refer to the use of technologies to determine performance without having to rely on individual site inspections and usually focus on *receptors*. Measuring proxy outcomes, such as using downstream water quality as an indicator of impact from DWWTSs, may be useful in determining performance; targeting DWWTS (*source*) inspections or triggering remedial actions. In Ireland, as part of the *Water Framework Directive* (WFD), the EPA and local authorities carry out a significant amount of surface water and ground water sampling.

The causal connection between the WFD sampling results and the impact of the DWWTSs in Ireland needs to be further defined. A research project is being established under EPA STRIVE to determine this causal link and may enable this type of approach being used in subsequent inspection plans.

In addition, the outcomes of the inspections will be analysed to establish any such relationship and a picture will be built over time.

vi. Follow-up Inspections

Such inspections are proposed to be undertaken as a follow-up to an initial inspection of the domestic waste water treatment system and includes, re-inspections (on appeal from the owner of a treatment system) and WSA inspections to confirm compliance or otherwise with an Advisory Notice. Such inspections shall be recorded and regarded as an inspection for the purpose of the Inspection Plan.

Third party inspections are proposed in some situations as part of the requirement to verify improvement or remedial works. Such reports would emanate from appropriately qualified person(s). In such instances the Inspection Plan would propose a statistical sample of random inspections of such systems.

vii. Pre-Sale Inspections

There is a legal requirement for the seller of a premises with a DWWTS to furnish a valid certificate of registration to the purchaser of the premises. Experiences from other countries suggest that prospective purchasers may also be looking for an inspection to be carried out prior to the completion of a sale by the seller.

REVIEW OF THE INSPECTION PLAN

The EPA proposes to undertake reviews of the risks presented by DWWTS on a regular basis notwithstanding the statutory requirement to review the Inspection Plan at intervals of not less than 5 years. The primary purpose of the review will be to ensure that the strategies implemented are successful in protecting human health and the environment. In doing so it will collect intelligence on those factors that may impact on existing risk assessments. This will involve re-examinations of such matters as the densities of DWWTSs, the capacities of pathways, and the sensitivities of receptors. The reviews, once undertaken will be followed-up by reconsiderations of the inspection strategies. Those reconsiderations will look at the cases for re-categorising risks and adjusting intervention tools, regulatory intensities and control priorities accordingly.

CONSULTATION PROCESS

It is anticipated that the proposed Inspection Plan will be available for public consultation. Interested parties and individuals will be invited to submit comments in writing to the EPA. The EPA will take into account all comments received in drafting the Inspection Plan.

The EPA will provide a copy of the final Inspection Plan to the Minister for the Environment, Community and Local Government and to each local authority by the end of 2012. It will also be published on the EPA website.

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LOCAL AUTHORITY ASSESSMENT OF ONSITE WASTEWATER TREATMENT SYSTEMS – PRACTICE, ISSUES AND OPTIONS

Seamus O'Brien

ABSTRACT

The existing Onsite Wastewater Treatment Systems have been identified in Environmental Protection Agency reports and River Basin Management Plans prepared for the implementation of the Water Framework Directive as a source of diffuse and point source pollution. Malfunctioning Onsite Wastewater Treatment systems are a threat to human health and the natural environment. This paper is based on research that aimed to establish what the current practices being used by Local Authorities are and what are the issues to be addressed. This was done via case study, focus group and email questionnaire. The study identified the scope of typical Local Authority inspections, what the follow up procedures were and the gaps in knowledge and issues raised. Issues addressed include alternative standards to be used for older houses and restricted sites, nature and scope of inspections, and identification of areas where guidance and clarification are needed. Options are outlined covering areas such as sharing resources and the need to have a system to raise awareness with the public. Areas where guidance and training have to be provided to staff that will be carrying out the initial inspections and dealing with non-compliances are also discussed.

Keywords: onsite wastewater treatment systems, local authority assessments, causes of non-compliant treatment systems, domestic systems.

INTRODUCTION

Onsite wastewater treatment systems (OSWWTS) have been identified in numerous water quality reports by the Irish Environmental Protection Agency as a source of diffuse and direct (point source?) water pollution. These conclusions have also been confirmed in the characterisation reports carried out for the River Basin Management Plans. This report states that unsewered properties account for 3% of the Nitrate and 7% of the Phosphorus load going to surface water (Western RBD, 2008). Given that the 2006 census shows that there are 458,876 unsewered properties in Ireland (CSO, 2007), the inspection and assessment of these systems is a sizable task. Historically, the planning system has been used as the main control for the type and standard of OSWWTS proposed for properties. The bulk of these systems are located in rural areas with the 2006 Census showing just 29,931 in urban classed areas. Approximately a third of all houses with OSWWTS have private non group water scheme wells. Given that the EPA estimate that up to half of all

faecal contamination cases of wells is a result of OSWWTS, the potential threat to human health and ground and surface water quality from malfunctioning systems is significant (Daly, 2009). It is important therefore to ensure that a fair and consistent approach to dealing with malfunctioning OSWWTS is taken by Local Authorities. Given the forthcoming new inspection regime, to become effective from early 2013, it was felt that a baseline of existing practice needed to be established and some of the issues currently facing local authorities raised. The initial aim of the dissertation that underlines this paper was to present a discussion document and possible options that would feed into the development of any further guidance or protocol.

METHODS

A literature review of the available information in relation to guidance, legislation, methods of assessment and background information was undertaken. Following on from this, it was decided that in order to establish the current practices being undertaken by local authorities, three different methods would be undertaken.

The initial method used was an email survey to each of the 34 Local Authorities in Ireland. A response was received from 23 of these Local Authorities with two stating they did not carry out any OSWWTS inspections; both were City Councils. This left a core of 21 Local Authorities that formed the basis for the analysis of issues and practices carried out by the Local Authorities.

To supplement this, an interview was carried out with Mr Colm O'Callaghan, Senior Executive Scientist with Cavan County Council. Cavan County Council is the only Local Authority not included in the European Court of Justice judgement against Ireland for not fully implementing of EU Directives 75/442/EEC and 91/156/EEC in case C-188/08. They are the only Local Authority to introduce Bye laws relating to OSWWTS in the Country, and their experience of the Bye laws as well as issues that occurred were added to the results of the analysis of the email questionnaire to give a fuller picture of the current practices.

The third method used involved obtaining feedback on the preliminary results of the study from a focus group. This focus group was made up of representatives of other Local Authorities, the EPA and I.T. Sligo. This group reviewed a draft discussion document that was an amalgam of the results of the analysis of the email questionnaire, interview and experience of having worked on the investigation and assessment of problem OSWWTS. The membership of the focus group included representatives from Local Authorities who had not replied to the survey but would be in areas of the country where ground conditions for percolation would be moderate to poor. This focus group helped to ensure that the discussion document included as wide a variety of voices and view points as possible and also helped to ensure that any element of bias in the draft discussion document was mitigated.

RESULTS

The email questionnaire focused on the current practices undertaken by Local Authorities. It started by looking at the number of Local Authorities undertaking proactive planned inspections and those undertaking inspections and assessments as a result of complaints. The results show that the minority (six of the twenty one Local Authorities) carry out proactive inspections and in the majority of these cases, they were carried out as part of other regulatory requirements e.g. OSWWTS inspections are carried out as part of protection plans for shellfish areas or drinking water supplies.

When asked about staffing levels to carry out the assessment/inspections of existing OSWWTS, only one local authority had full time staff carrying out the inspections. In the other twenty local authorities, the inspection/assessment of OSWWTS on a proactive or reactive basis was only one part of the staff members' duties. A breakdown of the time allocated for inspections by the fifty two part time staff in the twenty local authorities is presented in Table 1. This equates to 5.35 full time equivalent staff, and when the full time staff are added, it is equivalent to 7.35 full time staff for the twenty one local authorities. These staff were involved in carrying out 448 proactive/planned inspections and 295 complaint related inspections.

Table 1: The breakdown of the percentage time of staff given to OSWWTS issues

No of Staff Members	5	14	23	4	6
Percentage of time given	1 %	5 %	10 %	20 %	25 %

The local authorities were also asked to explain what the site inspection/assessment entailed. It was found that the following elements were classed as being standard in the vast majority of inspections carried out by local authorities:

- Visual inspection for signs of ponding.
- Examination of the curtilage of the dwelling.
- Check grey water drainage is connected to the system.
- Examination of adjoining land drains.

In relation to the use of dye testing, it was a less used option in planned inspections than it was in complaint driven inspections. This is reflected in the response to the question of the number of site visits carried out. The average number of site visits for compliant sites was a single site inspection; however for non compliant sites, the number of visits ranged from two to many as presented in Figure 1 below.

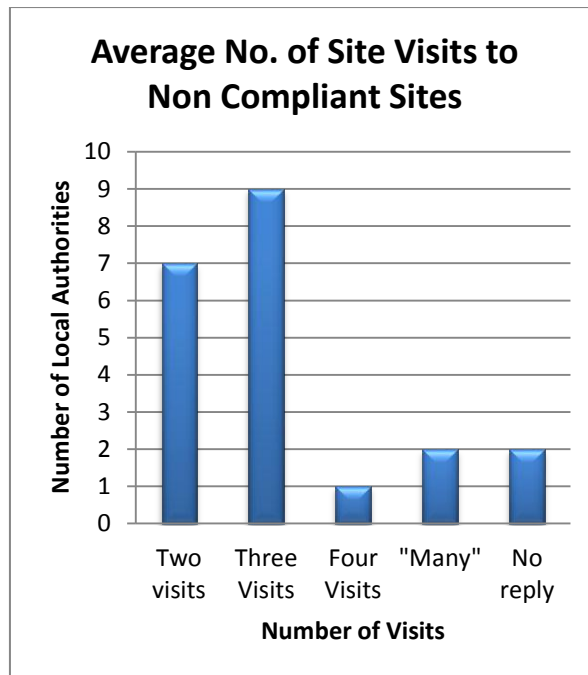


Figure 1: Average number of site visits to non-compliant sites

As can be seen from Fig. 1, non-compliant sites are more resource intensive.

The local authorities were then asked as to the causes of the non-compliances that they encountered.

They were asked to rank a set of options in terms of frequency of non-compliance that they encountered, and the results are as follows:

1. Ground conditions unsuitable for the hydraulic loading from the system.
2. Septic tank or treatment unit cracked and leaking.
3. Greywater connected to a soakaway and not to the treatment system.
4. Unlicensed direct discharge to surface water.
5. Pumps and other electrical or mechanical components faulty and not repaired.
6. System not desludged.
7. Percolation area compacted or built upon.

It is interesting to note that the local authorities who are carrying out proactive/ planned inspections based these inspections on regulatory requirements such as the shellfish and source protection plans. This shows a drive to use resources to meet regulatory requirements, and as a result, the limited available resources cannot always be aimed at areas of high risk due to poor soils and vulnerable groundwater for example. The ranked sources of non-compliance also highlight the need for proper installation and maintenance of treatment systems, with damaged tanks, misconnections and un-repaired components forming three of the top five reasons for non-compliance.

The local authorities were then questioned in relation to enforcement procedures for non-compliant systems and again were asked to rank how they deal with these homeowners:

1. A notice is issued under Section 12 of the Water Pollution Acts on the property owners to carry out certain works within a set timeframe.
2. Proposals are requested from the property owner
3. A programme of works is outlined to the householder and they are asked to complete them within a set timeframe and this is done in letter form.
4. A notice is issued under Section 23 of the Water Pollution Acts requesting information to be submitted.
5. In cases of discharge to a water course, the property owner is prosecuted immediately under Section 3 of the Water Pollution Acts.

These responses show an approach from local authorities that offers an opportunity for householders to remediate issues within a set timeframe and bring the system up to or as near as possible to the relevant standards. If the householder will not co-operate with the local authorities, then prosecution under the Local Government (Water Pollution) Acts 1977-2007 will have to be considered.

The local authorities were also asked to outline how they assessed proposals submitted to them for non-compliant sites. The local authorities were asked if they accepted proposals in line with the Environmental Protection Agency 2009 Code of Practice or if an alternative “Fit for Use” criteria would be applied.

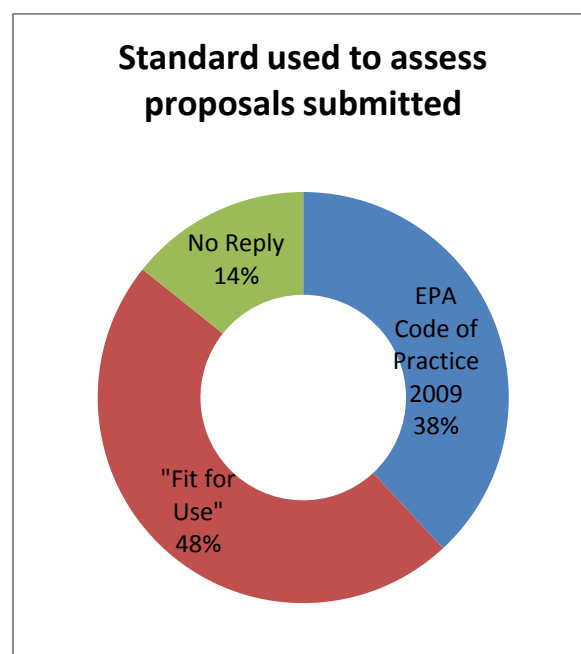


Figure 2: Breakdown of the standard used to assess proposals submitted

Fig. 2 indicates that almost 50% of respondents use a “fit for purpose” standard and almost 40% require all proposals to comply with the standards outlined in the EPA Code of Practice.

It should be noted that the 2009 Code of Practice (Section 6.6) allows for alternative standards to be used on sites.

In addition, Local Authorities outlined the following ranked criteria used to examine proposals and their impacts on the adjacent areas.

1. Public Health Issues e.g. drinking water wells.
2. Surface and ground water quality
3. Public Nuisance
4. Cost
5. Type of system proposed and details of the proposals.

These factors suggest that the design of treatment systems must keep in mind the above key priorities of public health, water quality and the avoidance of public nuisance. In terms of systems used, 57% of Local Authorities accept only Irish Agrément Board (IAB) and En12566 approved systems, with 29% accepting systems that had other sources of accreditation such as USEPA.

The Local Authorities were also questioned in relation to sludge management from OSWWTS. A noticeable difference of approach was taken by different Local Authorities with regard to the landspreading by farmers of their own septic tank waste on to their own agricultural land.

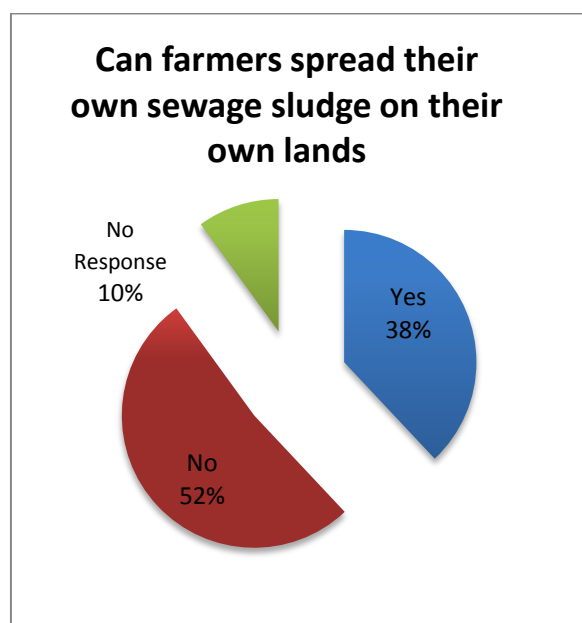


Figure 3: Percentage of Local Authorities that allow farmers spread on their own lands

Fig. 3 indicates that the majority of Local Authorities do not allow farmers to apply their own sludge on their own land, despite provision for such activity in the Use of Sewage Sludge in Agriculture Regulations 1998-2001. This may be due to the difficulty in enforcement of the Regulations, particularly in grassland situations. The questionnaire also

asked about the willingness of Local Authorities to accept domestic sewage sludge into their wastewater treatment plants. The response was that 80% of Local Authorities were willing to accept domestic sewage sludge, subject to available capacity. This raises the question as to whether Local Authorities will have sufficient capacity to deal with the considerable additional volume of waste generated if all rural houses were to desludge annually.

DISCUSSION AND CONCLUSIONS

The research carried out between the analysis of the questionnaire, interviews, focus group feedback and literature review has raised the following:

Standards & Guidance

As many houses have been built before the introduction of the 2009 EPA Code of Practice, it raises an issue as to the standard that is to be applied when assessing sites and remediation proposals. If a property does not have sufficient space to meet the separation distances required in the 2009 Code of Practice, what standard should be used? It is important that a consistent approach is taken by Local Authorities to ensure that a common standard or guideline is applied to all proposals. It is also important that the mitigation measures will allow the site to comply with Part H of the Building Regulations. These regulations set the 2009 Code of Practice as the standard for developments. Where the Local Authority sets a standard less onerous than the Code of practice, then it should not have implications for houses demonstrating compliance with the Building Regulations at a later date. This also applies to the standard of system that is acceptable; should only systems certified by the Irish Agrément Board or EN 12566 series of standards be accepted or should other systems certified by agencies such as the USEPA be also accepted?. Perhaps a combination of the two should be considered where IAB certified systems along with systems certified by a limited number of internationally respected agencies should be acceptable, with the ultimate decision resting with the Irish EPA.

Guidance to Local Authorities need to be developed as to the appropriate standards to be applied in certain cases e.g. restricted sites. This will not cover every non-compliant situation, but give guidance and help to establish consistency in the approach taken by Local Authorities. This guidance should be developed with an input from local authorities before any training is provided. A good training regime will be required to ensure that a standard inspection is applied and that standard methods are used. Additional information and guidance will need to be included in this training e.g. guidance on the use of tracing dyes, appropriate legislation to be used etc.

Resources

One of the biggest issues facing Local Authorities in dealing with the assessment of existing OSWWTS is the (lack of) resources available. As presented in Table 1, the existing available

resources for inspections is quite limited, with resources being deployed to meet regulatory requirements and address public health and water quality complaints. The proposed new inspection regime will prioritise areas for inspection within a functional area, but this may or may not overlap with areas where existing required regulatory inspections are being carried out. This will require additional resources in terms of staff, particularly in relation to follow-up of non-compliant sites, which are more resource intensive. Whether they are redeployed internally or employed on a contract or sub-contractor basis will need to be considered. An important element will be to ensure that the persons carrying out the inspections have the relevant expertise and training. It is proposed that the registration fee (€5 or €50) for domestic wastewater treatment systems will cover the costs of inspection at national level. It is doubtful however that this would be possible, taking into account inspector salaries and expenses, administration and 5 year duration. In a recent presentation to Engineers Ireland, it was indicated that if 450,000 households pay a €5 registration fee, the total fees for 5 years would amount to €2.25m. However, the cost of putting a technician and a van on the road for a year for each Local Authority would cost €2.38m per year (O'Rourke, 2012). When only rural counties are considered, this figure is reduced to €1.82m per annum. An approach may be to have one person covering a number of local authorities where there are known better soils and less vulnerable conditions and a full time person may be required in areas of poorer soils. The inspections will have to be targeted to maximise efficiency and where internal redeployment is used, may result in reduced work carried out in other areas.

Public awareness

Many members of the public have preconceived ideas as to how a septic tank or other treatment system works and what is best for it and who can empty it. Many of these ideas are inaccurate or incorrect. As was the case in Cavan County Council, some people thought that the regulations relating to septic tanks did not apply to them because they had purchased biological treatment systems.. In many cases, it may lead to unnecessary work being carried out to resolve issues that did not exist or the situation being exploited by different interest groups e.g. contractors, equipment suppliers etc. An information campaign should be organised at both a national and local level to address these concerns and outline clearly what is involved. Recently, an information leaflet has been developed by the Department of Environment, Community and Local Government; however further information and public awareness will be needed. This may also raise the need for grant aided funding to house owners to cover upgrade costs. The funding of such a scheme or its administration is something that needs to be examined so that it is seen to be fair and consistent.

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SESSION V CONSTRUCTED PASSIVE AND INTERMITTENT SYSTEMS

CURRENT STATE OF DECENTRALIZED WASTE WATER TREATMENT TECHNOLOGY IN DENMARK

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BACKGROUND

Wastewater treatment in Denmark has been a national concern and one of the areas targeted in order to improve environmental and water quality in the country. The government has taken steps to reduce the discharges and emissions of organic pollution and nutrients from wastewaters to achieve better quality of waters. In 1987, the government stated an action plan aimed at reducing nitrogen and phosphorus discharges to the aquatic environment of 50% and 80% respectively. The plan was intended to reduce the discharges originated from urban, agriculture and certain industrial sources. As expected, it also came with strategies for each of the targeted fields. About the same time the European Union (EU), through the directive 91/271/EEC of 21 May 1991 (eur-lex.europa.eu, 2012) also legislated on the topic and Denmark as a member country, implemented the requirements in 1994. As a result the country upgraded or constructed new systems for all the municipalities with populations above 5000 PE and implemented biological treatment with nitrogen removal to treatment plants designed for 1000 PE or more and limiting total nitrogen discharge to a yearly average limit of 8 mg l⁻¹. Regarding phosphorus, the stated a limit for system above 5000 PE that brought down to an annual average discharge of 1.5 mg l⁻¹. According to DANVA (2007) the country has 1078 wastewater treatment plants (WWTP) aimed at treating water for all the communities of more than 30PE, which can provide up to tertiary treatment. Although the national wastewater production volume is increasing the national trend is that the number of WWTP is decreasing in number by enlarging the capacity and centralizing treatment.

Once the urban areas wastewater discharges were controlled, the Danish Ministry of the Environment aimed at reducing the uncontrolled discharges of insufficiently treated wastewater and in 1997 issued the act 325 of May 1997 on wastewater treatment in rural areas. The Act recommended the treatment of wastewaters either by connecting the isolated dwellings and households to larger treatment systems or by the establishment of decentralized and autonomous wastewater treatment. Additionally to demand of the treatment of the wastewater generated, the act legislated on the level of treatment by defining four treatment classes in the rural areas, which defined specific water quality discharged. The four class classification system, limits the discharge, according to the nature

of the recipient waters and the water quality goal set by the local authorities and specified by regional water plans. The four classes aim at removing at least 95% of the organic matter (class O), removal of organic matter plus nitrifying up to 90% of the ammonia nitrogen (class SO), the removal of organic matter and the removal of 90% phosphorus (class OP) or the combination of demand the removal of 95% organic matter, 90% nitrification and the removal of 90% of the total phosphorous (class SOP).

Concomitantly with issuing the discharge limits, the Ministry of Environment and Energy produced a list of treatment systems for up to 30 PE that could meet the discharge requirements. According to the Ministry's estimates at that time there were around 90000 households that should comply with discharge limits set by the initiative. As a part of the Ministry's list included compact technical systems as well as low technology natural wastewater treatment systems. Concerning the low technology systems the Ministry issued guidelines for infiltration systems (Ministry of Environment and Energy, 2000), horizontal subsurface flow constructed wetlands (Ministry of Environment and Energy No 1, 1999), biological sand filters ((Ministry of Environment and Energy No 3, 1999), evaporative willow systems (Ministry of Environment and Energy No 25, 2003) and vertical flow constructed wetlands (Ministry of Environment and Energy No 52, 2004). Additional to the accepted systems by the Ministry, some municipalities have also authorized the establishment of systems not included among the national guidelines list, but that are already used successfully in neighboring countries in order to evaluate the performance and the removal capacity.

Table 1: Summary of site characteristics

Type	Organic matter	Total P	Nitrification
SOP	95%	90%	90%
SO	95%		90%
OP	90%	90%	
O	90%		

O: Removal of organic material

OP: Removal of organic material and phosphorus

SO: Removal of organic material and ammonia

SOP: Removal of organic material, phosphorus and ammonia

After the promulgation of the act, the municipalities began updating the local water plans and the establishment of decentralized systems to meet the goal and after more than ten years, the national goals have not been met and the municipalities are striving to fulfill the goal. The choice of decentralized systems selected may include infiltration systems, compact

technical systems, and the so-called “natural” wastewater treatment systems, such as constructed wetlands, biological sand filters, evaporative systems (willow systems) and other similar or patented systems. The criteria used for the system selection might include local limitations (can restrict the use infiltration systems), the discharge limits, area requirement for the systems, costs, and the owners wish. Once the owner decides which system fits his needs, the plans and the required paperwork that includes permits, descriptions and drawings are submitted for municipal approval. The establishment of the system cannot begin until all the permits are in place.

Table 2: System’s Class removal capacity

Class	Infilt.	Willow systems	Construct. wtlnd (SF)	Construct. wtlnd (VF)	Bio. sand filter	Techn. systems
SOP	X	X		(X)*	(X)*	X
SO	X	X		X	X	X
OP	X	X		(X)*	(X)*	X
O	X	X	X	X	X	X

*Only if fitted with chemical precipitation P removal units.

The last years as the municipalities finished the local water plans, the number of decentralized systems in Denmark is increasing due to its proven efficiency, low establishment costs and low operation and management requirements. An initial estimate indicates that the country has around 360.000 households with no treatment (Faldager I, 2011). Out of these houses an estimate of 96.000 must improve their wastewater treatment and due to modifications on local water plans additional 14.000 are to be included to the number of systems to be established. In total, around 110.000 Danish households must construct or improve their wastewater treatment capacity to meet the national discharge limits. An estimate of 87,000 of the systems must comply with the SOP class, the most stringent limit, which means that they must remove organic matter, nitrify and remove P.

SYSTEMS CHARACTERISTICS

As mentioned before, the Danish EPA has been involved in the developing and has funded research to evaluate the available systems and produce national guidelines. Since most of the settlements have already centralized and advance wastewater treatment, the guidelines are aimed at treating wastewaters for populations up to 30 PE. The EPA has guidelines for

Infiltration systems, Biological sand filters; willow based evaporative systems and constructed wetlands.

SOIL INFILTRATION

Soil infiltration is the preferred decentralised wastewater solution used in Denmark. Besides complying with the all the discharge classes, soil infiltration systems are economic, relatively easy to build, operate and maintain and does not have a large visual impact. Additionally, there is considerable experience in the design and building of this type of systems and therefore is favoured by owners, constructors.

Before establishing a soil infiltration system in Denmark some information is required, including characterization of the soil, local water table level and location of possible water supply wells. Some of the restrictions include:

- All the systems must be fitted with a certified sedimentation tank.
- An infiltration system cannot be placed closer than 300 m of a water supply well for more than 10 households.
- Additional distance restrictions apply, depending on hydro-geological characteristics of the site.
- An infiltration system cannot be established closer than 5 m from a natural water course.
- The bottom of an infiltration system must be placed 2.5 m and at least 1m above water table.

Additionally, there are some recommendations that include distance to buildings, trees and bushes, step terrain and about complying with the local building regulations.

The guidelines present suggestions relative to the place of establishment to avoid odours, recommendations about operation and maintenance of the system, about the use of pumps and installation of electrical connections and economy. Most of important, the guidelines provide detailed technical drawings regarding design, construction location and characteristics of all the components of the system. As an example some of the plans are presented in Fig. 1 and Fig. 2.

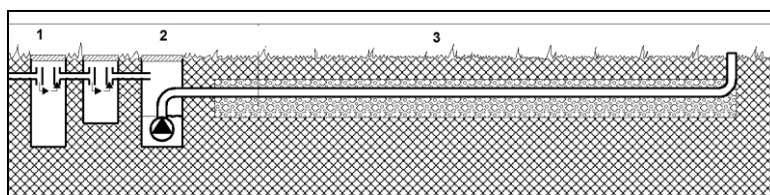


Figure 1: Diagram of a soil infiltration

1) sedimentation tank, 2) pumping well, 3) infiltration field.

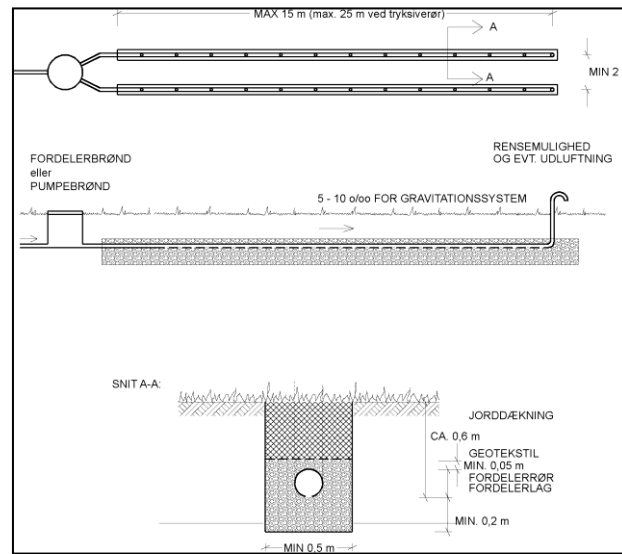


Figure 2: Diagrams showing the installation requirements of distribution pipes.

Soil infiltration systems are not required to be monitored and therefore very little data is available so that the performance can be evaluated.

