An investigation into the performance of subsoils and stratified sand filters for the treatment of wastewater from on-site systems

Literature Review

for project 2000-MS-15-M1
(The Hydraulic Performance and Efficiencies of Different Subsoils and the Effectiveness of Stratified Sand Filters)

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An investigation into the performance of subsoils and stratified sand filters for the treatment of wastewater from on-site systems

LITERATURE REVIEW

Authors

L. Gill, P. Johnston, B. Misstear and C. Ó Súilleabháin

Project Participants

Trinity College Dublin

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The Environmental RDTI sub-Measure of the Operational Programme for the Productive Sector (2000-2006) has been allocated indicative funding of £20m by the National Development Plan. The aims of the environmental research sub-programme are to:

- Contribute to a better environment by delivering applicable and relevant RTDI data and information based on high quality science and technology;
- Generate data, information and knowledge for improved management of the environment;
- Develop new techniques, methods and systems for measuring, recording and predicting the quality of the environment;
- Develop practical methods for the integration of environmental considerations into policies and programmes of the main economic sectors.

This project (2000-MS-15-M1) is administered on behalf of the Department of the Environment and Local Government by the Environmental Protection Agency, which has the statutory function of co-ordinating and promoting environmental research. The EPA is represented on this project by Frank Clinton, Margaret Keegan and Helen Walsh.

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1. INTRODUCTION

1.1 Background

Water is a resource which is under increasing risk from human activities with contamination arising from both ‘diffuse’ (generally agricultural) and ‘point sources’, the latter exemplified by farmyards (manure and silage storage) and septic tank systems (Daly, 1993). In areas where the subsoil permeability is too low to allow sufficient soakage of the effluent there is a risk to watercourses from effluent ponding. Groundwater, on the other hand, is especially at risk in areas where bedrock is close to the surface, where subsoils of high permeability underlie the site and where the water table is close to the surface. Prevention of groundwater contamination is of critical importance as, once contaminated, the consequences are usually longer lasting than for surface water owing to longer residence times; moreover groundwater remediation is often expensive, if not impractical.

In Ireland, the domestic wastewater from over one third of the population, or approximately 400,000 dwellings, is treated by on-site systems (Department of the Environment and Local Government (DoELG) *et al*., 2000). Both on-site systems and wells are necessary in the absence of public sewerage or water supply systems, yet they may pose problems on a small site because of the risk of contamination of the water source.

On-site systems can be divided into two broad categories: septic tank systems, which are in the majority, and secondary treatment systems systems. A conventional septic tank system comprises a septic tank followed by a soil percolation area. As an alternative to a conventional percolation area the effluent from a septic tank can also be treated by a variety of filter systems. Removal of most of the suspended solids from the wastewater is achieved in the septic tank and this is accompanied by a limited amount of anaerobic digestion of settled solids in the base of the tank. The effluent then undergoes further physical, chemical and biological treatment in the percolation area before disposal.

In cases where the subsoils render a site unsuitable for conventional septic tank systems, modifications to the treatment system are available. A filter, such as an intermittent sand filter, peat filter or constructed wetland, can be constructed downstream of the septic tank. The effluent would pass through one of these processes and on to a polishing filter prior to disposal. Another alternative is the installation of a package wastewater treatment system. These systems, which produce a good quality of effluent for disposal, generally consist of a primary settlement tank and an aeration tank followed by a secondary clarifier. The aeration tank contains the micro-organisms, either in suspension or attached to an inert media, which are responsible of degradation or organic matter.

In many cases a lack of understanding of the treatment and disposal processes involved in domestic wastewater treatment has lead to poor design, siting and installation of on-site treatment systems, resulting in contamination of groundwater and watercourses. These problems result mainly from unsuitable natural conditions being encountered at the site i.e. unsuitable soil and subsoil properties for the treatment and disposal of effluent. Domestic effluents contain many substances that are undesirable and potentially harmful to human health and the environment. Pathogenic bacteria, infectious viruses, protozoa, organic matter, ammoniacal compounds and a variety of toxic chemicals are all found in significant amounts in wastewater (Tipperary (SR) County Council (TCC) *et al*., 1998).

1.2 Previous Studies

In recent years two major reports have been published by government agencies in relation to on-site treatment systems and groundwater protection. The Environmental Protection Agency (EPA) guidance document *Wastewater Treatment Manual: Treatment Systems for Single Houses* (EPA, 2000) aims to provide guidelines for the selection, design, operation and maintenance of these systems to enable continued sustainable development to take place in Ireland while *Groundwater Protection Schemes (DoELG/EPA/GSI, 1999)* aims to maintain the quantity and quality of groundwater, and in some cases improve it, by applying a risk assessment-based approach to groundwater protection and sustainable development. If these two complementary approaches are applied to the conventional *source-pathway-receptor* model for environmental management, Figure 1.1, the receptor being an aquifer, well or spring, the former approach would be seen as contamination prevention by minimising the contaminant source through adequate design of primary treatment and distribution systems while the latter specifically assesses the ability of the subsoil, or pathway, to protect the groundwater.

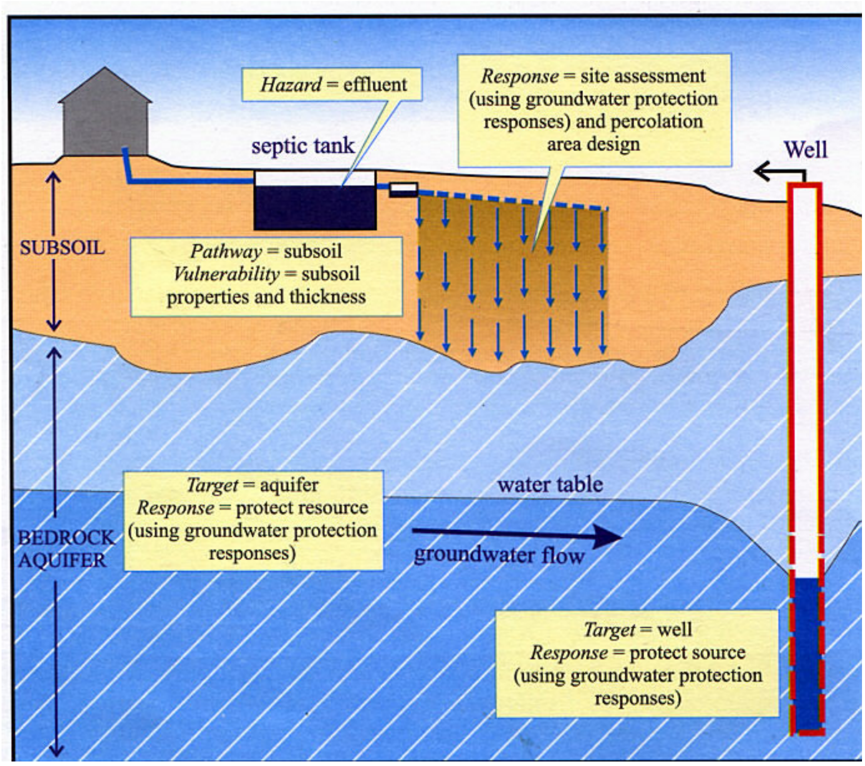

Figure 1.1 Schematic diagram showing how the elements of risk are applied to groundwater protection (DoELG *et al*., 1999).

1.2.1 Wastewater Treatment Manuals: Treatment Systems for Single Houses

A research study entitled Small Scale Wastewater Treatment Systems, co-ordinated by the Department of Civil Engineering, The National University of Ireland, Galway, was carried out between 1995 and 1997 under the direction of the EPA (EPA, 1998). The main objectives of this project were to assess small scale treatment systems and to establish guidelines for their future use, so as to ensure sustainable development. Following on from this study, and with regard to S.R.6: 1991 (*Septic Tank Systems, Recommendations for Domestic Effluent Treatment and Disposal from a Single Dwelling House,* which is published by the National Standards Authority of Ireland) the EPA published a guidance manual *Treatment Systems for Single Houses* (EPA 2000). The manual was prepared to provide guidance to planning authorities, developers, system manufacturers, designers, installers and operators on the design, operation and maintenance of on-site wastewater treatment systems for a single house. A single house system refers to a dwelling house of up to 10 people with toilet, living, sleeping, bathing, cooking and eating facilities.

The manual outlines the steps to be taken to assess the suitability of a site for a treatment system and a methodology for choosing the type of system and the optimum discharge route for the effluent. This is followed by information on the design, construction and maintenance of a septic tank, soil percolation area, intermittent filters, constructed wetlands and polishing filters. It also outlines the operation and maintenance, advantages and disadvantages of mechanical aeration systems.

1.2.2 Groundwater Protection Schemes

The Groundwater Protection Scheme (GWPS) framework is based on the concept of 'risk assessment' and 'risk management' (Misstear *et al*., 1998; DoELG *et al*., 1999). According to GWPS '*The risk of contamination of groundwater depends on three elements:*

> *the hazard afforded by a potentially polluting activity;*
>
> *the vulnerability of groundwater to contamination;*
>
> *the potential consequences of a contamination event.*

*…The hazard depends on the potential contaminant loading. The natural vulnerability of the groundwater dictates the likelihood of contamination if a contamination event occurs. The consequences to the targets depend on the value of the groundwater, which is normally indicated by the aquifer category (regionally important, locally important or poor) and the proximity to an important groundwater abstraction source (a public supply well, for instance)' (*DoELG *et al*.*, 1999).*

The vulnerability of groundwater depends on (i) the time of travel of the effluent to the target, (ii) the relative quantity of contaminants that reach the groundwater and (iii) the attenuation capacity of the medium through which the effluent travels. Based on this geological and hydrogeological assessment of groundwater vulnerability the Geological Survey of Ireland (GSI) rated vulnerability categories as outlined in Table 1.1. They also defined source protection areas, protection areas around major wells and springs, and resource protection areas, based on a classification of Irish aquifers into regionally important, locally important and poor.

The risk management element comprises a series of responses to potentially polluting activities. DoELG *et al.* (2000) also published a paper to be used in conjunction with both *Groundwater Protection Schemes* and *Wastewater Treatment Manuals: Treatment Systems for Single Houses* entitled *Groundwater Protection Responses for On-Site Wastewater Systems for Single Houses*. It outlines groundwater protection zones suitable for on-site wastewater treatment systems, and for those zones, recommends acceptable treatment systems, conditions and/or investigations depending on groundwater vulnerability, the value of the groundwater resource and the contaminant loading.

Vulnerability Rating	**Hydrogeological Conditions**				
	Subsoil Permeability and Thickness			**Unsaturated Zone**	**Karst Features**
	High permeability	Moderate permeability	Low permeability	(Sand/gravel aquifers only)	(< 30 m radius)
Extreme (E)	0 – 3.0 m	0 – 3.0 m	0 – 3.0 m	0 – 3.0 m	-
High (H)	> 3.0 m	3.0 – 10.0 m	3.0 – 5.0 m	> 3.0 m	N/A
Moderate (M)	N/A	> 10.0 m	5.0 – 10.0 m	N/A	N/A
Low (L)	N/A	N/A	> 10.0 m	N/A	N/A

Notes: N/A = not applicable.

Release point of contaminants is assumed to be 1-2 m below ground surface

Table 1.1 Groundwater vulnerability categories (adapted from DoELG *et al.*, 1999)

1.3 Objectives and Scope of Study

This desk study forms part of a larger EPA sponsored study which examines *The Hydraulic Performance and Efficiencies of Different Subsoils and the Effectiveness of Stratified Sand Filters Receiving Domestic Wastewater Effluent.* It is a continuation of previous studies on domestic wastewater treatment systems. While the recent EPA studies focused mainly on construction of treatment systems this project aimed to enhance the understanding of the processes involved and performance of different subsoils receiving domestic wastewater effluent from septic tanks and secondary treatment systems.

The study consisted of two sets of trials, one of twelve months duration and the other of eight months duration, designed to assess the following parameters:

- the hydraulic and wastewater treatment performance of two subsoils of known permeability receiving septic tank effluent;
- the hydraulic and wastewater treatment performance of two subsoils of known permeability receiving secondary treated effluent;

Accordingly, the project was divided into four main phases as follows:

Phase I: Initial literature review and site identification

Phase II: 12 month trials at two sites as follows,
<u>Site 1</u> - septic tank effluent discharged into a subsoil with T-value 15.
<u>Site 2</u> - secondary treated effluent discharged into subsoil with T-value 29

Phase III: 8 month trials at two sites as follows,
<u>Site 3</u> - septic tank effluent equally split between stratified sand filter and subsoil with T-value of 33
<u>Site 4</u> - secondary treated effluent equally split between stratified sand filter and subsoil with T-value of 52

Phase IV collation of results, analyses and final report preparation.

1.4 Literature Review Outline

Chapter 2 of this Literature Review describes the design, operation and efficiency of conventional septic tank systems. In Chapter 3 some of the treatment alternative systems are examined including attached growth systems, suspended growth systems and hybrid systems. Chapter 3 also considers the use of intermittent sand filters and stratified sand filters for secondary treatment of wastewater. Finally, in Chapter 4 the common contaminants found in septic tank effluent are summarised and the processes by which the subsoil attenuates these contaminants are outlined.

2. CONVENTIONAL SEPTIC TANK TREATMENT SYSTEMS

2.1 Introduction

Wastewater from individual dwellings in unsewered areas, generally rural, is managed by on-site treatment and disposal systems. Although a variety of on-site systems are available the most common system consists of a septic tank and a subsurface soil disposal field or percolation area (Figure 2.1). A septic tank is a buried watertight container that serves as a combined settling and skimming tank and as an unheated-unmixed anaerobic digester for domestic wastewater effluent (Metcalf and Eddy, 1991). It provides for separation of sludges and floatable materials from the wastewater and an anaerobic environment for decomposition of both retained and non-settleable materials within the scum layer. The septic tank effluent (STE), being highly polluting (containing faecal bacteria and high levels of nitrogen, phosphorous, organic matter and other constituents (Daly *et al*., 1993)), requires further treatment prior to discharge and is thus directed to a percolation area. In properly designed and constructed percolation areas advanced treatment is achieved for many wastewater constituents of concern through removal (e.g. filtration of suspended solids or sorption of phosphorous), transformation (e.g. nitrification of ammonium or biodegradation of organic matter) and destruction processes (e.g. die-off of bacteria or inactivation of viruses) (Siegrist *et al.,* 2000). Since conventional disposal fields are not suitable for all ground conditions, many alternative systems have been developed (EPA, 2000). The most suitable of these include mechanical aeration systems and sand intermittent filter systems.

Sufficient information must be obtained to determine whether a site is suitable for a conventional septic tank system i.e. septic tank and percolation area, or whether an alternative treatment system is required. The EPA recommends that this site characterisation consists of the following stages:

- a desk study, which collects any hydrological and hydrogeological information that may be available on maps etc. about the site and surrounding water resources (both surface water and groundwater);

- a visual assessment of the site, which defines the site in relation to surface features;
- a trial hole to evaluate the soil structure, depth to rock and water table;
- percolation tests.

(EPA, 2000)

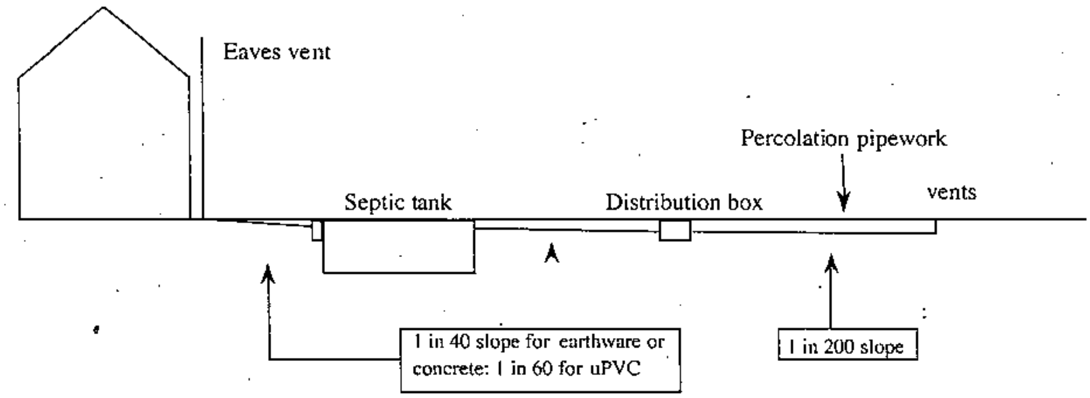

Figure 2.1 A conventional septic tank system (EPA, 2000).

2.2 Septic Tanks

2.2.1 Background

The first reported use of the household septic tank was in France in 1860 when John Louis Mouras and Abbe Moigno discovered that a ‘box’ placed between a house and its cesspool trapped excrement, reduced the amount of solids and produced a clarified liquid that more quickly entered the soil (Payne and Butler, 1995). It made its first appearance in the United States in 1883 when Philbrick introduced a two-chamber tank with an automatic siphon for intermittent effluent disposal to the residents of Boston. Septic tanks were introduced to England by Cameron in 1895 and the type in use today “are of a form that would be instantly recognisable by those early sanitary engineers” (Payne and Butler, 1995). Modifications to septic tank designs have been incorporated to improve the STE quality, particularly with respect to reducing the suspended solids concentration and to a lesser extent the Biochemical Oxygen Demand (BOD), thereby preventing accelerated clogging of soil adsorption systems.

2.2.2 Design of Septic Tanks

Whatever the mode of construction, in-situ or pre-fabricated, or shape, spherical, cylindrical or cuboid, septic tanks must be designed to:

- withstand corrosion;
- carry safely all lateral and vertical soil pressures;
- accommodate water pressure from inside and outside the tank without leakage occurring;
- remove almost all settleable solids in the effluent wastewater;
- prevent discharge of sludge or scum in the effluent and
- allow for the escape of accumulated gases.

It is important that septic tanks are correctly sized, based on the wastewater to be handled. Tanks that are properly sized and constructed provide highly effective treatment, capable of yielding effluent that is relatively free of fats, oils greases, solids and other constituents that can affect further treatment downstream (Bounds *et al.,* 1997). Septic tanks are normally sized based on the design flow with 1/3 of the tank volume designed to provide a 24 hour hydraulic detention time while the other 2/3 is set aside for scum and sludge accumulation. This effectively yields a total tank volume equal to 3 times the daily flow volume (Baumann *et al*., 1978 (cited in Siegrist *et al*., 2000) and USEPA, 1980). A factor of safety should be provided to allow for variations in wastewater loading and future changes in the character of the waste (Canter and Knox, 1985). Oversized tanks will not be cost-effective while undersized ones will produce poor quality effluent. The septic tank must be of sufficient volume to provide a hydraulic retention time in excess of 24 hours (EPA, 2000) at maximum sludge depth and scum accumulation to facilitate BOD and solids removal from the wastewater. Thus, an important factor in the design of a septic tank volume is rate of sludge accumulation, which is dependent on the number of occupants of the dwelling. It is important, therefore, when designing a septic tank to always consider the potential number of occupants rather than present occupation levels. Payne and Butler (1993) and the EPA (2000) recommend that the tank capacity be calculated using the following equation:

$$\mathbf{C = 180.P\ + 2000}$$

where C = the capacity of the tank (litres)

P = the design population

This assumes that the tank is desludged at least once in every 12-month period. In Ireland a minimum capacity of 2720 litres should be provided, which effectively equates to a design for a minimum population of 4 people (EPA, 2000). The installation of kitchen grinders increases the sludge load of the wastewater and so, in such situations, the septic tank capacity should be increased by 70 litres for each additional person (EPA, 2000). Table 2.1 outlines typical capacities and dimensions for rectangular tanks.

No. of Persons	**Capacity (Litres) C = 180 . P + 2000**	**Dimensions (m)**			
		Length		**width**	**depth**
		a*	d*	b*	c*
3	2720	2.2	1.0	1.0	1.2
4	2720	2.2	1.0	1.0	1.2
5	2900	2.4	1.0	1.0	1.2
6	3080	2.5	1.0	1.0	1.2

* refer to Figure 2.2

Table 2.1 Typical capacities of rectangular septic tanks (EPA, 2000)

Septic tanks may be constructed *in situ* of concrete or may be prefabricated from steel, reinforced concrete, glass fibre reinforced concrete or plastic. When steel is used as the construction material it must be treated with bitumen or other corrosion resistant substances. Despite a corrosion resistant coating, tanks still have the tendency to deteriorate at the liquid level (Canter and Knox, 1985). Research by the USEPA (1980) indicates that steel tanks have a short operational life, less than 10 years, due to corrosion. Enhanced quality control is associated with prefabricated tanks since post-construction testing can be carried out on the tanks in the factory. Regardless of the method of construction of the tank, it is essential that all joints are sealed properly to ensure water tightness.

Multi-chamber tanks are superior to single chamber tanks of same size producing effluent with up to 50% less suspended solids and BOD (Laak, 1980). However Laak also found that multi-chamber tanks without inlet and outlet baffles were 10-20% less efficient than single chamber tanks which were baffled. Baffles should be provided on the inlet and outlet of a septic tank to yield quiescent conditions within the tank and limit the disruption and re-entrainment of sludge and scum in the wastewater passing through the tank, thereby minimising suspended solids concentrations in the effluent. It should be noted, however, that a study by the Public health Service in the US cited in Bounds *et al*. (1997) stated that there was no conclusive evidence to suggest that there was any significant difference in the operation of the one and two compartment tanks. The EPA recommends that septic tanks should comprise two interconnected chambers to limit the discharge of solids in the effluent from the septic tank (Figure 2.2). The benefit of a two-chamber tank appears to depend more on the design of the tank than on the use of two chambers (Metcalf and Eddy, 1991). The benefits of two chamber over single chamber tanks are due largely to hydraulic isolation, and to the reduction or elimination of inter-chamber mixing (Canter and Knox, 1985). Mixing can occur by two means, water oscillation and true turbulence. Oscillatory mixing can be minimised by making chambers unequal in size, reducing the flow-through area and using an L-piece to connect chambers. In the larger first chamber, where most of the sludge accumulates, influent wastewater causes some mixing of sludge and scum with the liquid. More quiescent conditions, which are better for settling of low-density solids, exist in the second chamber as it generally receives the wastewater at lower velocities. These conditions, along with proper tank maintenance, lead to an effluent low in suspended solids entering the percolation area.

It is argued (Metcalf and Eddy, 1991) that a more effective way to eliminate the discharge of untreated solids involves the use of an effluent filter vault in conjunction with a single chamber tank (Figure 2.3). The vault contains a screen with a large surface area through which the effluent flows before disposal to the percolation area. Clogging of the screen is not generally a problem due to its large surface area but in the event of its occurrence, it is possible to remove and clean, or replace, the screen. An advantage of the effluent filter vault is that it can be installed in both existing and new septic tanks to limit the discharge of gross untreated solids.

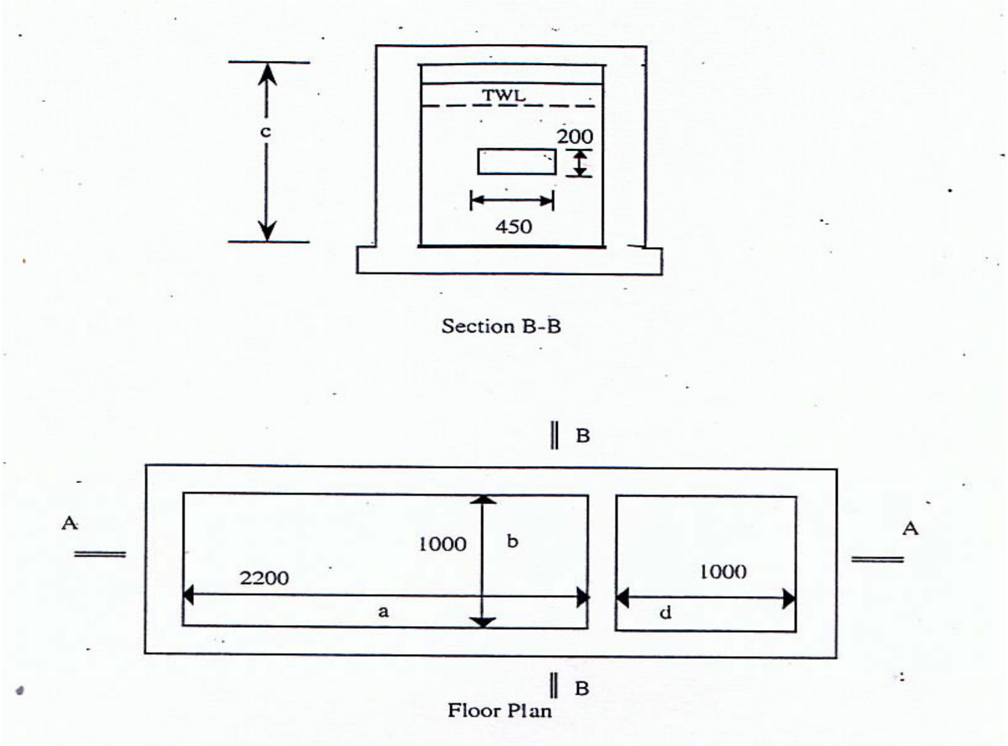

Figure 2.2 Diagrammatic layout of a septic tank (EPA 2000) (refer to Table 2.1)

Although the septic tanks discussed thus far are cuboid in shape, Canter and Knox (1985) reported that "current practices favour rectangular tanks although studies have shown little difference in performance between rectangular and cylindrical designs when sludge storage capacities were similar". Laak (1980) reported on a series of tests using a 27 hour detention time that found that the "outside shape of the tank, circular, cylindrical or rectangular did not appear to have had any significant effect" on the effluent quality.

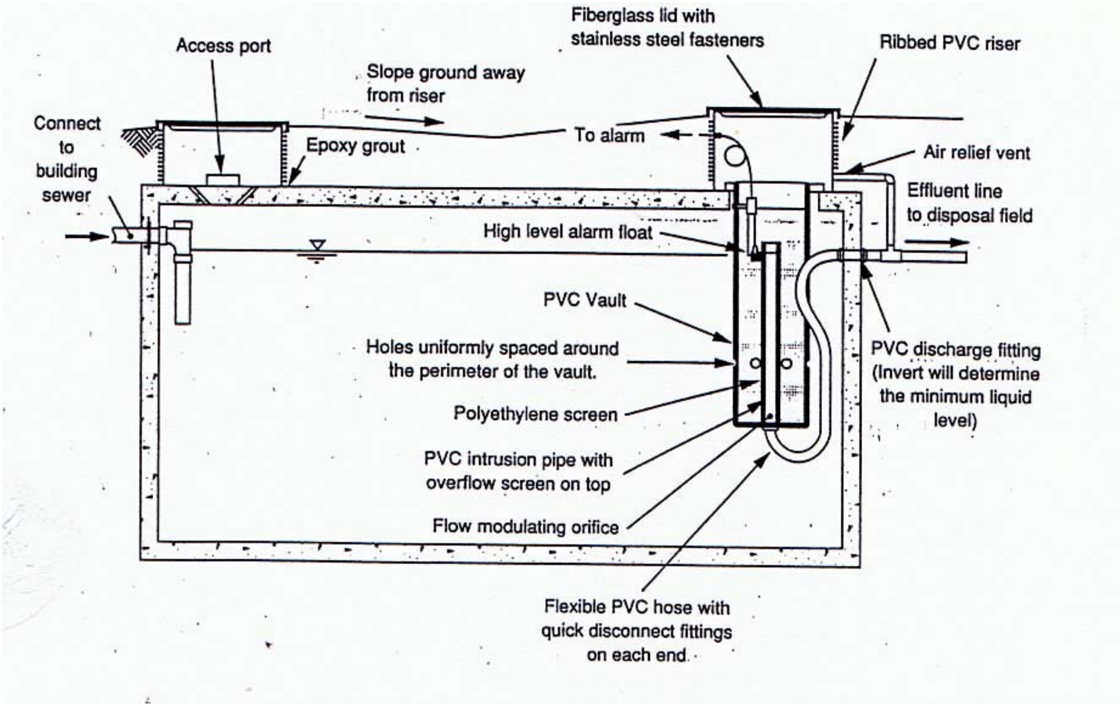

Figure 2.3 A single chamber tank equipped with filter vault (Metcalf and Eddy, 1991)

2.2.3 Septic Tank Operation

A septic tank provides quiescent conditions for settlement of solids and promotes the development of anaerobic conditions for treatment of both stored solids (sludge) and non-settleable constituents in domestic wastewater. The estimated biological and chemical characteristics of domestic wastewater in Ireland, which generally contains toilet flushings (black water or sewage) and washbasin, bathtub washings and kitchen waste (grey water), are highlighted in Table 2.2. The results of quantitative assessment of the physico-chemical quality of 28 septic tanks monitored in Co. Meath are shown in Table 2.3. It should be noted that the values in these tables are all time averaged and so do not truly represent the time distribution of septic tank influent (Table 2.2), which is typically characterised by erratic flow and concentration profiles due to the sporadic nature of discharges from the facilities within single households. Effluent concentrations from septic tanks do tend to be more constant, only varying gradually with time due to the buffering capacity of the septic tank volume (O'Luanaigh, 2003).

Parameter	Typical Concentration (mg/l unless otherwise stated)
Chemical Oxygen Demand COD (as O_2)	400
Biochemical Oxygen Demand BOD_5 (as O_2)	300
Total solids	200
Total Nitrogen (as N)	50
Total Phosphorus (as P)	10
Total coliforms (MPN/100ml)	10^7-10^8

*MPN Most Probable Number

Table 2.2 Characteristics of domestic wastewater from a single house (EPA, 2000)

Parameter	Concentration (mg/l)
BOD	264
COD	500
Suspended Solids	127
Ammonia	55.0
Total phosphorus	16.2

Table 2.3 Performance characteristics of 28 domestic septic tanks, regardless of tank configuration (adapted from Gray, 1995).

Concerns had been aired in the past that high concentrations of detergents and inorganic salts in grey water might upset the treatment processes within the tank and thus, in certain systems, the grey water is piped directly to a separate soil treatment system (Patterson *et al*., 1971). However, current opinion generally dismisses this premise (Grant and Moodie, 1995) suggesting that the average concentrations of detergents and inorganic salts in typical household effluent will not adversely affect the proper functioning of a septic tank. Consequently it is recommended that all wastes should be piped to the septic tank, although any type of runoff (rain from roofs, pavements etc.) should be diverted away from the tank in order to minimise the volume of waste to be treated, avoid dilution of the wastewater and prevent hydraulic overloading of the system. It should be noted, however, that research by Corey *et al*., (1978), cited in Converse and Tyler (1994), reported that it is possible that salts, such as those from water softener backwash, can reduce infiltration in percolation areas where a biomat has not formed.

There is a popular misconception that septic tanks treat domestic wastewater to a standard that does not further require treatment. They do, however produce a consistent effluent that is easy to treat. Septic tanks act primarily as a settlement chamber with only limited reduction of BOD and Suspended Solids (SS) content of wastewater which reduces the probability of clogging in the percolation area (Bouma, 1979). It has been reported that more than 45% of ultimate treatment can be accomplished in the septic tank (Bounds *et al*. 1997). The solids are stored in the tank and the liquid supernatant is allowed to overflow into the percolation area for further treatment. The sludge in the base of the tank is constantly being degraded as a result of anaerobic decomposition and so the net rate of sludge build-up is considerably reduced compared with the theoretical solids accumulation based on a mass balance across the tank. Nevertheless, the volume of sludge does increase with time and requires removal at regular intervals. While the time interval for cleaning varies with factors such as tank size and number of occupants in the house it is usually between 1 and 4 years (S.R.6: 1991), although once every 12-month period is recommended by the EPA (2000). A well constructed and maintained septic tank can remove between 15 - 30% of the BOD and retain between 50 - 70% solids. A typical effluent from a septic tank contains about 80 mg/l solids (Patterson *et al*., 1971; Goldstein and Wenk, 1972; TCC *et al.*, 1998; EPA, 2000;). Research in Ireland on two sites by Henry (1990) recorded average suspended concentrations of 160 and 198 mg/l respectively while research by Keenan (1983), also in Ireland, recorded an average suspended solids concentration of 138 mg/l. However Canter and Knox cite research by Viraraghavan (1976) who found that BOD and Chemical Oxygen Demand (COD) removal efficiencies were in the order of 50% while total suspended solids removal was less than 25%. They also refer to a study by Lawrence (1973) which recorded less than a 15% reduction in BOD and a 34-35% reduction of suspended solids. The settled solids (sludge) on the floor of the tank are partially digested by anaerobic micro-organisms with the liberation of gases, principally carbon dioxide (CO_2) and methane (CH_4). Oils, greases, fats and soaps in the wastewater float to the surface aided by these gases, forming a thick scum over the liquid mass and providing an indication that the tank is functioning properly (Daly *et al*., 1993). The degree of digestion depends on the size of the tank, frequency of cleaning and temperature (Payne and Butler, 1995). While Keenan (1983) and Henry (1990) did not measure the BOD reduction across the septic tank they reported average effluent concentrations of between 268 mg/l and 564 mg/l.

The anaerobic environment within the septic tank is largely ineffective in reducing the nutrient loading of the wastewater. Nitrogen in the influent wastewater is mainly in the form of organic nitrogen and ammonia, measured as total Kjeldahl nitrogen (TKN). Under anaerobic conditions, much of this organic nitrogen is converted to readily oxidisable ammonium ions (NH_4^+). A typical average wastewater influent TKN concentration is 38 mg/l [32% NH_4^+: 68% org-N] whilst average effluent TKN concentrations are recorded at similar levels around 40 mg/l [75% NH_4^+: 25% org-N] (Bauer *et al*., 1979). Keenan (1983) reported an average ammonium concentration of 51 mg/l while Henry (1990) reported average concentrations 28mg/l and 44mg/l. Therefore, although the septic tank environment promotes the conversion of organic nitrogen to ammonium, it is ineffective with respect to total nitrogen removal across the process. It should be noted that nitrate concentrations in septic tanks are very low and usually zero, due to the lack of aerobic conditions.

The anaerobic digestion process also converts most of the influent phosphorous, in the form of both organic and condensed phosphate (polyphosphate), to soluble orthophosphate which passes out in the effluent. Bauer *et al.* (1979) also reported average total phosphorus concentrations in influent wastewater to septic tank systems serving single houses of 25 mg/l [35% inorganic P (orthophosphate): 65% organic-P]. Salvato (1992) reported average orthophosphate concentrations in septic tank effluent to be 15 mg/l. A typical total phosphorus concentration entering the percolation area is about 15 mg/l (Canter and Knox, 1985) of which about 85% is in the soluble orthophosphate form (Bouma, 1979 and University of Wisconsin, 1978). Keenan (1983) recorded average total phosphorous concentrations of 16 mg/l while Henry (1990) recorded average orthophosphate concentrations of 29mg/l and 50 mg/l. While septic tank influent quality in Ireland differs from the more generally dilute sewage in North America, (Gray, 1995), the processes involved in the septic tank are the same and so it would be reasonable to expect similar behaviour of the nutrients under Irish conditions.