BIOLOGICAL SAND FILTER

Biological sand filters (BSF) have been in use in Denmark for decades, but official construction guidelines were only published by the Danish EPA in 1999 (Ministry of Environment and Energy, 1999). According to the guidelines BSF fulfils SO class and although the guidelines did not mention P removal or water-recirculation issues to improve total nitrogen removal, if the systems are fitted with P precipitation units some municipalities are accepting the construction of BSF systems as SOP class. The guidelines recommend a system with pre-treatment and followed by a lined bed with a surface area of 5m^2 per PE. The bed must be filled with a 0.8 m depth specific granulometry sand, where the biological processes will remove organic matter and nitrify. The system is fitted with drainage pipe placed on the bottom of the bed and engulfed with a layer of coarse gravel and separated from the filter by a geotextil membrane. The drainage pipes are connected to ventilation pipes that allowed atmospheric air to convect to the bottom of the bed that diffuses to the rest of the bed and permit the presence of aerobic bacteria in the filter. For loading the bed(s), the guidelines suggest a set of pressurised pipes placed on the surface also surrounded by coarse gravel. The loading of the bed is sequential which allows the diffusion of air in the unsaturated filter in between loading pulses. According to the guidelines the BSF should be covered by grass planted soil. The guidelines does not mention any restriction concerning placing in respect to buildings, but instead suggests some practical recommendation,

including a 5 m distance to buildings, water courses, lakes and slopes. According to the guidelines the only restrictions that must be observed are restriction related to national building codes.

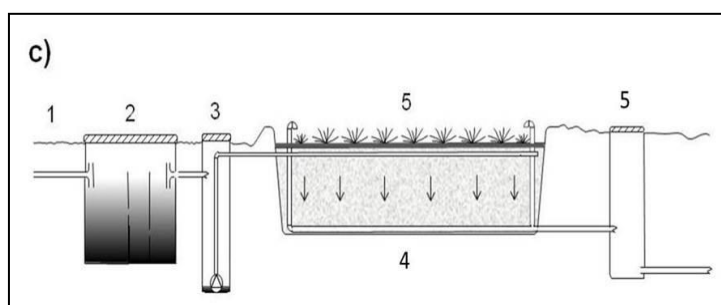


Figure 3: Biological Sand filter 1) inlet, 2) sedimentation tank, 3) pumping well, 4) bed, 5) outlet well. (Figure not to scale).

Performance of BSFs is in general satisfactory and can remove organic matter and nitrify influent waters well above 95%. Experiences with BSFs built with the possibility of recycling and P precipitation units also show that performance can be enhanced to remove total nitrogen and also to comply with P removal demands that can place BSFs in the SOP class.

CONSTRUCTED WETLANDS (CW)

Constructed wetland technology has been around for many years. In Denmark, CW came as a result of the need to comply with discharge demands in small municipalities. The first system was constructed in 1983, and ever since around 700 CWs have been built. The first systems built were subsurface flow constructed wetland and according to Brix et al. (2005) around 170 of them were built, but later on and due to the new requirements in discharge limits (i.e. higher nitrification rates) and the incapacity of the systems to comply with the discharge limits and in some cases not being able to meet the performance promised when constructed, are being phased out and/or being replaced for more effective technology. On the other hand, vertical flow constructed wetland establishment is on the rise and more than 500 of these systems have been established since the guidelines were published in 2005. Fig. 4 shows the evolutions of systems in Denmark.

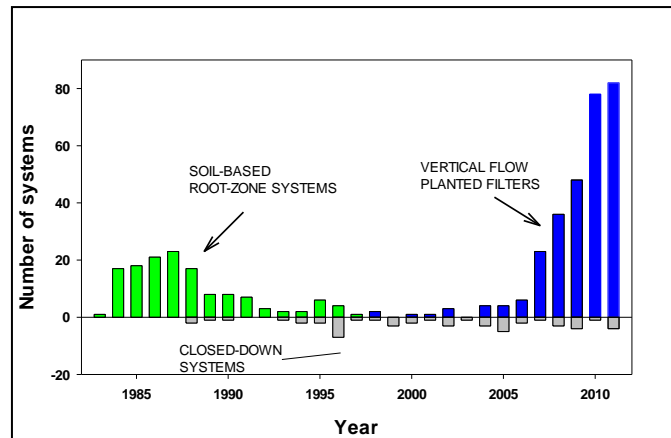


Figure 4: Number of Constructed wetlands established and number of CWs that have been closed down per year in Denmark in the period 1983-2011.

Horizontal flow constructed wetland (HFCW)

The initial design of the Danish constructed wetland systems was copied from the German design and recommendations. These initial systems were soil-based and the treated water was intended to pass through the root-zone of the plants as subsurface horizontal flow (Figure 5) and not exposed to the atmosphere. At the time, the system sellers claimed that as the reeds (*Phragmites australis*) develop, the roots and rhizomes, would increase the soil hydraulic conductivity to ensure subsurface flow of the treated wastewater and even boost the removal capacity of certain pollutants.

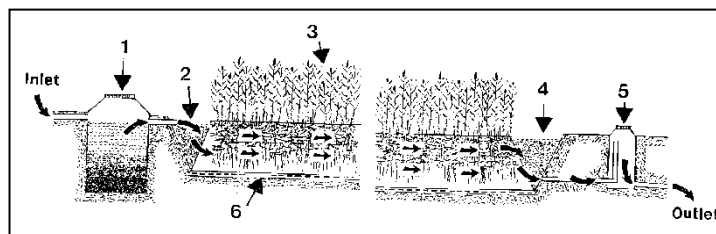


Figure 5: Danish reed bed constructed according to the root-zone concept. 1) Sedimentation tank; 2) Stone-filled inlet distribution trench; 3) Plants (*Phragmites australis*), 4) Stone-filled outlet collection trench; 5) Outlet regulation well. Modified from Brix (1987).

The geometrical design of the first systems was more or less square. However, as it became evident that flooding of the bed was a common problem, the geometrical design was modified and the aspect ratio (length to width ratio) was changed expecting to overcome problem, but the modification didn't prove effective.. The depth of these systems ranged between 0.6 to 1 m and the bottom was plastic lined. Local soil is used as the medium and the beds are planted with *Phragmites australis* using potted seedlings, clumps or rhizomes. The wastewater is pre-treated in a two- or three-chamber settling tank and discharged in

the bed to a stone-filled inlet trench. At the outlet a similar stone-filled trench is used to collect water from the reed bed.

The Danish EPA guidelines regarding HFCW state that they can fulfil class O, meaning that HFCW are accepted as a wastewater treatment solution, in places where only the removal of organic matter is needed. The guidelines have modified the initial designs in from the original components including the distribution systems, the media used for filling the bed and the treated water collected system. The “new” configuration stresses that the distribution and collection of treated waters must be effective and embedded by gravel. The guidelines also presented a granulometric curve that must be observed when selecting the filling media for the beds. Concerning size, the guidelines demand a minimum area of 5m²/PE.

The majority of the systems are constructed to treat domestic sewage from small villages in rural areas and as on-site systems for single households and farms. Often small villages have combined sewerage systems, and the reed beds therefore receive rainwater as well as sewage. Constructed reed beds are also used to treat wastewater from schools and institutions, camping sites, run-off from roads, and effluent from some food-processing factories. A few systems function as a tertiary step after conventional wastewater treatment systems.

Concerning building restrictions the guidelines do not present any special recommendations and only mentions that a 5 m distance to buildings, slopes and water course must be observed. The guidelines also stresses that the national building codes have to be obeyed. The maintenance issues mentioned in the guidelines refers to the sedimentation tank emptying, observing that there is no superficial flow, and that no harvesting must be done.

Although many of the HFCW have been replaced, there are still some that are operational and that are monitored at regular intervals. Performance results show that in spite of variable and often high concentrations of suspended solids and BOD in the inlet water, the concentrations in the effluent are consistently low and typically below the required discharge limits set by the Danish EPA and effluent BOD₅ concentrations are consistently less than 5 mg/L. Removal of nitrogen and phosphorus fluctuates from less than 30% and up to about 60% of inlet concentrations. The removal of NH₄-N is low due to the systems characteristics (saturated beds), and as expected nitrification limits total nitrogen removal.

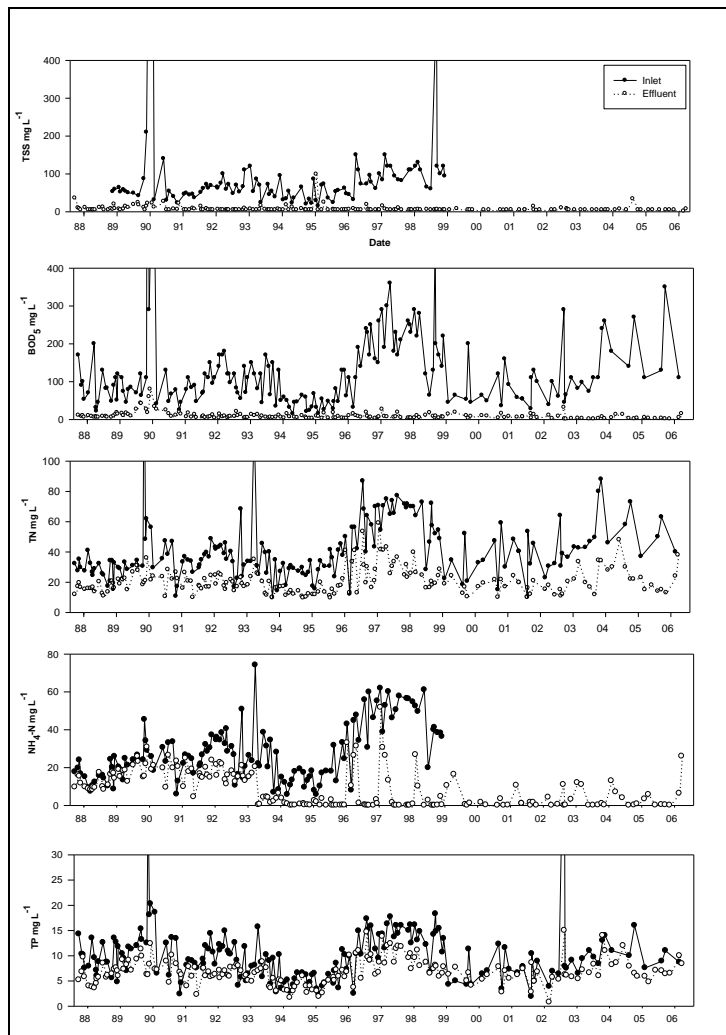


Figure 6: 20 year Performance results of a soil based horizontal flow constructed wetland constructed in Lyngby to treat water from approx. 600 PE.

Fig. 6 presents the results of around 20 year performance of a 4000 m² soil based system built in Lyngby to treat effluent water from 580 PE. The system receives water from a combined sewerage and is fitted with a stormwater amortiguation pond, primary treatment and two beds operating in parallel. The results show that regardless of the inlet variation and concentration of BOD and TSS the systems can consistently produce effluents of BOD and TSS below that can fulfil the discharge limits. The nitrification can be divided in two parts first between the establishment and the second section after 1993. During the first part nitrification capacity of the system was limited so in mid-1993, to improve the nitrification, the municipality decided to enhance the system with a fixed passively aerated system. As expected once the fixed film system was installed nitrification improved but not the total nitrogen removal, which is also expected since there are no biological conditions for denitrification. Phosphorus removal along the period has been very limited and in average is only around 30%. The system is still under operation and some modifications to the system have taken place, including the improvement of the primary treatment and the dismantling of the fixed film nitrification step.

Vertical flow constructed wetland (VFCW)

Now-a-days outlet criteria often include demands for nitrification and removal of phosphorus before discharge, even in rural areas (SOP class). The constructed wetlands systems established since 1998 are all compact vertical flow planted filters (Fig. 7) largely because of their high capacity to remove BOD and to nitrify the wastewater using a relatively small area (Brix and Arias, 2005a; 2005b). Recycling of effluent to the sedimentation tank improves and stabilises the performance of the system and enhances the removal of nitrogen by denitrification (Arias et al., 2005). Phosphorus can be removed by chemical precipitation in the sedimentation tank.

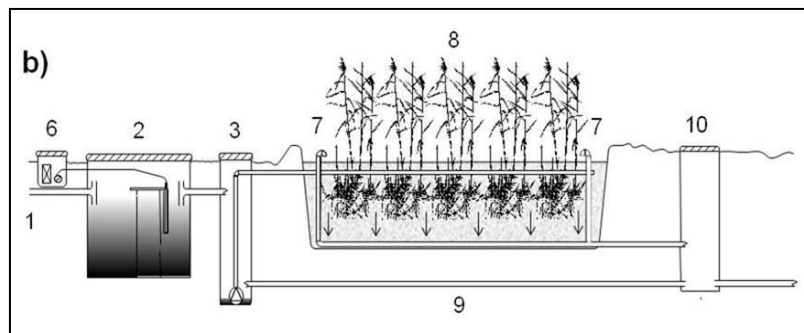


Figure 7: Compact vertical flow constructed wetland system with recirculation and a dosing system in the sedimentation tank for removal of phosphorus 1) inlet, 2) sedimentation tank, 3) pumping well, 4) bed, 6) P removal system 7) aeration pipes 8) plants; 9) recycling, 10) outlet well. (figure not to scale)

The VFCW were constructed according to the Danish construction guidelines (Brix and Arias, 2005a) and consist of one meter deep sand-filled (specific sand granulometry) planted beds (with *Phragmites australis*) with a surface area of around 3m²/PE. The beds are lined. The effluent from the sedimentation tank is pulse loaded on top of the bed and distributed homogeneously by means of pressurised pipes. The water trickles down through the unsaturated bed and is collected by drainage pipes at the bottom and evacuated from the bed. The guidelines prescribe the distribution system must be insulated with a layer of at least 0.15 m tree shreds or other material to avoid freezing during winter. The guidelines also recommend the recycling of treated effluent back to either the sedimentation tank or the pumping well improve performance in regards of total nitrogen removal and treatment of organic matter. The guidelines also required the use of specific granulometry for the sand used as filling media. Concerning place of establishments the guidelines do not set restrictions and only recommends following the national building codes. For maintenance the guidelines only mention the annual emptying of the sedimentation tank, spooling of distribution pipes if needed and the manual removal of possible invasive plant species on the bed. It also mentions that no harvesting should be done.

The Danish EPA guidelines approve the system as an SO class meaning that are effective removing organic matter and nitrifying. Further research and practice has shown that the

system can comply with SOP class when fitted with phosphorus chemical precipitation units and many municipalities in the country accept VFCW as and SOP solution.

Performance of VFCW has been monitored as most of the municipalities require follow up and sample analyses, especially the one with SOP class discharge limit. Removal of BOD and TSS from data collected of around 90 systems show that removal of both is higher than 95% regardless of inflow concentration season and system size. Nitrification in the same systems is consistently higher than 95% (figure 8). Total nitrogen removal is dependent on the recycling rates while P removal varies depending on the availability of P chemical precipitation units. The systems fitted with P removal units can comply with SOP class limits.

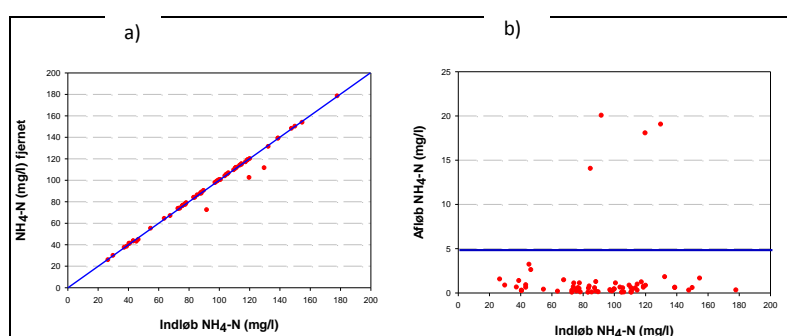


Figure 8: Ammonia nitrogen removal in ca 90 VFCW in Denmark a) inlet concentrations vs removal percentage, b) Inlet concentration vs. effluent concentration.

Willow based evaporative systems

Willow based evaporative systems is the result of research in energy plants that resulted in the establishment of sites with high evaporative capacity and the applicability in the wastewater management technology. During the last year around 2000 of these systems have been design and built across the country. The technology is spreading and now sites can be found in most of northern Europe and new research is being done in other parts of the world using different plants.

The Danish Guidelines differentiates two types of facilities, on that allows infiltration and the second the bed is lined and therefore no infiltration is possible. In general the willow wastewater cleaning facilities generally consist of c. 1.5 m deep high-density polyethylene-lined basins filled with soil and planted with clones of willow (*Salix viminalis* L.). The surface area of the systems depends on the amount and quality of the sewage to be treated and the local annual rainfall. For a single household in Denmark the area needed typically is between 120 and 300 m². The annual precipitation at the site of construction is an important dimensioning parameter. Settled sewage is dispersed underground into the bed under pressure. The stems of the willows are harvested on a regular basis to stimulate the growth of the willows and to remove some nutrients and heavy metals.

The main characteristics of the willow systems are:

- For a single household (5 PE) system, the sewage has to be pre-treated in a 2- or 3-chamber sedimentation tank with a minimum volume of 2 m³ before discharge into the willow system
- Closed willow systems are generally constructed with a width of 8 m, a depth of minimum 1.5 m, and with 45 degree slopes on the sides
- The bed is enclosed by a water tight membrane and wastewater is distributed underground within the system by a level controlled pump.
- A drainage pipe is placed in the bottom of the bed. The pipe can be used to empty water from the bed if salt accumulates after some years.
- One half or third of the willows are harvested every year to keep the willows in a young and healthy state with high transpiration rates

Willow systems with soil infiltration are dimensioned in the same way as closed willow systems. The willows will evaporate all wastewater during the growing season, but during winter some wastewater will infiltrate into the soil.

Wastewater must be properly pre-treated in e.g. a sedimentation tank before discharge to the system. The wastewater is distributed in the bed by a pump and a pressurised distribution pipe placed in the middle of the system. The distribution pipe is placed in a layer of 16-32 mm gravel or some other material with a high porosity. The distant end of the distribution pipe is placed in an inspection and cleaning well. The water level in the soil can be monitored in the well, and if it becomes necessary to pump high salinity water out of the system, this can be done here.

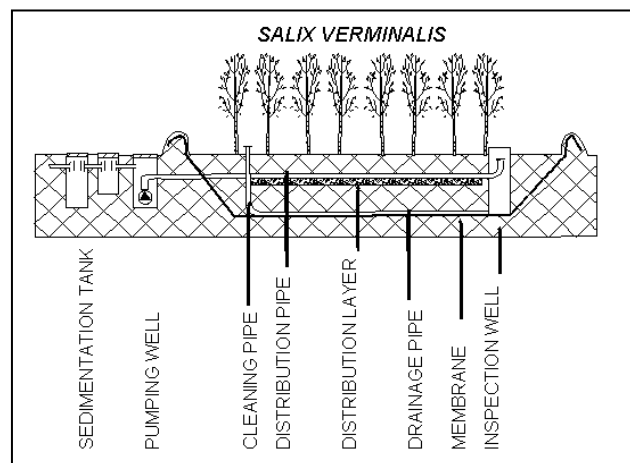


Figure 9: Sketch of a willow system with no outflow (evaporative system).

A drainage pipe is placed in the bottom of the bed used, if necessary to pump water out of the system in case of clogging or need for washing accumulated salts out of the system. The bed is filled with the original soil removed during the excavation of the site. Around the bed a 0.3 m high dike is built up to avoid water from the surroundings to enter the willow bed, and to allow

the accumulation of water on the surface during the winter. A standard system will have width of 8 m, a depth of 1.5 m, and the length will depend on the needed area.

The performance of the systems has shown that the first couple of years there is accumulation of water in the surface, but as time goes by the system, if properly design, can cope with the water without producing excess. One of the concerns for these systems has been the accumulation of slats in the beds that could harms the plants, but up to now and with the older system in operation for more than 15 years there are no reports of failed systems due to salinity. Further research is being done in this topic.

Other systems

There are a couple of other systems available although no national guidelines have been produced. These systems might include technology applied in the neighbouring countries that have been successfully test and include systems such as filtralite-P® filterbeds and Naturen® systems. These systems rely on artificial materials such as expanded light weight clay aggregate as filling media for the filters. In general these systems consist of a pre-treatment, similar to the previously described systems a fibre glass “container” to house the media sometimes called biofilter where the media is placed where pressurized water is distributed on the surface and trickles through the media and is collected and evacuated at the bottom of the filter. Some of these systems use specific binding material to remove phosphorus (Fig. 10) so they can meet SOP class so an extra section is built to house the P removal section that according to the manufacturer is around 10m³ for 10 years need and when the material has to be replaced. Systems that only have to meet O or SO class don not need to use special media and therefore are more compact and less expensive (Fig. 11).

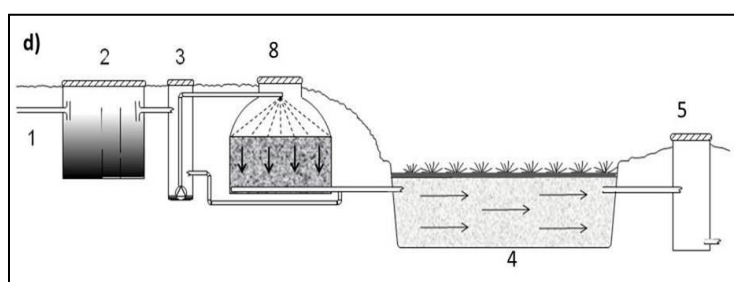


Figure 10: Filtralite P filter 1) inlet, 2) sedimentation tank, 3) pumping well, 4) Filtralite-P® based bed, 5) outlet well 8) LWA biofilter; 9) recycling. (*figure not to scale*)

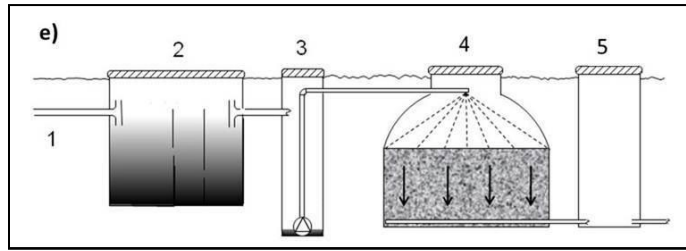


Figure 11: Naturen® system 1) inlet, 2) sedimentation tank, 3) pumping well, 4) LWA filter, 5) outlet well. (*figure not to scale*).

Performance for these systems meets the limits but since they are relatively new there is no long time research that can predict the performance of the specific media.

FINAL REMARKS

All the solutions presented if properly built and managed can meet the discharge limits to which they are designed. Small modifications such as recycling or the addition of po removal units can even increase the performance and meet other class limits. According to the data gather (not presented here) systems such as BSF and VFCW can be built even smaller and shallower maintaining good performance and consequently reducing building costs and footprint.

The calculated costs for building these systems show that for the final user the implementation of a decentralized solution is around 20% lower than connecting to a centralized wastewater treatment facility and additionally the costs and taxes after operation does make the solution competitive.

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THE TREATMENT OF ON-SITE WASTEWATER USING WILLOW BED EVAPOTRANSPIRATION SYSTEMS IN IRELAND

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ABSTRACT

Willow bed evapotranspiration systems can treat on-site wastewater effluent with the advantage that, if sized correctly, they produce no discharge either to ground or to surface water. Willow beds are a viable alternative to conventional treatment methods, when for example, in situ soil is of too low a permeability to allow for treatment / disposal via percolation or when a discharge of treated/untreated effluent to a watercourse is not permitted. 11 willow bed systems have been constructed as pilot trials at houses around County Wexford to treat the domestic wastewater produced. The systems have been built with the idea of evaluating different parameters with respect to overall performance including effluent type (septic tank or secondary treated effluent), willow varieties and aspect ratios of the basins. The effluent flow into the basins, water level, rainfall and evapotranspiration have all been monitored on each site to assess the effectiveness of the system. The overall aim is to produce guidelines for the design of such on-site treatment systems. In parallel to the full scale systems, mesocosm experiments have been set up in Dublin. These experiments have investigated the evapotranspiration rate of four different willow varieties against reference evapotranspiration while also monitoring the effects of the application of three different effluent types onto each variety. Results have indicated that the application of both primary and secondary effluent has significantly increased the evapotranspiration rates for all the varieties of willow, although there has been no noticeable difference in evapotranspiration between varieties.

Keywords: on-site wastewater, willow beds, zero discharge, evapotranspiration, crop coefficient

INTRODUCTION

Ireland has over one third of its population using on-site wastewater treatment systems (CSO, 2007), mostly consisting of septic tanks discharging effluent into a subsoil percolation area. However, in some regions the subsoil is of too low a permeability to allow for treatment by percolation, so an alternative is required. County Wexford, which is located in the south east of Ireland is one such region where there are extensive areas of such clayey subsoil. In the last decade, during the construction boom, the local authority in Co. Wexford granted hundreds of planning permissions for single houses, for which on-site treatment consisted of a septic tank followed effectively by the discharge of effluent to a nearby stream. Many of these streams and rivers throughout the county are being shown to suffer from poor water quality. Hence, in order to improve this situation and find a more appropriate solution to the treatment and disposal of on-site effluent in such areas the council implemented a pilot trial to investigate the use of zero discharge willow bed systems at a number of different sites.

The design and monitoring of the willow beds was carried out by a research team from Trinity College Dublin. The four year trial is being carried out on eleven different full-scale systems at eight different sites throughout the county. The willow bed design is similar to the systems outlined by Gregersen and Brix (2001), which have proved successful in Denmark, but have been based upon realistic Irish rainfall, evaporation and on-site effluent production statistics. The system consists of a lined basin refilled with excavated soil and then planted with willow cuttings. The primary objective of the project is to determine the optimum size required to treat a known volume of effluent and corresponding volume of rainfall at any given location. Other parameters including aspect ratio, effluent types and willow variety are also being compared. The latter two parameters have also been investigated in more detail in a series of mesocosm experiments that compare four different willow varieties against the application of three different effluent qualities.

METHODS

Full Scale Systems

Background

Eleven full scale systems have been constructed in Co. Wexford to treat wastewater effluent from single house dwellings. The sites are distributed throughout the county in areas which are known to have marl (i.e. pure clay) subsoils. The systems were designed with deliberate variations between key parameters (effluent type, willow species, plan area, aspect ratio and effluent distribution) as seen on Table 1, in order to provide comparisons and determine sensitivity to these in terms of overall performance.

Construction

On each site the basin was dug out to a depth of 1.8 m. The slopes of the sides of the basin were kept as straight as possible in order to maximize the volume of the system. The layer of top soil was kept separate from the subsoil. Any sharp stones were then removed from the basin to help aid against punctures. Following this, a 1.5 mm, 180 g per m² non-woven geotextile was installed. A 0.5 mm Low Density Polyethylene (LDPE) impermeable membrane was then laid on top of the geotextile layer, with the joins being welded together to ensure a watertight seal. An excess of 0.38 m along the top edge of the basin was required, in order to maintain water-tightness along the berm. A second geotextile layer was then laid on top of the LDPE to aid against punctures from willow tree roots and stones. An inspection/pump well was installed at one end of each willow bed. The 300 mm diameter plastic pipe was placed, standing vertically upon a concrete tile such that it protruded over the top surface of the willow bed. Slots were cut in the side of the pipe, which was then wrapped in geotextile to allow for ingress of water while preventing clay blocking the slots.

The effluent is distributed throughout the systems via a number of rigid 110 mm dia. percolation pipes, which are laid in a 0.3 m layer of 20 mm washed gravel at the bottom of the basin. The pipes are laid at 3.0 m centres as shown in Fig. 1. The effluent from the septic tanks or package treatment systems is either gravity fed or is pumped into the head of the pipe network from where it flows by gravity.

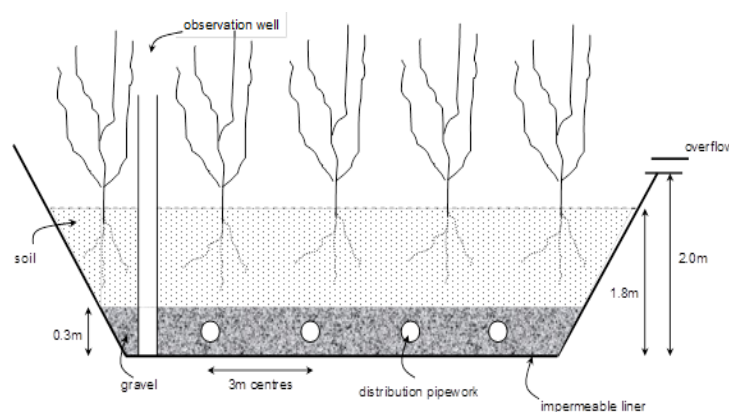


Figure 4: Willow bed schematic

The soil that had been excavated for the basin was then backfilled into the lined hole, starting with the subsoil and finished with the layer of topsoil. A 0.3 m high berm was constructed around the edge at the top of the bed incorporating the geotextile and impermeable membrane to act as an overflow protection in the instance of flooding, due to heavy rainfall in the first year for example, or in winter time.

Willow Planting

The Danish EPA Guidelines (Miljøstyrelsen, 2003) stated that the planting of multiple different varieties is imperative in order to resist disease and parasites. Hence, from a pool of 8 willow varieties, each treatment system was planted with 3 different varieties of willow (see Table 1) in a predetermined pattern. The cuttings were planted at a density of 3 per metre square.

Monitoring

The flow into each willow bed was monitored continuously using either tipping bucket flow recorders for the gravity fed systems or via a level meter situated in the sump for the pumped sites (Thalimedes, OTT Hydrometry). The water level in each basin was monitored continuously using a pressure gauge sensor (Orpheus, OTT Hydrometry) situated in the inspection sump. Chemical and microbiological analysis was carried out on the effluent and water in the inspection sump on a monthly basis for each treatment system. Weather stations (Casella) were erected at each site to continuously monitor and collect rainfall data as well as other meteorological parameters (wind speed, air temperature, humidity, solar radiation) from which the reference evapotranspiration (ET_o) could be determined.