Studies have also shown that removal of viruses, bacteria and micro-organisms within the tank is negligible (Patterson *et al.*, 1971; McCoy and Ziebell; 1975; Canter and Knox, 1985). Particulate organic matter, which contains complex molecules such as carbohydrates, proteins and lipids, are broken down under anaerobic conditions in the septic tank into simpler soluble compounds which will pass out of the tank with the effluent. The biological conversion of organic matter in the anaerobic environment of the septic tank occurs in three stages:

- the hydrolysis of insoluble high molecular-mass compounds into soluble organic compounds suitable for use as a source of energy and carbon,
- the bacterial conversion of these compounds into identifiable lower-molecular-mass intermediate compounds such as organic acids,
- the bacterial conversion of these intermediate compounds into simpler end products, principally methane and carbon dioxide.

It is clear, therefore, that a typical septic tank effluent is of poor quality, with a high pollution potential. It thus requires further treatment prior to discharge to watercourses or groundwater.

2.3 PERCOLATION AREAS

2.3.1 Design and Operation of Percolation Areas

Once the effluent leaves the septic tank it enters a distribution box from where it is channelled into an engineered distribution trench for discharge into the subsoil where it undergoes further biological, chemical and physical treatment. Attempts to distribute the flow equally between trenches or areas of a bed using distribution boxes and 100 mm diameter perforated drain pipe are commonly made, but have been shown to be ineffective (Otis *et al*., 1978 cited in Siegrist *et al*., 2000). The soil treatment system, or percolation area, is the most important component of the conventional septic tank system as it is here that the majority of treatment occurs. Research has shown that greater than 90% removal efficiencies can be achieved for organic constituents (BOD, COD and S.S.), micro-organisms and viruses by filtration, sorption and biodegradation processes. However, the removal of nutrients is more limited (USEPA, 1980; Jenssen and Siegrist, 1990; Van Cuyk *et al*., 2001). Purification efficiencies in soil treatment systems can be very high yielding near complete removal of faecal coliform bacteria and greater than 4 log (99.99%) reduction in viruses (Emerick *et al*., 1997 (cited in Siegrist *et al*., 2000); Stevik *et al*., 1999 and Van Cuyk *et al*., 2001). Viraraghavan and Warnock (1976) reported that the soil had the ability to reduce 75-90% of the total suspended solids, BOD, COD and soluble organic carbon in the STE.

It is important, at this juncture, to distinguish between percolation areas and soakaways as they are often confused. A soakaway or soakage pit is basically a deep hole in the ground into which the STE flows for disposal. Since effluent is released over a small area there is a danger of clogging, causing ponding, or the treatment capacity of the soil being exceeded leading to groundwater contamination by untreated effluent. By the very mode of their construction (i.e. excavation) the depth of the treatment medium (subsoil) between the source and target (groundwater) is being depleted. Soakaways are not, therefore, "a satisfactory alternative to percolation areas” (SR:6, 1991) and should not be used. Typically a percolation area consists of a series of narrow, relatively shallow trenches or mounds filled with a porous medium (usually gravel). The following equation is used for drainage trench area calculations (Payne and Butler, 1993):

$$A = P \times V_p \times 0.25$$

where,

A = Floor area (m^2) of subsurface drainage trench

P = The number of people served by the tank

V_p = the percolation value of the subsoil (sec/mm)

A typical trench system is shown in Figure 2.4. Trenches are shallow, level excavations usually about 800 mm deep and 450 mm wide. The bottom is filled with 20-30 mm washed gravel over which the percolation pipe, maximum length of 20 m per trench, is laid at a 1 in 200 slope. The pipe is then surrounded by more of the gravel which is covered with a suitable semi-permeable geotextile to prevent backfill from clogging the aggregate. About 20 m of 100 mm piping is required per person (EPA, 2000). The porous medium is used to:

- maintain the structure of the trenches;
- provide partial treatment of the effluent by facilitating biomat formation;
- distribute the effluent to the infiltrative soil surfaces and
- provide temporary storage capacity during peak flows.

(Kreissel, 1982)

Research by Amerson *et al*. (1991), cited in Converse and Tyler (1994), reported that it appeared that the dust from the aggregate used in percolation trenches was a major factor in changes in infiltration. Van Cuyk *et al.* (2001) argued that the use of gravel in percolation trenches can have a detrimental effect on the infiltration zone permeability by:

- blocking pore entries;
- becoming embedded in the soil matrix;
- yielding fines that are deposited in pore entries or
- by focusing wastewater constituents as a result of the reduced permeability of the above effects.

Based on these potential adverse effects of gravel on infiltration capacity it was calculated that aggregate free percolation trench area could be constructed with an area 40% to 50% less than that required for gravel systems (Siegrist *et al.*, 2000).

Percolation areas disperse the effluent over a large area close to the ground surface, as far as possible from the watertable and bedrock. Hence, the treatment capacity of the subsoil is maximised and the risk of ponding is reduced where the infiltration and/or permeability is low. Proper design of percolation areas is critical to the successful operation of septic tank systems.

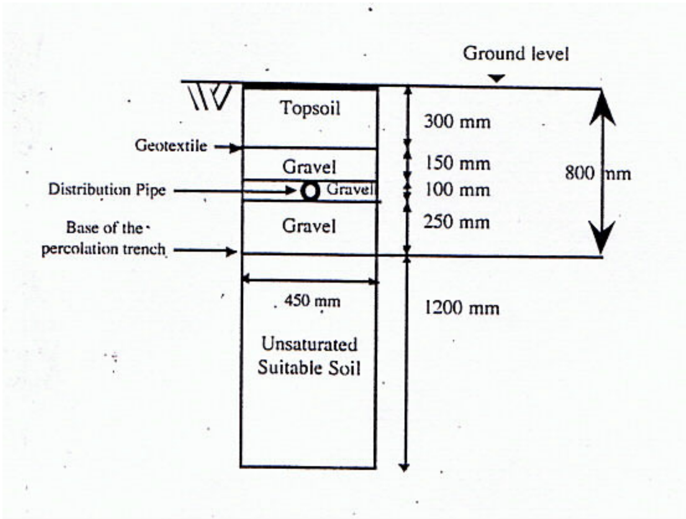

Figure 2.4 Section of a percolation trench (EPA, 2000).

Work by Cotteral and Norris (1969) and Laak *et al.* (1974) identified that the most important factors affecting the performance of a soil absorption system are subsoil properties, biomat formation and loading rate. Subsoil properties will be examined in detail in Chapter 4.

2.3.2 Biomat Formation and Development

The performance expectations of subsurface wastewater infiltration systems include long-term wastewater infiltration and adequate wastewater renovation prior to groundwater recharge (Siegrist and Boyle, 1987). Soil clogging, a phenomenon known to occur as a result of wastewater infiltration, is a function of the organic and solids loading rate from the effluent. Siegrist (1987) lists the main soil and wastewater characteristics that influence the occurrence and rate of progression of soil clogging as: soil temperature, moisture content, aeration status, applied effluent composition, hydraulic loading rate and method of application. Soil clogging has the affect of reducing the hydraulic conductivity and in many cases it is the hydraulic conductivity of the biomat, and not the subsoil, that becomes the controlling variable in wastewater infiltration rates (McGaughey and Winneberger, 1964 – cited in Wilhelm *et al*., 1994; Converse and Tyler, 1994). If the native sediments are finer-grained than the gravel surrounding the distribution pipe, a 2 to 5 cm thick mat forms over time below the gravel (Anderson *et al.*, 1982 – cited in Wilhelm *et al*., 1994). However, where a subsoil receives a highly treated effluent (BOD and SS < 20mg/l) a biomat will not form because the amount of organic matter is very low (Converse and Tyler, 1997). This affects the infiltration rate with research showing that secondary treated effluent, sand filter effluent in this case, infiltrated at rates 7-12 times greater than septic tank effluent (Loudon *et al*., 1998). Converse and Tyler (1994) concluded that highly pre-treated wastewaters could be applied at 2-6 times greater than that recommended for septic tank effluent and possibly at rates equal to the soil saturated hydraulic conductivity. They also report (1997) that there is undocumented evidence to suggest that infiltration rates may increase due to increased biological activity from worms and other organism activity due to the aerobic nature of secondary effluent.

The clogging zone or biomat forms at the soil–gravel interface along the base and wetted sides of the percolation trench as suspended solids, equal to or larger than the soil pore size, are trapped. While some degree of soil clogging can enhance wastewater treatment through physical/chemical and biochemical process severe clogging can lead to hydraulic dysfunction, anoxic soil conditions and diminished wastewater treatment (Otis, 1985; Siegrist, 1987). Bacteria and other micro-organisms start to grow on the particulate matter due to extensive and lengthy contact between the wastewater constituents and the porous matrix (Figure 2.5). It is within this zone that the majority of the biological activity occurs, and it is here that the processes of decomposition of suspended materials, bacterial build-up and decomposition of organic material by bacterial action continually modify the infiltrative capacity. Mats have been observed to retain as much as 99.9% of the original coliform population over a distance of less than a foot (McCoy and Ziebeil, 1975). The zone is formed by three distinct phases:

- Physical: where solids in the effluent physically clog the soil pores;
- Chemical: where soil colloids swell as a result of chemical processes

- Biological: where bacteria or bacterial breakdown products reduce pore size.

(Patterson *et al.*, 1971)

Research by Orlob and Butler (1955) on five Californian soils concluded that the infiltration capacity of the soil absorption systems was controlled by the nature of the biomat and not the permeability of the soil. Although the biomat penetrates into the soil subsurface, the majority of it is located on the surface of the soil. This leads to reduced permeability, more uniform infiltration and a concomitant unsaturated flow almost regardless of hydraulic loading (Van Cuyk *et al.*, 2001). Wastewater induced clogging increases the soil biogeochemical activity and can enhance, sorption, bio-transformation and die-off/inactivation processes (Siegrist, 1987; Siegrist *et al*., 1991). The physical method of straining, whereby the movement of material greater than void size is inhibited, also forms an important treatment process within the biomat. The biomat plays an important role in reducing the numbers of faecal bacteria in the percolating effluent (McCoy and Ziebell, 1975) and has also been reported to effectively remove other effluent constituents by various sorption reactions (Laak, 1974; McCoy and Ziebell, 1975 and Miller and Wolf, 1975). Removal of pathogens and other constituents may be less than predicted or desired if the clogging zone development is retarded. Such retardation could occur if the influent to the percolation trench is of a highly treated quality (secondary treatment effluent, for example). Conversely, excessive clogging can be detrimental causing hydraulic dysfunction, anaerobic soil conditions and reduced purification (Van Cuyk *et al,* 2001). It should be noted, however, than localised anaerobic conditions develop due to the high O_2 demand and moist conditions Wilhelm *et al*., 1994).

Over time the biomat develops a dynamic equilibrium in properly loaded and maintained systems. The rate of biomat development has been correlated with the amount of suspended solids and O_2 demand in the effluent (Siegrist and Boyle, 1987). As effluent solids accumulate, leading to formation and growth of the biomat, mineralised constituents and particulate material which have been reduced in size are carried away with the percolating water and gases produced from biological conversion of the waste are released (Metcalf and Eddy, 1991). Simultaneously, the development of the biomat can lead to progressive ponding in the trench due to decreased infiltration. The increase in hydraulic head associated with this increase in ponding depth has been shown to compensate for the increased resistance to infiltration (Siegrist and Boyle, 1987). This demonstrated that aggregate depth was important to maintain the hydraulic performance of the percolation trench with increasing ponding. Siegrist *et al.*, (2001), however, argue that most systems operated under continuous use at a design rate of 10 to 50 l/m^2/d will eventually clog to such a degree whereby hydraulic failure will occur.

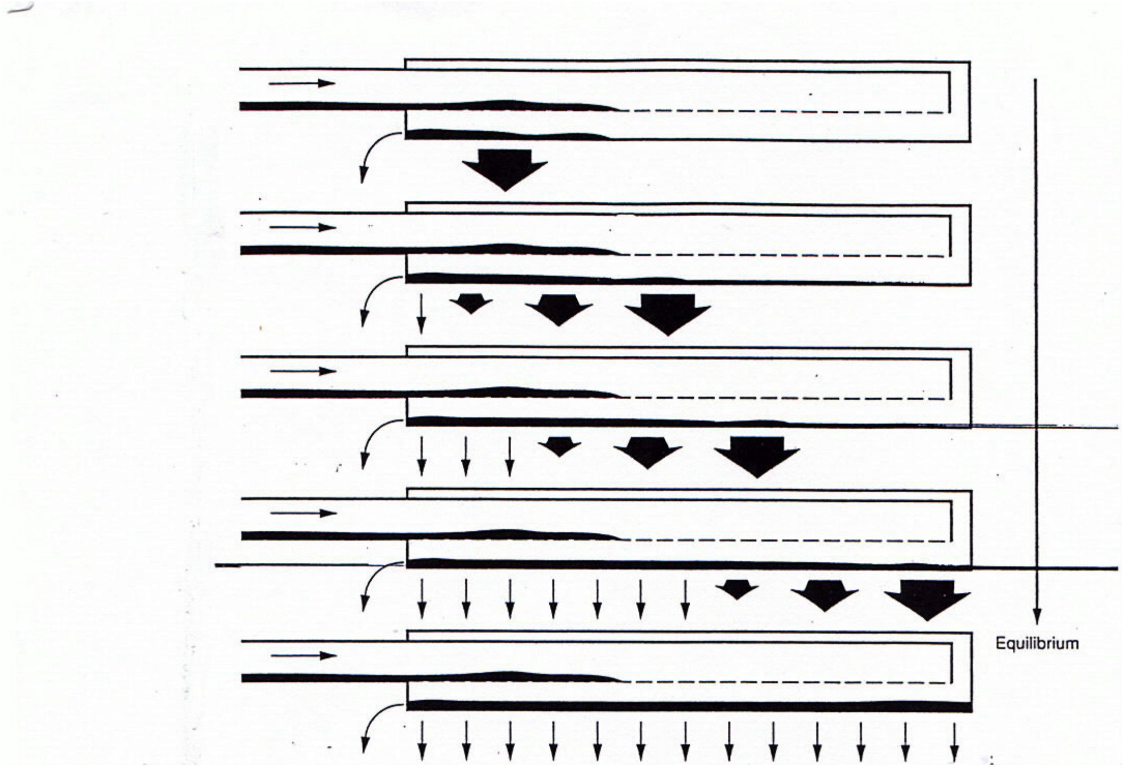

Figure 2.5 Progressive development of a biomat in a percolation trench (Kreissel, 1982)

2.3.3 Loading Rates

Satisfactory performance of percolation systems requires careful consideration of applied wastewater composition and hydraulic loading rate to prevent excessive clogging. The filter field loading rates are limited by either the hydraulic conductivity of the soil or the interaction of the biomat and soil at the trench interface (Bouma, 1975), Lack of consideration of these parameters can lead to hydraulic dysfunction and diminished wastewater treatment. Canter and Knox (1985) report on three types of loading regimes that can be utilised: continuous ponding, dosing and resting, and uniform application without ponding. Continuous ponding has the effect of increasing the effective infiltrative area by submerging the sidewalls of the trench and increasing the hydraulic gradient across the infiltrative surface which may increase the infiltration rate. However, this method lead to anaerobic conditions in the trench which can cause subsequent problems in terms of both hydraulic flow and biological decomposition. Dosing and resting overcomes some of the problems associated with the continuous ponding method. This is where a "reserve" percolation area is built so that it is possible to rest each percolation area in turn for several months at a time. This has the effect of encouraging reaeration of the trench and degradation of the clogging mat thus prolonging the effective life of the system (Otis, 1984). In the uniform application without ponding method, the liquid is distributed over the entire infiltrative surface at a lower rate than the soil infiltrative capacity thus preserving unsaturated aerobic conditions and minimising resistance of the clogging mat. In Ireland, the EPA recommend that the percolation trenches receive an even flow of effluent at a loading rate of 20 $l/m^2/d$ which takes into account the effect of the biomat on subsoil infiltration and ponding.

While a high degree of treatment normally occurs in the biomat at current loading rates higher hydraulic loading rates and non-uniform distribution methods can result in a malfunction in the system. Research by Owens *et al.*, (1997) highlighted the failure of a serially loaded septic tank percolation area due to continuous anaerobic conditions resulting from continuous saturation by effluent. Many studies have shown that a large percentage of bacteria remain near the infiltrative surface when effluents are applied to a porous media. However, if hydraulic loading rates are too high or the dosing frequency is too low, microbes can be transported to lower regions in a soil matrix, posing a treatment concern in systems with a shallow water table (Van Cuyk, 2001). Siegrist and Boyle (1987) reported that the findings of research they carried out was consistent with prior research (Jones and Taylor, 1965; Laak 1970; Hargett, 1982; Kristiansen, 1982; Pell and Ljunggren, 1984 and Siegrist *et al.*, 1985) demonstrating that soil clogging development was accelerated both: at higher hydraulic loading rates with a constant effluent; and with increasingly concentrated effluents at a constant hydraulic loading rate, leading to system failure.

2.4 Failure of Septic Tank Systems

The ability of septic tank systems to effectively treat domestic wastewater to a degree that minimises risk of contamination of water resources depends on suitable site selection, competent system design and effective system maintenance. Problems associated with septic tank systems are common and generally due to insufficient attention to detail in any one of the areas alluded to above, as summarised in Table 2.4.