Table 3: Summary of design parameters being compared between 10 full-scale systems

	Constructed	Effluent ^a	Area (m ²)	Aspect ratio (length: width)	Distribution	Willow species
1	May 2009	SE	570	1.6 : 1	pumped	<i>Bjorn, Tora, Jorr</i>
2	May 2009	SE	296	4.6 : 1	pumped	<i>Bjorn, Tora, Jorr</i>
3	April 2010	STE	420	2.9:1	pumped	<i>Tora,,Thorhild, Tordis</i>
4	May 2010	STE	464	1.8 : 1	gravity	<i>Tora, Thorhild, Olof</i>
5	April 2010	STE	24	2.7 : 1	gravity	Native Irish species
6	July 2010	SE	900	1.4 : 1	gravity	<i>Tora, Tordis, Olof</i>
7	July 2010	SE	900	9.0 : 1	gravity	<i>Tora, Tordis, Olof</i>
8	Sept. 2010	SE	340	3.4 : 1	gravity	<i>Tora, Tordis, Olof</i>
9	April 2011	STE	520	5.2 : 1	pumped	<i>Tordis, Sven, Inger</i>
10	May 2011	STE	560	8.8 : 1	gravity	<i>Tora, Tordis, Inger</i>
11	July 2012	SE	1000	4.4:1	pumped	<i>not yet planted</i>

Water Balance

The water budget for each system was carried out throughout the monitoring period on the basis of influent flows combined with rainfall and evapotranspiration rates. ET_o was calculated from the meteorological parameters using the Penman-Monteith equation (FAO, 1998). The product of the water level in the sump and the void ratio of the soil in the basin provided the volume of water in the system. The actual evapotranspiration (ET_{willow}) from the willow bed for a given time period was then calculated using a water balance equation. A crop factor can then be determined by comparing ET_{willow} to ET_o .

Mesocosm Experiments

Cylindrical containers of height 1000 mm and diameter 540 mm were placed at an open site at the Trinity College Botanic Gardens in Dublin, and filled with layers of gravel (75 mm), sand (440 mm) and topsoil (460 mm). A 30 mm inspection plastic inspection pipe was also inserted into each container to allow for measurement of the water level. Each pipe had 10 mm diameter holes drilled at the bottom to allow for the ingress of water from the gravel layer (see Fig. 2).

Table 4: Classification of willow varieties

Willow Variety	Clone
Tordis	((<i>Salix schwerinii</i> x <i>S. viminalis</i>) x <i>S. viminalis</i> .)
Sven	(<i>Salix viminalis</i> x <i>S. schwerinii</i>)
Inger	(<i>Salix triandra</i> x <i>S. viminalis</i>)
Torhild	((<i>Salix schwerinii</i> x <i>S. viminalis</i>) x <i>S. viminalis</i>)

Four willow varieties were then planted; Tordis, Sven, Inger and Torhild, which are all subspecies of *Salix viminalis* (see Table 2), one plant per container.

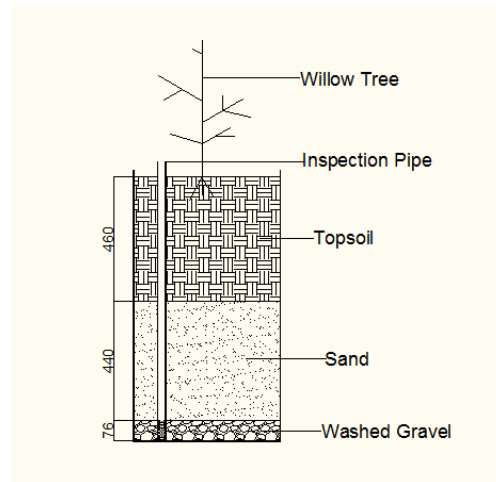


Figure 5: Cross sectional detail of mesocosm

Water sampling and analysis

A plant from each variety was then given an application of primary (septic tank) effluent, secondary treated effluent or rain water (as a control). The dosage was 2.2 litres per week which equated to the approximate areal hydraulic loading rate that one willow tree would receive in a full-scale willow bed. The recipes for the synthetic effluent were based upon recipes by Peebles and Mancl (1998) but then adjusted so the effluent reflected the concentrations more usually found in Irish septic tank (i.e. primary treated) and secondary treated on-site effluent (see Curneen and Gill, 2012). A water balance of the systems was carried out, in a similar manner to the full scale systems. However, for the mesocosm experiments, the level was recorded manually at regular intervals instead of automatically (as with the full scale systems). The actual evapotranspiration for that time interval was then determined using the water balance equation, and respective crop factors calculated.

RESULTS

Full Scale Systems

Climate variation

The climatic variation between 5 of the total 8 sites has been compared over a 12 month period. System 5 was omitted due to some data missing during the monitoring period and Systems 9, 10 and 11 omitted as data was only started to be collected in Spring 2012. The reference evapotranspiration varied between 420 and 480 mm annually over the 5 sites (see Fig.3) and there was no obvious pattern between the evapotranspiration rate and geographical location within the county. The same applies to the wind speed measurements on the 5 sites, with the mean wind speeds between sites varying from 1.37 m/s to 2.68 m/s.

However, for the rainfall there does appear to be a noticeable difference between north Wexford (WB 3, 6/7) and south Wexford (WB 1/2, 4, 8), see as shown on Fig.3, with higher rainfall recorded in the north of the county.

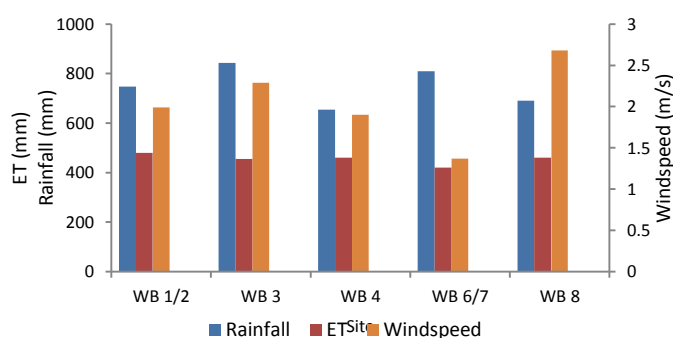


Figure 6: Meteorological data for 5 sites across Co. Wexford (March 2011-March 2012)

System performance

Monitoring of the full scale systems is currently ongoing. The willow systems which have been established for 2 or more growing seasons have so far performed as intended, operating with zero discharge whilst receiving all the household effluent in addition to rainfall. For example, Fig. 4 shows the monitoring results of daily water level within System 1 and the daily rainfall. It is interesting to note the effect that large rain events have on the water level within the basin which can be clearly seen. These jumps are being analysed as one technique to determine the relevant net void ratio within the systems. The difference in plan area with respect to performance can also be seen when comparing System 1 against System 2. Both systems are situated on the same site; however, it is evident from Figs. 4 and 5 that the larger willow bed system (System 1) could cope more effectively with the hydraulic loading, showing much lower levels for longer towards the end of the growing season. It was also better able to assimilate the larger rainfall events in winter.

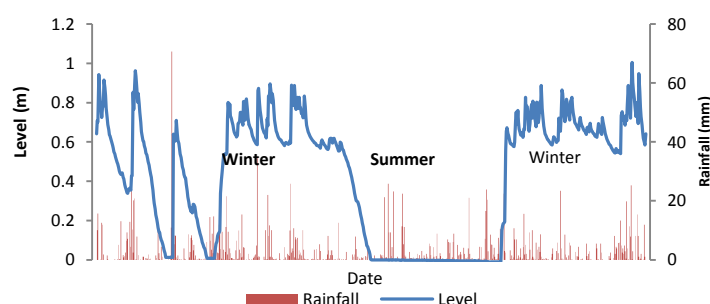


Figure 7: Levels and Rainfall data for System 1, Rathmakee, Co. Wexford

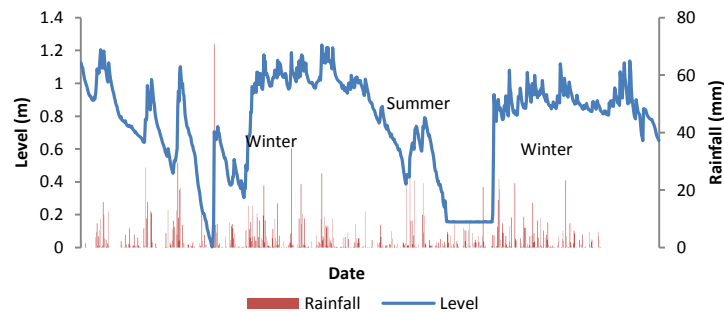


Figure 8: Levels and Rainfall data for Willow System 2, Rathmakee, Co. Wexford

System 4, which was located in the north of the county and built a year later, has also performed as intended. Although the water levels inside the system did not fully drop to zero in the first summer due to the willow trees being in the first growing season and hence relatively small (see Fig. 6), the system was able to contain all the receiving rainfall and effluent over the winter, hence maintaining zero discharge. As with Systems 1 and 2 the effect of large rain events on the water level can again be seen.

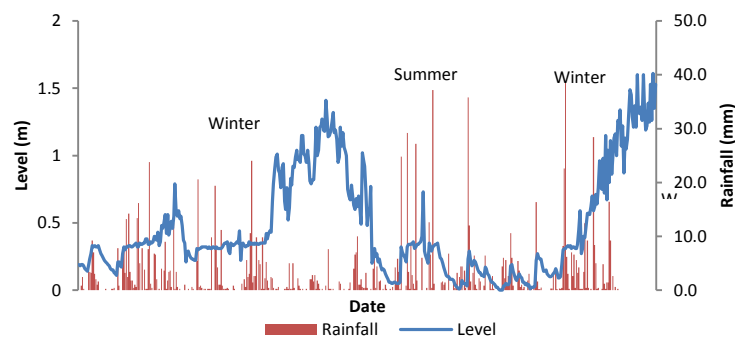


Figure 9: Levels and Rainfall for System 4 Killenagh, Co. Wexford

Soil core samples from the willow systems are being analysed as an alternative method to determine the relevant net void ratio of the backfilled soil in each basin. Once this is determined for each system, the volume of water at any instant can be calculated from the continuous level monitoring at the inspection well. From this, a water balance can be applied to the system to determine the actual evapotranspiration across any given time period.

Daily effluent quantities discharged by the single houses into the three most mature operational systems can be seen in Table 3. It should be noted that these quantities, although significantly lower than the EPA Code of Practice design value of 150 litres per capita per day (Lcd), compare favourably with other recent research on on-site wastewater production in Ireland as, determined by Gill *et al.*, 2009.

Table 5: Selected effluent quantities

System	Flow (L/d)	Person Equivalent (Lcd)
2	281	70.3
3	230	115.0
4	452	113.0

Table 4 shows some examples of the effluent quality being discharged into the systems as well as the water quality at the inspection wells. This shows that significant reductions in organics, nutrients (nitrogen and phosphorus) and indicator bacteria (*E. coli*) had occurred by the water quality taken from the inspection sumps with respect to the STE being discharged into each system. Some of this reduction will be due to rainfall dilution but a significant amount is also due to natural attenuation as well as uptake by the trees for growth. However, it is noticeable that there was a much more limited change in chloride concentrations between the influent and inspection sump and on some sites a slight increase occurred. It is expected that the background chloride concentrations will increase over the years as chloride should not be significantly picked up by the willows nor naturally attenuated and so these levels will need to be monitored in order to determine what levels might start to inhibit tree growth.

Table 6: On-site effluent (STE) and willow bed inspection well (WB) water quality parameters

	Sample Type	COD (mg/l)	Total N (mg/l)	Ortho-P (mg/l)	Chloride (mg/l)	<i>E. coli</i> (MPN/100ml)
2	STE	498	76.7	6.24	67.0	4.61x10 ⁶
	WB	11	4.7	0.16	38.4	6.49x10 ²
3	STE	477	78.0	5.99	37.8	6.49 x10 ⁶
	WB	29	6.7	0.02	25.5	9.8 x10 ²
4	STE	383	58.1	4.15	47.7	6.5 x10 ⁵
	WB	55	6.3	0.64	58.7	1.19x10 ²
9	STE	396	54.5	6.47	27.6	2.9 x10 ⁶
	WB	30	3.7	0.06	34.7	4.23x10 ²

Finally, some early lessons have been learned at this stage of the trials with respect to the construction of these systems. It seems that weeding is imperative in the first growing season in order to remove the competition for light, space and nutrients. In fact, even more effective was the laying of a weed proof barrier on top of the backfilled soil. It is also important to plant the willow cuttings as early as possible in the growing season (ideally in March), in order to give them the benefit of a full first season's growth before effluent is introduced to the system as this will enhance their chances of surviving the initial winter period. This was particularly apparent at one site (System 4) whereby the immediate inflow

of effluent had a detrimental effect on the willow cuttings resulting in the majority dying off, which effectively set the performance of the system back 2 years.

Mesocosm Experiments

Climate

The average monthly weather data along with the monthly ET_0 at the site location for the 2010 and 2011 growing seasons are shown in Fig. 7. As can be seen, the climatic data for the two growing seasons were quite similar with a few notable exceptions. The average temperature during June and July was noticeably higher for 2010 and the evapotranspiration rate for August 2011 was comparatively low due to unseasonable weather. During the 2011 growing season time frame (June to Sept) the total rainfall was 372 mm. The average temperature for the period was 13.4°C and the reference evapotranspiration at the site was 313 mm. It can be seen from Fig. 7 that the air temperature had an almost direct correlation with the reference evapotranspiration with the exception of August as a result of unseasonal weather. For comparison, the total rainfall for the 2010 growing season was 302 mm with an average temperature of 15.6°C and a reference evapotranspiration of 331 mm.

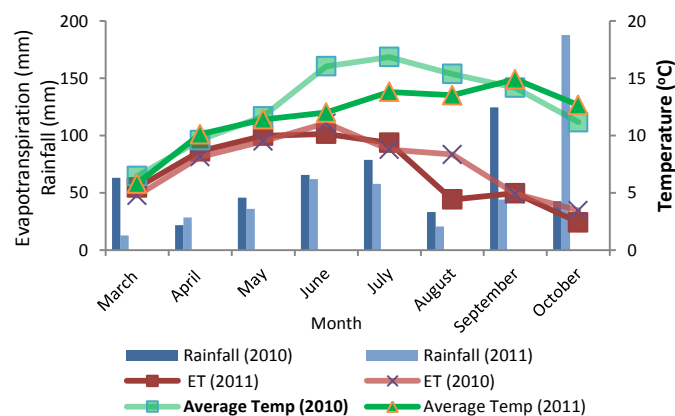


Figure 10: Monthly meteorological data for 2010 and 2011 growing seasons

Evapotranspiration and crop coefficient values for the two growing seasons can be seen in Figs. 8 and 9. The pattern for the 2010 and 2011 seasons is quite similar, although the actual evapotranspiration values are lower for the 2011 season. There was no discernible difference in the evapotranspiration between the four willow varieties, with the exception of the Torhild being applied with secondary treated effluent. This was due to poor growth and development issues which persisted from the start with the plant.

The benefit to evapotranspiration by adding synthetic effluent (both primary and secondary) is apparent for the 2010 and 2011 seasons (Figs. 8 and 9). Over the 2010 growing season it was observed that the trees applied with primary treated effluent had slightly higher evapotranspiration (1165, 1129, 1030 and 1087 mm) compared to the trees receiving secondary treatment (1064, 995, 962 and 410 mm), and both of these well outperformed the trees receiving neither primary nor secondary effluent (544, 867, 579 and 656 mm). The crop coefficients followed the same pattern. The 2011 growing season resulted in much lower evapotranspiration values but a slight increase in crop coefficient values. Again, the trees under primary treatment (734, 667, 590 and 800 mm) had slightly higher ET_{willow} values compared to the trees under secondary treatment (665, 602, 517 and 286 mm) and both these outperformed the trees receiving no treatment (445, 476, 362 and 453 mm), with the exception of the Torhild variety receiving secondary treated effluent.

At the time of writing (August 2012) the trees during the current season appear to have produced improved evapotranspiration rates with some of the trees having already surpassed 2011 results, with 8 weeks (approx.) of the growing season still remaining.

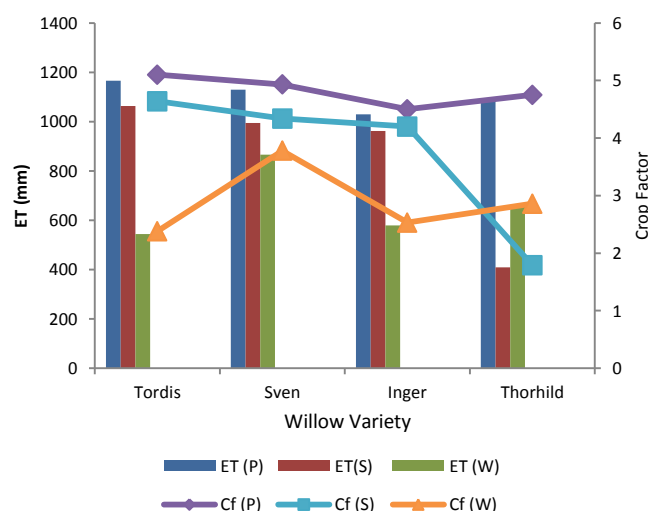


Figure 11: Evapotranspiration rates and crop coefficients for 2010 season

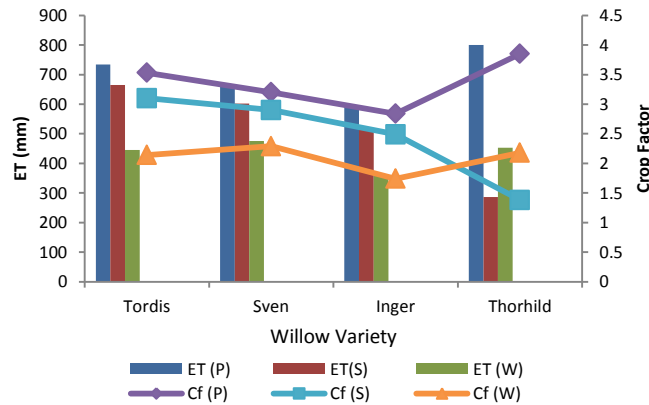


Figure 12: Evapotranspiration rates and crop coefficients for 2011 season

DISCUSSION AND CONCLUSIONS

Full Scale Systems

The full scale systems are all now constructed and are being monitored. The sites in which the willows have been growing for two or more seasons are showing good potential in terms of operating as sustainable solutions for the treatment and disposal of on-site effluent in areas of low permeability subsoils as shown in Figs.4, 5, 6. Systems 1 and 2 in particular have produced positive results, with all effluent and rainwater being disposed of via evapotranspiration in the 2011 season.

Several lessons have been learnt during the trials on full-scale system to date. There is evidence that the initial growth of willows was hampered by weed growth, primarily on Systems 6 and 7 and it has been noted that willows need to be growing for a full season before effluent is added to the system. In addition, the background chloride levels in the willow basins will need to be monitored for several years to assess whether it eventually reaches a level to inhibit tree growth.

Mesocosm trials

The mesocosm trials have proved very interesting to compare directly the evapotranspiration from different willow varieties against strength of effluent applied. The increase on evapotranspiration brought about by the addition of effluent matches the findings from other studies, e.g. Guidi *et al.* (2008), Gregersen and Brix (2001) and Martin (2007). The results for the 2010 growing season also compare favourably with results obtained in Italy by Guidi *et al.* (2008) for willows in their first growing season, especially in the case of the willows treated with effluent or fertilisation. However, the total evapotranspiration across the second season for the willows in Ireland was more muted than the Italian based study which reported a 50% increase on the first season. The crop factors for the second season's growth are more comparable with 3.18 being reported in

the Italian study compared to 2.9 here in Ireland. The crop coefficients in these mesocosm trials also compare favourably to the crop coefficients determined by Bialowiec *et al.* (2007) used to treat landfill leachate in Poland.

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THE ATTENUATION CAPACITY OF CONSTRUCTED WETLANDS TO TREAT DOMESTIC WASTEWATER IN IRELAND

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ABSTRACT

An Irish EPA-funded project was recently carried out to investigate the removal efficiency of constructed wetlands in treating chemical and microbial wastewater contaminants found in domestic wastewater effluent. This paper provides a summary of the key findings, which have contributed to specific design guidelines included in the recently published EPA Code of Practice (2009). Two horizontal subsurface flow reed beds were constructed on separate sites in Ireland - one to provide secondary treatment and the other to provide tertiary treatment for single house domestic effluent. Nitrogen removal was found to be poor across both reed beds, with only 29% removal of TN across the secondary treatment bed and 41% removal across the tertiary treatment bed, with little distinctive seasonal change. Removal of Total P in the beds averaged 45% and 28% respectively. In addition, plant uptake of P was poor with any subsequent harvesting having negligible impact on controlling phosphorus accumulation. Significant removal of organic matter, in the form of COD, and E.coli was achieved in the secondary treatment reed bed, in line with other packaged wastewater systems. Reduction in hydraulic load due to evapotranspiration is, however, insignificant in a temperate Irish climate.

Keywords: On-site wastewater; horizontal subsurface flow reed bed; removal efficiency; nitrogen; phosphorus; *E.coli*; bacteriophages

INTRODUCTION

Constructed wetland systems (CWS) are regarded as a simple, low energy, low cost, aesthetically pleasing and, most notably, sustainable solution to many wastewater treatment applications such as urban and agricultural runoff, and on-site domestic effluent disposal. There has been a significant increase in the use of CWS for on-site wastewater treatment purposes in Ireland over the last decade in particular but there still remains a

knowledge gap as to how efficiently they perform under certain climatic conditions and how reliable and viable they are as both a secondary and tertiary treatment option.

The most common type CWS in use in Northern Europe is the horizontal subsurface flow (SSF) reed bed which has shown consistently good removal patterns in BOD, suspended solids (SS) and pathogenic organisms at certain organic loading rates; In many studies the influent concentrations of BOD are generally < 150mg/L (Thurston *et al.*, 2001; Vymazal, 2002; García *et al.*, 2003) which is rather diluted compared to the septic tank effluent (STE) levels recorded in this study and previous others (O'Súilleabháin, 2004; Gill *et al.*, 2007) in Ireland which range between 150 and 400 mg/L. Studies carried out in Mediterranean and sub-tropical climates have also shown that temporal and seasonal effects play a significant role in organic removal in horizontal SSF reed beds (Solano *et al.*, 2003; Headley and Davison, 1999).

Nutrient removal in such systems has proved to be variable due to the complex interaction of a range of parameters causing changes in nutrient supply, uptake or release of chemical substances and biological activities of micro organisms and plants (Kadlec, 1999). Total nitrogen (TN) removal rates reported for these systems for example, have ranged from high removals of over 90% (Søvik and Mørkved, 2008) to removals as low as 11% (Kuschik *et al.*, 2003). Equally, SSF reed beds generally do not remove high amounts of P from wastewater. A summary of the performance efficiency of such wetlands in a range of European countries showed mean total phosphorus (TP) removals of between 26.7 and 61.4% (Vymazal, 2002), most probably as a result of the different media types used and the complex dynamic interactions occurring internally in wetland systems.

Like subsoil treatment, SSF reed bed systems are known to offer a suitable combination of physical, chemical and biological factors for the removal of pathogenic microorganisms. Processes such as sedimentation, filtration, sorption to organic matter (OM), exposure to natural biocides excreted from plants, retention in biofilms and natural die-off are key to the fate of organisms such as bacteria and viruses during their passage through wetlands. Limited attention has been paid to the efficacy of these reed beds with respect to pathogen removal but of the few studies carried out targeting indicator organisms such as faecal coliforms, bacteriophages and other viral indicator organisms (Thurston *et al.*, 2001, Vega *et al.*, 2003) removal efficiencies of 90 – 99.95% (i.e. 1 log-unit – 3.4 log-unit) have been recorded. The fate of microbial pathogens in these systems is significantly influenced, however, by various factors such as climatic conditions, hydraulic retention time (HRT) and the design specifics of the wetland itself (García *et al.*, 2003).

In this field study, two horizontal SSF reed beds were constructed on-site, the first to provide secondary treatment and the second to provide tertiary treatment of the incoming domestic wastewater. A comprehensive analysis, including a detailed water balance study,

was carried out to investigate the efficiency of each of the beds in removing a range of chemical and microbial contaminants.

METHODS

Wetland design and construction

Two horizontal SSF reed beds were designed based on a first-order BOD₅ model (Kadlec and Knight, 1996) and constructed on-site for monitoring over their first 26 months of operation. The first reed bed (RB1), provided secondary treatment of domestic STE, while the second (RB2) operated as a tertiary treatment system following pre-treatment in a rotating biological contactor (RBC) –Fig.1.

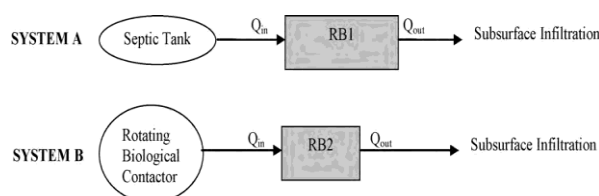


Figure 1: Process flow diagram of systems including RB1 and RB2

Their dimensions (see Table 1) were based on the documented average Irish daily wastewater generation per capita at the time of 180 L d⁻¹ (EPA, 2000) and an actual serving population of 3 people for RB1 and 2 for RB2. Note the daily wastewater figure has since been reduced to a more realistic value of 150 L p⁻¹ d⁻¹ in the new EPA Code of Practice (EPA, 2009).

Table 1: RB1 and RB2 design characteristics.

Reed Bed ID	Influent Type	Plant Species	Dimensions l x b x h (m)	Area (m ²)
RB1	STE	<i>Phragmites australis</i>	5.8 x 2.6 x 0.6	15
RB2	SE	<i>Iris & Typha</i>	4.0 x 1.0 x 0.6	4

Both beds were sealed with a single sheet of impervious butyl rubber liner and filled with washed limestone gravel of 5-15 mm diameter. At the inlet and outlet zones, 15-30 mm gravel was spread locally to reduce the threat of clogging and improve effluent distribution. *Phragmites australis* were then planted in RB1 and a mixture of *Typha latifolia* and *Iris pseudacorus* were added to RB2 in blocks of 4 m⁻².

INSTRUMENTATION AND ANALYSIS

Measurement of wastewater production on both sites across the duration of the monitoring period was achieved by placing a tipping bucket flow-gauge under each of the outlets of an effluent distribution box located upstream of each reed bed. The disparity in the hydraulic loading rate (HLR) between the two systems was revealed (Table 2), with RB2 receiving, on average, nearly twice the daily flow per unit surface area to that of RB1 over the duration of the monitoring, although both secondary and tertiary treatment reed beds were over-designed when comparing the respective HLRs to design values (EPA, 2000), owing to the lower than anticipated on-site hydraulic loads.

Table 2: RB1 and RB2 mean hydraulic parameters.

Reed Bed ID	Influent Type	Inflow (L/d)	HLR (mm/d)	Design HLR (mm/d)
RB1	STE	327.3	21.8	36
RB2	SE	136.9	34.2	90

A weather station (Campbell Scientific) was erected on each site to record the requisite meteorological data for a rigorous assessment of the water budget throughout the monitoring period. In conjunction with the calculation of nominal hydraulic retention times (nHRT) across each bed, Rhodamine WT (RWT 20% w/v solution) was used as a tracer for the measurement of actual hydraulic retention time using a calibrated submersible fluorometer in the outlet sump.

Composite water samples at the reed bed inlet and outlet points were collected on average once every 2 to 3 weeks over the monitoring period. Laboratory analysis comprised testing all samples for COD, ammonium (NH₄-N), nitrate (NO₃-N), nitrite (NO₂-N), total nitrogen (TN) and PO₄-P using a Merck Spectroquant Nova 60[®] spectrophotometer and associated USEPA approved reagent kits, and total coliforms (TC) and *E. coli* using the Idexx Colilert[®]-18 analysis method.

Stable isotope (¹⁵N) trials. To **study the relative importance of the various nitrogen pathways** and removal mechanisms in wetlands, a ¹⁵N stable isotope tracer study was carried out on both reed beds using labelled ammonium chloride, ¹⁵NH₄-N (Cambridge Isotope Laboratories). The stable isotope was added in solution to each wetland alongside the RWT and composite effluent samples then taken every day for a subsequent period of 40 days (RB1) and 64 days (RB2), far greater than the actual HRT measured. All samples were analysed in the laboratory for the full suite of nitrogenous compounds and were then partitioned into organic suspended (org-N) and dissolved (NH₄-N, NO₃-N and NO₂-N) forms for analysis of ¹⁵N/¹⁴N ratio using **continuous flow-isotope ratio mass spectrometry** (Thermo Delta^{plus} CF-IRMS).

Phosphorus uptake study. During the summer of the third year of operation, representative samples of the reeds in RB1 (*Phragmites australis*) and RB2 (*Typha latifolia* and *Iris pseudacorus*) were collected to quantify the level of P-uptake by the macrophytes. A total of 9 plants were dug up across each bed to ensure representative samples. The density of reed growth was measured in each section using a 1 m×1 m quadrant. The dried biomass samples were then weighed and phosphorus extraction was carried out using the acid digestion method (APHA, 1998). The P concentrations were subsequently measured in a Varian ICP (Inductively Coupled Plasma) instrument.

Viral tracer experiments. To investigate the fate and transport of enteric viral pathogens in both reed beds, multivirus injection experiments were conducted on each reed bed over a 12-day period using bacteriophages MS2, ΦX174 and PR772. MS2 has been used as a surrogate for coxsackievirus and norovirus given its similarities in structure and isoelectric point (IEP) whereas Jin *et al.* (1997) suggest ΦX174 is an accurate model for poliovirus due to having the same IEP and exhibiting the same attachment behaviour. Less is known about the response of PR772, other than that it is very closely related to PRD-1 at the genome level. A profile and description of each of the three phages can be found in a number of studies (Lytle *et al.*, 1991, Collins *et al.*, 2006). Each of the three bacteriophages was injected into a 1L solution of distilled water, which was in turn spiked into each bed via the dual inlet pipe configuration. Effluent samples were collected from the bed's outlet pipe by an automated sampler every six hours and returned to the laboratory for phage assaying.

RESULTS

Water Balance. In incorporating the correct crop coefficients in the daily water balance, the mean flow exiting RB1 was found to be 349.9 L/d, slightly greater than the mean inflow of 327.3 L/d. Likewise, a similar pattern emerged on RB2 where the mean outflow from the reed bed was measured at 149.2 L/d, exceeding the mean inflow of 136.9 L/d. It was found that both beds did not make a significant difference to the incoming hydraulic loads, acting to increase RB1 and RB2 winter flows by 6.4% and 7.2%, and summer flows by 0.5% and 1.7%, respectively.

HRT. Two tracer studies were carried out on both reed beds at the end of the first year of sampling and the second during the final phase of sampling. These results indicated similar actual HRTs (RB1 ≈ 6.5 days; RB2 ≈ 5 days) to the calculated nominal retention times (nHRT),

indicating that any potential channeling or dead zones were largely absent through the beds.

Organic matter (OM) removal. Inspection of COD concentrations at the inlet and outlet of each reed bed (see Table 3), showed a reasonable and similar level of mean OM removal across both (RB1 = 67%; RB2 = 55%). Analysis of the temporal variation in COD load removal across RB1 and RB2 indicated a steady increase in treatment performance over time for both. This pattern developed as a result of the beds undergoing an adaptation period, with ever increasing biofilm on the gravel media responsible for extracting and digesting organic compounds. However, little seasonal variation in treatment efficiency suggested that in a temperate climate like that of Ireland, temperature does not play a major role in the kinetics of the system, at least not during its early years of operation.

Nitrogen removal. Table 3 shows that $\text{NH}_4\text{-N}$ constituted the highest fraction of TN concentrations at both the inlet and outlet of RB1. The efficiency of N-removal in the bed appears to have been limited by both slow rates of mineralisation, with only about half of the Org-N fraction converted to $\text{NH}_4\text{-N}$ and little nitrification owing to the predominant anoxic environment of the bed. No discernible pattern with regard to seasonal nitrogen removal could be observed although removals were at their highest during the first year of reed bed operation. Analysis of the TN reaction rates (i.e. removal efficiency) over time also revealed a slight declining trend over the first three years of operation.

Table 3: Average influent and effluent nitrogen loads from RB1 and RB2.

	COD		TN		org-N		$\text{NH}_4\text{-N}$		TKN		$\text{NO}_3\text{-N}$		$\text{NO}_2\text{-N}$	
	(mg/l)	(g/d)	(mg/l)	(g/d)	(mg/l)	(g/d)	(mg/l)	(g/d)	(mg/l)	(g/d)	(mg/l)	(g/d)	(mg/l)	(g/d)
RB1 In	514	182	105.5	38.6	26.5	10.1	74.9	27.5	101.4	37.6	3.9	1.0	0.2	0
RB1 Out	195	61	76.9	27.3	13.1	4.7	61.0	21.1	74.1	25.8	2.8	1.5	0.05	0
RB2 In	193	32	92.8	15.6	25.0	4.1	22.1	4.0	47.1	8.1	37.9	5.1	7.8	2.4
RB2 Out	107	15	63.9	9.2	11.8	1.8	20.7	2.6	32.5	4.4	26.7	3.9	4.7	0.9

The TN load removal (41%) in RB2 was poor especially given that the effluent from the RBC was partially nitrified and that nutrient removal and denitrification in particular, is often a key focus for tertiary treatment. However, the inability of the RBC to mineralize all org-N to $\text{NH}_4\text{-N}$ and then fully nitrify the effluent did compromise the subsequent potential for denitrification in the reed bed. While some nitrification may have been possible in the isolated aerobic zones of the bed it is more likely that adsorption to sediment particles was responsible for any further removal in $\text{NH}_4\text{-N}$.

The results of the ^{15}N stable isotope tracer study carried out on RB1 are plotted on Fig. 2 against the parallel results from the RWT tracer to indicate the HRT. The results showed that the $\text{NH}_4\text{-N}$ was not naturally enriched with respect to $\delta^{15}\text{N}$ values passing through the reed bed which indicates that much of the influent $\text{NH}_4\text{-N}$ from the septic tank was passing

straight through the reed bed without being taken up in any biologically mediated reactions. This was confirmed from the tracer study carried out in parallel. Following the peak, the lag and then distinct rise in the $\delta^{15}\text{N}$ values for the suspended org-N fraction shows, however, that some of the $\text{NH}_4\text{-N}$ had been biologically assimilated into organic nitrogen (biomass or plants). The tail on the $\text{NH}_4\text{-N}$ trace after the RWT trace has decayed sharply away may be indicative of the $\text{NH}_4\text{-N}$ that has been taken into organic form and then released again as soluble $\text{NH}_4\text{-N}$ – an example of so-called nitrogen “spiraling” (Kadlec *et al.*, 2005). The results of the trial showed that after 50 days only 44% of the spiked ^{15}N had been recovered, indicating that the rest had been taken up by the plants or lost by other means (denitrification etc).

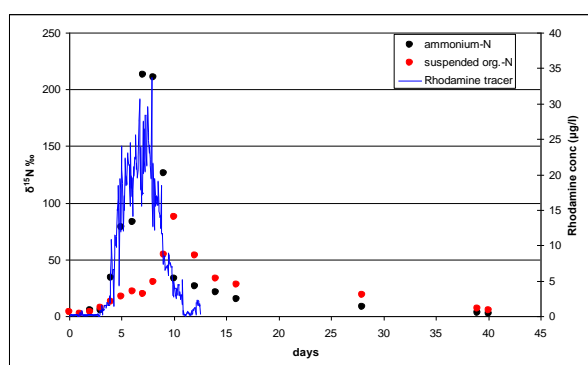


Figure 3: ^{15}N values for RB1 nitrogen fractions and RWT tracer.

On RB2, elevated org-N $\delta^{15}\text{N}$ values in the RBC effluent indicated that its origin was from the biomass in the plant which has synthesised the soluble $\text{NH}_4\text{-N}$ from the waste – i.e. it was not ammonium molecules in the household influent passing straight through the process. There was a significant enrichment of the soluble N species ($\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$) through the reed bed which is indicative of the process of denitrification. Again, the elevation of the org-N $\delta^{15}\text{N}$ values in the reed bed effluent indicated the “spiraling” of the nitrogen through the reed bed through different phases. After 65 days only 28% of the spiked ^{15}N had been recovered which indicated that the remainder must have been taken up by the plants or lost by other means (denitrification etc).

Phosphorus removal. Removal of P, in its soluble $\text{PO}_4\text{-P}$ form, was found to be on average 45% which was surprisingly higher than TN removal - SSF reed beds are thought generally to have a greater potential to remove N than P (Vymazal, 2002). The temporal variations of $\text{PO}_4\text{-P}$ effluent concentrations showed the same pattern as the influent values and it appears that adhesion sites were still readily available after the 26 months of monitoring. Removal in $\text{PO}_4\text{-P}$ over time followed a similar trend to TN whereby the performance efficiency of the bed appeared to drop after year 1 and stabilise over the next 18 months.

Summer removal rates were also seen to be slightly higher than those of winter possibly due to uptake by plant growth which is subsequently re-admitted to the bed upon die-off creating a P-sink. A strong linear correlation between $\text{PO}_4\text{-P}$ surface loading and removal suggested a consistent removal of P throughout the bed and that plentiful adsorption sites were still available after 26 months of monitoring (Fig. 3).

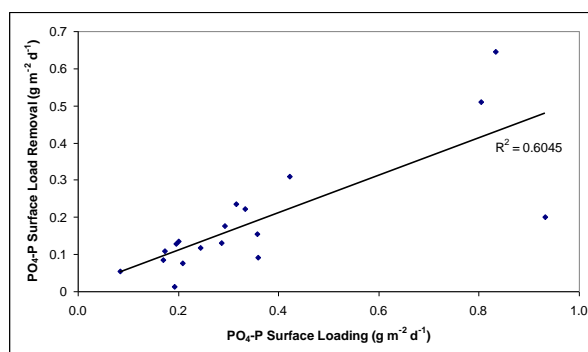


Figure 3: Relationship between $\text{PO}_4\text{-P}$ surface loading and surface load removal rate.

The total P mass in the *Phragmites australis* reeds (stems and roots) in RB1 at the end of the monitoring period was calculated to be 250 g, two thirds of which was associated with the stems. This accounted for 10% of the total mass of P removed (2.50 kg) over the duration of monitoring. If the annual above ground stem matter was completely harvested as a method to control P, it would equate to just 8.4% of the annual total P-load to the reed bed.

An average P-load removal of 22% through RB2 was less than half the rate achieved by RB1 owing to the reduced surface area for adsorption and a greater average HLR. From a seasonal perspective there was a recognisable disparity in average removal rates between summer and winter with greater performance achieved during the warmer months when plant growth would be at its maximum.

It should also be highlighted that mean $\text{PO}_4\text{-P}$ concentrations in the effluent remained very high at 23 mg/L which significantly exceeds any typical discharge consent to freshwater receiving waters. The results of the analyses of the *Typha latifolia* and *Iris pseudacorus* samples taken from RB2 at the end of the monitoring period showed a total P mass of 52 g associated with the reeds compared to a total of 0.167 kg of P removed over the duration of monitoring. Hence, the P in the living roots and stems accounted for 31% of the mass of P removed. If the stems and leaves on this tertiary treatment system were harvested at the end of each growing season, these results show that it would also equate to only 1.3% of the annual total P-load to the reed bed.

Bacterial removal. In RB1, mean removal rates in TC and *E. coli* were found to be similar at 1.8 (98.5%) and 1.4 log-units (96%), respectively. Removal of *E. coli* (1.7 log-units) was slightly greater than for TC (1.3 log-units or 94.6%) in RB2 (Table 4).

Table 4: Average influent and effluent *E.coli* concentrations from RB1 and RB2.

	<i>E.coli</i> conc. (MPN/100mL)	Removal (Log-unit)
RB1 In	7.44×10^5	1.4
RB1 Out	2.80×10^4	
RB2 In	1.10×10^4	1.7
RB2 Out	2.39×10^2	

Analysis of the temporal log-removal rates in RB1 showed both indicator organisms to follow the same pattern throughout the duration of the monitoring with little seasonal or annual variation evident. A very similar trend was mapped in RB2. Results taken from additional intermediate sampling points placed at the middle section of the reed beds showed there to be an exponential decrease in the concentration of both coliform species with longitudinal distance (TC: $r^2 = 0.947$; *E. coli*: $r^2 = 0.977$), mirroring findings in previous studies (Williams *et al.*, 1995; Decamp and Warren, 1999).

Bacteriophages. The results from the viral tracer experiment on RB1 showed the first detected breakthrough of virus to be PR772 after 90 hours (3.8 days) - similar to the initial rhodamine breakthrough, followed by Φ X174 (shown in Fig. 4) at 108 hours (4.5 days) and MS2 at 126 hours (5.3 days). The peak breakthrough times of both Φ X174 and PR772 also coincide very closely with that of rhodamine at 160 hours (6.7 days). An exceptionally high percentage recovery was noted for phage Φ X174 at 96% as opposed to MS2 (74.8%) and PR772 (10.6%), indicating potential suitability of the former as a viable wetland biotracer.

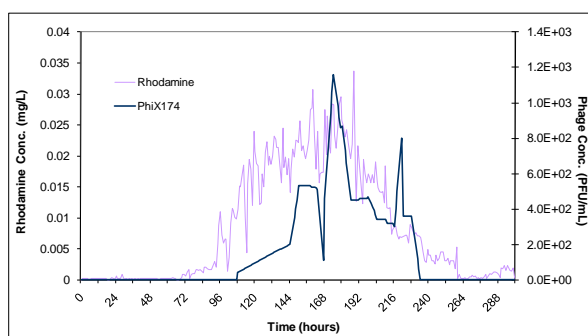


Figure 4: Breakthrough curve for phage Φ X174 on RB1 (rhodamine curve also plotted)

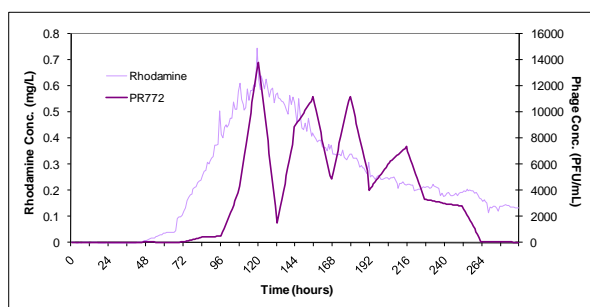


Figure 5: Breakthrough curve for phage PR772 on RB2 (rhodamine curve also plotted)

The initial breakthrough of each of the phages on RB2 was akin to RB1 with PR772 (Fig. 5) again showing first after 48 hours (2.0 days) – very similar to the initial rhodamine breakthrough. This was followed by ΦX174 at 72 hours (3.0 days) and MS2 at 84 hours (3.5 days). The peak breakthrough times of both ΦX174 and PR772 again, as in RB1, coincided very closely with that of rhodamine at 160 hours (5.0 days). Phage recovery, however, showed some contrasting results with respect to RB1. While MS2 recovery was relatively similar for both beds, ΦX174 was seen to drop from 96.0% to 76.9% in RB2. On the other hand the percentage recovery of PR772 was found to significantly increase from 10.6% to 87.5%. While little of this phage was retrieved in RB1 where the septic tank effluent being treated was high in organic chemicals, the cleaner effluent in RB2 did little to prevent significant transport of PR772 to the outlet.

DISCUSSION

Horizontal SSF constructed wetlands in on-site wastewater treatment

This study was used to validate the use of horizontal SSF wetlands as both a secondary and tertiary treatment option for on-site wastewater treatment in Ireland and inform the design guidance given in the new legislative EPA Code of Practice (2009). The Code of Practice specifies an area of 5 m² per population equivalent (p.e.) for a secondary treatment SSF gravel media constructed wetland (reed bed) and 1 m² per p.e. for a tertiary treatment reed bed. This study has shown that wetlands designed with a length to width ratio of 3:1 should promote good hydraulic distribution and thus the optimal pollutant residence times in such treatment systems. However, these treatment systems do not seem to reduce significantly the on-site effluent hydraulic load due to evapotranspiration - a common misconception held in Ireland. Tertiary treatment reed beds could be used to target nitrogen removal as they provide the right conditions for denitrification if receiving nitrified effluent from a secondary treatment plant. However, significant N-removal through secondary treatment

reed beds receiving septic tank effluent is unlikely. Finally, both the secondary and tertiary treatment reed beds removed relatively small P-loads over the course of their first few years of operation, leaving high concentrations of P in the effluent. This performance is only likely to deteriorate over time as the precipitation and adsorption sites become more saturated. Aside from this, plant uptake and subsequent harvesting will not have a significant impact as a method to control phosphorus accumulation in these treatment systems.

CONCLUSIONS

- Removal of COD was found to consistently range 55-70% in both reed beds whilst significant *E.coli* removal (>97.5%) was also recorded. Despite such high removal of the latter, it is clear that circa 239 MPN/100mL still remain in the discharging effluent of the tertiary treatment bed and as such direct discharge to surface water is not viable thereafter.
- N removal was found to be poor across both reed beds, with only 29% mean removal of TN across the secondary treatment bed and 41% mean removal across the tertiary treatment bed. ¹⁵N stable isotope tracing revealed that a high proportion of NH₄-N from the septic tank effluent was passing straight through the secondary treatment reed bed without taking part in any biogeochemical processes whilst the remaining NH₄-N was shown to being biologically assimilated into org-N and then released again as soluble NH₄-N on a cyclic basis. On the tertiary treatment reed bed, the chemical and isotope analysis suggested that only limited denitrification was occurring in the anoxic environment of the bed. Influent NH₄-N and org-N, in contrast, were merely changing form on a cyclic basis through processes such as mineralization, immobilization and plant uptake.
- Removal of P in the secondary treatment bed was found to be more than double the removal achieved across the tertiary treatment bed due to a combination of reduced surface area for adsorption and a greater mean HLR. Further investigations in the P-uptake study revealed that the roots and stems of three different macrophyte species only contained limited amount of phosphorus with respect to the annual wastewater load meaning that plant uptake and subsequent harvesting should not be considered to be a significant sustainable long-term method to control phosphorus.
- Recovery of bacteriophages ΦX174 in both the secondary and tertiary treatment reed beds was shown to be high indicating their potential suitability as viral indicators in SSF reed bed treatment systems and highlighting the poor capacity of reed beds to successfully remove viral micro organisms. Sorption is thought to be the most important process controlling the movement of the phages in the wetlands given that the brief nature of the injection trials would render viral inactivation to be an unlikely event. PR772 displayed a much more contrasting trend in both reed beds with the high removal rate of the phage in the secondary treatment bed possibly owing to hydrophobic interactions with the abundant OM present, causing increased attachment. The larger diametric size of PR772 to the other tested phages would also have made this hydrophobic effect more pronounced.

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CONSTRUCTED ON-SITE SAND FILTERS AS SECONDARY AND TERTIARY EFFLUENT TREATMENT PROCESSES

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ABSTRACT

Full scale sand filters were constructed to treat effluent from two separate on-site wastewater treatment systems in Ireland; one a septic tank, the other a naturally-aerated peat filter. The respective effluents were pumped into intermittently dosed, stratified sand filters. Samples were taken at different layers within the sand filters as well as different depths in the subsoil beneath the filters which were tested at various hydraulic loading rates and analysed for chemical and bacteriological determinants. As a result of the trials, the recommendations for design hydraulic loading rates in Ireland were 30 L/m² day for filters receiving septic tank effluent and 60 L/m² day for filters receiving secondary treated effluent. A full scale trial was then set up to compare parallel tertiary treatment filters receiving on-site wastewater effluent; one constructed from indigenous sand media the other from a recycled glass media. On average the stratified glass filter performed similarly to the stratified sand filter for all of the monitored parameters. However, the overall mean nitrogen load removal from the glass filter proved to be higher compared to the sand filter, due to the higher level of organics reaching the bottom of the filter. Phosphorus removal throughout the 3 year trial was similar through both filters (which was surprising given the poor phosphorus adsorption properties of the glass) with no apparent reduction in removal efficiencies over time.

Keywords: sand filter; on-site wastewater treatment; recycled glass; nutrient removal.

INTRODUCTION

Ireland has over one third of its population using on-site wastewater treatment systems, mostly consisting of septic tanks discharging effluent into a subsoil percolation area. In situations where a septic tank installation is not suitable, some form of secondary treatment system can be installed such as mechanically-aerated systems or filter systems to improve the quality of the effluent before discharge to the subsoil. One of the options promoted in new Code of Practice by the Irish EPA (EPA, 2009) is the use of an intermittent stratified

sand filter either as a secondary treatment unit or as a tertiary treatment / polishing filter in place of the percolation area. The immediate advantage of such a system is that it requires a much smaller plan area compared with the equivalent plot needed for a percolation field.

The aerobic conditions in sand filters are maintained through the intermittent application of the effluent and oxygen consumption is balanced by the renewal of the air phase with atmospheric air by the means of convective and diffusive exchanges through the surface (Boller et al. 1993). Studies have also shown that slight improvements to treatment performance of filters can be gained by small-volume, short hydraulic flushes as opposed to more frequent, larger volume doses (Darby et al. 1996; Rodgers et al. 2006). Phosphorous removal in sand filters is primarily due to the mineral content of the sand used and is controlled mainly by adsorption and mineral precipitation reactions. Two full scale stratified sand filters were constructed to treat effluent from two separate on-site wastewater treatment systems, one a septic tank, the other a naturally-aerated peat filter and studied in order to define suitable design criteria for Ireland.

Other full scale trials have also been set up to evaluate recycled glass in a single pass filter receiving on-site wastewater effluent in comparison to a filter constructed from indigenous sand media. At present almost all of Ireland's recovered glass is exported to the UK whilst specialist silica required for sand filters is often imported from overseas. The use of indigenous recycled glass as a filter media would thus reduce energy and transport costs addressing the concept of sustainability and complimenting the EU Waste Management Directives.

METHODS

Site and filter construction

Site A A house with 4 PE discharging into a 4000 litre septic tank. The sand filter was designed on the basis of previous research in the US (Nichols *et al.*, 1997). It was constructed with a surface area of 3 m by 2 m, depth of 1.05 m and comprised of layers of sand, decreasing in particle size with depth from 1.0 to 0.1 mm diameter (see Fig. 1). The effluent was pumped onto the sand filter via a pressure distribution manifold from a pump sump with float switch. The distribution manifold was 38 mm diameter plastic pipe in a closed loop comprising four parallel laterals at 0.5 m spacing. The effluent was discharged via 3 mm diameter orifices set at 200 mm centres throughout the manifold which ensured an even distribution across the surface area of the filter. The subsoil beneath the filter into which the effluent percolated had a T-value of 33 (equivalent to a field saturated hydraulic conductivity (k_{fs}) of 0.13 m/day).

Site B A house with 4 PE discharging into a naturally-aerated peat filter (*Puraflo*®, Bord na Mona) installed downstream of a 4000 litre septic tank. The stratified sand filter was

identical to the filter constructed at Site A. The subsoil beneath the filter had a T-value of 52 (k_{fs} of 0.08 m/day).

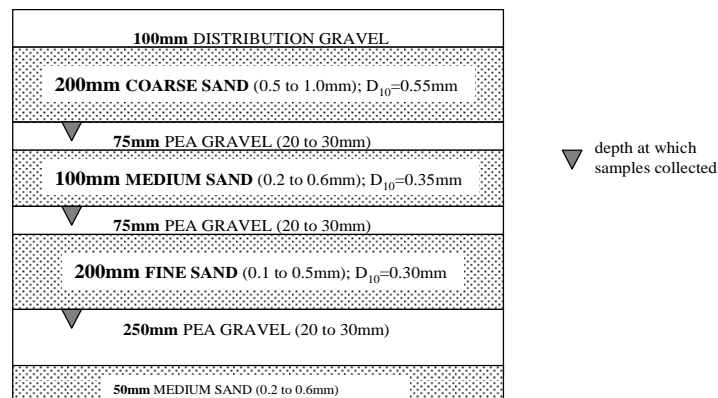


Figure 1: Cross section through stratified sand filter

Site C The study site consisted of a package secondary treatment unit (BioCycle™) treating effluent from a single house containing a family of 5 (2 adults and 3 children) which also operated as Bed & Breakfast service with an allowance for up to 6 additional guests. The effluent from the BioCycle™ was pumped to two unvegetated stratified sand and glass filters in parallel acting as tertiary treatment polishing units as shown on Fig. 2. Both the sand and glass filters were designed as per the EPA Code of Practice (EPA, 2009) on the basis of a design hydraulic loading rate of 60 L/m².d with the resulting total area of the two tertiary treatment filters of 15 m², divided into two tanks 3.0 m x 2.5 m in plan. The effective depth for treatment in the filter tanks was 0.9 m which was stratified into layers from the top down as follows: 100 mm gravel, 200 mm media (granite sand/glass), 75 mm gravel, 100 mm limestone, 75 mm gravel, 300 mm media (granite sand/glass) and 50 mm gravel. The gravel layers (28 mm single grade pea gravel) separated the different layers of media in the tank and ensured an even distribution of the influent throughout. The effluent was discharged to the tanks via a pressurised 32 mm diameter manifold, consisting of 9 arms in parallel, with a series of 3 mm holes, 300 mm apart to ensure even distribution over the entire surface area of the filter.

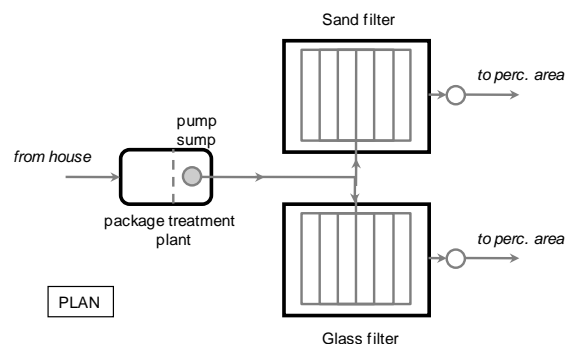


Figure 2: Schematic plan of stratified sand and glass filters.

The particle sieve analyses are summarised for the media used in each of the filters in Table 1. This highlights the higher uniformity coefficients for the granite and limestone sands sourced locally for Site C compared to the specialist sand shipped in for the filters constructed at Sites A and B. The layer of limestone sand on Site C was incorporated into both filters specifically to target the removal of phosphorus.

Table 1: Particle size distribution for filter media

	D ₁₀ (mm)	D ₆₀ (mm)	D ₆₀ /D ₁₀
Coarse (Sites A&B)	0.55	0.68	1.24
Medium (Sites A&B)	0.35	0.52	1.48
Fine (Sites A&B)	0.30	0.41	1.38
Glass (Site C)	0.23	0.53	2.30
Limestone (Site C)	0.58	0.95	1.64
Granite (Site C)	0.15	0.58	3.87

SAMPLING AND ANALYSIS

Gravity cups were installed within the sand filters (three per level) to capture effluent en route down through the sand (or glass) matrix so that the treatment performance with respect to each media layer could also be determined. Ultrasonic level sensors (Siemens Milltronics) and data loggers were installed within the pump sump chambers to monitor the hydraulic loading and the dosing frequency on each filter. Suction lysimeters (Soilmoisture Equipment Corp.) were installed at Sites A and B to collect samples of percolating effluent in the subsoil beneath the filters. Rainfall, wind speed, temperature, solar intensity and relative humidity were also monitored at each site using a weather station (Campbell Scientific).

All filters were operated in a standard unsaturated single-pass mode (i.e. with no recycle). The overall treatment efficiency of the filters in parallel was compared at different organic and hydraulic loading rates. Sampling was carried out on a biweekly basis on both tanks influent and effluent and throughout their respective depths during the trial periods. All samples were analysed for pH, conductivity and temperature on site and then chemical and microbiological parameter analysis in the laboratory. Chemical oxygen demand (COD), ammonium (NH₄-N), nitrite (NO₂-N), nitrate (NO₃-N), Total nitrogen and ortho-phosphate (PO₄-P) were analysed using the Spectroquant Nova 60[®] spectrophotometer and associated reagent kits whilst total coliforms and *E. coli* were quantified using the Colilert[®] method.

ADSORPTION ISOTHERMS

The phosphate adsorption capacity of the different media (sand and recycled glass) was compared before it was added to the filters by the determination of the Freundlich and Langmuir isotherms. Adsorption isotherms were determined by preparing solutions of potassium ortho-phosphate and potassium nitrate to give concentrations of 320, 160, 80, 40, 20, 10, 5 and 2.5 mgP/L. 5g of each sand / glass sample were put into 125ml Nalprene bottles followed by 100 ml of the respective solution. The samples were then sealed and fixed to a slowly rotating wheel which kept the sample and solution continually mixed for a period of 20 hours. The concentration of the solution was then analysed for ortho-phosphate before and afterwards to calculate the percentage adsorbed.

RESULTS

Sites A and B. The performance of the stratified sand filters was analysed by sampling within the filter beneath each layer of sand. The filters were subjected to a normal hydraulic loading for the first 10 months at which point the effluent which was being diverted to parallel percolation trenches in both sites was blocked to promote an increased loading onto the sand filters, as follows:

Site A - normal loading rate = 28.6 L/m² day (dosing 3.5 times per day)

- high loading trial = 57.2 L/m² day (dosing 8.3 times per day)

Site B - normal loading rate = 41.0 L/m² day (dosing 7.3 times per day)

- high loading trial = 97.9 L/m² day (dosing 18.8 times per day)

At Site A piezometers in the sand filters revealed no significant head at the lower hydraulic loading rate of 28.6 L/m² day but a continually rising head at the higher loading rate of 57.2 L/m² day, increasing to 0.7 m above the base after five weeks. This indicates that such a loading rate was too high for a filter receiving septic tank effluent with such a fine sand layer at the base, discharging into subsoil with a T-value of 33. The organic loading rate during the normal hydraulic loading period was 41.1 gCOD/ m² day. At Site B, even though the subsoil had a slower percolation T-value of 52, no head level above the base was measured during either loading period indicating much reduced biofilm development due to lower organic loading compared with Site A. The organic loading rate during normal operation at Site B was 8.8 gCOD/ m² day

Analyses of nitrogen species are presented for the sand filters on both sites (Fig. 3). On Site A, at the lower hydraulic loading rate of 28.6 L/m² day, significant nitrification has occurred within the filter with subsequent denitrification suggested in the finer sand lower down. However, at the higher loading rate (approaching the EPA design value of 60 L/m² day) nitrification was not complete by the base of the filter with much of the nitrogen still in an

ammoniacal form. At Site B, the septic tank effluent was nitrified in the peat filter before being dosed onto the sand filter. No particular evidence of denitrification can be seen in the sand filter presumably due to the combination of unsaturated, aerated conditions in the filter with low organic loads and a shortage of carbon, thus inhibiting the denitrifying bacteria. This pattern was similar at the higher loading levels although there was evidence of denitrification again in the lower fine sand layer arising from localized saturation at this higher hydraulic loading rate. Figure 3(a) reveals a much larger drop in total nitrogen occurred in the upper sand layer receiving the high organic load than in the corresponding layer dosed with secondary treated effluent.