Problem	Immediate Cause
Odour	Inadequate ventilation of drains Blocked drainage field Inadequate drainage field
Backing up of sewage and surface flooding	Sagging or blocked inlet drains Blocked drainage field Inadequate drainage field Tank full of sludge
Solids discharged from tank	Tank full of sludge Inefficient or undersized tank
Local watercourse pollution	Blocked drainage field Inadequate drainage field Tank full of sludge Deliberate overflow connection made Proliferation of tanks discharging to land which quickly drains to watercourses
Groundwater pollution	Drainage field operating properly but system in unsuitable location Proliferation of tanks in sensitive area
Tank full of groundwater/tank lifts	High water table

Table 2.4 Symptoms and immediate causes of septic tank system problems (after Payne and Butler, 1995)

3. ALTERNATIVE SECONDARY TREATMENT PROCESSES

3.1 Introduction

Where a site has been deemed unsuitable for the construction of a conventional septic tank treatment system, the installation of a secondary treatment system can be considered. These systems may take the form of mechanical aeration systems, filter systems and constructed wetlands. Of interest to this project, and discussed in this chapter, were the package treatment plants which come in the form of mechanically and passively aerated modules. These provide secondary treatment of septic tank effluent or, as in the case of mechanical aeration systems, are an alternative to the septic tank treatment system. They are always succeeded by a polishing filter and, where used as an alternative to septic tank systems, are preceded by a primary settlement tank (EPA, 2000). The primary settlement tank provides for separation and retention of settleable solids and floatable materials, reduces the size of the subsequent biological treatment stage by up to 50%, reduces oxygen demand and thus the power requirement in mechanical aeration systems and buffers against shock loads (Metcalf and Eddy, 1991). Typical removal efficiencies of 50-70% suspended solids and 25-40% BOD are achievable. The primary purpose of a polishing filter is to reduce the concentration of micro-organisms in the treated wastewater prior to disposal (EPA, 2000).

3.2 Package Treatment Systems

Secondary treatment systems are designed to provide a controlled environment for the accelerated microbial degradation of organic matter and, in some cases, nutrients. In the case of mechanical aeration these systems, where designed for single dwellings, usually contain a primary settlement tank, aeration tank and secondary settlement tank (clarifier) in a fully enclosed compact unit. Passively aerated systems, which generally just contain modules of aerated media, are installed downstream of the septic tank and effluent is discharged to a polishing filter, usually the subsoil. In the aerated chamber carbonaceous organic matter (represented by CHONS) in wastewater is converted by heterotrophic micro-organisms into microbial biomass, thereby releasing both nutrient containing compounds along with carbon dioxide in general accordance with the following stoichiometry :

$$COHNS + O_2 + \text{nutrients} \longrightarrow CO_2 + NH_3 + C_5H_7NO_2 + \text{other end products}$$

$$C_5H_7NO_2 + 5O_2 \longrightarrow 5CO_2 + 2H_2O + NH_3 + \text{energy}$$

(Metcalf and Eddy, 1991)

The microbial biomass is then separated from the treated wastewater by sedimentation in a settlement tank where it can be periodically drawn off as sludge from the base of the tank. Nitrification, the conversion of ammoniacal products to nitrate by autotrophic bacteria, is only achievable in the aeration chamber once most of the carbonaceous matter has been assimilated by heterotrophic bacteria. The nitrifying autotrophic bacteria are much slower growing than the heterotrophs and require a suitable environment downstream of the main carbonaceous removal zone where organic loads are low and both dissolved oxygen and ammonium concentrations are high. Nitrification is considered in more detail in Section 4.2.3.

Phosphorous, once released from organic matter, will remain as soluble ortho-phosphate which will pass out in the effluent without any significant reduction (Fitzgerald, 1995). It is possible to remove orthophosphate from wastewater by either biological or chemical treatment processes, however, both methods require involved engineered treatment design and operational demands unsuitable for such small-scale applications (Stocks *et al.*, 1994). Secondary treatment plants do achieve a degree of phosphorous reduction by a combination of sedimentation of particulate constituents and bacterial assimilation. Typical removal efficiencies across such works are 50-65% total phosphorous and 10-20% orthophosphorous depending the type of secondary treatment process employed (Gill, 1999 : Smith, 1999).

Secondary treatment systems are generally classified according to the type of biological treatment process:

- Attached growth — biofilms form as clusters of cells adhered to an inert media, e.g. rotating biological contactor (RBC).
- Suspended growth — the bacterial culture is suspended in a "mixed liquor" and group together to form "flocs", e.g. sequencing batch reactor (SBR).
- Combination — an inert media is present in a flooded reactor resulting in a combination of attached and suspended growth treatment processes, e.g. biological aerated filter (BAF).

3.2.1 Attached Growth Systems

Attached growth systems consist of an inert medium on which micro-organisms grow as a biofilm with typical depths of 60-3000µm (Hawkes, 1983) in a complex ecosystem of bacteria, protozoa, fungi, invertebrates etc. Treatment is achieved by aerobic degradation of the wastewater constituents when they are brought into contact with the heterotrophic bacteria within this microbial biofilm (Figure 3.1). Hence, it is important that the media provides a maximum surface area to volume ratio for optimal biofilm-wastewater contact. As the film only develops on surfaces that receive a constant supply of nutrients, the effectiveness of the system in promoting maximum wetting of the media surface area is an important factor affecting performance (Gray, 1999). The first stage of treatment is the adsorption of organic nutrients onto the film. Fine particles in the wastewater are flocculated by extra-cellular polymers secreted by the attached heterotrophic micro-organisms and adsorbed onto the surface of the film, where, along with organic nutrients which have been physically trapped, they are broken down by extra-cellular enzymes excreted by the bacteria and fungi (Gray, 1999). Soluble nutrients in the influent and also those resulting from this extra-cellular enzymatic activity are directly adsorbed by the biofilm and synthesised. The by-products of this aerobic degradation process diffuse out into the liquid wastewater phase.

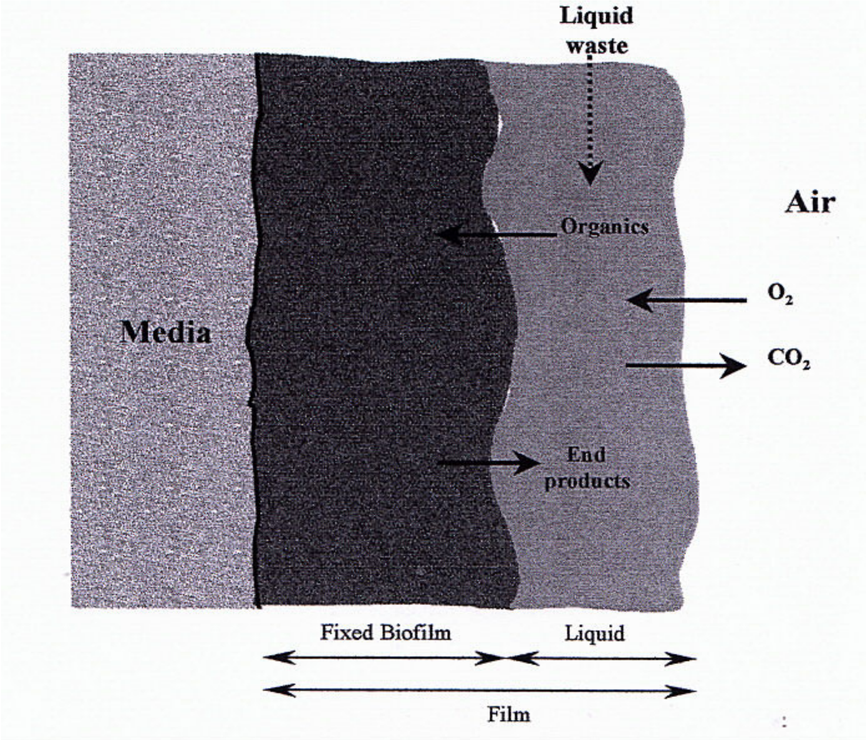

Figure 3.1 Schematic representation of the structure of biofilm in a fixed-film reactor.

The mass transfer of substrates (organic matter and oxygen) into and within the biofilm regulates the rate of reaction of the micro-organisms. External mass transfer involves the transfer of the substrates from the bulk liquid through a laminar fluid layer into the biofilm. If external mass transfer is the limiting factor in the microbial degradation process (for example, if the wastewater contains only a low concentration of organic matter), the observed reaction kinetics within the biofilm would be first order characterised by the following equation :

$$\frac{dX}{dt} = kX$$

where, X= concentration of organic matter (mg/l)

k = first order rate coefficient (day^{-1})

(Mihelcic, 1999)

The biofilm development is thus directly proportional to the concentration of organic matter in the influent wastewater, and, increasing the influent organic matter to a level at which it no longer is the limiting factor to growth would result in an exponential increase in the microbial population.

Internal mass transfer is the resistance from the biofilm itself against substrate diffusion from the surface into the cell clusters. The thickness of the biofilm is critical in determining the internal rate of mass transfer with respect to dissolved oxygen which is continually being depleted as it diffuses through the thickness of biofilm. The depth to which oxygen will penetrate depends on a number of factors such as composition of the biofilm, its density and the rate of respiration within the film itself, and has been estimated to be anywhere between 0.06 and 4.00 mm (Gray, 1999). However, it is generally acknowledged that the critical biofilm thickness at which internal mass transfer becomes limiting, is approximately 0.15 mm (Shieh, 1982). Hence, if the biofilm is less than 0.15 mm thick it can be assumed that it is fully penetrated by oxygen and the entire biofilm is not oxygen limited maintaining aerobic conditions. This is the most efficient situation for such a secondary wastewater treatment process whereby the reaction kinetics are defined as zero order as characterised below:

$$\frac{dX}{dt} = k$$

An increase in influent organic matter concentration, therefore, will not result in increased biofilm activity. If, however, the biofilm is greater then 150 µm thick, oxygen becomes the limiting factor in microbial respiration and the reaction kinetics go from zero order to half order as shown.

$$\frac{dX}{dt} = kX^{1/2}$$

In this situation not all the organic matter is utilised and an anoxic, and subsequently anaerobic environment is established inside the biofilm moving towards the media.

As the biofilm increases in thickness, the adsorbed organic matter is metabolised before it can reach the micro-organisms near the media surface leaving no external organic source available for cell carbon. These micro-organisms then enter into an endogenous phase of growth where they are forced to metabolise their own protoplasm without replacement due to shortage of available substrate. They lose their ability to cling to the media surface and are subsequently washed off. As a result a new biofilm layer starts to grow. This phenomenon is known as 'sloughing' and is a function of both hydraulic loading, which accounts for localised shear velocities on the biofilm and variations in organic and nutrient loading, which account for biofilm metabolism (Metcalf and Eddy, 1991; Sawyer and Hermanoniczs, 1998).

An example of an attached film process used for small-scale wastewater treatment applications is the Rotating Biological Contactor (RBC) (Figure 3.2) which consists of a series of closely spaced (20-30 mm) flat or corrugated circular discs, normally plastic, on a slowly rotating horizontal shaft in a closely fitting contoured tank, often referred to

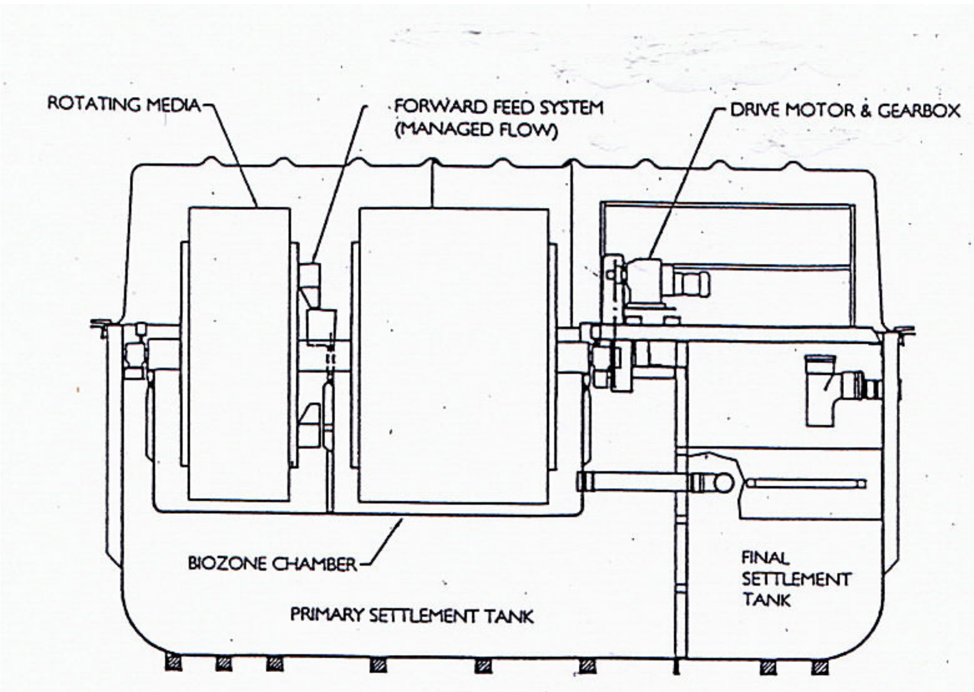

Figure 3.2 Schematic diagram of RBC system (Fitzgerald, 1995)

as the biozone (Metcalf and Eddy, 1991; Fitzgerald, 1995; Gray, 1999). Approximately 40% of the surface area of each disk is in contact with the sewage at any one time (Payne and Butler, 1993). The required surface area of the discs can be determined by using an organic loading rate of 5g BOD/m^2.d of settled sewage and a per capita loading of 40g

BOD/d of settled sewage (EPA, 2000). The discs are mounted perpendicular to the effluent flow and rotate at about 1-2 rpm (Fitzgerald 1995). Some RBC designs incorporate primary settlement tanks while all include secondary settlement tanks, which require frequent desludging. Most units provide automatic intermittent desludging whereby the settled solids are pumped back to the primary settlement tank for storage with the primary sludge which should be removed every 2-3 months (Payne and Butler, 1993 and Fitzgerald, 1995). To ensure effective performance, RBC units should be covered to protect the biofilm from the weather, eliminate flies, control odour and reduce the load on the motor turning the rotor due to wind. Covering also insulates the system, reducing heat loss and increasing the rate of oxidation (Metcalf and Eddy, 1991; Gray, 1999).

Once the RBC is operational, micro-organisms attach themselves and grow onto the surfaces of the rotating discs, forming a biofilm. Once the biofilm is established the rotating discs have two main functions:

- they facilitate contact between the biofilm and the organic matter in the wastewater followed by contact with the atmosphere for adsorption of oxygen, and
- they create high hydraulic shear on the biofilm removing excessive solids from the discs – keeping a thin biofilm for maximum efficiency of penetration - and maintaining these sloughed solids in suspension so that they can be carried to the clarifier.

(Metcalf and Eddy, 1991)

The discs are usually arranged in groups in compartments separated by baffles to minimise short circuiting, reduce the effect of surges and to simulate plug flow conditions (Fitzgerald, 1995; Gray, 1999). The first upstream sections receive a higher organic load and thus produce higher levels of solids than later sections. Where nitrification is desired, it can be achieved by extending the number of compartments such that the organic loading is low enough in the final compartment downstream to provide favourable conditions for the slower growing nitrifiers. An example of the RBC system in Ireland is the Biodisc® by Klargetser Environmental Ltd. Other examples of fixed-film small-scale package plants include the Bioclear® filter system using random packed plastic media, the Puralo® system which uses a coarse peat mixture media and the Envirocare P6 (or Titan Biotec®) system with a structured plastic media.

PEAT FILTERS

Of especial interest to this project was the performance capabilities of peat based attached growth systems as it was decided to install this type of system on the two research sites specified for secondary treatment of domestic wastewater effluent. There were a number of reasons why it was decided to install this type of system, namely the Puraflo® system:

- It was purported to produce a good quality effluent,
- As it is passively aerated the absence of an aerator suggested a reliable, low maintenance system that had cheap running costs,
- The peat media had a estimated lifespan of 15 years ,
- It was awarded the Agrément Certificate which certifies that it is satisfactory for the purpose defined and meets the requirements of the 1991 Building Regulations,
- It was the system most recommended by consultants and local authorities who were contacted during the site identification phase of the project,
- Bord na Móna assisted greatly in the identification of sites and the provision of information on their system and the project area as a whole, and
- As the systems were purchases for research purposes, Bord na Móna offered a discount on the units.

Peat has been found to be an effective medium for the treatment of septic tank effluent. Rock *et al.*, (1982) recorded an 83% reduction in COD and a 90% reduction in TSS across a peat filter. Research by Rock *et al.* (1984) carried out under laboratory conditions found that 30cm of sphagnum peat compacted to a density of 0.12 Mg/m^3 reduced the BOD_5 and SS concentration in STE by greater than 95% and 90% respectively. Further column tests by Viraraghaven and Rana (1991) showed BOD and TSS reductions of greater than 90% respectively. Rock *et al.* (1984) and Viraraghavan and Rana (1991) recorded COD removal of only 72% and 80% respectively but this was due to a COD contribution by the peat itself which was expected to decrease with time. Similarly Talbot *et al.*, (1996) recorded an average COD reduction of 78%. Brooks *et al*. (1984) tested three sphagnum peat filters, two lined and the other discharging to the subsoil, under field conditions. They reported reduction in BOD_5 and COD of greater than 90% and 80% respectively. McKee and Brooks (1994) reported reduction of 89% or greater in BOD concentrations on 11 out of 12 peat systems tested. Research carried out by Lindbo and MacConnell (2001) and Monson Geerts *et al.* (2001) on Puraflo® systems recorded greater than 90% reduction in both TSS and BOD. Monson Geerts and McCarthy (1999) reported that peat filters installed at the Northeast Regional Correction Centre (NERCC) had consistently removed greater than 90% BOD and TSS. Talbot *et al*. (1996) recorded an average reduction of 97% and 92% in BOD_5 and TSS respectively. This reduction in organic load occurs in the top of the peat where the more aggressively growing heterotrophic bacteria are dominant while nitrification takes place deeper down in the peat provided aerobic condition prevail (Henry, 1996).