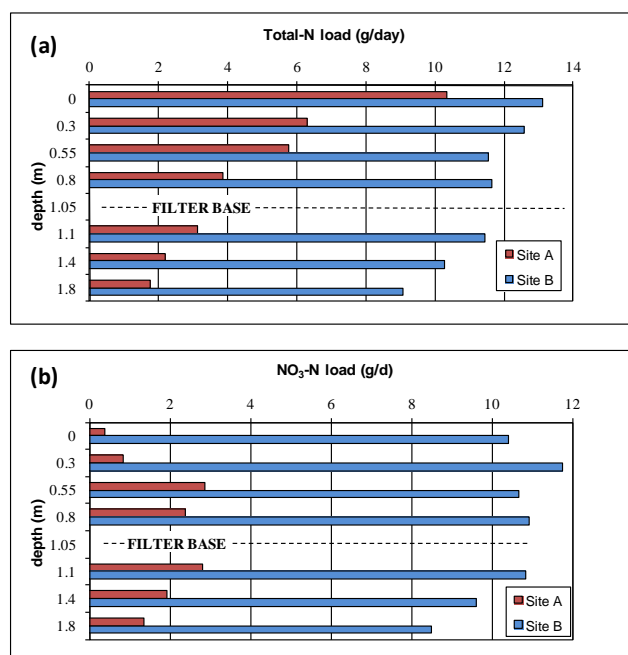


Figure 3: Comparison of (a) total nitrogen and (b) nitrate removal down through the filters and underlying subsoil.

The concentrations of ortho-phosphate passing through both filters were noticeably reduced as seen in Fig. 4. A plot of the process efficiency of both filters with depth revealed an almost total removal of phosphate giving a correlation coefficient between phosphorous loading and removal (k) = 1.16 for hydraulic loadings in the range of 24-57 L/m² day. However, at the higher loading rate of 97.9 L/m² day the removal efficiency appeared to fall away indicating that the residence time in the filters may not have been sufficient for the adsorption removal process to take place effectively. This indicates that, in terms of phosphorous removal using this sand specification, an upper design hydraulic loading rate of 60 L/m² day should be applied. Samples of the three different sands used in the sand filters were analysed using X-ray diffraction analysis (Phillips PW1720) to reveal the respective mineral composition. Apart from the expected predominance of quartz, goethite (Fe₂O₃) was also found in the coarse and medium sands but not in the fine sand sample. Hence, the iron oxide would have acted as a site for cation exchange with the soluble phosphate. This is validated by the results which show that the largest removals were found in the medium

sand layer (see Fig. 4) which was only 100 mm thick but contained the highest levels of goethite. The results from the Langmuir adsorption isotherm experiments on the medium sand yielded estimates that the phosphate adsorption capacity of the sand filter under a typical on-site ortho-phosphate load of 3.5 g/day would be expected to last over 370 days. This was confirmed on both filters where the performance of the filters with respect to phosphate removal showed no noticeable decrease up to the end of the fieldwork after about 340 days of operation. This does have obvious implications, however, for the long-term performance of phosphate removal in such sand filters.

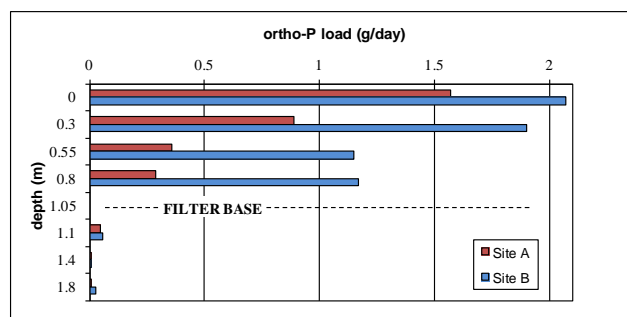


Figure 4: Average PO₄-P loads as a function of depth through the two stratified filters and underlying subsoil.

Finally, analysis of *E. coli* removal through the sand filters and into the subsoil below at the different loading rates for the septic tank effluent (Fig. 5), revealed a correlation between overall removal and hydraulic loading rate: removal coefficient $k = -8.7$ at 28.6 L/m² day and $k = -4.6$ at 57.2 L/m² day. Complete removal occurred at the lower loading rate whilst there were still viable concentrations of *E. coli* discharging to the subsoil at the higher rate. In general, the main removal of bacteria occurred in the top few centimeters of the sand as reported in other studies. Interestingly, no such correlation for *E. coli* against hydraulic loading rate was found for Site B, with all samples below the limit of detection by the base of the sand filter irrespective of loading rate.

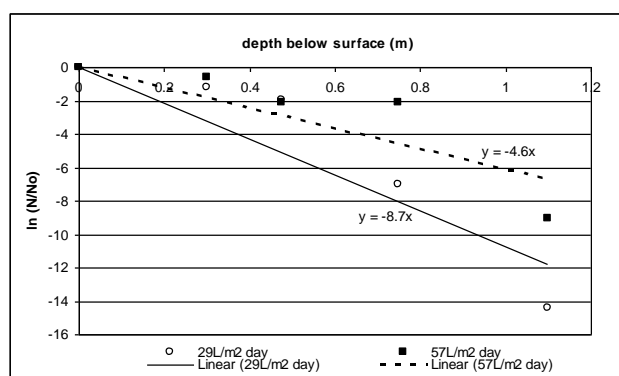


Figure 5: Removal of *E. coli* with depth through the sand filter and subsoil (Site A).

N = number *E. coli*, N_0 = original number from septic tank

Site C. The average hydraulic loading rate on the filters across the trial period was 567 L/d ($38 \text{ L/m}^2\text{.d}$) but with relatively high variations throughout the trial due to the nature of the Bed and Breakfast trade whereby the population of the house regularly fluctuated between 5 to 9 people on a daily basis. The filters were dosed on average 8 times per day throughout the trial.

The average influent COD loadings from the wastewater treatment unit across the trial were $108 \pm 59 \text{ g/day}$ ($144 \pm 58 \text{ mg/L}$). The overall removal efficiency (assessed from the final effluent quality from each filter) showed that the sand and recycled glass filter media performed equally well on average over the course of the study, returning average COD removal rates of 75 % and 73% between the sand and glass respectively. The average effluent loads from the sand and glass filters were $28 \pm 21 \text{ g/d}$ ($34 \pm 24 \text{ mg/L}$) and $27 \pm 21 \text{ g/d}$ ($36 \pm 30 \text{ mg/L}$). Comparisons between the sand and glass throughout the tank depth showed that 52% of COD removal occurred within the first 300 mm filter layer in the sand filter compared to 30% across the glass. In the remaining 100 mm limestone layer and 200 mm sand/glass filter the sand and glass showed further removal rates of 22% and 44% respectively. This shows that the sand media was more efficient than the glass under the higher effluent COD conditions in the first 300 mm filtrate layer. However, this difference in COD removal between the two filters became muted with depth as the COD concentrations of the percolating effluent reduce with depth and the kinetics should become more externally controlled through the biofilm and move from half to first order.

The phosphorus adsorption capacity of the crushed glass, granite sand and limestone sand established by the adsorption isotherms indicated that the glass has very little adsorption capacity or affinity for phosphate. The comparison of the sand samples showed that the limestone sand has a high affinity for phosphate compared to the granite sand. The average P loading onto the filters from the BioCycleTM secondary treatment unit was measured at $6.4 \pm 4.3 \text{ g/d}$ ($11.0 \pm 4.1 \text{ mg/L}$). On average the sand and glass filters removed 51% and 40% $\text{PO}_4\text{-P}$ respectively. The average effluent loads from the sand and glass filter were $3.4 \pm 3.5 \text{ g/d}$ ($5.6 \pm 4.0 \text{ mg/L}$) and $3.8 \pm 3.5 \text{ g/d}$ ($6.5 \pm 4.5 \text{ mg/L}$). The relative removals of $\text{PO}_4\text{-P}$ in the sand filter across the top, limestone and bottom layers were 38%, 52% and 10% respectively in comparison to the glass filter at 40%, 55% and 5% respectively. The 40% removal across the top layer of glass is somewhat surprising in the context of the isotherm results but is thought to be associated with the maximum biomat activity in this area of the filter due to the highest organic loading. Approximately 54% of the $\text{PO}_4\text{-P}$ removal occurred in the limestone layers which was not as impressive as had been hoped for in the design of the filters. The final effluent concentrations from both filters exceeded a level that would be acceptable, for example for surface water discharge. The trend for the removal efficiency of $\text{PO}_4\text{-P}$ over time for both filters was only slightly negative (see Fig. 6), with no significant reduction in removal efficiency apparent over time, which might be evidence that the filters (and in particular the limestone layer in each filter) was becoming saturated over this relatively short period. Calculations using the results of the Freundlich isotherm and mean

loadings of P on the filter suggest that the glass filter should have become saturated after approximately 15 months for example.

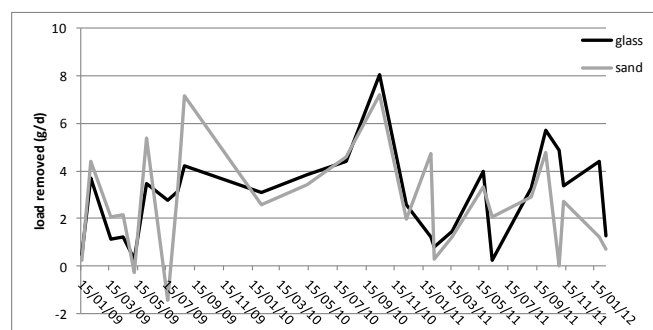


Figure 6: PO₄-P load removal over time through the stratified sand and glass filters.

Fig. 7 displays an overview of the average nitrogenous loads as total nitrogen (TN), organic nitrogen (Org-N), ammonium (NH₄-N), nitrite (NO₂-N) and nitrate (NO₃-N) as a function of tank depth.

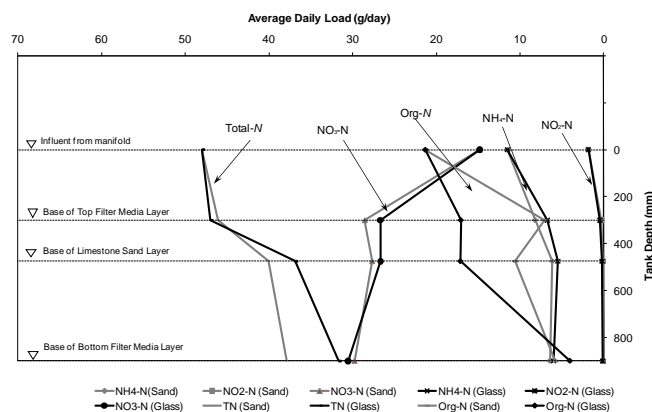


Figure 7: Nitrogen loadings as a function of depth through the sand and glass media filters.

Nitrogen removal can be seen to have occurred throughout the tank depth with daily average total nitrogen (TN) removal rates across the sand and glass units of 16% (average effluent discharge of 35.1±26.2 g/d, 49.3±21.8 mg/L) and 28% (average effluent discharge of 29.5±19.1 g/d, 43.0±16.7 mg/L) reduction respectively from average TN loading onto the filters of 48.1±26.5 g/d, 54.6±18.3 mg/L. The higher TN removal in the glass filter is interesting and can be explained by higher organic loads percolating down to the limestone glass layer and lower glass layer as discussed earlier (i.e. 52% of COD removal occurred within the first 300 mm filter layer in the sand filter compared to only 30% COD removal across the first glass layer). These higher organic concentrations in the lower layers of the filter provided food for the heterotrophic denitrifying bacteria in the layers where there were also high nitrate concentrations with much of the nitrification occurring in the upper layer in both filters, as can be seen in Fig. 7. In this zone the Org-N was also being reduced through mineralisation and converted to NH₄-N which offsets somewhat the decrease in NH₄-N loads as a result of the nitrification process.

E. coli removal in both the sand and glass was approximately the same at 3 log removal with more than 2 log removal being achieved within the first 0.2 m layer in both filter media at 99.6% and 99.0% respectively (Fig. 8). A further average 1 log removal occurred down through the remaining layers of both filters. It should be noted however, that there were still concentrations of *E. coli* of the magnitude 10^3 MPN/100ml in the final effluent from both filters which would not be suitable for direct discharge to groundwater. When assessing the removal rate as a function of hydraulic loading, there was no evidence of reduction in *E. coli* removal rates with increased loading conditions, as might have been expected. The overall removal coefficient k for the sand and glass filters (from a plot of $\ln(N/N_0)$ versus filter depth) was -7.2 and -7.6 respectively, at the average hydraulic loading rate of $38 \text{ L/m}^2 \cdot \text{d}$ throughout the trial period.

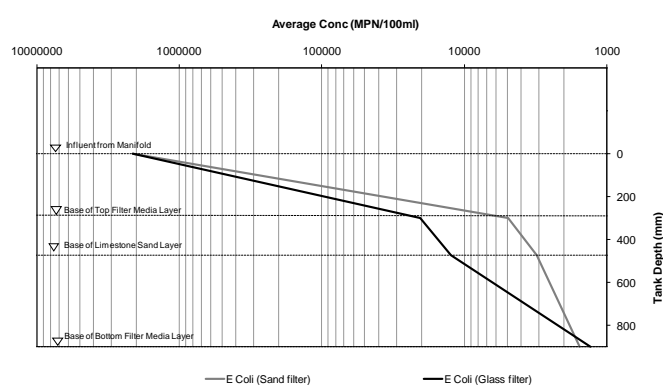


Figure 8: *E. coli* removal as a function of tank depth.

CONCLUSIONS

The detailed study on the two stratified sand filters receiving different quality effluent revealed that both treatment systems greatly reduced the chemical and biological loading rates of domestic wastewater effluent discharging directly to the subsoil below, reducing the potential for groundwater pollution. However, the results indicate that for septic tank effluent the recommended hydraulic loading rate on relatively slowly draining subsoils should not exceed $30 \text{ L/m}^2 \text{ day}$ both due to ponding problems in the base of the filter and also to ensure the effective removal of enteric bacteria. This loading rate has been adopted in the EPA Code of Practice (EPA, 2009). Ortho-phosphate removal in the filters is related to the mineral composition of the sand used and these trials showed that a maximum loading rate of $60 \text{ L/m}^2 \text{ day}$ appeared to be the optimum for phosphate removal. However, it should be acknowledged that the removal efficiency would reduce over time as the potential adsorption sites fill.

The comparison between the tertiary treatment stratified glass and sand filters showed that, on average they performed similarly for all of the monitored parameters. However,

within the filters there were some differences in process kinetics between the two systems which led to some interesting results with respect to nitrogen removal. Although the overall organic matter removal through the two filters was very similar, there was a much slower breakdown of organics through the top layer of the recycled glass filter than the sand filter which provided a carbon source for higher levels of denitrification in the lower parts of the filter where the nitrate concentrations were more optimal. Hence, the overall mean nitrogen load removal from the glass filter proved to be 18.6 g/d compared to the only 13.0 g/d through the sand filter.

Phosphorus removal was higher through the sand filter than the recycled glass filter, 51% and 40% respectively, which was to be expected due to the poor phosphorus adsorption properties of the glass. The limestone layer in both filters only accounted for approximately half of the P removed which was disappointing as laboratory studies had shown it to have excellent phosphorus adsorption properties. There was no strong evidence that the limestone layer was starting to approach saturation. Finally, on average 3 log removal of enteric microorganisms through both filters using the indicator bacteria *E. coli* was shown which, although reasonable compared to other tertiary treatment systems, still left significant numbers of bacteria in the effluent which would not be suitable for direct discharge into groundwater. These trials show that the use of recycled glass as a filter media as replacement for sand could be promoted due to its similar performance which would complement sustainability concepts.

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SESSION VI RISK ASSESSMENT/DISPOSAL OPTION

THEORY AND PRACTICE OF SOIL AND SITE INVESTIGATION FOR WASTEWATER DISPERSAL SYSTEMS IN THE UNITED STATES

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ABSTRACT

The nature of soil and site investigations for dispersal components of onsite wastewater systems varies considerably across the U.S. The variability results from state, regional and local permitting agencies that have jurisdiction over these activities. This paper presents an overview of the variability among jurisdictions within North Carolina and other states. Further, it provides an account of methods used to perform site evaluations and contrasts them with methods used in Ireland. Procedures for identification of limiting soil conditions are described along with methods for addressing them using appropriate design parameters to protect public health and the environment. The role of system management for risk minimization is highlighted within this context.

Keywords: on-site wastewater treatment; soil and site evaluation; nutrient removal.

INTRODUCTION

In 2007, the U.S. EPA reported that 20% of U.S. households used onsite wastewater treatment systems (septic systems) with 47% of systems serving suburban communities and 50% serving rural residences. Highest percentage use is in the South (46% of homes), followed by the Midwest (22%), Northeast (19%) and West (13%) (U.S. EPA, 2007).

Of note is that the older population centers such as New York City, Boston, and Philadelphia have low use since central sewers were installed in these areas more than 100 years ago. Major growth in the West following World War II was accompanied by significant public works sewer projects. Growth in the Southeast began in the 1960s and urban areas were extensively sewered. Southeastern rural areas have experienced significant growth in the last three decades as farm land has been converted to subdivisions. Onsite systems were installed in these developments with the assumption that centralized sewer would

eventually become available. Now that the U.S. EPA has officially embraced it as a cost-effective and long-term option, onsite systems are now viewed as a critical component of our wastewater treatment infrastructure (U.S. EPA, 1997). Only with proper management can these or any wastewater system remain viable and environmentally sound.

GOVERNING RULES AND REGULATIONS

Rules governing onsite wastewater treatment systems in the U.S. are issued at the state or local level. Implementation may also occur at either the state or local level. North Carolina uses state-level rules except where local jurisdictions have adopted their own more stringent regulations.

In North Carolina, county environmental health personnel are authorized as agents of the state to enforce the regulations in their jurisdiction. A centralized intern training program is conducted by the state to promote consistency in rule implementation and enforcement. In reality, despite these efforts, there is often variability across counties and regions. With three distinct geological settings (coastal, piedmont and mountain), some professionals contend that state rules should vary accordingly. Anecdotal evidence indicates that local or county variability in rule implementation is not unique to North Carolina.

Concerns about effects of nutrient over-enrichment of ground and surface waters have resulted in increased scrutiny of rules related to onsite wastewater treatment systems. A recent survey conducted by the State Onsite Regulators Alliance (SORA) provided information regarding implementation of rules specifically related to nitrogen and phosphorus inputs from septic systems. Results indicate that nitrogen is more widely regulated than phosphorus (Table 1). The higher level of concern regarding nitrogen is due to its greater mobility in groundwater (State Onsite Regulators Alliance, 2012).

Table 1: Number of U.S. States with rules governing Nitrogen and Phosphorus inputs from onsite systems

Status of Rules	Nitrogen	Phosphorus
Number of states with existing rules	25	10
Number of states where rules are being considered	9	4
Number of states with local rules	18	7
Number of states where N or P are not currently viewed as a problem	8	28
Number of states with some perceived problems but no rules at this time	8	8

SOIL AND SITE EVALUATIONS FOR SINGLE-FAMILY RESIDENTIAL SYSTEMS

An informal survey by the author of a dozen states revealed that all respondents described soil morphology in either auger borings or pits excavated on the proposed site. Parameters for description generally included the following:

- For each soil horizon
 - Texture
 - Structure
 - Consistence
 - Color
- For each soil profile
 - Depth to soil wetness or seasonal high water table indicators (color and/or redoximorphic features)
 - Depth to restrictive horizon(s)
 - Topography and landscape position
 - Presence/absence of organic soils
- For the site
 - Available space for system installation and repair area (if required) including required setbacks

- Other proximal features such as large capacity wells, large wastewater flows, utilities, location of structures, drives and parking, artificial drainage, etc.

A methodical approach to site evaluation is necessary, particularly with respect to Long Term Acceptance Rate (LTAR) assignment. This eliminates subjectivity and allows for consistency among professionals. Such an approach has been developed by Dr. David Lindbo of NC State University. Lindbo's method includes micro-adjustment (up or down) of LTAR on the basis of site characteristics and soil morphology. The specific parameters for and order of consideration are: slope (%), texture, silt content, mica content, coarse fragment content, structure (including grade, size and type), consistence (based on most restrictive horizon), redoximorphic features (water table), soil depth, landscape position, and slope description (linear, concave, convex). One additional parameter described as "local understanding" accounts for experiential knowledge (Lindbo et al., 2007; Lindbo et al., 2005).

Use of percolation ("perc") tests for soil evaluation in the U.S. varies. In North Carolina, perc tests were commonly used well into the 1970's until more emphasis was placed on soil morphology. Rules adopted in 1982 included requirements for specific morphological descriptions instead of perc tests because of concerns over methodological variation across jurisdictions. All but a few counties in Missouri have also discontinued use of percolation tests. Rhode Island employs them when professional opinions vary. Ireland's approach of requiring specific methodology promotes consistent application of the percolation test and imparts validity to its use. Saturated hydraulic conductivity (Ksat) is often required for second opinions or deeper investigations. However use of Ksat data (perc tests) remains highly controversial due to inherent soil variability. One scientist has stated that these measurements should only be used to confirm the soil morphologic LTAR and reduce it if needed (A. Amoozegar, personal communication).

The State of North Carolina provides a standardized form listing parameters for soil and site evaluation, but does not mandate its use on the local level. The specific nature of Ireland's Site Characterization form is impressive. It compels the evaluator to respond in some way to every possible concern relative to system establishment. Presence of or proximity to features of concern are addressed in the rules. If local authorities adhere to those, encroachment on features that may be adversely affected by effluent dispersal can potentially be avoided. However, an indication of their presence is not specifically solicited as it is on the Irish form.

QUALIFICATIONS FOR PERFORMING SOIL AND SITE EVALUATIONS

According to Ireland's guidance, the Local Authority (LA) maintains lists of persons deemed qualified to perform site evaluations, indicating that these are private-sector professionals who have received specific training and certification. This is also the case in some U.S. states where certified or licensed private sector professionals must be engaged by the landowner. These persons may be soil scientists, geologists, engineers, or sanitarians. Some states have requirements for specific certification to perform soil evaluations, regardless of what other professional credentials one might have. Many states, including North Carolina authorize public sector employees (typically Environmental Health Specialists) to perform soil/site evaluations for both single family residential and small commercial systems. This is likely a relic from the early to mid twentieth century when information on use of septic systems was developed and disseminated by the U.S. Department of Agriculture (USDA). USDA endeavored to provide farm families with guidance on individual waste treatment systems. As rural areas established local health authorities, sanitarians were trained to design and site septic systems for both farm and non-farm applications. The sanitarians were also responsible for enforcement action if enabling legislation was enacted. The wisdom of endowing public sector personnel with authority to both design *and* permit septic systems is an interesting topic for consideration. While an agency with permit oversight and enforcement jurisdiction might have the capacity and expertise to perform site evaluations and system design, concentrating instead upon design *review* might preclude potential conflicts of interest when diligent enforcement may someday be required.

PERFORMANCE OF ONSITE SYSTEMS

Citing various sources including Nelson et al, (1999), U.S. EPA estimated that between 10 and 20% of systems do not adequately treat waste (U.S. EPA 2002). In 1997, the U.S. Department of Commerce reported that half of systems in service in 1997 were more than 30 years old and thus more likely to malfunction (U.S. DOC, 1997). System performance studies have been conducted for many years in North Carolina. Reported malfunction rates varied considerably: 11.4% (DEH, 2002), 5% (Lindbo et al., 1998), 13.3% (Deal et al., 2007), and 21% (Hinson et al., 1994). Notably, the methods of site selection and malfunction determination varied considerably and reported rates are not directly comparable.

One factor in performance is system age since capacity tends to decline over time. The question of how long systems should last is often posed. Theoretically, systems should continue to provide a high level of treatment for as long as sixty years. In practical terms, system performance and longevity is dependent upon multiple factors related to overall

management. Proper siting, design, installation, use, operation, maintenance and inspection are individually and collectively relevant. Lindbo et al., (1998) evaluated relative performance documented in two separate surveys of systems in northeast North Carolina. Although system age and climatic conditions were factors in improved overall performance from 1991 to 1995, adoption of more stringent siting criteria and pro-active efforts of the local Management Entity in problem identification and correction played a prominent role. This highlights the importance of diligent management to ensure optimum performance.

Ensuring proper management is a considerable challenge. For the vast majority of “conventional” systems, the owner is solely responsible for maintenance. If maintenance occurs at all, it is rarely tracked. Use of more complex technologies typically is (but not always) accompanied by some sort of tracking mechanism. Mechanisms used in the U.S. vary from the very simple to the quite complex. States surveyed indicated that although web-based reporting is used on a limited basis, reports are typically submitted via hard copy with results entered into electronic databases. Tracking system management is most often conducted by public sector entities. Responsible management entities that actually perform, document and report O&M activities for single family residential systems may be from either the public or private sector.

ALTERNATIVES FOR OVERCOMING SOIL AND SITE LIMITATIONS

It is critical to recognize that there are sites where it is inappropriate to use onsite wastewater treatment systems. If an individual system is not advisable, effluent from multiple residences can be collected via STEP or STEG systems and conveyed to a common treatment system and dispersal component. Centralized sewers are appropriate if population densities justify the expense and public opinion favors the approach. States were polled regarding alternative technologies used to overcoming site limitations similar to those commonly encountered in Irish landscapes, i.e., – shallow depth to groundwater, impermeable clays or rock. The results of this informal poll are discussed here with a strong note of caution that the suggested alternatives are used across a wider variety of sites than may be indicated here. The states responding to the survey use a minimum separation to limiting conditions that ranges from 0.2 to 1.2 m for seasonal high water tables and 0.2 to 1.8 m. for bedrock or other extremely restrictive horizons. A summary of alternatives for different soil and site limitations is presented in Table 2.

Table 2: Potential alternative onsite wastewater treatment and dispersal options for selected soil and site limitations

Potential Alternative	Limitation		
	Clayey subsoil	Shallow Subsoil Above Karst Aquifer	High SHWT
Pretreatment	x	x	?
Lower LTAR	x	?	
Areal fill/pressure distribution	x	x	x
Areal fill/gravity distribution	x	x	x
Timed-dosed pressurized drainfield (LPD)	x		
Shallow narrow drainfield		x	x
At-grade system		x	x
Subsurface drip dispersal	x	x	x
Surface drip dispersal	x	x	x
Surface spray dispersal	x	x	x
Bottomless sand filter			x

Clayey sub-soils dictate that hydraulic loading be carefully considered. The LTAR must be adjusted accordingly to allow effluent to be treated and subsequently move through the finer texture. Installing the drainfield in the more-permeable topsoil while still using a LTAR appropriate for the subsoil will maximize treatment while accounting for hydraulic capacity. The addition of a pretreatment component results in cleaner effluent which may be more easily accepted by the soil. An areal fill system (Figure 1) is an above-grade soil treatment area designed and installed such that the entire infiltrative surface is located above the original ground elevation using suitable imported soil material for fill. Such systems may use gravity, pressure-dosed gravity or low-pressure distribution. A final cover of suitable soil stabilizes the completed installation and supports vegetative growth.

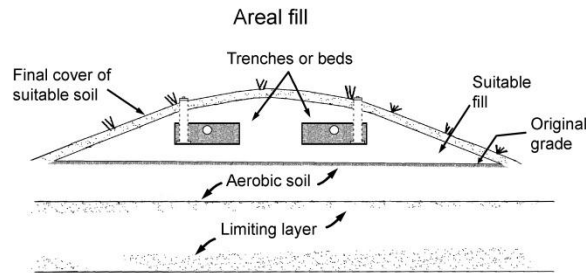


Figure 1: Cross section of an areal fill dispersal component (CIDWT, 2009)

Use of shallow systems such as “at-grades” is a similar option installed such that some part of the infiltrative surface is located at or below the original ground elevation using suitable imported soil material for fill. The excavation is 0 to 6” into native soil. This option utilizes gravity, pressure-dosed gravity or low-pressure distribution with the orifices of the distribution pipe above the original ground elevation. Again, a final cover of suitable soil stabilizes the completed installation and supports vegetative growth; (Figure 2). Using drip dispersal (either subsurface or surface) allows effluent to be applied in ‘micro-doses’.

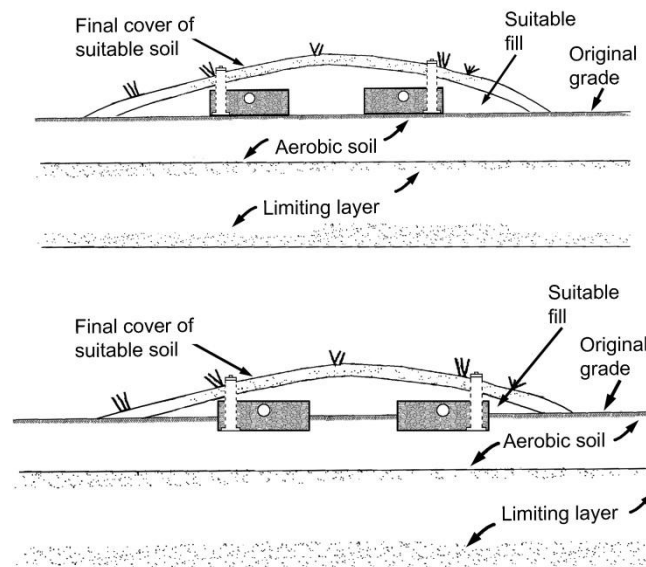


Figure 2: Cross section of an at-grade dispersal component (CIDWT, 2009)

Shallow sub-soils above karst aquifers present the challenge of preventing preferential flow of inadequately treated effluent into groundwater. The addition of pretreatment under these conditions must be designed to include a denitrification regime to ensure that nitrate levels are reduced. Areal fill, at-grade, low pressure distribution (Figure 3) or shallow narrow drainfield systems (Figure 4) may address separation issues, but (by their nature)

may increase nitrate levels if denitrification cannot be designed into the system. Subsurface drip, surface drip or surface spray dispersal present more opportunities for use assuming site-specific parameters are addressed and if allowed by local regulation.

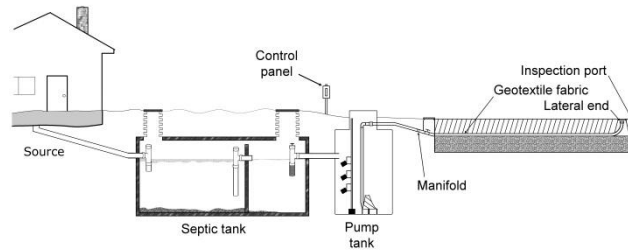


Figure 3: Cross section of a system using low pressure distribution (CIDWT, 2009)

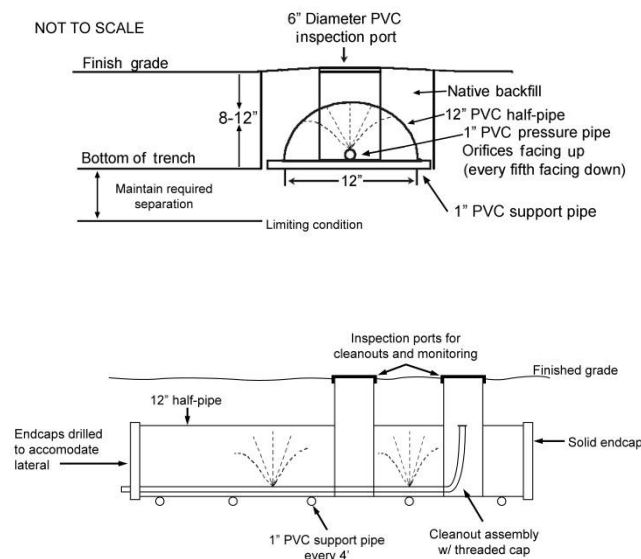


Figure 4: Cross sections of a shallow narrow drainfield dispersal component using low pressure distribution (CIDWT, 2009)

If seasonal high water tables are the limiting factor, there is obvious risk for groundwater contamination. Using shallow installations is preferred for all systems, but particularly critical under these conditions. Bottomless sand filters have been successfully used in coastal states, but typically include a pretreatment component with a denitrification configuration.

FUNDING DECENTRALIZED INFRASTRUCTURE

When systems malfunction or are deemed out of compliance, the cost of repair or replacement is typically borne by the system owner. While some low- or no-interest loan programs are available for decentralized projects, municipal system upgrades and repairs typically garner the vast majority of available public funding. The U.S. EPA Clean Water Act established the Clean Water State Revolving Fund (CWSRF) program which uses federal and state contributions to provide loans, refinancing, purchasing or guaranteeing local debt for water quality projects. States have flexibility in resource allocation on the basis of their particular environmental needs. Statistics from the 2009 CWSRF report indicate that only 0.02% of dollars spent on wastewater infrastructure supported individual/decentralized sewage projects (U.S. EPA 2010) and this number has not changed significantly in the recent past.

Since 20% of U.S. households use these types of systems, the disproportionate nature of funding allocation is clear. The reasons for the disparity are indistinct, particularly in light of the 1997 Response to Congress which highlighted the benefits and potential costs/savings related to individual/decentralized options. The report outlined the barriers that must be overcome to effectively implement decentralized technology. These included:

- Lack of knowledge and public misperception
- Legislative and regulatory constraints
- Lack of management programs
- Liability and engineering fees
- Financial barriers (including a reference to the CWSRF's disparate funding context) (U.S. EPA 1997).

Nearly sixteen years later, the same barriers obviously have yet to be surmounted despite industry efforts to do so. Ireland will surely encounter the same challenges as the country implements new initiatives related to its system inspection program.

CONCLUSION

The considerable variability among regulations related to onsite wastewater treatment system permitting is rooted in the history of the industry in the U.S. Local and state environmental health programs originated as agricultural initiatives that co-evolved with public health departments. The regulatory structure within each state and locale has subsequently been molded by economic, political and social forces. Groups such as the Consortium of Institutes for Decentralized Wastewater Treatment (CIDWT), the National

Onsite Wastewater Recycling Association (NOWRA) and the State Onsite Regulators Alliance (SORA) advocate a more standardized approach to all aspects of system management in the U.S. and Canada. These groups promote consistency through education, practitioner training, and enhanced interaction between regulatory, academic and private sector professionals. Their activities represent the most tangible effort to overcome the barriers identified in the 1997 Response to Congress.

Establishment of robust dispersal components for onsite wastewater treatment systems requires that sites first be properly assessed to identify limitations. Systems must then be designed with attention to those limitations and carefully installed to meet rigorous standards. Long-term operation, monitoring maintenance and inspection must then be diligently performed and tracked. Effective risk management necessitates assiduous attention to these details. While these practices have yet to be consistently implemented in the U.S., increased communication among industry sectors warrants cautious optimism. The need for reasonable allocation of public sources of funding is a pre-requisite for sustainable implementation of individual/decentralized infrastructure.

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THE USE OF GEOSPATIAL MODELLING IN DETERMINING STRATEGIES FOR ON-SITE WASTEWATER TREATMENT IN AREAS OF LOW PERMEABILITY SUBSOIL

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ABSTRACT

The domestic wastewater of over one third of the population in Ireland is treated by on-site treatment systems. However, if there is insufficient permeability and/or subsoil depth to take the effluent load, surface ponding may occur resulting in a risk of effluent discharge and runoff of pollutants to surface water. The aim of this project is to create a decision support tool to identify solutions and better manage on-site wastewater in such areas. Geospatial modelling is being developed for testing in four counties: Wexford, Limerick, Leitrim and Sligo. Solutions are being developed for various scenarios, incorporating human settlements, the physical environment comprising geology, land cover, hydrology and meteorology, and infrastructure such as transportation and utility networks. Initial outputs indicate that onsite wastewater treatment systems are aligned with road networks, forming ribbon developments, and are frequently clustered. Modelling at this stage suggests that such is the number and distribution of onsite systems in low permeability areas that they pose a high risk of surface water pollution.

Keywords: on-site wastewater treatment; geospatial modelling; low permeability subsoil.

INTRODUCTION

Ireland's population is a little over 4.5 million. Around 1.75 million live in rural areas (CSO, 2012), with the majority of rural dwellers depending on onsite wastewater treatment systems (OSWTs) for domestic wastewater disposal. Septic tank systems form the bulk of OSWTs employed, with the result that over 400,000 septic tank systems are in use today. To put this into context, the ratio of septic tanks to head of population in Ireland is 1:11. The figure for England and Wales is 1:65.

If located and constructed within procedures laid down by the Sustainable Rural Housing Guidelines for Planning Authorities (DEHLG, 2005) and the EPA's Code of Practice (2009), septic tank systems should present minimal threat to water quality. However, systems located on low permeability subsoil require particular attention. If the effluent loading on the subsoil is too high in these areas,

or there is insufficient permeability in the subsoil to take the effluent load, surface ponding may occur with associated health risk. There will also be a risk of effluent discharge and runoff of pollutants to surface water resulting in possible eutrophication of waterways. A further consideration is legacy septic tank systems, whose construction and performance under less stringent guidance such as *Recommendations for effluent treatment and disposal from a single dwelling house* (EOLAS, 1991) may serve to compound the issues presented by low permeability subsoils. The number of legacy septic tank systems is significant. Almost 340,000 existed in 1991, and it is reasonable to assume that a substantial number are still in operation.

This research will help to define the extent of the problem of existing septic tanks and give advice on how to ameliorate sites. The geospatial modelling will be tailored to evaluate a series of different strategies that could be considered by Local Authorities with respect to planning decisions and the treatment and disposal of wastewater effluent in areas of low permeability subsoil.

METHOD

Geospatial modelling is conducted through Geographic Information Systems (GIS). ESRI's ArcGIS 10 is used to evaluate several strategies that could be considered by Local Authorities. The strategies, listed below, will be compared on both a cost-benefit and environmental impact/sustainability basis:

- Connect houses to proximate existing main sewer system for wastewater treatment in a centralised treatment plant;
- Construct conventional or small bore sewerage through clusters of single house developments, taking the wastewater to a decentralised plant with consented discharge to water courses;
- Use closed storage tanks with regular tankering of effluent to central wastewater treatment works (suitable for holiday homes or new developments with mandatory water saving technologies);
- Use on-site technologies such as willow systems, Low Pressure Pipe (LPP), Drip Distribution (DD) or ecosanitation;
- Discharge effluent through an imported media filter, which percolates down into more permeable subsoil or bedrock.

The principal datasets upon which modelling processes are applied are:

- Topography;
- Soils and subsoils;
- Depth to bedrock;
- Likelihood of inadequate percolation;
- Rivers and lakes;
- Urban and rural sewered areas;
- Wastewater treatment plants;

- Hydrometric stations;
- XY coordinates for septic tank systems;
- Road networks.

Glenbrien and Ballymurn in County Wexford are the foci of this paper. An Post's GeoDirectory has been employed to map OSWTSs in the area, with the assumption that any mapped location outside a sewered area will of necessity have an onsite wastewater treatment system. Thus, each GeoDirectory data point (represented by a yellow dot on the maps) is assumed to represent a septic tank for the purposes of this research. Initially, XY coordinates for septic tanks are mapped over: rivers; roads; wastewater treatment plants; layers indicating likelihood of inadequate percolation to groundwater; and urban and rural sewered area polygons. Once relationships have been established, high risk zones are identified. Modelling processes are then concentrated on these high risk areas.

Kernel density analysis and OSWTS aggregation is employed to identify hotspots within study areas, in order to enable prioritisation of localised assessment.

Once the modelling architecture is tested, coding the model to provide a user-friendly transition from input to potential solution will be implemented. Although at an early stage, it is anticipated that coding development will progress as follows:

- Configure model software with appropriate parameters.
- Deploy and test models on ArcGIS Desktop.
- Port models to ArcGIS Server.
- Design appropriate User Interface.
- Performance tune geoprocessing models for the ArcGIS Server environment.
- Deploy and test models on ArcGIS Server.

RESULTS

The modelling architecture developed thus far is illustrated in Figure 1. From an initial user-specified input a series of linked geoprocesses are designed to interrogate the input and indicate an appropriate solution(s).

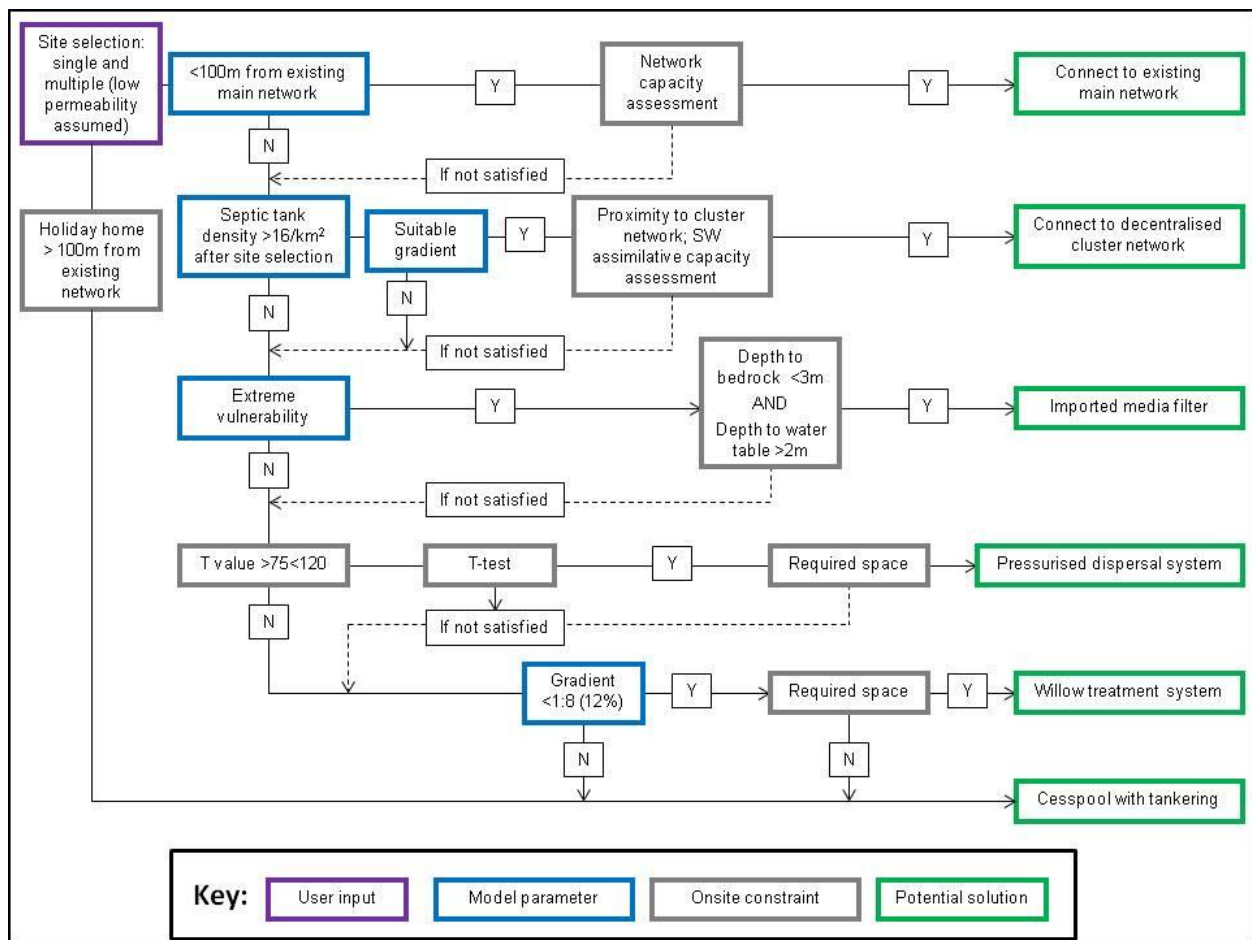


Figure 1: Modelling architecture.

The result of each process is dependent on a series of input variables. Some variables are generated by preceding geoprocessing stages, while others are created by further user inputs through a user interface to be developed once the modelling framework has been affirmed. While each stage of the modelling involves a stand-alone process, coding will automate the geoprocessing stream.

More than one solution may be viable for a given site and so solutions will be ranked on cost and sustainability in order to aid the decision-making process. At this stage of the research however, costs and sustainability have yet to be factored in, although each of the geoprocesses relevant to the model parameters has been applied to the data acquired thus far.

The geoprocessing outputs illustrated in this paper show examples of three of the model parameters: likelihood of inadequate percolation, distance from an existing network and septic tank density. At this stage of the project on-site constraints have not been factored in.

POTENTIAL SOLUTIONS

The first step is to identify on-site systems in areas of low permeability subsoil which therefore have a likelihood of inadequate percolation. This uses a recently developed EPA map, *Likelihood of*

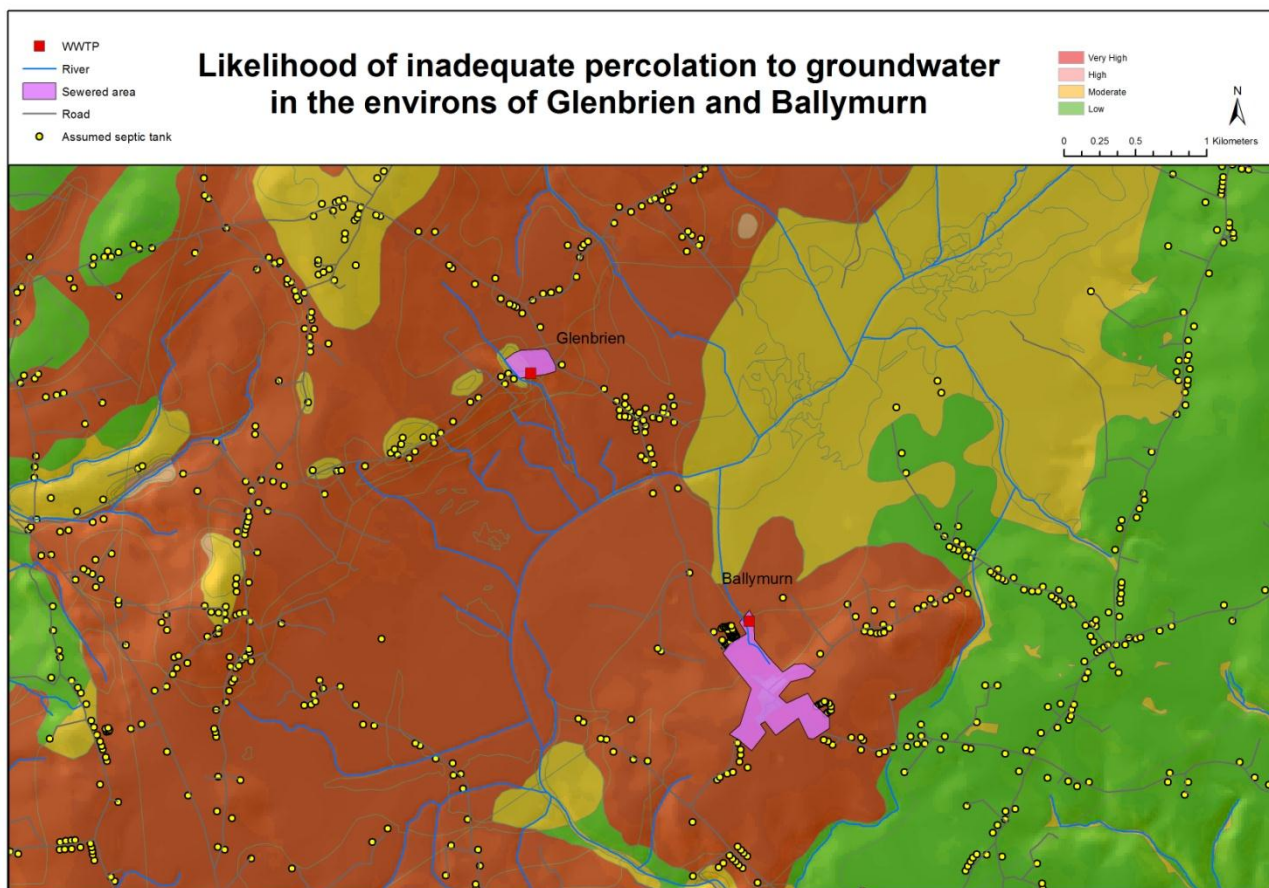


Figure 2: Likelihood of inadequate percolation to groundwater.

Inadequate Percolation to Groundwater, which has combined soil data (from Teagasc) with subsoil permeability, aquifer and vulnerability data (from the Geological Survey of Ireland) to categorise mapped areas into *low*, *moderate*, *high* and *very high* risks.

The *very high* category means >85% likelihood of inadequate percolation to groundwater and these are the areas focussed on for this project. An example is shown in Figure 2 for Glenbrien and Ballymurn in County Wexford, where OSWTSs are overlaid on the EPA risk map.

Connect to existing network

The distance from an existing sewer network is the first parameter considered in the model architecture. If there is a centralised facility close enough to dispose of wastewater without recourse to an OSWTS, the feasibility of connecting to that network should be investigated. An assessment of

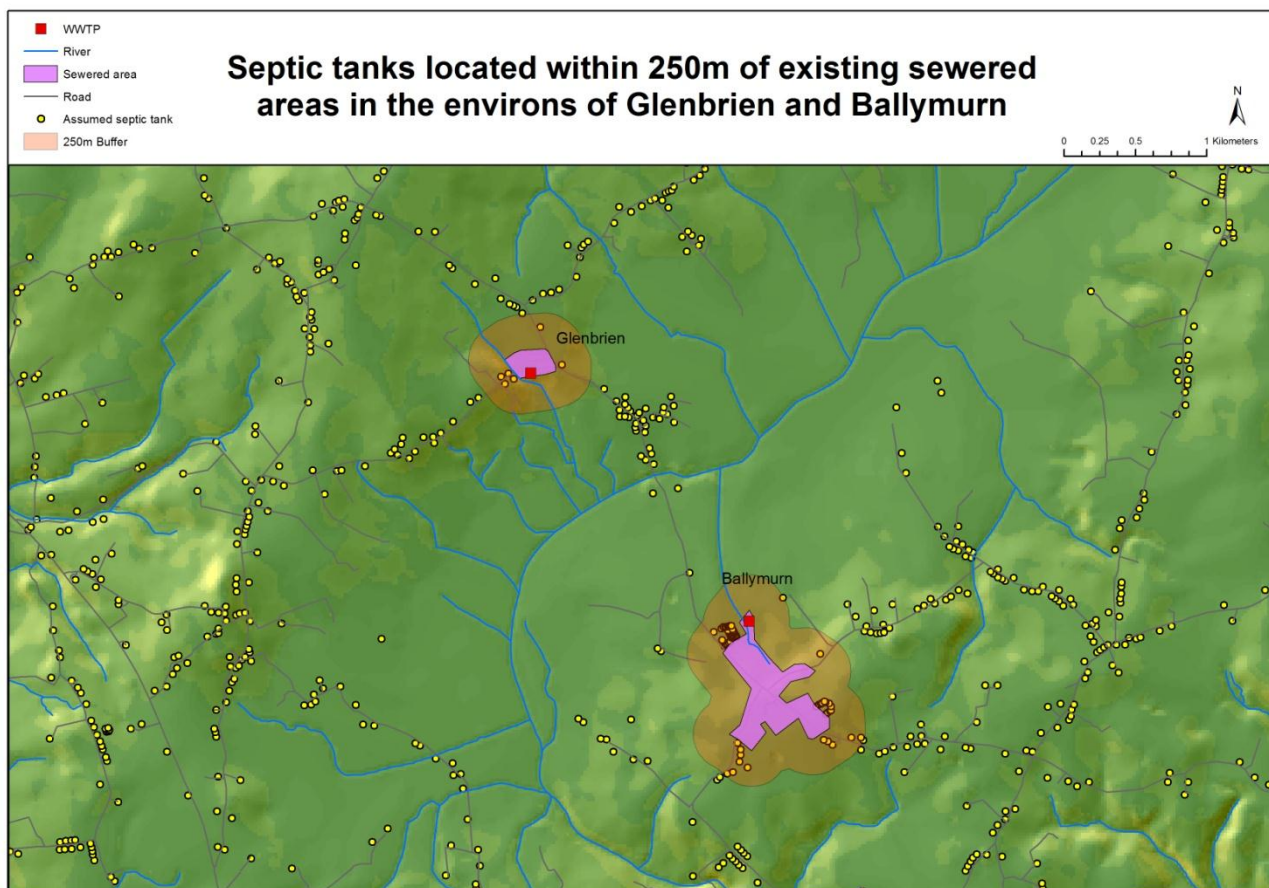


Fig. 3. Buffering indicating proximity of septic tanks to existing sewerage networks.

existing capacity of the centralised facility, as well as the engineering design of how to get the effluent to the sewer network, will then need to be made. Figure 3 illustrates the buffering process based around the existing decentralised WWTPs in Glenbrien and Ballymurn. Buffering enables the identification of OSWTS locations or potential development sites within a specified distance of an existing sewerage network. In this example 250 m is the radius, indicated by the beige zones, but users will be able to specify custom distances. For example, 100 m is indicated in the model architecture of Figure 1, which reflects a bylaw in County Cavan that states that any dwelling within 100 m of mains sewerage must be connected. Once locations are inputted, it will be possible to estimate the cost of connection. However, if the model indicates that it would be economically prohibitive to connect, or the existing treatment plant and/or sewerage capacities are insufficient, the next model stage is initiated: assessment of OSWTS density.

Connect to decentralised network

The density of OSWTs in the areas of inadequate percolation is then calculated in order to assess whether it would be feasible to cluster several OSWTs together into a small bore sewer network to discharge to a decentralised treatment plant with licensed discharge to surface water. This would have the advantage (particularly from the Local Authority's perspective) of only having a single consent to manage which covers several houses.

OSWTS density is also an important factor when assessing the risk of groundwater and/or surface water contamination. Although a properly functioning OSWTS and percolation area can attenuate most pathogens before the effluent reaches groundwater, high density installations can cause nitrogen contamination (EPA, 2009). Alternatively, high density in zones of low permeability subsoils can lead to significant surface water contamination due to the cumulative pollutant loading. Research on on-site systems in Ireland constructed and operating according to the Code of Practice (EPA, 2009) and in free draining subsoils has shown that densities of approximately 300 systems per km² could be acceptable with

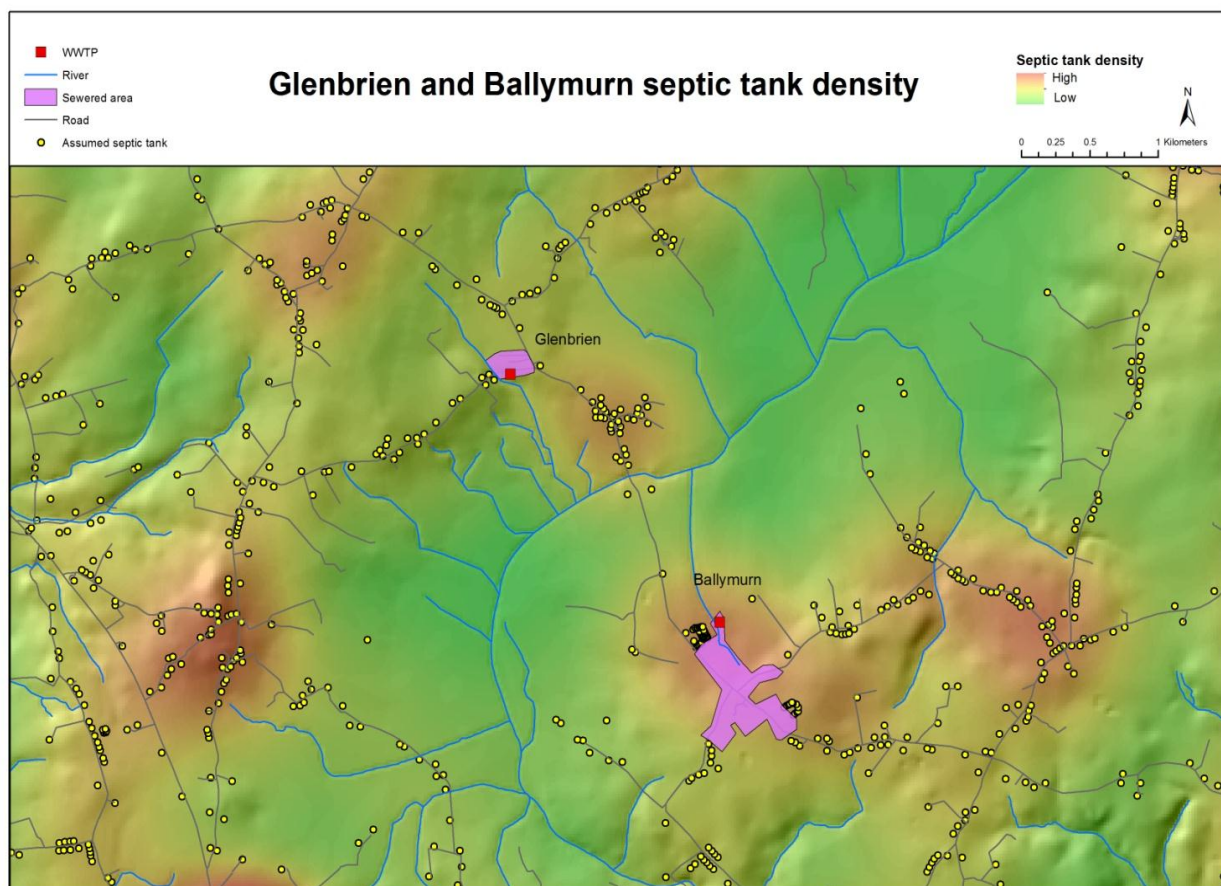


Figure 4: Septic tank system density.

regards to nitrogen loading (Gill, 2007). However, often the nominal density threshold chosen to indicate potential vulnerability is much lower to take account of poorly designed and constructed systems, as well as unsuitable subsoils (in particular, low permeability subsoils - the focus of this study). For example, the US EPA designation is that any density greater than 16 systems/km² constitutes a 'region of potential contamination' (Yates, 1985). Moreover, anecdotal evidence in Ireland suggests that Local Authorities consider 8 systems/km² to represent the nominal critical threshold.

For the purposes of this model however, the density of OSWTSs and proximity to a water course are critical parameters when assessing the feasibility of a small bore sewerage upgrade through clusters of single house developments, to convey the partially treated effluent to a decentralised plant with consented discharge. Figure 4 illustrates septic tank densities around Glenbrien and Ballymurn. Green areas indicate densities in the order of 0-5/km², green/yellow areas indicate 10-16/km² and orange/red areas indicate densities in the order of 40/km². As an example, Figures 2 and 4 show that the settlement immediately to the south-east of Glenbrien is situated in an area with *very high* likelihood of inadequate percolation to groundwater with a high OSWTS density of 32/km². In addition, the distance of the nearest system to the main watercourse in this area, the River Sow, is only 60 m. As such, this would indicate a settlement that might be a candidate for clustering, although more detailed design and on-site assessment would be required to confirm this.

If the OSWTS density is less than the threshold value (yet to be determined) which indicates whether clustering OSWTSs is worth investigating further, the next tier of potential solutions are initiated which moves into solutions at a single house level.