The 90% reduction in NH_4-N measured by Rock *et al*., (1982) was reflected by a corresponding increase in NO_3-N. Viraraghavan and Rana (1991) recorded a 95% reduction in NH_4-N but never quantified the reduction in total N. While near complete nitrification was achieved within the system examined by Rock *et al.* (1984) there was less than 10% reduction in total N. It was found, however, that substantial (62%) denitrification occurred under anoxic conditions when there was a readily biodegradable organic carbon source present. Lindbo and MacConnell (2001) also found that while almost complete nitrification occurred within the Puraflo® system there was no reported denitrification. However, Monson Geerts *et al*., (2001) recorded a 41% reduction in total N during the winter and a 30% during the summer across a Puraflo® system. Under field conditions it was found that greater than 60% reduction in total N was achieved in three peat filters examined with denitrification suspected as the primary removal process (Brooks *et al*., 1984). Research by McKee and Brooks (1994) showed an average total N removal of between 21 and 86% depending on the source of the peat. Analysis of the peat filter effluent at the NERCC showed between a 2% and 67% reduction in total N. While the peat medium is a carbon source, relying on this would result in its accelerated decomposition. However, it was suggested (Rock *et al.*, 1984) that STE not only provided the essential nutrients for denitrification under anaerobic conditions but also a more readily source of organic C thereby minimising the need to utilise the peat as a C source. This was corroborated in the field studies of Brooks *et al*. (1984).

Research by Rock *et al.*, (1982) recorded an initial reduction in P across a peat filter of greater than 70% but this reduced to 32% in the third year of operation. However, it must be considered that this filter was underlain by sand. Apart from an approximate 10% reduction in the total phosphorous attributed to microbial assimilation no substantial removal of P by the peat medium was recorded in the laboratory trials by Rock *et al*. (1984). They concluded that P removal was dependent on the presence of Fe Al and/or Ca in the peat. Similar reductions were measured by Viraraghavan and Rana (1991). Lindbo and MacConnell, (2001) report little or no reduction in phosphorous concentration across Puraflo® modules examined. Similarly Monson Geerts *et al*. (2001) measured a reduction in P of between 3 and 11% which was probably due to microbial assimilation. Under field conditions it was found that, in three filters tested, a 58%, 62% and 96% reduction in total P was achieved. However, this was not attributed to the peat but to the protective layer of sand above the liner in the first two cases and the characteristics of the subsoil in the other case (Brooks *et al*. 1984). However, research by McKee and Brooks (1994) found that for 12 peat filters examined the peat was responsible for a 60 to 65% reduction in total P. Analysis of the peat filter effluent at the NERCC showed between 23% and 61% removal of phosphorous.

Research by Rock *et al*., (1984) showed substantial removal of total and faecal coliforms in 30, 60 and 90cm columns of peat dosed with STE. Rock *et al.* (1982), Brooks *et al*. (1984), Viraraghavan and Rana (1991), McKee and Brooks (1994), Talbot *et al*. (1996), Monson Geerts and McCarthy (1999), Monson Geerts *et al*. (2001) and Lindbo and MacConnell (2001) reported greater than 99% removal of total coliforms and faecal coliforms by peat filters.

While Henry (1996), Monson Geerts (2001) and Lindbo and MacConnell (2001) highlight the significant reduction in BOD, SS and enteric bacteria across the Puraflo® system it is clear that there is little reduction in the nutrient load. The aerobic environment of the Puraflo® promotes almost complete nitrification but does not facilitate denitrification. While significant P removal was recorded in some peat filters it is dependent on the presence of cations such as Fe, Al and Ca. It should be noted that the Puraflo® modules consist of a fibrous peat media that is the by-product of the fuel preparation process of the peat burning industry in Ireland. In this process the fibrous material in the harvested peat is removed while the rest of the peat matrix is used as a fuel. It is this part of the peat that is most likely to contain the cations required for phosphate removal.

3.2.2 Suspended Growth Systems

In suspended growth systems the microbial population and the wastewater are maintained in a mixed suspension under aerobic conditions. Aerobic conditions in the reactor are achieved by the use of diffused or mechanical surface aerators, which also serve to ensure complete mixing. The combination of adequate dissolved oxygen levels and high concentrations of organic matter in the influent wastewater produces a high rate of microbial activity and inherent organic matter degradation.

The relationship between influent organic matter concentration and the concentration of micro-organisms in the reactor is fundamental to the successful operation of a suspended growth system (Figure 3.3). If wastewater with a high concentration of organic matter is placed in a batch reactor, the micro-organisms present require an initial start-up period to acclimatise to their nutritional environment, after which they multiply rapidly in the presence of oxygen and nutrients (log-growth phase). This growth period coincides with the period of maximum organic matter removal. As numbers continue to increase, the rate of increase in microbial mass starts to decline due to limitations in the food supply. At this stage the micro-organisms enter an endogenous phase since there are not sufficient concentrations of available substrate.

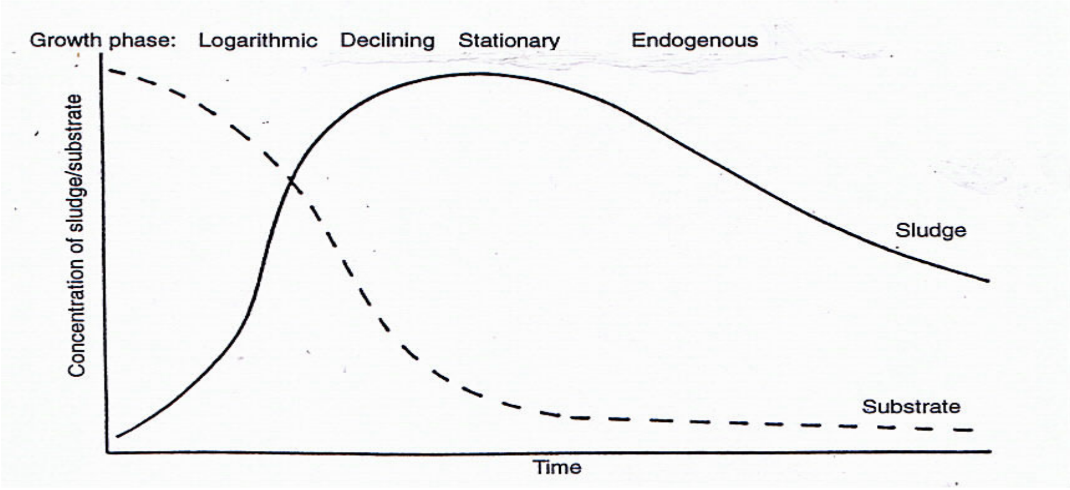

Figure 3.3 The microbial growth curve (adapted from Metcalf and Eddy (1991) and Gray (1999)).

During this phase, a phenomenon known as lysis can occur in which the nutrients contained within the dead cells diffuse into the liquor supplying any remaining viable biomass with a food source. It is during this phase that nitrification can be initiated since the heterotrophic organisms are not able to compete effectively. Most suspended growth treatment processes are continuous flow systems and hence an equilibrium concentration of bacteria must be maintained in the reactor to achieve the optimum growth characteristics (and thus organic matter degradation). While it is important that the micro-organisms decompose the organic matter as quickly as possible, it is also important that they form a satisfactory floc, which is a prerequisite for the effective separation of the biomass in the clarifier. It has been observed that the settling characteristics of the biological floc are enhanced as mean residence time of the cells (sludge age) in the system increases (Metcalf and Eddy, 1991; Gray, 1999). If, however, the sludge age is greater than an optimum period, around 10 days, a reduction in settling characteristics can be observed due to the development of filamentous bacteria which effectively reduce the density of the flocs by branching between them and creating a more open structure.

Removal of organic matter by the suspended micro-organisms within the reactor involves three mechanisms:

- adsorption and also agglomeration onto microbial flocs;
- assimilation, and
- mineralisation.

(Gray, 1999)

It is possible, by varying operating conditions, to engineer the treatment system so as to favour either assimilation or mineralisation. With assimilation, which is the conversion of organic matter to new microbial cell material, organic matter is removed by its precipitation as biomass. This results in an increased requirement for sludge separation and disposal. If, however, conditions were to favour mineralisation, which is the complete oxidation of organic matter, the volume of organic matter would be reduced under endogenous respiratory conditions, resulting in lower sludge handling costs but higher aeration costs (Gray, 1999). After the aeration tank the wastewater flows into a clarifier where quiescent conditions allow for settlement of the sludge, most of which is returned to the aeration tank to maintain its microbial concentration. To maintain equilibrium conditions the quantity of sludge drawn off should equal new biomass growth. An example of this type of process is the Enviropak® by Simon Allen Ltd.

The Sequencing Batch Reactor (SBR) is a fill-and-draw suspended growth treatment system in which all the unit processes are carried out sequentially in a single reactor (Metcalf and Eddy, 1991). Since the SBR provides batch treatment of wastewater, it can accommodate the wide variations in flow rates which are typically associated with single households (EPA, 2000). It is also possible to vary the length of the reaction cycles and amount of aeration in order to achieve nitrification and also enhanced phosphorous removal by switching between anaerobic and aerobic conditions. Table 3.1 (EPA, 2000) outlines the design criteria for an on-site SBR process for a single dwellings.

Parameter	Range
Total tank volume	0.5 - 2.0 times average daily flow
Number of tanks	Typically 2 or more
Solids retention time (days)	20 - 40
Aeration system,	Sized to deliver sufficient oxygen during aerated fill and react stage
Cycle times (hr)	4 – 12 (typical)

Table 3.1 Design criteria for the SBR process (EPA, 2000)

The SBR treatment process involves a five-step cycle:

(i) Fill — Primary effluent enters the reactor allowing the liquid level to rise from about 25% of capacity (at idle) to 100%. Air diffusers can be on or off. Normally lasts about 25% of the full cycle.

(ii) React — During this stage the air diffusers are on allowing for the completion of aerobic reactions initiated in the fill stage. Typically takes up 35% of total cycle time.

(iii) Settle — Air diffusers turned off allowing solids separation to occur, providing a clarified supernatant to be discharged. Normally more efficient than in a continuous flow systems as the reactor contents are completely quiescent.

(iv) Draw — Removal of clarified wastewater from the reactor. Time required varies from 5 to 30% of the total cycle time.

(v) Idle — In a multi-tank system it provides time for one reactor to complete its fill cycle before switching to another unit. Not an essential phase - sometimes omitted.

(adapted from Metcalf and Eddy, 1991)

Another important step that greatly affects SBR performance, although it is not included in the five basic process steps, is sludge wasting. Wasting is essential to maintain a given food-to-micro-organism ratio in the reactor in order to preserve the efficiency of the system. There is no set time period within the cycle dedicated to wasting as the amount and frequency of wasting is determined by performance requirements although, in general, sludge wasting usually occurs during the settle or idle phase.

3.2.3 Hybrid Systems

In a hybrid system the support media is fully submerged in wastewater in an aeration tank which receives oxygen from diffusers at the base of the tank (Figure 3.4). Owing to the high specific surface area of the media, large amounts of biofilm can develop and the biofilm that periodically sloughs off remains in suspension for a time thereby continuing the treatment of the wastewater, making these filters very efficient (Fitzgerald, 1995). In the water industry, there are two main types of hybrid systems referred to using the terminology, BAF (biological aerated filter) and SAF (submerged aerated filter). A SAF is a continuous flow system with settlement of the sloughed biomass achieved in a succeeding clarifier from which sludge is periodically drawn off. A BAF has no secondary clarifier and relies on a backwash sequence, typically once per day, to keep the biofilm thin enough

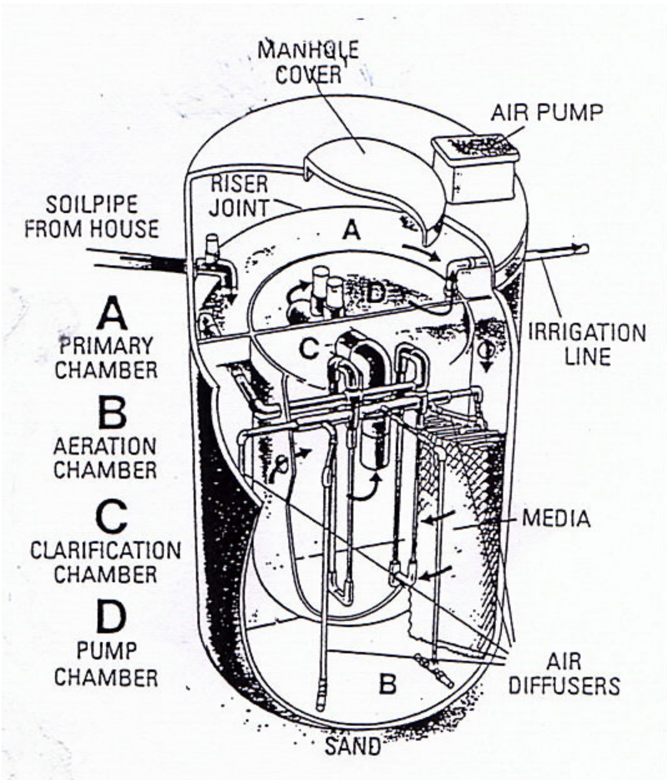

Figure 3.4 A biological aerated filter (BAF).

prevent sloughing and hence maintain high quality effluent. The backwash sequence involves taking the filter off-line and pumping clean effluent back through the media at high rates together with vigorous aeration. This has the effect of creating high shear velocities and also knocking the media particles together to loosen the excess biomass. This dirty washwater is then pumped back to the primary settlement tank where the solids co-settle with the influent wastewater. These BAF systems are more suitable for large-scale works where space is at a premium than the small-scale applications of interest in this study due to the period required off-line for the backwash (which requires a balancing tank or parallel BAF for continuous treatment), requirement for clean effluent storage tank, the complicated sequencing of the backwash and the overall extra energy requirement involved. Hence, for small-scale treatment systems (usually propriety package plants), the SAF system is adopted, although it should be noted that in some cases the term "BAF" is used to describe any generic hybrid system (EPA, 2000). Examples of such package treatment plants include the bioCycle®, Septech® 2000 (Shay Murtagh Ltd.) Bio-crete® (Delanet Concrete) and Biofilter® (FM Environmental Ltd.). The general recommended design of the surface area of media can be determined using an organic loading rate of 5g BOD/m^2.d of settled sewage and a per capita loading of 40g BOD/d of settled sewage (EPA, 2000) although each propriety system will have its own manufactures specifications.

3.3 Intermittent Sand Filters

3.3.1 General Principles

Intermittent sand filters (Figure 3.5) can be used for secondary treatment of domestic wastewater in areas where local subsoil conditions inhibit the use of conventional soil percolation systems. They are essentially shallow beds of graded sand ranging in thickness from 600 to 900 mm. The filter sand is sandwiched between a 200 mm thick layer of distribution gravel at the top and a layer of filter gravel to prevent outwash or piping of the sand at the bottom (Metcalf and Eddy, 2003; USEPA, 1999, EPA, 2000; Appendix A).

Figure 3.5 Schematic diagram of an intermittent sand filter (EPA, 2000)

STE is pumped intermittently three to four times a day onto the surface of the sand through 25 mm diameter perforated pipes embedded in the distribution gravel (EPA, 2000). The treated liquid can either be collected in an underdrain system below the filter, from where it is discharged to a polishing filter, or allowed to infiltrate into the subsoil below, if it is of sufficient depth. In the latter case there is no filter gravel required at the base of the sand filter. An impermeable liner is used to seal off the sides of the filter and, where the polishing filter is offset, the entire intermittent filter must be enclosed in this liner (EPA, 2000). It is critical that the filter is constructed with sufficient thickness of sand at the effluent collection point so that the capillary fringe at the filter base does not infringe on the zone required for treatment (USEPA, 1985). There are two types of intermittent filter commonly used, soil covered and open (EPA, 2000). In soil covered filters, a geotextile is used to separate the soil cover from the distribution gravel while, with open filters, the gravel distribution layer is exposed at the surface although they are often covered with plastic for improved maintenance and insulation (Metcalf and Eddy, 2003; EPA, 2000). While filters may be partially buried or constructed above ground, it is preferable that they are completely buried with the top of the distribution gravel at ground surface (EPA, 2000).

3.3.2 Filter Grading

The sand grain-size is fundamental to the operation of the filter, affecting the quantity of wastewater that may be filtered, the rate of filtration, the penetration depth of particulate matter and the quality of filter effluent. While, theoretically, sand with a large specific surface (m^3/m^2) is desired to support the large bacterial populations required for the treatment of STE, too fine a sand limits the quantity of wastewater that may be successfully filtered due to early filter clogging (USEPA, 1985). Ellis and Aydin (1995) reported a greater presence of bacteria in a finer sand filter, D_{10} = 0.17 mm, than a coarser one, D_{10} = 0.45 mm (where D_{10}, the effective grain size, is the size corresponding to the 10% finer than line on the grain-size curve). While coarse sand results in longer filter life (Farooq and Al-Yousef, 1993), too coarse a sand lowers the STE retention time resulting in inadequate treatment. Filter efficiency does not depend on grain-size alone, however, but also on the degree of sorting, or uniformity coefficient, of the filter sand (Whitehead, 2004). The uniformity coefficient is the ratio of the grain-size that is 60% finer by weight, D_{60}, to the effective grain-size, D_{10} (Terzaghi and Peck, 1967).

The EPA (2000) recommends an effective grain-size for soil covered and open filters in the range 0.7–1.0 mm and 0.4–1.0 mm, respectively, and a uniformity coefficient of less than 4. The USEPA (1999) recommends a similar uniformity coefficient but a grain-size, regardless of filter type, in the range 0.25–0.75 mm. In Scandinavian countries, sand filters are constructed with washed sand which has a D_{10} between 0.5 and 1.2 mm and a uniformity coefficient of less than 3.5 (Appendix A). A uniformity coefficient of less than 4 guarantees a well sorted filter minimising the problem of saturated zones created by abrupt textural changes which can limit oxidation, promote clogging and reduce the action of the filter to a mere straining mechanism (USEPA, 1985). Trials in France (Liénard et al., 2001) found that the optimum grain size for treatment performance was a D_{10} between 0.25 and 0.40mm with a uniformity coefficient range between 3 and 6, ensuring that the fines content should not exceed 2.5-3% (by mass).

3.3.3 Treatment Processes and Loading Rates

Treatment of STE by an intermittent sand filter is a result of physical, chemical and biological processes (all of which are dealt with in more detail in Chapter 4). Suspended solids are removed by filtration while other constituents are removed by adsorption (Metcalf and Eddy, 2003). Although physical and chemical processes play an important role in the removal of particulate matter, it is the biological processes, such as nitrification, denitrification and BOD removal that are most important (USEPA, 1999). The most important environmental factors that determine the effectiveness of the treatment processes are temperature and re-aeration. Temperature directly affects the rate of microbial growth, chemical reactions, adsorption mechanisms and several other factors contributing to the treatment of STE, with better performance recorded in warmer climates (USEPA, 1985).

Various studies have shown that single stage intermittent sand filters, organically and hydraulically loaded at around 5-20 gBOD/m^2d and 40-100 l/m^2d respectively, attain removal efficiencies of 90% COD, 95% BOD, 30% total nitrogen, 40% total phosphorous and 2-4 log removal of faecal coliforms (Sauer *et al.*, 1976; Pell *et al.*, 1990; Darby *et al.*, 1996; Widrig *et al.*, 1998; Van Buuren *et al.*, 1999). According to the USEPA (1999) sand filters produce a high quality effluent with typical concentrations of 5 mg/l or less of BOD and SS, as well as nitrification of 80% or more of the applied ammonia. Research carried out at higher hydraulic loading rates of 110 l/m^2d on village scale applications (Bahgat *et al*., 1999) recorded a lower mean ammonium removal of 60%. The same study also discovered that the increase in total oxidised nitrogen in the filter was less than the corresponding decrease in ammonium, indicating the occurrence of denitrification in anaerobic regions of the filter. These anoxic regions could be a reflection of organic matter build-up in the filter that leads to saturated conditions, or, they could be due to the inability of oxygen to diffuse to the lower regions of the filter (Van Buuren *et al*., 1999; Bahgat *et al*. 1999). The hydrogen ions released (H^+) during nitrification reaction leads to a drop in pH which could have a detrimental on further nitrification but would be favourable to pathogen die-off (Gerba and Bitton, 1984; Wood, 1995). A study carried out at even higher hydraulic loading rates at 250 l/m^2d (Mottier *et* *al.*, 2000) looking at primary treated effluent (comparable to septic tank effluent in ammonia concentration but with lower COD concentrations) discharged into infiltration basins, consisting of 2m deep dune sand, achieved 90% COD removal and almost complete nitrification. However, poor removal of faecal coliforms and streptococci were also reported due to high pore water velocities.

The EPA (2000) recommends a hydraulic loading rate of 40–100 l/m^2d with soil covered filters adopting the lowest hydraulic loading value. Higher loading rates produce anaerobic conditions, which have a negative influence on performance, and increase filter cleaning frequency (USEPA, 1985; Van Buuren *et a*l., 1999). The hydraulic loading rate may have to be adjusted depending on the concentration of various constituents in the STE. As the organic matter and nitrogen in the STE are subject to aerobic microbial processes in the filter, it is essential that the oxygen available within the filter, and the re-aeration rate of the filter, are sufficient to meet the demands of these processes. For that reason the USEPA (1999) recommends an organic loading rate of between 2.45 - 9.78 $gBOD/m^2d$. A higher loading rate in the region of 12 g BOD/m^2d is reported in a study of Scandinavian sand filters (Appendix A). One study found that if COD loads exceeded 50 g/m^2d there was a noticeable drop off in faecal coliform removal and problems with clogging (Van Buuren *et al.*, 1999). This is corroborated by a study which also found that ponding occurred on sand filters (D_{10}=0.29mm) at organic loading rates in excess of 47 $gCOD/m^2d$ (Darby *et al.*, 1996). Another study, looking at the effect of intermittent dosing frequency to optimise the oxygen transport into the filter, calculated a maximum oxygen flux into a 0.85mm effective size sand filter of 55 gO_2/m^2d (Schwager and Boller, 1997).

The aerobic conditions in the sand filter are maintained through the intermittent application of STE. 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 which is maximised by intermittent infiltration to maximise the convective renewal of the air phase (Boller *et al.*, 1993). However, some designs also incorporate ventilation pipes down through filter to improve aeration at depth (Washington State, 2000). In general, studies have 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 (Boller *et al.*, 1993; Darby *et al.*, 1996) which concurs with experience on other biofilm processes such as large-scale trickling filter applications. Dosing frequencies of between 4 and 24 times per day have been reported in the above studies and it would appear that a design value in the region of 12 doses per day gives the best treatment performance, although this is obviously also dependant on the overall hydraulic loading rate.

Phosphorous removal in sand filters is controlled mainly by adsorption and mineral precipitation reactions as has been covered in Section 4.4.2. The adsorption capacity of a sand is regulated by the occurrence of natural minerals such as iron, calcium and aluminium but also affected by the chemical characteristics of the effluent (Redox potential and pH) within the filter. Although several studies have shown sorption of phosphate onto calcareous sands (Section 4.2.2), in certain cases non-calcareous sands have been shown to be even more effective at phosphate removal, for example, in an aluminium enriched sand (Robertson, 2003). The lack of buffering of the septic tank effluent in the non-calcareous sands promotes lower pH levels of the effluent within the sand which in turn favours precipitation of the phosphate onto the aluminium sites. For sands with poor adsorption characteristics (which can be determined by simple laboratory experiments to establish the Freudlich amd Langmuir isotherms) the amendment of the media with a high phosphate sorption capacity material, such as fly ash could be considered (Cheung and Venkitachalam, 2000).

The removal of microbiological organisms in sand filters has been measured is several of the above studies (Van Buuren *et al*., 1999; Mottier *et al.*, 2000) indicating a fall off in performance at higher hydraulic loading rates. However, the application of a 0.8m sand filter as a polishing filter for the disinfection of secondary treated effluent has been shown to be effective at hydraulic loading rates of 165–350 l/m^2d giving 5.0-3.8 log removal of faecal coliforms respectively (Salgot *et al.*, 1996), with final effluent concentrations of 10-147 CFU/100ml. Several detailed studies have also looked at the removal of both virus and bacterial pathogens and indicators in laboratory sand columns. In general the removal of both bacteria and viruses has been shown to occur within the first 0.3m depth of sand (Gross, 1990; Hua *et al.*, 2003) which has also been confirmed by a few in-situ trials carried out in the sandy subsoils (Nicosia *et al.*, 2001) and dunes (Schijven *et al.*, 1999) using dosed bacteriophages (viruses). The retention of micro-organisms by the sand is generally regulated by physico-chemical sorption onto the media and also filtration and sorption on the biofilm (Sélas *et al.*, 2002). The effectiveness of this retention is dependent on the pH of the effluent and presence of any cations, which have been shown to effectively shield viruses and thus facilitate their transport through the media (Zhaung and Jin, 2003). Equally, studies have shown far greater virus removal under unsaturated conditions compared to saturated conditions due to higher sorption and also inactivation at the air-water interface (Jin *et al.*, 2000). This result was corroborated for the removal of pathogenic bacteria which was shown to be higher at lower water saturations due to accumulation of bacteria at the air-water interface (Schäfer *et al.*, 1998).

Several trials have also looked at recirculating sand filters (Van Buuren *et al*., 1999; Christopherson *et al*., 2001) whereby a fraction of the effluent is pumped back to the join the influent normally at a recirculation ratio in the range of 3:1 – 5:1. This generally has the effect of increasing the hydraulic loading rate and dosing frequency but reducing the concentration of influent which can improve the mass transfer characteristics through the biofilm. Often recirculating sand filters are designed to target specific compounds such as nitrogen where the recirculated nitrified effluent can be passed through an anoxic zone for example, to achieve denitrification before re-entering the filter.

3.3.4 Stratified Sand Filters

All intermittent sand filters and mechanical aeration systems must be succeeded by a polishing filter (EPA, 2000). Only one of these types of filter, the stratified sand filter, will be considered during the course of this literature review as it is an objective of this project to examine its efficiency both as a secondary treatment process and a polishing filter for domestic wastewater. Only one paper (Nichols *et al.,* 1997) examining the efficiency of stratified sand filters as a secondary treatment system of STE has been uncovered during the course of this literature review, although Washington State (2000) has produced a set of guidelines for their design and operation. It is important to note that the research summarised here (Nichols *et al.,* 1997), was carried out under American conditions and, therefore, absolute values will differ from those experienced in Ireland. The sand filter consisted of three layers of sand, decreasing in coarseness with depth, separated by layers of pea gravel (Figure 3.6). All the sand layers had a uniformity coefficient of 1.7.

100 MM **DISTRIBUTION GRAVEL**
200 MM coarse sand ***(0.4 TO 1.4* MM*)* ; D_{10} = 0.56 MM**
60 MM **PEA GRAVEL (20-30 MM)**
120 MM medium sand ***(0.1 TO 0.5* MM*)* ; D_{10} = 0.29 MM**
30 MM **PEA GRAVEL (20-30 MM)**
180 MM fine sand ***(0.1 TO 0.5* MM*)* ; D_{10} = 0.18 MM**
200 MM **PEA GRAVEL (20-30 MM)**
50 MM **SAND TO PROTECT PVC LINER**

Figure 3.6 Schematic cross section of a stratified sand filter (adapted from Nichols *et al.,* 1997)

The sides and bottom of the filter were separated from the natural soil by a 0.76 mm PVC liner and the distribution layer was overlain by untreated building paper and soil cover. The Washington State stratified sand filter is more or less the same design at a hydraulic loading rate of 49 l/m^{2}d, although, interestingly recommends the addition of vertical vent tubes at 200mm centres to improve aeration throughout the depth of the filter.

The design hydraulic loading rate of the filter was 51 l/m^2d over a surface area of $33.4m^2$. For the first 17 months the filter was loaded to one quarter of its design capacity at 12.5 l/m^2d which resulted in almost complete nitrification of inflow ammonium after an initial acclimatisation period of 25 days. Analyses of the influent nitrogen species revealed a 87% NH_4-N to 13% organic-N split, with an average NH_4-N concentration of 42.7 mg/l, which equates to a nitrogenous loading rate of 3.7kg total-N/person/year. When the filter was subsequently loaded to its design capacity for 12 months, the effluent ammonium to nitrate ratio showed an increase to average levels (as N) of 6.2 : 20.5 mg/l respectively indicating partial nitrification. It was also observed that nitrification ceased at wastewater temperatures of less than 4^oC.

Phosphorous removal was greater than 98% for the first 25 days but thereafter the percentage removed was greatly reduced, due to the unavailability of sorption sites, until eventually inflow and outflow phosphorous concentrations were very similar. The average orthophosphate loading rate during the study was 1.2 kg ortho-P/person/year.

This stratified sand filter also removed, on average throughout the trials, 90% of the organic carbon and 99.97% of the faecal coliforms. While such process efficiencies for organic carbon removal and faecal coliform numbers treated by the stratified sand filter are similar to those obtained using intermittent sand filters, the degree of nitrification in the stratified sand filter is a lot more favourable.

4. CONTAMINANTS AND SUBSOIL ATTENUATION PROCESSES

4.1 Introduction

Site selection is fundamental to the successful design and operation of domestic wastewater treatment systems and hence their capacity to contaminate groundwater and surface water, since further treatment of effluent from the engineered section of the system is dependent on the attenuation capacity of the subsoil. This chapter begins with an outline of subsoil characteristics, which includes an outline of the BS5930 classification scheme and a synopsis of the background to Irish subsoils. The factors and processes contributing to the contaminant attenuation in the subsoil and the main contaminants in STE are then examined. Finally, the contamination of groundwater in Ireland by STE is reviewed.

4.2 Subsoil Characteristics

As defined in the Groundwater Protection Scheme (DoELG *et al.*, 1999), s*ubsoils* are the most important feature in influencing groundwater vulnerability to pollution. They act as a protecting filter layer over groundwater, their effectiveness depending on type, permeability and thickness. Subsoils are the 'loose' unlithified sediments present between topsoil and bedrock. The word 'soil' is the general term used by engineers to describe all 'Quaternary deposits', 'drift' and 'overburden'. However, it is useful to distinguish between the topsoil (the upper metre or so affected by biological and weathering processes) and the underlying subsoil, as the latter is of most relevance in attenuating contaminants from on-site wastewater systems.

There are a number of ways (geological, soil science and geotechnical engineering) of describing properties of subsoils, i.e. the material and mass characteristics which give the subsoil name – particle size distribution, plasticity and dilatancy, density/compactness and discontinuities. The method used to describe subsoils during the course of this project will be the geotechnical approach based on BS5930.

4.2.1 Using BS5930 to Describe Subsoils

Under BS5930, the British Standard Code of Practice for Site Investigations, subsoils are described primarily on the basis of their material characteristics (which give the subsoil name) such as particle size distribution (including texture), plasticity and dilatancy; and the mass characteristics such as density/compactness, bedding and discontinuities (Daly and Swartz, 1999). The material and mass characteristics are factors relevant to the assessment of permeability and attenuation capacity of subsoil and thus to groundwater or surface water vulnerability. Work carried out by the GSI to-date suggests that it will be possible, with further research, to relate the subsoil name obtained from using BS5930 to a broad permeability class.

Particle size Distribution

A subsoil consists of rock particles, mineral grains and sometimes organic matter, together with variable amounts of air and water. The rock particles and mineral grains are split into groups depending on size, as shown in Figure 4.1. The relative proportion of sand, silt and clay particles in the subsoil is the most important factor influencing the permeability, i.e. fine textured subsoils have low permeability while coarse subsoils have high permeability, all other things being equal.

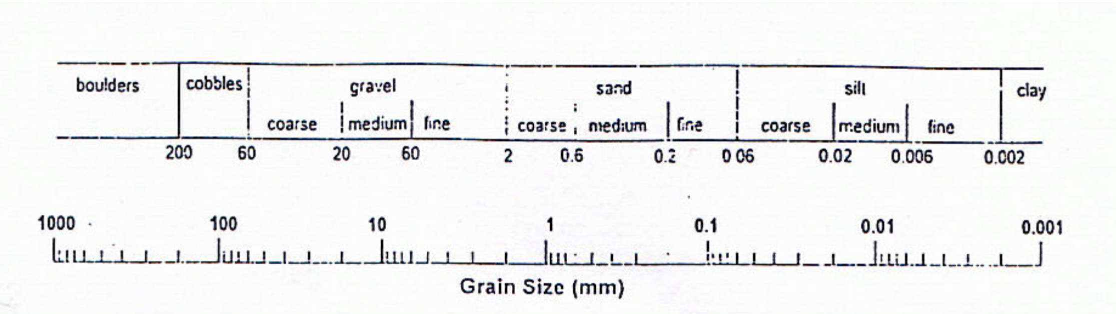

Figure 4.1 Main subsoil size groups (BS5930).

Density/Compactness

The more dense or compact the subsoil, the lower the permeability.

Discontinuities

Discontinuities in the subsoil provide preferential flow paths for percolating liquids, reducing the attenuation capacity of the soil and thus increasing the likelihood of contamination. Typical discontinuities include fissures/cracks caused during deposition of the sediment; by weathering; by plant roots; by soil fauna or by parting due to the structure of the subsoil.

Bedding

Far from being a uniform matrix, subsoils often contain beds, laminations or lenses of different sediments. This heterogeneity can have a far reaching impact on water and contaminant movement in the subsurface. For example, if a low permeability clayey sediment underlies a coarse grained sediment of greater permeability the vertical movement of liquid through the subsoil would be hindered resulting in ponding or horizontal spreading of the liquid which could jeopardise water courses.

Plasticity

Plasticity is the ability of a material to deform, or change shape, without breaking when subject to an external force or pressure. The greater the plasticity of a subsoil, the higher the clay content, as one of the main properties of clay is that it is highly plastic.

Dilatancy

Dilatancy, which describes the reaction of subsoil to shaking, can be used to assess the relative silt and clay content of a subsoil. There are three terms that describe a sample’s reaction to shaking or patting: rapid, slow and none. A sample has a rapid reaction if water quickly rises to the surface, making the surface shiny and wet looking. If the sample is subsequently squeezed, the shiny appearance goes away quickly. Samples have a slow reaction if it requires vigorous shaking to notice a change and there is little further change after squeezing. Generally sands and silts react rapidly in a dilatancy test while a clay will not react.

4.2.2 Background to Irish Subsoils

A high proportion of the lowland area of Ireland is covered with a significant thickness of Quaternary deposits. These have an important impact on the underground part of the hydrological cycle, either as aquifers or in the way they affect water moving into underlying rock aquifers. The Quaternary period is the most recent geological period, lasting from 1.6 million years ago to the present. It started as a period of global cooling in northern Europe, marking the onset of what is commonly known as the *Ice Age*.

The effect of the Quaternary deposits on the groundwater movement is largely a function of their permeability and, to a lesser extent, their thickness. Owing to their extensive nature and relatively high storage and in spite of their variable permeability, these deposits are an important source of baseflow in the summer, particularly in areas with no rock aquifers (Daly E., 1985). The Quaternary deposits of highest permeability are the various types of sands and gravels, namely glaciofluvial sands and gravels and esker sands and gravels. Where these deposits are sufficiently thick, extensive, saturated and clean they are considered to be aquifers in their own right. Where these deposits are not sufficiently extensive, or perhaps saturated, they are still important as they will allow a high proportion of recharge water to enter an underlying rock aquifer with which they are in hydraulic continuity. The onset of the warmer postglacial period (about 14,000 years ago) saw the disintegration of the ice sheets and the initiation of peat development in the resultant lake basins.

The main subsoils found in Ireland are:

Tills

Till, also known as boulder clay, is sediment deposited by or from glacier ice. It is the most commonly and wide-spread Quaternary subsoil type (TCC *et al.*, 1998). Tills are often tightly packed, unsorted, unbedded, possessing many different particle and clast sizes and types which are often angular or subangular.