Imported media filter

The next potential solution is to evaluate whether the depth of low permeability subsoil is shallow enough such that it could be replaced by a more suitable imported media (soil or sand) through which effluent percolates down into more permeable subsoil or bedrock zones. This is assessed by incorporating the mapped areas of extreme vulnerability which will indicate if the site is likely to have a depth to groundwater of less than 3 m. On-site assessment would then be required to determine the subsoil profile, depth of unsaturated zone and likely bedrock permeability.

Pressurised dispersal systems (LPP or DD)

Another solution that may be suitable at a single house level is the use of pressurised effluent dispersal systems such as Low Pressure Pipe (LPP) or Drip Dispersal (DD) which may prove to be acceptable in subsoils with T-values greater than 75. This is currently the focus of an EPA funded project whereby such systems are being tested in the field in areas of low permeability to assess a maximum T-value subsoil in which they would be applicable. Once this upper bound on T-value is available it can be incorporated into the decision support model.

Zero discharge solutions

Finally, if all the above potential solutions are ruled out for a site in a low permeability subsoil area then it may be possible to use a zero effluent discharge solution, such as a zero discharge evapotranspiration system (often known as a willow system) or a large sealed cesspool tank in which the effluent collects until it is periodically tankered away to a centralised treatment facility. Both these solutions have their own constraints which will need to be factored into the model; for example, available land area on site for the willow systems (as they typically require a large surface area) and frequency of tank emptying required versus cost for the cesspool solution.

DISCUSSION AND CONCLUSION

A preliminary modelling architecture has been constructed and several linked geoprocesses have been identified that will assist in determining strategies for on-site wastewater treatment and disposal in areas of low permeability subsoil. The modelling remains complex however and is designed to be a decision support tool; the final decisions are still dependent on on-site constraints to refine output solutions.

There are several aspects of the modelling that require further development. For example, modelling to assess the assimilative capacity of surface waters has not been developed at this stage, and a mechanism for evaluating sustainability has yet to be incorporated. Subsequent coding of the model is also required to enable web-based application. Another aspect that needs to be accounted for within the modelling process is an automatic assessment of how much spare capacity might there be in existing centralised facilities. The model will also be able to indicate whether new planning applications should trigger remedial actions due to an increase in OSWTS density or hydraulic load.

With further model development, the identification of potentially high risk areas (such as that found south-east of Glenbrien) will become more expedient. In addition, the model could be adapted to assist Local Authorities in deciding the optimum locations for new decentralised WWTPs and identifying which facilities should be upgraded should the need

arise. As such, the model is aiming to be a flexible wastewater decision-making toolset for Local Authority planners and managers to evaluate alternative strategies and reduce the risk posed by current domestic OSWTS solutions.

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REGULATION OF DISCHARGES FROM SEWERAGE TREATMENT SYSTEMS SERVING SINGLE HOUSES

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INTRODUCTION

The Northern Ireland Environment Agency (NIEA) of the Department of the Environment is responsible under the Water (Northern Ireland) Order 1999 for promoting the conservation of water resources and the cleanliness of water in waterways and underground strata. NIEA is responsible for the implementation and enforcement of environmental regulations, many of which require operators and their activities to be authorised and monitored where appropriate. The Agency is required to achieve full cost recovery for such regulatory functions through the application of fees and charges on customers we regulate. One of the ways in which NIEA exercises these functions on behalf of the Department is by controlling effluent discharges to waterways or the underground stratum, through the granting of discharge consents.

Applications for single domestic dwellings have averaged almost 3000 per year since their introduction. These require thorough scrutiny before a consent can be granted in order to protect the aquatic environment. Where an application is for discharge to the underground stratum, (soakaway) the applicant must arrange for soil percolation tests to be carried out. The results are submitted to NIEA to determine if the ground is suitable and also the length of the soakaway that will be required to operate an efficient soakaway.

LEGISLATION

On 24 August 2001, the Water (Northern Ireland) Order 1999 ('the Water Order') replaced the Water Act (Northern Ireland) 1972. Under Article 7 of the Water Order, it is necessary to obtain the consent of the Department of Environment to make a discharge from a house or other premises, to a waterway or to the underground stratum. This is a legal requirement and failure to do so is an offence under the legislation. The requirement to obtain consent to discharge applies to all proposed discharges and also to pre-existing discharges, irrespective of the date of commencement. Since 24 August 2001, all new discharge consents have been granted under the terms of the Water Order. However, discharge consents previously granted under the terms of the Water Act that were 'live' at 24 August 2001 became valid under the terms of the Water Order.

Under the terms of Article 11 of the Water Order, and in line with Government Policy on the 'Polluter Pays' principle, the Department has the powers to raise, through a scheme of fees and charges, sufficient funds to cover the Department's costs relating to:

- i. the processing of all applications for discharge consent, and also

- ii. the monitoring of all discharge consents which are checked by NIEA for compliance with consent conditions under the Department's compliance assessment monitoring programme.

The application fees element of the charging scheme was implemented on 29th October 2001, and applies to all persons making application for consent to discharge, for review of an existing consent or for the transfer of an existing consent. Under Schedule 1 paragraph 8 of the Water Order, a person selling or transferring ownership of consented premises is required to notify the Department within 21 days of the transfer occurring. NIEA charges a fee to the person who will take on responsibility for the discharge consent, to cover the administrative cost of preparing and re-issuing the consent.

REVIEW

There has been significant concern raised by local stakeholders during the River Basin Planning communication processes surrounding the impact of septic tanks on water quality. Based on this and the Agency's desire to review the regulatory process in terms of the Better Regulation Agenda, a review was launched in 2009, with SNIFFER (Ref) and North South Share (Ref) projects determining the evidence base. Three discrete work packages were undertaken for these projects, covered under the following headings:

- i. Diffuse source septic tanks assessment - NS Share
- ii. Legislative review/Responsibilities - SNIFFER
- iii. Review of Impacts – SNIFFER

The conclusions and recommendations from all three reports are presented below;

REVIEW OF DIFFUSE SOURCE SEPTIC TANKS ASSESSMENT

Information on waste water treatment in Northern Ireland is recorded in several databases, belonging to NIEA and other agencies, for operational, regulatory and reporting purposes. However, the baseline for this project was the NIEA Water Management Unit's Domestic Consents Database. Due to the poor quality of the data in this database, specialist expertise in the area of address matching and database cleaning was required for this project. Automatic address matching was completed on NIEA Water Management Unit's Domestic Consents Database. The automatic address matching process has successfully matched 47.5% of the records in the Domestic Consents Database. The remaining 52.5% (55,301 records) must undergo interactive cleaning. A customised interactive cleaning methodology is provided in a separate document – 'Database Cleaning Report and Methodology'. The process is built around QuickAddress Batch but other resources are also used including GIS mapping, NIEA's gazetteer website and the original paper-based applications.

RECOMMENDATIONS

1. Interactive address matching on the remaining unmatched records must now be undertaken by NIEA. The time and effort required to complete interactive cleaning is considerable. It has been estimated by the contractor that one person could validate the database to between 65 and 75% over a period of up to two years. Options to consider would be date limiting, i.e. limiting back to a specific year.
2. Whether or not NIEA elect to undertake the lengthy interactive matching process the current domestic consenting application process should be reviewed to ensure that future entries into the Domestic Consents Database contain all the required information. Addresses should be entered in their full, postally-correct state and spatial references should be included. QuickAddress Batch, GIS and NIEA's gazetteer tool can be used for this purpose. This would require a simple redesign of the IT package and an IT business out.
3. At present, less than half of the records in the Domestic Consents Database have verified spatial coordinates and can be mapped. Therefore, consideration should be given to applying the methodology developed by the Western River Basin District (WRBD) to generate a map of unsewered properties.

LEGISLATIVE REVIEW/RESPONSIBILITIES

Objectives

- A. To examine legislative requirements and responsibilities and identification of best practice in relation to On Site Wastewater Treatment systems (OSWTS) and,
- B. To review the scientific literature on the impacts of OSWTS and small Wastewater Treatment Works (WWTW) on water quality, identify methods used to estimate / quantify the nutrient loadings/impacts in discharges from small WWTW on water quality, and provide recommendations on a suitable methodology.

Key findings and recommendations

1. Existing OSWTSs are located on a GIS database that will facilitate a rational apportionment of risk in relation to other environmental pressures and sensitivities;
2. More formal arrangements are made for a systematic review of OSWTSs. This should prioritise those systems that are judged to pose the greatest risk of pollution. Such measures should include steps to ensure all systems are adequately maintained;
3. For new properties, planning and environmental regulation are better integrated such that measures to mitigate environmental impact are considered both at the earliest stage in the planning process and when final approval is given;
4. Responsibility for the preparation of applications should be placed on the applicant in order to reduce the administrative burden on regulators and to avoid their having to give specific guidance to individual applicants. It is intended that completion of

applications will rely on independent guidance on effluent treatment, so that all factors relevant to an application are considered and appropriate measures taken before the consent to discharge application is made. If possible, the use of an approved agent should be made mandatory;

5. A set of qualification criteria for an approved agent or adviser on effluent treatment should be developed. These criteria should include training and a specification for soil percolation testing. This should be supplemented with a code of practice and an approved list of advisers/assessors who will act as agents in the application for consents to discharge.
6. A database should be developed for the results of site assessments and associated decisions, including details of sites assessed as unsuitable for OSWTs (and the reasons for this assessment);
7. Where codes of practice are not already in place a revised approach to the regulation of OSWTs should be incorporated into a formal policy towards the administration and regulation of OSWTs which would also include a code of practice.
8. The concept of licencing domestic consent needs considered for risk modelling under Better Regulation principles.

REVIEW OF IMPACTS

Objectives

The objective of this project is to carry out a review of the impacts of OSWTs and small WWTW and in particular;

- A. The loadings – and in particular, the nutrient (nitrogen and phosphorus) loadings – and the microbial loadings associated with OSWTs discharges and small WWTW.
- B. The cumulative loadings of discharges from OSWTs and small WWTW within catchments.
- C. The impacts of discharges from OSWTs on water quality and ecology, with particular regard to the issue of eutrophication, and to factors affecting the risk of significant discharge impacts (e.g. size/nature of discharge and receiving water, chemical form of nutrient, discharge/timing/consistency).
- D. The relationship between OSWTs maintenance and the impact of discharges on water quality.
- E. Elucidate methods used to estimate/ quantify the nutrient loadings in discharges from OSWTs and small WWTW and their impacts on water quality, and provide recommendations on a suitable methodology for undertaking those tasks.

Key findings and recommendations

1. The impact of OSWTs on water quality and ecology is varied and unpredictable. Lack of information on the distribution and condition of these systems has been the main obstacle in assessing the contribution of OSWTs to water pollution.

Furthermore a lack of research relating specifically to the impact of OSWTs on a national scale has meant that assumptions have had to be made based on general water quality monitoring data. The use of such data is limited in that the impact of OSWTs cannot be isolated from other pressures.

2. Impact of septic tanks arises from the discharge, directly or indirectly, of three main contaminants: nitrates, phosphates and microbiological pollutants. In terms of impacts, nitrates and phosphates are more of a concern for ecology, whereas microbiological contaminants can have more serious consequences for human health if allowed to enter drinking water supplies.
3. Significance of OSWTs vary between and within countries, in general contaminant loadings in Northern Ireland are lower than in Scotland, but the importance of OSWTs appears to be higher. Lack of data for ROI makes a comparative assessment difficult.
4. Microbiological contaminants are of greatest concern of the three pollutants due to their implications for human health and as a result of the high concentrations which are able to reach surface and ground waters in comparison to those from other sources. Numerical modeling studies carried out by SNIFFER (2006) indicate that impacts are highest in Northern Ireland, where OSWTs contribute over 40% of diffuse microbiological contamination in comparison to less than 25% in Scotland. Comparable data for ROI are not available.
5. The contribution of OSWTs to nitrate pollution is relatively small when compared to other sectors. This ranges from approximately 4% in Northern Ireland and the Republic of Ireland to 6% in Scotland.

Outcome

NIEA has now completed a review of how applications for single domestic dwelling consents to discharge are handled and has redesigned the application process bearing in mind the findings of the projects above. The new processes are aligned to deliver against the ethos set out in the current draft Government White Paper on Better Regulation and apply to two distinct discharge types:

1. Application for Consent to Discharge to the Underground Stratum; and
2. Application for Consent to Discharge to a Waterway.

The restructuring and revision of application fees for single domestic dwellings has been developed with the aim of recovering the full costs associated with applying the legislation.

REVISED REGULATION PROCESS

Planning and Consent

Applicants should be aware that all discharges of sewage effluent require consent under The Water (Northern Ireland) Order 1999, which is independent of planning legislation. It is recommended that Water Order consent is applied for before planning permission. This

ensures that WMU will have assessed the site's suitability for sewage disposal before being asked by Planning Service to respond to the planning consultation. Should the applicant bypass WMU and apply for planning permission, WMU will not be in a position to respond to the planning consultation until an application for consent under The Water (Northern Ireland) Order 1999 has been determined, which may delay the planning decision.

Policy Statement

The Department's preferred option for sewage disposal is discharge to public sewer. Where the applicant can demonstrate that discharge to public sewer is not practicable, the sewage effluent should be discharged to the underground stratum via a sub-surface irrigation system, provided the ground conditions are suitable. Where the ground conditions are unsuitable, site improvement works should be undertaken to determine whether discharge to underground stratum can be accommodated. Discharge to a waterway may be allowed where the ground conditions are unsuitable for discharge to underground stratum.

Application for Consent to Discharge to the Underground Stratum

Implementation of the new processes will reduce the technical and administrative burden on both the applicant and the Department in processing applications for consent to discharge to the underground stratum. This has resulted in a small reduction to the application fee.

Where the applicant can demonstrate that discharge to a public sewer is not practicable, the sewage effluent should be discharged to the underground stratum via a sub-surface irrigation system, provided the ground conditions are suitable.

The application must clearly demonstrate that the proposal meets all the criteria listed below so that consent to discharge can be granted based on standard consent conditions. The specific details of the criteria required for Assessed Registration are consistent with extant legislation, current industry Codes of Practice and applicable British Standards.

1. Connection to the Northern Ireland Water Limited foul sewerage system is not feasible or practicable, or is not within 30m from the curtilage of the dwelling.
2. The discharge is solely of sewage from a single domestic dwelling and contains no surface water drainage.
3. The septic tank to be installed is certified as manufactured to BS EN 12566-1 or the packaged wastewater treatment plant to be installed has BS EN 12566-3 certification and has a removal efficiency for Biochemical Oxygen Demand of at least 95%.
4. Any sub-surface irrigation system is designed and installed in accordance with Departmental guidance, which is consistent with BS 6297:2007 + A1:2008.
5. Ground conditions are appropriate to offer effective treatment and dispersal of any sewage effluent discharged (percolation test average Vp value to be > 15 <100). Any

percolation test must be undertaken by someone who is familiar with the requirements of BS 6297:2007 + A1:2008.

6. Any drainage field is appropriately sized in relation to the reported percolation test results.
7. The discharge is not within 50m of any potable water supply.

Application for Consent to Discharge to a Waterway

The Department must be satisfied that any treatment system proposed will provide adequate protection to the receiving waterway.

It is proposed to move towards non-numeric consents for discharges to waterway from single domestic dwellings. Therefore any consent issued for systems not in place at the time of application will require compliance with the relevant British Standards (currently BSEN12566:2005 Part 3).

The requirements for discharge quality are based on the size of the catchment area of the waterway into which the discharge is to be made. In general, the smaller the catchment area of the waterway at the discharge point, the lower the flow in the waterway, and hence the lower the dilution available at the point where the effluent will be discharged. Therefore tighter discharge limits must be imposed for lower flowing waterways to reflect the fact that there is less available flow to dilute the effluent to a level which enables achievement and maintenance of the quality objectives for the waterway.

As stated, it is proposed to move towards non-numeric consents; therefore an applicant must demonstrate that any proposed treatment system can achieve a minimum percentage reduction in pollutant loading. This is measure in terms of Biochemical Oxygen Demand (BOD) removal.

The proposed treatment system must carry the appropriate CE marking, described in Annex ZA.3 of BSEN 12566-3:2005, demonstrating the appropriate treatment efficiency. The laboratory undertaking the testing must be able to demonstrate that they are a notified body for Part 3 of BSEN 12566:2005. As stated the treatment efficiency required will be determined by the dilution afforded by the proposed receiving waterway. This information will be obtained in the form of a catchment size.

The table below details the required minimum treatment efficiencies in relation to catchment size

Table 1: Required Treatment Efficiencies in Relation to Catchment Size

Catchment size of receiving waterway at discharge point	Minimum treatment efficiency* required (% Biochemical Oxygen Demand (BOD) reduction)
Greater than 2km ²	95%
Less than and including 2km ²	97.5%**

** Treatment efficiency based on a maximum influent loading of 500mg/l, as stipulated by Annex B 3.2 of BSEN 12566:2005 Part 3.*

***or 95% followed by an appropriate form of tertiary treatment designed to further reduce BOD concentration to give an overall reduction of at least 97.5%.*

It is recognized that there are treatment technologies available for which the appropriate British Standard has not yet been agreed and adopted. This may include systems such as peat filters, sand filters, willow filtration and reedbeds. Where such systems are to be considered, the applicant must satisfy the Department that the treatment system proposed will provide at least the level of pollutant removal required for the size of the catchment into which the discharge is to be made.

Where a discharge already exists but is not causing a visible impact on the receiving waterway, numeric discharge standards will be set commensurate with the optimum performance of the existing treatment system. No upgrade of the treatment system will be required. However no standard less stringent than a 40 mg/l Biochemical Oxygen Demand and a 60 mg/l Suspended Solids should be set.

Should an existing discharge be determined as causing a visible impact on the receiving waterway, the system will require upgrade to a system compliant with the requirements of BSEN 12566:2005 Part 3, which demonstrates the appropriate treatment efficiency.

Considerations

Apart from the site requirements laid out in this document, NIEA has other considerations to take into account before determining whether or not to consent a discharge. These include the density of dwellings in the immediate vicinity which are not on mains sewerage, or any Natural Heritage (a Directorate of NIEA) designations which may restrict the site. It is recommended that the applicant uses the services of a suitable engineer/architect to undertake:

1. Completion of the application and sign off.
2. The site suitability examination.
3. Design and installation of the sewerage system serving the domestic premises.

It may also be useful to commission the services of a professional hydrogeologist to assist with site suitability assessment.

It is an offence under the Water (Northern Ireland) Order 1999 to knowingly or recklessly make a statement which is false or misleading in any material particular for the purpose of obtaining a discharge consent. The Department will not hesitate to instigate enforcement action should an applicant be found to have willingly provided such information.

Failure to install the sewage treatment facilities in the manner described in the application (or in the case of existing systems to accurately describe the current condition on site) or to maintain as per the consent conditions shall be considered as non-compliance and therefore an offence under of the Water (Northern Ireland) Order 1999. Where the Department deems that this is the case, the Department's consent to discharge may be declared invalid and appropriate enforcement action taken.

Maintenance

Satisfactory long-term performance of the drainage field depends on correct operation and maintenance of the upstream system. The septic tank or sewage treatment works should be regularly desludged, and maintained and operated according to the manufacturer's instructions. The septic tank or sewage treatment works should be inspected regularly (at least annually). The drainage field should be inspected on a monthly basis to check that it is not waterlogged and that effluent is not backing up into the upstream system. Particular care should be taken to avoid compaction or disturbance of the area over and around the drainage field. Maintenance information should be recorded and retained by the building owner and occupier.

Monitoring & Compliance

In order to offer effective monitoring and enforcement of compliance with consent conditions, a robust and proportionate monitoring program is required. Elements of an effective monitoring and compliance protocol include:

- The consent holder maintaining any Septic tank/soakaway/package wastewater treatment plant as a condition of the consent.
- A maintenance and service log of any package wastewater treatment plant being kept and offered to officers of the Department for inspection on request.
- A record of de-sludging of any system being kept and offered to officers of the Department for inspection on request.
- A program of compliance inspection visits by NIEA implemented to a selected number of new and existing single domestic discharges each year.

Selection of sites for compliance monitoring will include both new installations and existing systems.

- Inspections of newly installed systems will ensure installation and operation in accordance with application details and consent conditions. The inspection shall include visual inspection of all elements of septic tank or sewage treatment system.
- Inspections of a number of existing systems will help monitor operation in accordance with the consent conditions. The inspection shall include visual inspection of all elements of septic tank or sewage treatment system and a request to view documentation of de-sludging and maintenance.

CURRENT NIEA INSPECTION PLAN

Sites are currently prioritized on the following basis for compliance monitoring:

- Inspections to catchments that have historical pollution issues;
- Where domestic sewage effluent is considered an issue within a Local Management Area Action Plan under the Water Framework Directive River Basin Management Plans.

45 Water Quality inspectors employed on behalf of NIEA are trained and equipped to undertake the investigations on a regional basis throughout Northern Ireland. Currently there are approximately 1000 inspections of septic tanks as a result of applications or pollution issues undertaken on an annual basis with approximately 3000 undergoing some level of intervention, from prosecution, to Enforcement Notice, to Warning Letter and advice.

FUTURE PROPOSALS

NIEA are currently developing a risk based methodology using GIS to include the following parameters:

- Overlay Bad and Poor Water Framework Directive classified areas
- Overlay Areas of Special Scientific Interest, Special Protection Areas and Special Areas of Conservation
- Overlay Local Management Areas (LMAs) and map against priority LMAs, e.g. Kingsmill waterbody.
- Unconsented properties – Discovered by mapping sewerage and unsewered areas, overlaying a database of all known residential properties in the area, and checking against consent records; or by inspection; or incident reports.

This methodology will provide a specific number of targeted “higher risks sites that NIEA will incorporate into a wider inspection and monitoring plan for the regulation of discharges.

ENFORCEMENT/INTERVENTION

Under the Water (NI) Order 1999, it is an offence to discharge effluent to waterways or water contained in any underground strata without the consent of the Department of the Environment (DOE). Discharge of effluent includes any sewage effluent from residential premises. Domestic consent holders are notified when their discharge is found to be in breach of their consent conditions and are asked to investigate the reason for any exceedances, and to supply the Department with details of their plans to remedy the situation.

Where consent holders do not remedy the situation within a reasonable timescale, dye tests may be performed and/or samples may be collected with a view to prosecution for breach of consent.

Failure to comply with the conditions of a consent is an offence under Article 7(6) of the Water Order which can on summary conviction lead to a maximum fine of £20,000 or a maximum prison sentence of three months (two years on conviction on indictment) or both. As Domestic properties are considered low risk sites the Department is mainly reactive to any reports of unconsented domestic discharges or domestic discharges believed to be breaching their consent. Reports can come from members of the public, NIEA staff and authorised Water Quality Staff or members of other organisations who find problems in the course of their work.

Proactive monitoring is currently driven by river quality data held by NIEA. Areas which are in river catchments classified as poor under the Water Framework Directive come under the scope of targeted programmes aimed at finding problems with domestic discharges in the area in addition to industrial, water utility and agricultural discharges. The Domestic Consent Breach Matrix, table 2, below directs the action taken by the Agency when Consent Holders breach their consent.

Complaint received/issue noted	Dye test and/or sample pass			
Investigation to include dye test and/or sample taken	No further action			
	Dye test and/or sample fail	no response	no response	Dye test and/or sample pass
	Advisory Letter	Warning Letter	Enforcement Notice	No further action
			dye test and/or sample taken at end of notice period	Dye test and/or sample fail
		response received & timescale for works agreed	Advisory Letter	Prosecution
				Dye test and/or sample pass
		dye test and/or sample taken at end of agreed period	Enforcement Notice	no further action
				Dye test and/or sample fail
				Dye test and/or sample pass
				Prosecution

Table 2: Domestic Consent Breach Matrix

REFERENCES

Code of Practice: “*A Guide For Users Of Sewage Treatment Systems*”, ISBN 978-1-903481-13-4.

Council Directive 92/43/EEC on the Conservation of natural habitats and of wild fauna and flora (The Habitats Directive)

Council Directive 2009/147/EC on the conservation of wild birds (The Birds Directive)

British Water Code of Practice, Flows & Loads 3, Sizing Criteria, Treatment Capacity for Small Wastewater Treatment Systems (Package Plants), ISBN 1 9034810 5 8

BS 6297:2007+A1:2008, Code of practice for the design and installation of drainage fields for the use in wastewater treatment. (plus any future amendments)

BS EN 12566-3:2005+A1:2009 Small wastewater treatment systems for up to 50 PT. Packaged and/or site assembled domestic wastewater treatment plants (plus any future amendments).

Building Regulations (Northern Ireland) 2000 (specifically Part N).

Environmental Protection Agency Code of Practice (2009), Wastewater treatment and disposal systems serving houses (p.e. ≤ 10)

Environmental Protection Agency, Draft consultation on technical guidance for the registration of exempt discharges of sewage effluent to surface and groundwaters under the Environmental Permitting Regulations, 2010.

“NIEA procedure on dealing with Water (Northern Ireland) Order 1999 applications for discharge consents to small waterways” – **internal NIEA document**

The Conservation (Nature Habitats, etc.) Regulations (Northern Ireland) 1999

The Construction Products Regulations 1991

The Controlled Activities Regulations (2005) – Scotland (SEPA) 2009, WFD96 Review of the Legislative Requirements and Responsibilities Relating to On-Site Wastewater Treatment Systems and Their Impact on Water Quality: Work Package A, B and C. Project funders/partners: SNIFFER, Northern Ireland Environment Agency, Scottish Environment Protection Agency, Environmental Protection Agency (EPA) Ireland, Northern Ireland Water, Department of the Environment Northern Ireland Planning and Environmental Policy Group.

North South Shared Aquatic Resource; Further Characterisation WP2.2: Refined diffuse source septic tanks assessment; Database Cleaning Report and Methodology

A RISK BASED METHODOLOGY TO ASSIST IN THE REGULATION OF DOMESTIC WASTE WATER TREATMENT SYSTEMS

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ABSTRACT

The Environmental Protection Agency (EPA) is proposing to use a risk based methodology to assist in the apportionment of inspections of domestic waste water treatment systems, taking account of the relative risks to human health and the environment from pollutants such as microbial pathogens, molybdate reactive phosphate (MRP) and nitrate. The resulting methodology is based on the source-pathway-receptor model for environmental management and uses available soils, geological, hydrogeological and settlement density data in a Geographic Information System (GIS) environment. The methodology involves estimating the pollutant load generated by domestic waste water treatment systems (DWWTs) based on housing density, the average number of people per house, typical phosphorus (P) and nitrogen (N) load per person, and typical reductions in concentrations in the 'tank' component of DWWTs. Hydrogeological information and maps of soils, subsoils and bedrock are used to evaluate the movement and attenuation of pollutants arising from DWWTs. The resultant pollutant load arising is applied to the land environment and predictions are made on the consequences. The broad resultant scenarios are: i) the area can enable satisfactory treatment of domestic waste water effluent in the subsoils; ii) inadequate percolation may occur with the possibility of ponding and movement of effluent into surface water without adequate treatment; or iii) effluent could enter groundwater without adequate treatment. In circumstances where treatment and disposal of effluent is unlikely to be satisfactory, the resulting molybdate reactive phosphorus (MRP) and nitrate load is 'added' to the effective rainfall in the case of surface water, and to the recharge in the case of groundwater, arising across a 1 km² grid country-wide, and a concentration is estimated for these areas. These concentrations are then compared to relevant environmental standards for phosphate and nitrate. The risk is then ranked countrywide into four categories (i.e. low, medium, high and very high) for both phosphate and nitrate based on the estimated concentrations in each 1 km² grid area. Microbial pathogens are taken to be influenced by pathway factors in a similar manner to phosphate. The land surface of the country is sub-divided into four categories: i) relative risk of ponding and pollution of streams by MRP and microbial pathogens via the surface pathway due to inadequate percolation; ii)

relative risk of pollution of streams and wells by MRP and pathogens via the subsurface pathway due to inadequate attenuation; iii) relative risk of pollution of streams by nitrate via the surface pathway due to inadequate percolation; and iv) relative risk of pollution of streams and wells by nitrate via the subsurface pathway due to inadequate attenuation. The resulting maps should not to be used for predicting precise impacts; they are intended to show relative risk on which an inspection regime can be based.

Keywords: domestic waste water treatment systems; risk and risk ranking; phosphate; nitrate; microbial pathogens; surface water; groundwater.

INTRODUCTION

The *Water Services (Amendment) Act, 2012* (S.I. No. 2 of 2012) provides for the regulation and inspection of domestic waste water treatment systems (DWWTs) across Ireland. Under the Act, the Environmental Protection Agency (EPA) is responsible for making a National Inspection Plan having regard to relevant risks to human health and the environment. In response to this requirement, the EPA has developed a methodology for organizing inspections of DWWTs based on the source-pathway-receptor (S-P-R) model for environmental management (EPA, 2012). The specific purpose of the methodology is to assist in apportionment of the inspections relative to the risk presented by DWWTs.

DWWTs located, constructed, installed and maintained in accordance with the current best practice guidance generally provide adequate treatment and disposal of domestic waste water. However, where the location, construction, operation and/or maintenance are inadequate, impacts may occur. The methodology, which is summarised in this paper, focuses on the issues that may arise in the areas that are problematical with regard to inadequate percolation and/or insufficient attenuation. Three pollutants were taken as representative of the threat posed by discharges from DWWTs to human health and water quality – molybdate reactive phosphate (MRP), nitrate and microbial pathogens.

DEVELOPMENT OF RISK-BASED METHODOLOGY

The development of the methodology was influenced by:

- the current understanding of the hydrological and hydrogeological settings present in Ireland;
- results of research on DWWTs and hydrogeology undertaken in Ireland;
- data and map information available as GIS datasets.

The analysis of spatial data in a GIS is critical to the methodology. The mapping scale of approximately 1:40,000 enables comprehensive regional assessments to be undertaken, but is insufficient to enable local variations to be captured. All calculations are undertaken across a 1 km² grid country-wide.