Glaciofluvial Deposits

Glaciofluvial sands and gravels are deposited by running water and so represent the stagnation and decay of the ice sheets. The gravels are usually stratified and clasts usually have rounded edges, being polished rather than striated. These deposits, which give rise to a variety of different landforms, including eskers and kames, are generally thickest in areas close to the major halt stages of the various ice sheets. It is in these regions that most of the important sand and gravel aquifers have been located and are likely to be located in the future (Daly E, 1985).

Glaciolacustrine Deposits

Glaciolacustrine deposits, which consist of sorted gravel, sand silt and clay, are normally found in wide flat plains, or in small depressions in the landscape. Due to their mode of deposition they are usually associated with glacial lakes. Deltas, which are formed as sediment deposited at a river mouth, usually contain interbedded sands and gravels, which dip lakeward, and 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 clay and silts.

Alluvium

Alluvium is a product of river flow and flooding and is usually of sand/silt grade. It may however contain gravel beds depending on its location. Because it occurs in river flood plains alluvium is not frequently encountered in septic tank installation, however any house built close to a river may be close enough to the flood plain for it to be considered as a potential site for a septic tank system (TCC *et al*., 1998).

Peat

Peat, which once covered 16% of the land surface of Ireland (TCC *et al*., 1998), consists mostly of vegetation which has only partially decomposed. This vegetation fills in marshes, ponds and other lakes carved out and left by Quaternary ice sheets. Thus, in Ireland, peat usually overlies badly drained glaciolacustrine silts and clays.

4.3 Factors and Processes Contributing to Contaminant Attenuation in the Subsoil

4.3.1 Physical Factors and Processes

(i) Permeability

For effective treatment of STE by the subsoil the permeability of the porous medium is critical as it controls flow of the percolating effluent and thus contact time between the STE and soil particles and associated biofilms. Laminar flow through a homogeneous isotropic saturated or unsaturated soil is described by Darcy's law (Domenico and Schwartz, 1998):

$$v = -K\frac{dh}{dl}$$

where, v = the volumetric flow rate per unit surface area (m/s)
K = the hydraulic conductivity (m/s)
dh/dl = the hydraulic gradient

For convective transport of pollutants in the subsoil, the linear velocity of the water rather than the Darcy velocity should be considered (Bouwer, 1984). The linear seepage of water in a porous medium can be approximated as:

$$v_m = \frac{v}{n_e}$$

where, v_m = linear seepage (m/s)
v = Darcy velocity (m/s)
n_e = effective porosity, i.e. the porosity available for fluid flow

The hydraulic conductivity, which may be considered as a measure of the ease with which liquid flows through a given medium, is dependent both upon the physical properties of the flowing liquid and the characteristics of the transmitting medium. The physical properties of STE, i.e. viscosity, density and specific weight, are practically constant, so the hydraulic conductivity may be considered as a function of the permeability of the medium alone. The main soil properties that affect permeability are its degree of saturation and subsoil geometry.

Unsaturated flow through the vadose zone, i.e. between the percolation trenches and water table, is important in downward movement and attenuation of microbial and chemical pollutants. These unsaturated conditions, essential in terms of residence time of the STE and promotion of an aerobic environment for treatment processes, can be achieved by application of a suitable loading rate in the percolation trenches which is a minute fraction of the soil's saturated hydraulic conductivity. In the saturated zone the driving force for groundwater flow is hydraulic head, defined as the sum of the elevation head and the pressure head. When a soil is saturated, all of the pores are water-filled, pressure head is positive and conductivity is maximal. When the soil dries out some of the pores become air-filled and thus the conductive portion of the soil's cross-sectional area diminishes. These air-filled pores are assumed to act like solid particles inhibiting fluid flow (Hillel, 1998). As desaturation develops, the first pores to empty are the largest ones, thus confining flow to the smaller less conductive pores. These large empty pores must then be circumvented by the percolating fluid increasing flow path tortuosity. Thus, it is essential to remember that while Darcy's law can be used to describe flow in the unsaturated subsoil, the unsaturated hydraulic conductivity, $K\theta_v$, is not constant. Furthermore, under unsaturated conditions the pressure heads are less than atmospheric, i.e. water is held in suction – capillary forces bind water to the soil particles, and effluent flow, therefore, may occur either as film creep along the walls of wide pores or as tube flow through narrow water-filled pores. The closer proximity of the STE to the solid phase in the unsaturated material and the longer residence time can be expected to enhance removal of pathogens and chemicals from the percolating fluid.

Although there is not a universally accepted working relationship between grain-size and permeability, widespread research (Hazen, 1892; Loudon, 1952; Norris and Fidler, 1965; Masch and Denny, 1966; Summers and Weber, 1984; Shepherd, 1989; Schuh and Cline, 1990; Alyamani and Sen, 1993) has shown that a strong correlation exists between permeability and some representative grain diameter. While permeability of coarse gained sediments is greater than fine grain sediments, all other things being equal, grain-size alone does not dictate the permeability of the sediment as orientation of the particles, degree of sorting within the sediment and presence of preferential flow paths also have an effect. Research has shown that the permeability generally decreases for poorly sorted sediments (Hazen, 1892; Fraser, 1935; Carman, 1939; Krumbein and Monk, 1942; and Beard and Weyl, 1973). It must be noted with respect to unsaturated flow, however, that at low volumetric water contents the relationships that hold true in saturated flow may be invalid. For example, at lower volumetric water contents, coarse materials may have very few saturated pores and could thus have a lower unsaturated hydraulic conductivity than finer grained sediments which would have more saturated pores (Fetter, 1994).

PREFERENTIAL FLOW

Long continuous openings in the soil matrix have long been regarded as very important in the preferential movement of water through the soil profile. Preferential flow is a rapid transient physical phenomena occurring in the larger and predominantly vertical continuous soil pores (Di Pietro *et al*., 2003). Preferential flowpaths also influence solute transport through natural soils (Larson, 1999). The importance of macropores in providing preferential flowpaths through the subsoil for percolating effluent is illustrated in Table 4.1. Williams *et al*., (2000) outline various studies that showed that contaminants appeared

Soil Type	**Retention**	
	Intact	**Disturbed**
Silt loam	78.0	99.8
Sandy loam	21.0	95.0

Table 4.1 Retention of *E. coli* in Kentucky soil cores (two soil types) either intact (macropores present) or sieved and mixed (macropores absent), based upon the ration of the concentration of cells in the leachate compared to the concentration in the irrigation water (Wood, 1995).

faster at a given soil depth than would be predicted if the water flowed through the entire volume of soil due to the prominence of preferential flowpaths. It should be noted, however, that not all large voids are preferential flowpaths as some are hydrologically effective in chanelling flow through the soil while other are not. It is reported (White, 1985; Kung, 1990) that flow along preferential paths is the cause of rapid movement of dissolved and suspended matter through soil. While the importance of macropores is confirmed by Weiler and Naef (2003) they found that of the macropores present in their study area only a few contributed significantly to preferential flow. Williams *et al.*, (2000) observed that the risk of groundwater contamination is increased by the presence of active preferential flowpaths. Kung (1990) found that preferential flowpaths were the dominant flow pattern in a sandy vadose zone monitored. Öhrström *et al.* (2004) observed dye movements through preferential flowpaths rather than the total soil matrix in an unstructured sandy loam. Beven and Germann (1982) report that the presence of preferential flow paths (pores formed by soil fauna, pores formed by plant roots, cracks and fissures and natural soil pipes) may lead to spatial concentrations of water flowing through unsaturated soil which would have important implications for the rapid transport of STE through the vadose zone with limited treatment. They also found that the impact of macropores is governed by the water supply to the macropores, the water flow in the macropores and the water transfer from the macropores to the surrounding soil matrix. Weiler and Naef (2003) conclude that the volume of water received by macropores alters the percolation depth and transport of solutes. Allaire-Leung (2000) reported on how the increase in tortuosity of macropores reduced the effect of these preferential flowpaths.

(ii) Filtration

Particle size distribution also plays an important part in the removal of suspended solids, including bacteria, from the STE by acting as an effluent filter. There are three filtration mechanisms (McDowell-Boyer *et al*., 1986):

- Surface Filtration – occurs at soil surface when particles are too large to penetrate the soil resulting in biomat formation,
- Straining – particles small enough to enter the soil pores are removed by mechanical straining as the effluent percolates through the subsoil,

- Physico-Chemical Filtration - this occurs when very small particles, i.e. where the ratio of soil grain diameter to that of the particulate is greater than twenty, are retained if the attractive forces predominate when the particles collide with the soil.

Siegrist *et al*. (2000) refer to research by Updegraff (1983) who found that straining becomes an effective mechanism when the average cell size is greater than the grain size d_5 (d_5 is the particle diameter at which 5% of the particles in mass are smaller and 95% of them are larger). Hagedorn (1984) reports that bacterial travel is limited by physical straining or filtration, with the degree of retention inversely proportional to the particle size of the soil. However, Johnson and Atwater (1988) report on a study which found that a coarser–textured sand is just as effective overall as a loamy sand in removing coliform bacteria, although the fine-textured material is more effective in the first 15 cm. Filtering starts with the trapping of the larger suspended particles at the surface or at some depth. Individual particles may be blocked in the pores or several particles may interact to form a bridge in the pore that prevents further movement of these particles in the direction of flow. Once movement of the larger suspended particles has been blocked, these particles themselves begin to function as a filter and trap successively smaller suspended particles (Canter and Knox, 1985). Bouwer (1984) reported that bridging occurred when the diameter of the suspended particle was larger than 0.2 times the diameter of the particles constituting the porous medium. Depending on how particles in the porous medium were packed, bridging also occurred if the diameter of the suspended material was more than 0.07 times that of the particles in the medium. When the size of the suspended particles was less than 0.07 times the particle size of the medium, the suspended particles moved through the medium without bridging or blocking.

4.3.2 Chemical Factors and Processes

(i) Adsorption

Adsorption is a factor in the removal of phosphates, ammonium, organic compounds, bacteria and viruses from STE. It is an important phenomenon in soils that contain clay as the very small size of clay particles, their generally platy shapes and the occurrence of large surface area per given volume make them ideal adsorption sites (Gerba and Bitton, 1984). The iron, aluminium and hydrous oxides coating the subsoil clay minerals and magnesium-hydroxy clusters or coating on the weathered surfaces of ferromagnesium minerals provide excellent sorption sites (Miller and Wolf, 1975).

Adsorption is the physical and/or chemical process in which a substance accumulates at a solid-liquid interface (Mihelcic, 1999). It results from the differential forces of attraction or repulsion occurring among molecules or ions of different phases at their exposed surfaces (Hillel, 1998). During the process of adsorption a chemical species passes from one bulk phase to the surface of another where it accumulates without penetrating the structure of this second phase. Desorption refers to the reverse of the process of adsorption (Burchill *et al*., 1981). Because adsorption removes contaminant from the fluid phase, even if only temporarily, it acts to slow the movement of the contaminant through the subsoil. The term "retardation" is thus commonly used to describe the effects of contaminant adsorption and a retardation coefficient assigned to the adsorbate (the substance being adsorbed). It is important to make the distinction between adsorption, which is a superficial attachment or repulsion, and absorption, which involves the transfer of a molecule from one phase to another, via their interface, resulting in the alteration of the composition of the second phase. Adsorption can be divided into two categories: chemical adsorption and physical adsorption - although as both can occur simultaneously in soils it is often impossible to distinguish between them.

Chemical adsorption, or chemisorption, involves valence forces of the type which bind atoms to form chemical compounds of definite shapes and energies (Burchill *et al*., 1981). It tends to occur at specific adsorption sites, and does not proceed past the monolayer stage, i.e. all of the adsorbed molecules are in contact with the surface layer of the adsorbent. Chemisorption can progress as either an endothermic or exothermic chemical reaction, but strong chemical bond formation is often associated with large exothermic heats of reaction (releasing large quantities of energy to the environment). As an adsorptive molecule approaches the adsorbent surface, an energy barrier has to be overcome for reaction to take place; hence chemisorption, and desorption, usually involve an activation energy. Chemical reaction at a surface may actually prevent the original species ever being recovered.

Physical adsorption is a rapid, non-activated process which occurs at all interfaces. Transport processes, like diffusion or fluid flow to the interface, are rate-determining, the heats of adsorption are relatively low, and the chemical nature of the adsorptive species is essentially preserved in the processes of adsorption and desorption (Burchill *et al*., 1981).

Clay particles are of colloidal nature and, when dry, neutralising counter-ions are attached to their surfaces. When wetted they display negatively charged surfaces that attract and adsorb cations (Bouwer, 1984). A hydrated colloidal particle of clay or humus forms a micelle, in which the adsorbed ions are spatially separated, to a greater or lesser degree, from the negatively charged particle. Together, the particle surface, acting as a multiple anion, and the "swarm" of cations hovering over it, form an electrostatic double layer (Hiemenez, 1986; Harter, 1986).

The cations of the double layer can be replaced by other cations introduced in the STE through a process called cation exchange which may be regarded as a form of adsorption (adsorption mainly applies to organic compounds while ion exchange applies to inorganic compounds). The smaller the cation and the higher its charge, the more strongly it is adsorbed and the more readily it replaces more weakly held cations (Bouwer, 1984). Of the more common cations in soil systems, Li^{+} is the most weakly held, followed in order of increasing bond strength by Na^{+}, H^{+}, K^{+}, NH_4^{+}, Mg^{2+}, Ca^{2+} and Al^{3+}. Thus, if a percolating solution contained more Ca^{2+} than that in the original subsoil, the Ca^{2+} would replace K^{+}, Na^{+}, and the other more weakly bound cations held in the adsorptive mantle surrounding the clay particles until the cations adsorbed to the clay were again in equilibrium with the cations in solution. Cation exchange reactions are rapid and reversible; therefore, the composition of the exchange complex responds to frequent changes in the composition and concentration of a percolating solution. The composition of the soil's exchange complex in turn governs the soil's pH, as well as swelling and flocculation-dispersion tendencies (Hillel, 1998).

The total cation-exchange capacity (measured in milliequivalents of cations per 100 grams of soil) of a soil depends not only on its clay and organic matter content but also on the type of clay present. The cation-exchange capacity of clay minerals ranges from around 10 meq/100g for kaolinite to about 90 meq/100g for montmorillonite while the cation-exchange capacity of stable organic matter in the soil may exceed 200 meq/100g. The cation-exchange capacity of soil materials, therefore, generally ranges from essentially zero for sands and a few meq/100g for light textured soils to about 60 meq/100g for clay soils (Bouwer, 1984).

(ii) Precipitation

Precipitation is the separation of an insoluble product when two solutions are mixed together. It occurs in soils when the soluble ortho-phosphate ions (PO_4^{3-}) present in percolating STE, or sorbed onto soil colloids, react with ions in the soil solution. The nature of the product and the efficiency of this precipitation process, described as phosphate fixation, depends on the cations present and the pH of the soil.

In strongly acidic soils there are sufficient aluminium, iron and manganese ions in solution to cause the precipitation of all dissolved phosphate ions. Zanini *et al.* (1998) report that constant nitrification also generates acidity which can increase the number of cations present – in theory nitrification uses 7.14 moles of alkalinity (as $CaCO_3$) per mole of NH_4 oxidised. Because of its large specific surface (m^2/m^3) the freshly precipitated phosphate has a degree of solubility; this solubility decreases over time due to the growth of larger particles of precipitate at the expense of smaller ones as a result of the dynamic exchange between the precipitate and its dissolved ions.

In alkaline soils, phosphates quickly react with calcium to form a sequence of products of decreasing solubility, e.g. phosphate can react with calcium carbonate to form monocalcium phosphate, dicalcium phosphate and then tricalcium phosphate which is less soluble. Although tricalcium phosphate is quite insoluble, it may undergo further reactions to form even more insoluble compounds such as hydroxy-, oxy-, carbonate- and fluorapatite compounds (Brady and Weil, 1996).

4.3.3 Biological Factors and Processes

Anaerobic conditions can exist in the vadose zone, an otherwise aerobic region, due to its unhomogeneous anisotropic nature. This enables both aerobic and anaerobic biological transformations to occur, such as organic matter decomposition, nitrification and denitrification.

(i) Organic Matter Decomposition

Microbial decomposition of organic matter proceeds most rapidly in the aerobic zones where micoorganisms in the subsoil use the oxygen present as an electron acceptor during the decomposition of the substrate (McCarty *et al*., 1984). Organic compounds in the percolating STE are subject to enzymatic oxidation resulting in carbon dioxide, water and energy production in a process that can be represented by:

$$R-(C,4H) + 2O_2 \qquad CO_2 + 2H_2O + \text{energy}$$

Aerobic organisms cannot function in localised anaerobic zones within the subsoil treatment system so anaerobic or facultative organisms, such as methanogenic bacteria, become dominant. Since anaerobic decomposition is a slower process, pockets of partially decomposed organic matter can often accumulate in the subsoil. Anaerobic decomposition releases relatively little energy for the organisms involved and produces a wide range of partially oxidised organic compounds, such as organic acids, alcohols and methane gas (Brady and Weil, 1996), some of which have a detrimental effect on the subsoil "micro-environment" by inhibiting flora and fauna growth while others, notably methane gas, contribute to the greenhouse effect. Anaerobic decomposition is a three step process as follows :

(i) Hydrolysis — conversion of insoluble high molecular-weight organic matter to soluble organic matter by hydrolytic bacteria,

(ii) Acidogenesis	conversion of soluble organics to a range of organic acids and alcohols, hydrogen and carbon dioxide by acidogenic bacteria,
(iii) Methanogenesis	conversion of hydrogen and ascetic acid to methane and carbon dioxide by methanogenic bacteria

(Brady and Weil, 1996)

The rate limiting step is considered to be the methanogenesis due to the slower growth rate and higher sensitivity of the methanogenic bacteria.