RISK CHARACTERISATION

The risk characterisation is based on the combination of the following S-P-R elements (see Figure 1):

- Pollutant load from each DWWTS, derived from typical discharge concentrations and quantities.
- Pathway susceptibility, which includes consideration of attenuation by physical, biological and chemical processes. Two pathways are considered: surface and subsurface.
- Cumulative load entering the surface water or groundwater environment derived from DWWTS density and estimation of attenuation.
- Dilution of load at the water receptor.
- Risk ranking using estimates of predicted pollutant concentrations at the receptor.
-

POLLUTANT LOAD (SOURCE CHARACTERISTICS)

The pollutant load is derived by combining typical effluent quantity and quality from each DWWTS. The average input of pathogens, MRP and nitrate from an individual DWWTS, prior to treatment in the subsoil or polishing filter, are given in Table 1.

RECEPTORS OF CONCERN

The main receptors of concern are human health from direct contact with microbial pathogens, surface water from eutrophication and/or polluted groundwater being used as a private water supply (*e.g.* untreated well water).

Table 1: Data for the calculation of overall load

Input Parameter	Input Value
Pathogen Load (<i>E. Coli</i>)	5,000 -1 million per 100 ml
Phosphorus load in kg per person/year (in liquid discharge leaving Septic Tank)	0.5

Nitrogen load in kg per person/year (in liquid discharge leaving septic tank)	2.7
Persons Per House ⁴	2.8
Density of Houses	Variable

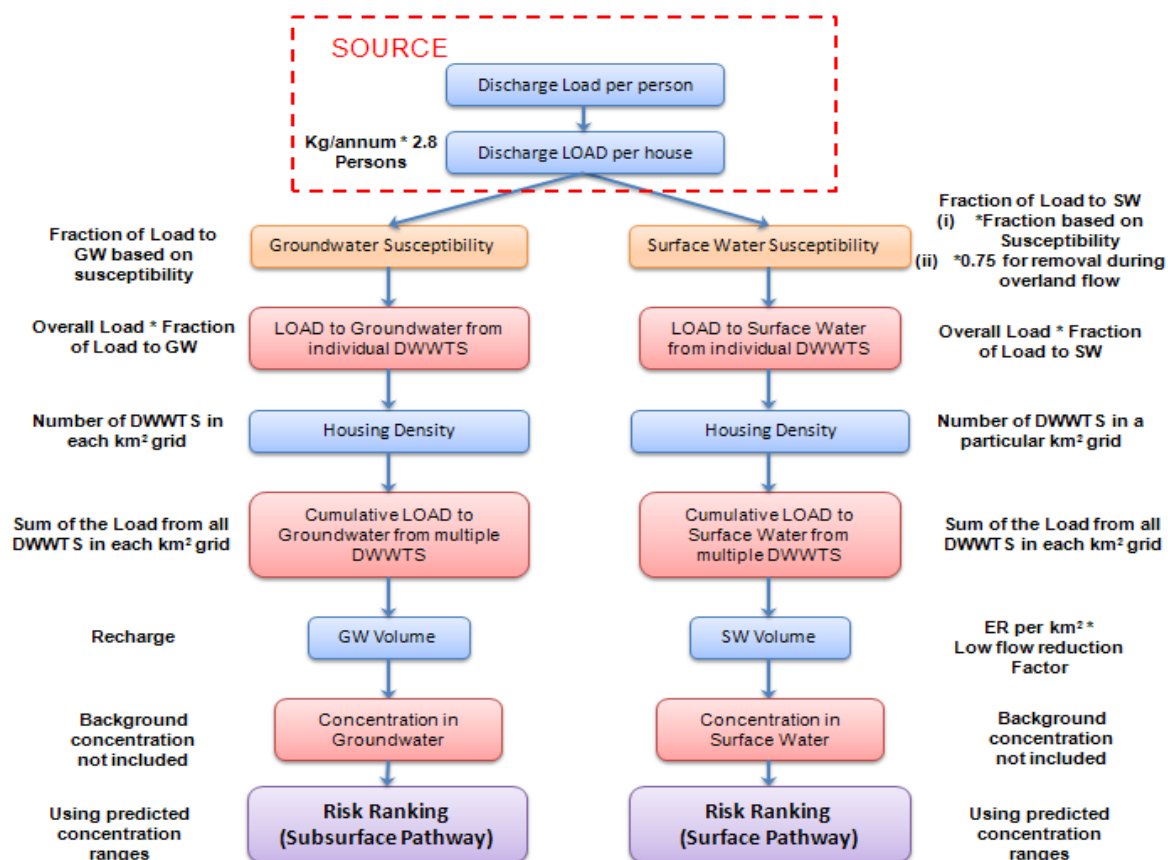


Figure 1: Outline of methodology for risk rank

⁴ Based on CSO data.

PATHWAY SUSCEPTIBILITY

The pathway is the link between the source of pollution and the receptor, and can either be at or close to the surface or deeper underground, or a combination of all three. Natural vertical and horizontal pathways for effluent migration are determined by the on-site geology and hydrogeology, particularly by the nature of the soils, subsoils and underlying aquifers. Artificial pathways may include drainage ditches, land drainage pipes and stream culverts.

The characteristics of both the surface and subsurface pathways are defined by the '**pathway susceptibility**', which is a measure of the degree of attenuation between source and receptor.

Susceptibility matrices were developed in the context of two over-arching environmental scenarios:

- **Inadequate percolation**, which may result in surface ponding of effluent, bypass directly to surface water and the associated threats to human health and surface water quality, and
- **Insufficient attenuation** (subsurface treatment of the effluent), which may result in directly polluting groundwater/drinking water supplies (wells and springs), and/or indirectly impacting on surface water.

The likelihood of inadequate percolation arising at a site, due to presence of soils, subsoils and/or bedrock with a low permeability, is subdivided into four categories – low, moderate, high and very high. A national map illustrating the likelihood of inadequate percolation is shown in Figure 2. Based on an evaluation of the hydrogeological settings present in Ireland and practical site assessment experience, the probability of finding inadequate percolation or inadequate depth to water table within these categories is given in Table 2. By combining the information in Figure 2 with the probabilities in Table 2, it is concluded that approximately 39% of the total land surface of the country is likely to have inadequate percolation for all or some part of the time.⁵

⁵ Detailed information on the combination of data layers used to derive the inadequate percolation map is given in EPA (2012).

Table 2: The probability of finding inadequate percolation for each susceptibility category

Susceptibility Category	Probability of finding inadequate percolation within a category
Low	<5% of sites.
Moderate	Approximately 25% of sites
High	Approximately 50% of sites
Very High	>80% of sites

The likelihood of inadequate attenuation in soils and subsoils depends on the pollutant type. Within the scope of the data layers used for this assessment, the likelihood of pathogens or molybdate reactive phosphorus (MRP) reaching a groundwater or surface water receptor is determined by the same factors: type of aquifer (bedrock or sand and gravel); and depth of soil/ subsoil (as derived from vulnerability maps). Therefore the pathway susceptibility is similar for both pollutants. Three relative categories of susceptibility are taken to apply to pathogen and MRP susceptibility – ‘very high’ or ‘high’ where groundwater vulnerability is classed as ‘extreme’, and ‘low’ in all other cases, as the subsoil cover overlying the bedrock receptor is considered to provide sufficient protection. Approximately 61% of the country is in the ‘low’ category, with 15% in the ‘very high’ category.

Three susceptibility categories are considered as sufficient to apply to nitrate susceptibility. Susceptibility was considered to be ‘very high’ where dry soil and infiltration occurs readily, ‘moderate’ where de-nitrifying bedrock and high permeability subsoils are in evidence, or ‘low’ where wet soils and all other situations overlying de-nitrifying bedrock are found. Approximately 30% of the country is in the ‘very high’ category, with 1% in the ‘moderate’ category.

ESTIMATING CUMULATIVE LOAD

For each 1 km² grid area, the load of pollutants (MRP and nitrate) available to reach either groundwater or surface water was derived by the sum of MRP and nitrate load per person (Table 1) and the number of DWWTSs, assuming 2.8 persons per DWWTS. This load was then applied to the pathway characteristics in each grid square.

Surface Pathway

Where inadequate percolation is dominant and the effluent ponds and/or is likely to be piped to streams/ditches, it is assumed that a proportion of the MRP and nitrogen load – 25% – is removed by, for instance, plant uptake in the ponded areas, attenuation in the soil and some percolation in dry weather. While this is an arbitrary figure that requires validation, the purpose of the methodology is to produce a risk ranking rather than predicted impacts. In addition, it is highly likely that there will be some reduction, and use of this factor has assisted in distributing the final risk categories more logically.

The pollutant load from individual DWWTS reaching surface water by the surface pathway in each grid square is therefore derived from the MRP and nitrate loads given in Table 1, the factors given in Table 2, with a further reduction of the calculated load by 25% due to attenuation at the surface before a water receptor is reached. The total load in each 1 km² area can then be estimated by multiplying by the number of houses.

Subsurface Pathway

The factors applied to enable an estimation of attenuation in the geological materials as discharged effluent percolates underground are given in Table 3.

Table 3: Factors applied to estimate contaminant load from individual DWWTS reaching groundwater

Groundwater Pathway	Input Value
	% of load leaving septic tank that will reach receptor
LOW Susceptibility for MRP/Pathogens	0
HIGH Susceptibility for MRP/Pathogens	10
VERY HIGH Susceptibility for MRP/Pathogens	90
LOW Susceptibility for Nitrate	10
MODERATE Susceptibility for Nitrate	15
VERY HIGH Susceptibility for Nitrate	30

It is assumed, for instance, that where the susceptibility of groundwater to percolation of microbial pathogens and MRP is 'low', no pathogens or MRP will reach groundwater. Where the susceptibility is high, it is assumed that the effluent will percolate through at least 1 m of subsoil, with a consequent significant reduction of MRP concentrations and pathogens (this is based on research by Gill *et al* (2009)). Where the susceptibility is very high, little attenuation of MRP is considered to occur. With regard to nitrate, significant attenuation in the biomat is assumed to occur; this reduces the loading proportions given in Table 3.

RESULTANT CONCENTRATION AT RECEPTOR

Once pollutants reach the water receptor, further attenuation occurs due to dilution.

Surface Pathway

The main factor determining the degree of dilution in each km² grid area is effective rainfall. Therefore, the estimated pollutant load was added to the effective rainfall to provide a projected concentration. As DWWTSs are likely to have a greater impact on surface water during low flow conditions than at other flow conditions (Macintosh *et al*, 2011), a low flow reduction factor of 0.22 was applied to the average flow volume in an attempt to account for low flow conditions; this factor is based on the average Q90/Q50 flow ratio that might be expected in Irish conditions, and was calculated using the EPA's HydroTool⁶. Pollutant concentrations for both MRP and nitrate in each 1 km² area are derived from these calculations.

Subsurface Pathway

With regard to groundwater, the water quantity in each 1 km² area providing dilution is obtained from the Geological Survey of Ireland's recharge map, thereby enabling pollutant concentrations in groundwater to be estimated.

RISK RANKING

The final step in deriving relative risk maps is comparison of the predicted concentrations at the receptor with appropriate environmental standards for MRP and nitrate (Table 4). Microbial pathogens are considered to be influenced by pathway factors in a similar manner to MRP. For MRP, two of the range boundaries are based on the environmental quality standard (EQS) of 0.035 mg/l, that forms the boundary between good and moderate status river water bodies, and 0.025 mg/l, which forms the boundary between high and good

⁶ Used by EPA for estimation of flow in ungauged catchments

status. In the case of nitrate, the categories are based on boundaries used by the European Environment Agency for cross European comparison (EPA, 2010).

While the approach outlined here uses results from the prediction of pollutant concentrations, the maps are not intended to be used for predicting precise impacts; they are intended to show relative risk on which an inspection regime can be based.

Table 4: Molybdate Reactive Phosphorus and nitrate concentrations used in deriving surface pathway and subsurface pathway risk ranking

Risk Ranking				
	<i>Low</i>	<i>Medium</i>	<i>High</i>	<i>Very High</i>
MRP concs. (mg/l P)	<0.015	0.015- 0.025	0.025- 0.035	>0.035
Nitrate concs. (mg/l N)	<2	2-3.6	3.6- 5.6	>5.6

RESULTS OF RISK RANKING PROCESS

Maps for County Meath illustrating the results that arise from this process are shown in Figures 3, 4 and 5. Figure 3 is a map illustrating the location and density of houses in unsewered areas. Figure 4 shows the relative risk of surface water pollution (streams) from MRP and pathogens in DWWTS waste water *via* the surface pathway. Figure 5 shows the relative risk of water pollution (streams and wells) from MRP and pathogens in DWWTS waste water *via* the subsurface pathway. With regard to nitrate, all of County Meath is in the 'low' risk ranking category.

The intention is that maps will be available at 1:40,000 scale for each local authority to help focus inspections on the relevant issues. The percentage areas country-wide in each relative risk category are given in Tables 5 and 6.

Table 5: Percentage areas in the different relative risk categories nationally for MRP and pathogens

Relative risk category	MRP & Pathogens	
	Streams <i>via</i> surface pathway	Streams and wells <i>via</i> subsurface pathway
Low	64.5	91.3
Moderate	10.8	4.2
High	6.6	1.9
Very High	18.1	2.6

Table 6: Percentage areas in the different relative risk categories nationally for nitrate

Relative risk category	Nitrate	
	Streams <i>via</i> surface pathway	Streams and wells <i>via</i> subsurface pathway
Low	99.7	99.93
Moderate	0.17	0.04
High	0.05	0.02
Very High	0.07	0.01

CRITICAL SOURCE AREAS (CSAs)

Critical source areas (CSAs) are the intersection of hydro(geo)logically susceptible areas and pollutant sources in catchments (where transfer pathways and pollutant sources coincide). Research findings show that small areas of a catchment (5-20%) may generate a disproportionately large amount of pollutants; location of these areas enables the focusing of mitigation resources in areas which will yield the most benefit.

The approach outlined in this paper follows the Critical Source Area methodology, which may, in the future, be adapted to considering other pollution sources.

SUMMARY AND CONCLUSIONS

1. The area of the country where there is 'inadequate percolation' for some or all of the year due to poorly permeable soil, subsoil and/or bedrock is relatively large – 39%. These areas present a significant challenge in terms of ensuring that discharges from DWWTSs are adequately treated such that they do not pose a risk to human health and the environment. However, the risk depends not only on a problematical pathway but also on the potential loading from DWWTSs. When this is taken into account, it is concluded that the risk arising from MRP and microbial pathogens is 'very high' in approximately 18% of the country.
2. While there are areas of the country where there is a threat to groundwater from phosphate, nitrate and/or microbial pathogens – almost 3% of the country is in the 'very high' risk category –, practical engineering solutions could generally resolve or alleviate any potential problems.
3. Therefore, in general, DWWTSs pose a greater risk to surface water than to groundwater.
4. Microbial pathogens pose a threat to human health in circumstances where there is a significant likelihood of direct contact either from effluent at the surface or in untreated water from private wells in vulnerable areas.
5. Phosphate is the main pollutant posing a threat to the environment, particularly to surface water, either where there is inadequate percolation or where there is inadequate attenuation prior to entry of waste water into bedrock aquifers, particularly karstified (cavernous limestone) aquifers. While the cumulative pollutant load arising from DWWTSs will be minor compared to urban waste water treatment systems and agriculture at river basin scale, it can be significant in certain physical settings at small catchment scale.
6. The threat posed by nitrate from DWWTSs is low at catchment scale and at the scale of this assessment – 1 km² – due to dilution; however, in exceptional circumstances, at site-scale (a few hectares), a high density of houses can cause localised plumes with elevated nitrate concentrations in groundwater.
7. The approach outlined in this paper and described in more detail in EPA (2012) will enable the EPA and local authorities to adopt a risk-based approach to assist in the selection of DWWTSs for inspections, whereby the level of inspection will be proportionate to the risk posed to human health and the environment.

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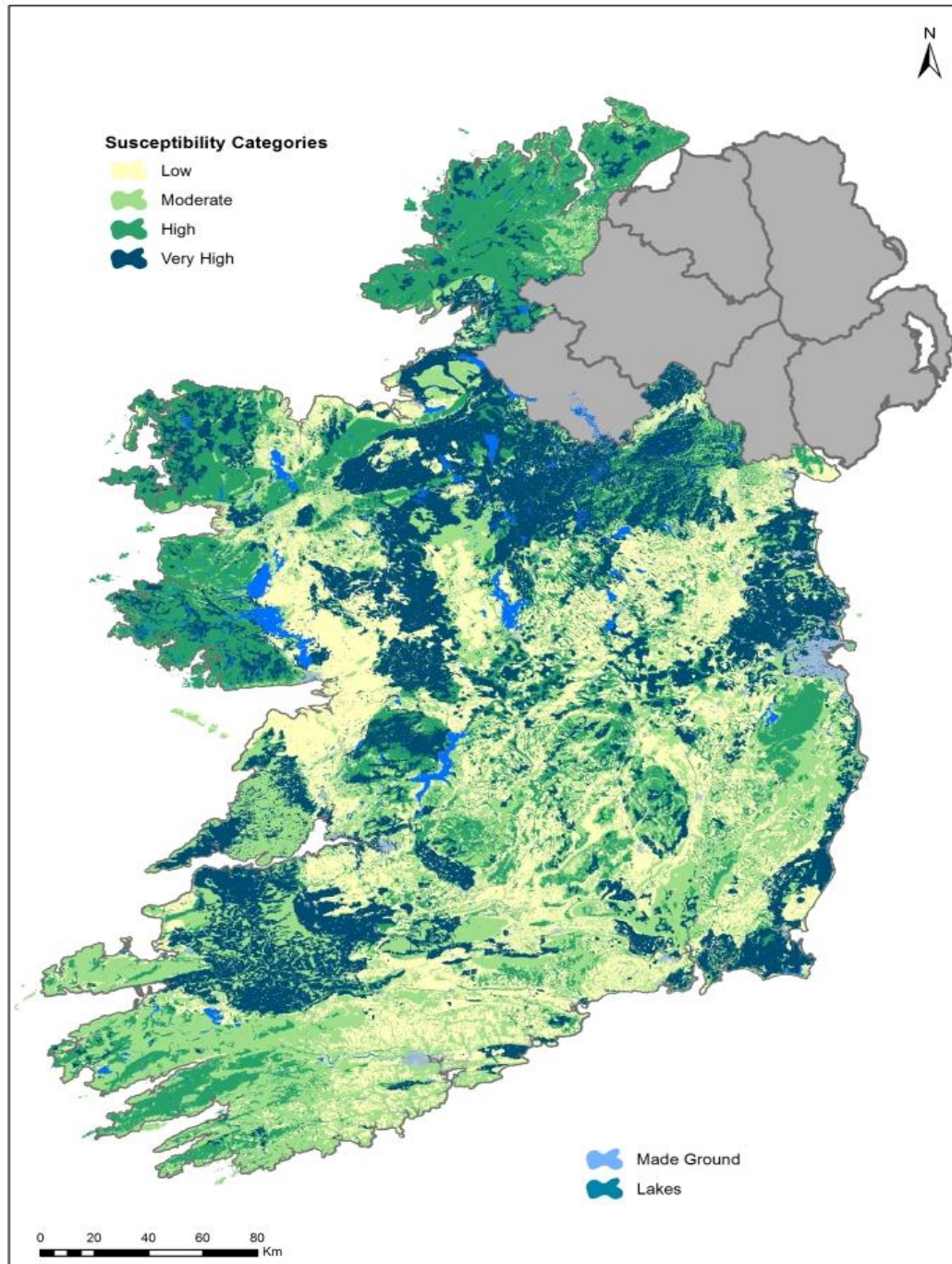


Figure 2: Map illustrating the distribution of susceptibility categories for inadequate percolation. Data captured at 1:40,000 scale. [This map summarises the relevant hydro(geo)logical parameters that characterise the surface ‘pathway’ for water in the source-pathway-receptor framework.]

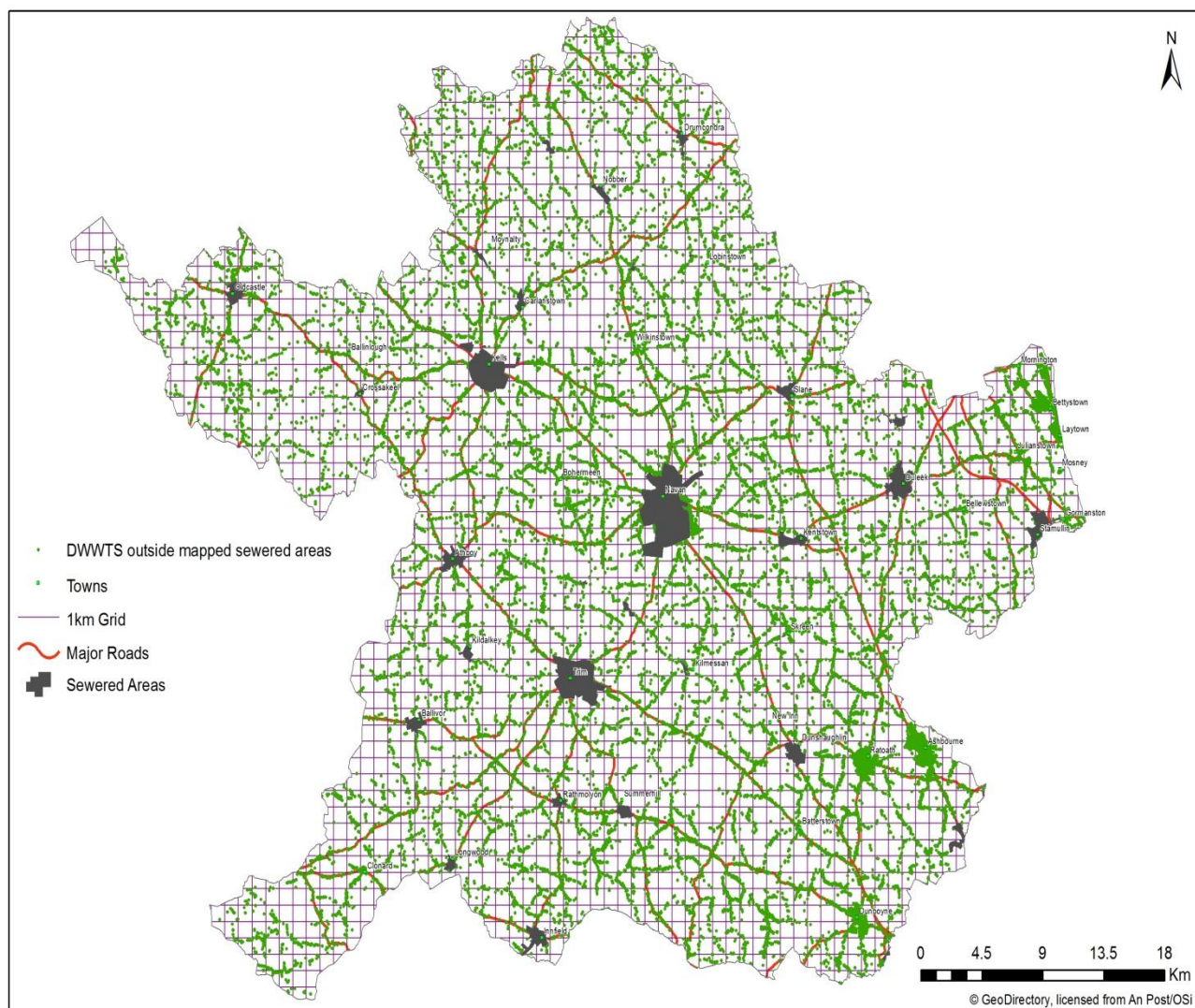


Figure 3: Map of housing locations and density in County Meath (Maps of the sewered areas in Dunboyne, Rathoath, Ashbourne, Laytown and Bettystown were not available and so are not shown.)

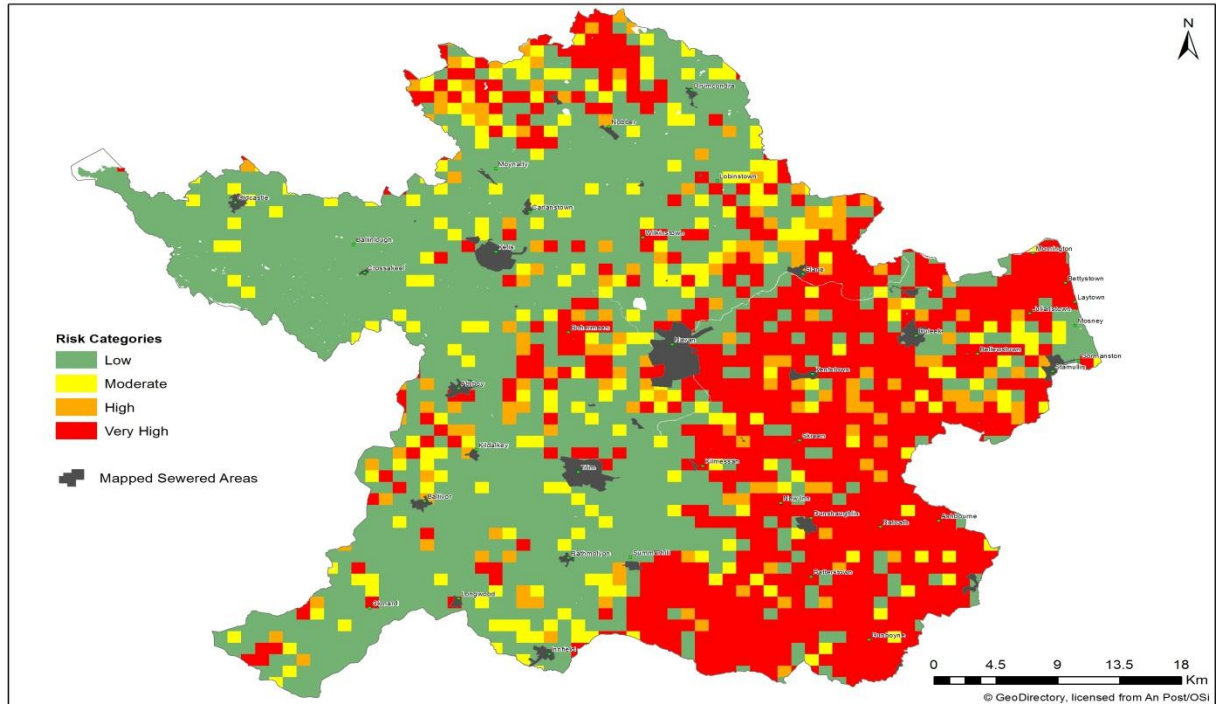


Figure 4: Relative risk of water pollution (streams) from MRP and pathogens in DWWTs waste water *via* the **surface pathway** in County Meath

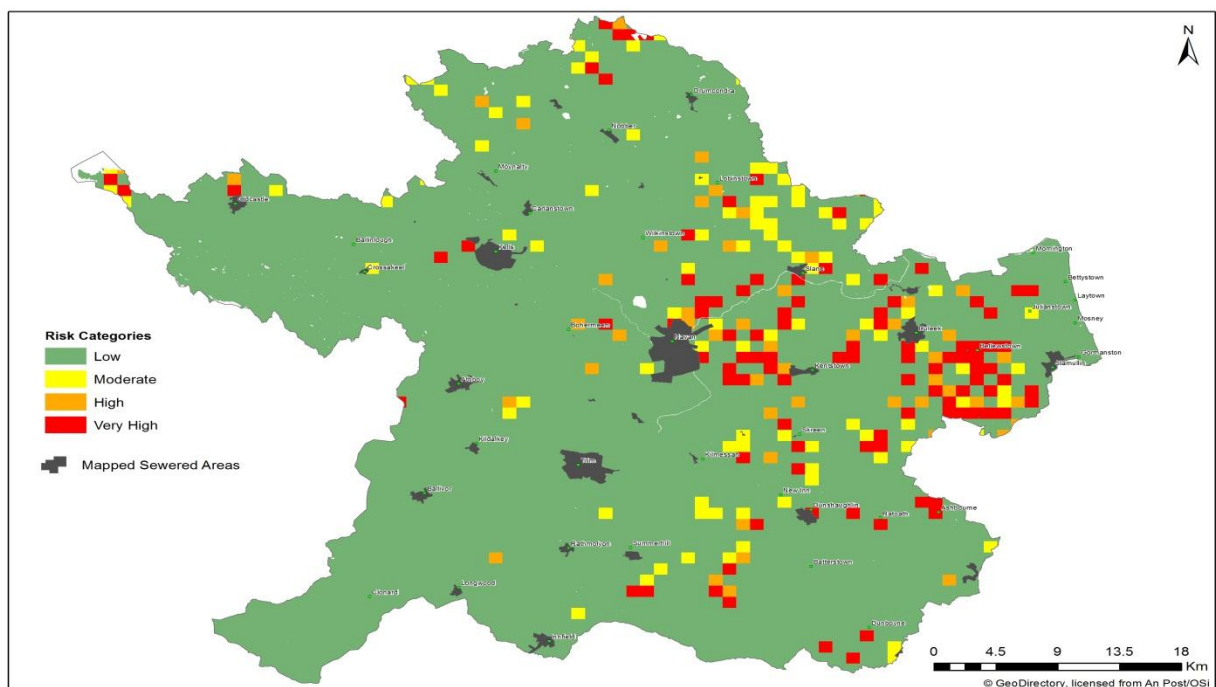


Figure 5: Relative risk of water pollution (streams and wells) from MRP and pathogens in DWWTs waste water *via* the **subsurface pathway** in County Meath