(ii) Mineralisation, Nitrification and Denitrification

The nitrogen content in STE (as described in section 2.2.3) is typically 70% to 90% ammonium and 10% to 30% organic nitrogen (Lance, 1972; Nilsson, 1990 (both cited in Siegrist *et al*., 2000) and Gold and Simms, 2000). Organic nitrogen contains amine groups which are broken down by soil micro-organisms, by a process called mineralisation, into simple amino compounds which are then hydrolised releasing nitrogen in the form of ammonium (NH_4^+). The reduction in the ammonium concentrations in the STE as it percolates through the unsaturated subsoil is accompanied by an increase in nitrate (NO_3^-) concentration brought about by the process of nitrification. Nitrification can be limited by low temperatures, insufficient oxygen or by lack of alkalinity (Van Buuren *et al.*, 1999). Nitrification is a process used by only a few genera of autotrophic bacteria as a means to generate energy as the energy yield for the reaction is low resulting in very slow growth rates (Wood, 1995). It is a two step process with the first step, the rate limiting step, consisting of the conversion of ammonium to nitrite (NO_2^-) by nitrosomonas sp.. This reaction is followed closely by the conversion of nitrite to nitrate by another group of autotrophic bacteria, nitrobacter, resulting in a net increased soil acidity through the production of hydrogen ions (H^+).

If the percolating nitrate solution enters an anaerobic pocket, or zone of reduced oxygen concentration, where there is appropriate bacteria and a supply of readily available carbon source in the form of organic substrate present, it will undergo denitrification. Denitrification involves a series of reactions used by a wide range of facultative anaerobic heterotrophs in which oxidised forms of nitrogen are used as alternative electron acceptors and nitrate is reduced to gaseous nitrogen (NO, N_2O or N_2). The proportion of the three main gaseous products seems to be dependent on the prevalent pH, temperature, degree of oxygen depletion, and concentration of nitrate and nitrite ions (Brady and Weil, 1996). N_2 is generally produced under anaerobic conditions while under conditions of low pH, reduced oxygen and high nitrite and nitrate it is generally nitric oxide gas (NO) and nitrous oxide gas (N_2O) that are produced.

4.4 Key Contaminant Groups in Septic Tank Effluent

4.4.1 Organics and Suspended Solids

Biodegradable organics in either dissolved or suspended form can be characterised by biochemical oxygen demand (BOD) or chemical oxygen demand (COD) which measure the amount of oxygen required for biochemical and chemical oxidation respectively. Volatilisation and adsorption, followed by microbial degradation are the main processes for removal of soluble biodegradable organics in the subsoil (Siegrist *et al.*, 2000). Suspended solids, including organic and mineral matter, can be removed through a combination of physical straining and biological degradation processes (Reed *et al.*, 1994). Laboratory columns simulating the unsaturated zone often remove 80 to 90% of organic C in STE when aerobic conditions are maintained suggesting aerobic oxidation and retention to be the main removal processes (Wilhelm *et al.*, 1994b). As outlined in Section 4.3.1, subsoils act as effective porous media biofilters in removing suspended matter within a specific particulate size range. The large specific surface of individual soil particles and subsoil organic matter provide high potential for biofilm development which is of great importance in the breakdown of organics in the percolating STE (Bouwer, 1984). Micro-organisms use O_2 as the electron acceptor in the oxidation of organic C to CO_2 (Wilhelm *et al.*, 1994a).

Van Cuyk *et al*. (2001) report that the optimum biological degradation potential of the system is only achieved after an acclimatisation period of 2 to 3 months.

4.4.2 Inorganic Constituents

(i) Nitrogen

The fate of the nitrogen introduced in the STE is dependent on its initial form as well as biological and chemical activity in the subsoil treatment system. The removal mechanisms for nitrogen include mineralisation, volatilisation, adsorption, cation exchange, incorporation into microbial biomass, nitrification and denitrification. In a properly installed and operating system the predominant nitrogen retention reaction is ammonium adsorption while the predominant transformation reaction is nitrification (Siegrist *et al*., 2000). The two forms of nitrogen that are of concern to the pollution of groundwater and surface water are ammonium and nitrate. Ammonium is toxic when present in high concentrations while nitrate presence in drinking water has been linked to methaemoglobinaemia (blue baby syndrome) in infants and it is also assumed to promote eutrophication in estuarine environments (Harman *et al*., 1996).

Ammonium ions can be discharged directly from a septic tank to the percolation trench or they can be formed by the mineralisation of organic nitrogen, contained in the STE, in the upper layers of the soil system. Prolonged adsorption of NH_4-N appears most effective under anaerobic conditions, where nitrification is inhibited with Canter and Knox (1985) reporting that under the anaerobic conditions normally prevailing directly below the percolation trench, ammonium ions are readily adsorbed onto negatively charged soil particles. They are also adsorbed by organic colloids in the soil (Jenssen and Siegrist, 1988). After the adsorption capacity of the first few inches of soil is reached, the ions in the percolating STE will travel further to find unoccupied sites if anaerobic conditions persist. Cation exchange may also be involved along with adsorption in the retention of ammonium ions in the subsoil; however, just as the adsorption capacity of the soil can be exceeded, there are only a finite number of exchange sites available in the subsoil. The adsorption of NH_4-N can range from 2mg/100g in sandy soil to 100mg/100g in fine grained soils with a

30% clay content (Jenssen and Siegrist, 1988). Vermiculite and similar clays are shown to strongly fix NH_4-N (Wilhelm *et al.*, 1994a). The fraction of the cation exchange capacity (CEC) of the subsoil that may be used to adsorb ammonium depends on the concentration of the other cations in the STE as these cations, especially divalent cations such as Ca^{2+} and Mg^{2+}, compete with ammonium for exchange sites. It is important to note that the ammonium adsorbed by the soil CEC is only temporarily immobilised because it can be readily oxidised to nitrate in the presence of oxygen (Lance, 1984, Jenssen and Siegrist, 1988). However, several experiments have demonstrated that approximately 50 to 85% of fixed NH_4-N may be unavailable or only slowly available to nitrifying micro-organisms (Nommik and Vahtras, 1982). It can be difficult to quantify NH_4-N retention as it can be masked by oxidation and other interactions (Wilhelm *et al.*, 1994b). Ammonium can also be incorporated into microbial cell tissue or, if the pH becomes alkaline, volatilisation may occur, i.e. ammonia (NH_3) gas released, although nitrogen removal by these mechanisms is only minor (Lance, 1984; Canter and Knox, 1985).

Under the aerobic conditions of the subsoil treatment system ammonium will undergo nitrification. Micro-organisms in the subsoil use O_2 as the electron acceptor in the oxidation of NH_4-N to NO_3-N. Wilhelm *et al.*, (1994a) outline both field and column experiments which indicate that almost complete nitrification occurs within 1m of the distribution pipe and within a few hours of exposure to O_2. In some well-aerated systems, complete nitrification of effluent occurs within 0.5m of the biomat below the percolation trench (Whelan and Barrow, 1984). Jenssen and Siegrist (1988) report that nitrification, which is normally very rapid and is pH (7-8.5 desirable) and temperature (8-10°C desirable) sensitive, can occur in the first 30 cm of the subsoil. In properly constructed systems complete nitrification can be achieved (Siegrist *et al.*, 2000); however, one to two months are required from system start-up to generate a full population of nitrifiers (Van Cuyk *et al.*, 2001). As nitrification is temperature dependent, it is reasonable to assume that complete nitrification might not occur year round due to seasonal effects. Research carried out by Harman *et al.* (1996) in a sandy subsoil receiving STE from a school showed nitrate concentrations increasing from less than 0.1 mg(NO_3-N)/l in the septic tank to 112 mg(NO_3-N)/l at the water table while ammonium concentrations decreased from 128 mg(NO_3-N)/l in the tank to less than 1 mg(NO_3-N)/l at the water table. In this case, because nitrate levels observed at the water table were similar to effluent ammonium levels, almost complete nitrification of ammonium appeared to occur with little ammonium being removed by other processes. Wilhelm *et al.,* (1994b) report on research they carried out in which the concentration of nitrate at the watertable approximated 70% of the original N in the STE. Since nitrate is a negatively charged ion it is not attracted to the negatively charged soil colloids and as such is readily leached to the groundwater. Nitrate can be removed from the subsoil by denitrification, as discussed in Section 4.2.3. The occurrence of this process requires the presence of an anaerobic zone and an adequate biodegradable carbon source (Jenssen and Siegrist, 1988). However, due to aerobic oxidation the wastewater is usually devoid of sufficient organic C to promote denitrification in properly functioning septic tank treatment systems (Wilhelm *et al*., 1994a and 1994b).

(ii) Phosphorous

Phosphorous is generally the limiting nutrient for algal growth in many aquatic ecosystems. The main source of phosphorous in STE is household detergents. Elevated phosphate levels can stimulate plant and algae growth which can eventually result in the eutrophication of surface water-bodies (Schindler, 1977). As most phosphorous is retained in the subsoil by adsorption and precipitation it is not generally a problem to groundwater in Ireland and is not likely to be in the future (Kilroy *et al*., 1998). This is because the maximum admissible concentration for drinking water (2.2mg/l P) is rarely approached, except in areas of gross contamination. However, as P concentrations of only 20µg/l may trigger eutrophication groundwater may act as a pathway for P to surface water targets. As a result septic tanks have been implicated as a source of phosphate in some lakes where dwellings are located nearby although the evidence suggests that a significant amount of STE phosphorous is immobilised in the vadose zone sediments (Zanini *et al*., 1998).

Phosphorous is mainly present in the STE as orthophosphate, dehydrated orthophosphate and organic phosphorous (Siegrist *et al*., 2000). Bouma (1979) reported on studies that found that more than 85% of total phosphorous in the STE was in the soluble orthophosphate form. The organic phosphorous in the effluent can be subject to mineralisation, in a way similar to organic nitrogen, resulting in the production of soluble inorganic phosphate (Brady and Weil, 1996). Phosphate is generally immobilised within a few metres of the percolation trench and significant movement of phosphate is rare (Reneau *et al*., 1989). As a result contamination of groundwater with phosphorous is seldom investigated or reported. However, studies have identified phosphorous leaching in areas with sandy soil, shallow water tables and in phosphorous saturated soils (Breeuwsma *et al*., 1995). Phosphorous attenuation in the subsurface is controlled by soil adsorption and mineral precipitation reactions which can be considered in two general categories: initial adsorption reactions and much slower precipitation reactions that regenerate additional adsorptive surfaces occurring both from phosphate in solution and from phosphate previously sorbed (Lance, 1984). The types of reaction that fix phosphorous in relatively unavailable forms differ from soil to soil and are closely related to soil pH. In acid soils these reactions involve mostly Al, Fe or Mn, either as dissolved ions, as oxides, or as hydrous oxides. In alkaline and calcareous soils, the reactions primarily involve precipitation as various calcium phosphate minerals or adsorption to the iron impurities on the surfaces of carbonates and clays. At moderate pH values, adsorption on the edges of kaolinite or on the iron oxide coating on kaolinite clays plays an important role (Brady and Weil, 1996). Arias *et al*., (2001) report that at pH levels greater than 6 the reactions are a combination of physical adsorption to Fe and Al oxides and precipitation as sparingly soluble calcium phosphates. At lower pH levels they suggest that precipitation as Fe and Al phosphates becomes increasingly important. It is clear, therefore, that the capacity of a subsoil to fix phosphate is dependent on the presence of Fe, Al, Mn and Ca. Robertson (2001) found that the organic matter in STE causes iron in soils to become soluble. As a result the phosphorous is precipitated out of solution forming stable coating on the soil particles.

Soils vary greatly with respect to P-sorption capacity with a quartz sand receiving STE probably becoming saturated after a few months while the sorption capacity of a weathered sand or fine grained soil may hold for a period of 10 years or more (Jenssen, 2001). With respect to sands the P-removal efficiency is often high initially but then decreases after some time as the P-sorption capacity of the sand is used up (Arias *et al*., 2001). In calcareous terrain, acidity generated by the oxidation reactions stimulates carbonate mineral dissolution resulting in near-neutral pH and Ca enrichment in the plume (Robertson, 2003). Analysis by Wilhelm *et al*., (1994b) on STE in a calcareous subsoil recorded the sorption of PO_4-P followed by its precipitation with Ca as an amorphous mineral that later crystallised. As a result most of the P in the STE was removed in the first three months after leaving the septic tank. In general phosphate retention is greater in acidic settings than in neutral or basic settings so the possible lowering of soil pH is not expected to increase phosphate mobility (Wilhelm *et al.*, 1994a). They therefore concluded that calcareous sand seemed to have a good ability to limit PO_4-P mobility. This corroborated previous findings by Whelan (1998). Phosphate attenuation on noncalcareous sites monitored by Robertson (2003) appeared to be as direct result of the development of acidic conditions and elevated Al concentrations which subsequently caused the precipitation of Al-P minerals. The potential for P sorption of a porous medium is dependent on the mineral composition, which determines the metal types and content, and the degree of weathering of the particle surfaces, which renders the metals in an oxide ot hydrous state where they are able to react with phosphorous compounds (Jenssen, 2001). It is also dependent on the grain size of the medium. Analysis by Zhu *et al.*, (2003) on the phosphorous sorption characteristics of a light-weight aggregate of expanded clay found that the finer the grain size the higher the P sorption. They also found that the influence of temperature on P sorption was greater for the coarser grained aggregate. While capacity for P sorption is finite, a given soil, however, does not have a fixed capacity to remove phosphorous as these reactions are dependent upon many factors such as the phosphorous concentration in the soil solution, soil pH, temperature, time and the concentration and type of ions in the STE (Lance, 1984).

Mineral precipitation reactions may be affected by flow velocity but are less likely to be influenced by loading history (Harman *et al*., 1996). Lance (1984) found that by decreasing the hydraulic loading rate from 0.45 m^3/m^2d to 0.1 m^3/m^2d, phosphate removal in sand columns was increased from about 10% to about 80%. Harman *et al.* (1996) report on research carried out at three test sites in sandy subsoil, two in operation for six years and the third for twelve years. Phosphate attenuation in the unsaturated zone of both six year old sites was nearly complete with less than 0.1 mg/l being recorded at the water table. At the older site, however, removal of phosphate was found to be less than 50%. The research, carried out on a system in operation for 44 years, lead them to the conclusion that the attenuation of phosphate in the subsoil was controlled by mineral precipitation rather than sorption because:

- a comparison of the maximum sorption capacity of the soil and the phosphate sorbed suggested that the sorption capacity of the soil had been reached,
- the phosphate concentration recorded at the water table over a three year period was relatively constant and significantly lower than the STE phosphate concentration suggesting steady-state conditions. If adsorption was the main attenuation process phosphate concentration at the water table would increase as sorption sites were used up,
- sorption capacity did not change significantly within or below the phosphorous peak in the subsoil suggesting that the high concentration of phosphorous was more likely a result of a zone of phosphorous precipitation.

Precipitation may provide unlimited capacity to fix phosphate, provided that the solution and soil chemistry is conducive to the reaction (Jones and Lee, 1979).

4.4.3 Pathogens

With an estimated 200,000 wells and springs in use in Ireland (Wright, 1999) the survival of pathogens under unsaturated and saturated conditions is a major concern in the protection of groundwater resources. Pathogens commonly found in STE include enteric bacteria, at sustained concentrations, and viruses and protozoa, at highly variable and episodically released levels (Cliver, 2000). The most important pathogenic bacteria and viruses that might be transported to groundwater include *Salmonella* sp., *Shigella* sp., *Escherichia coli* and Vibrio sp., and hepatitis virus, Norwalk virus, echovirus and coxsackievirus (Abu-Ashour *et al.*, 1994). Viruses and protozoa are not continuously present at high densities, but rather are shed during disease events and thus the concentration in the STE can vary from zero to values of the order of 10^6 organisms/100ml or more (Siegrist *et al.*, 2000). While some reference is made to viruses and protozoa in this literature review they were outside the scope of the project as the main concern of the project was the persistence of enteric bacteria with subsoil depth.

Bacteria are single-cell prokaryotic organisms containing a colloidal suspension of proteins, carbohydrates and a cytoplasm. The cytoplasm contains ribonucleic acid (RNA) and deoxyribonucleic acid (DNA) which control protein synthesis and reproduction respectively. Reproduction is usually by binary fission, although some species reproduce sexually or by budding (Metcalf and Eddy, 2003). Pathogens commonly found in domestic effluent include some enteric bacteria which are gram-negative, facultative anaerobes with a wide variety of metabolic capabilities, including anaerobic respiration using NO_3^- as the electron donor, that inhabit the intestinal tracts of humans and animals (Campbell, 1993). Their presence in groundwater, therefore, is not conclusive of anthropic contamination. Where present, however, bacterial pathogenic organisms of human origin typically cause diseases of the gastrointestinal tract, such as typhoid, gastroenteritis and cholera (Table 4.2). As it is not feasible to test samples for the presence of all pathogenic bacteria, samples are tested for the presence of indicator species under the premise that their presence is suggestive of the presence of human pathogens.

The fate of pathogens in subsoils is primarily governed by their transport and persistence. Bacteria and virus survival in the subsurface environment is in the order of a week to several months (Wilhelm *et al*., 1994a). Reneau *et al*. (1989) report that pathogenic bacteria and viruses in soil are generally reduced to minimal numbers within two to three months although enteric bacteria have been known to survive up to five years under some unusual environmental conditions. A few studies (Gerba *et al.*, 1984) have been carried out on the comparative survival between different pathogenic microorganisms in groundwater as shown in Table 4.3. In subsoil treatment systems inactivation/die-off, filtration and adsorption can be extremely effective attenuation processes in the removal of pathogens from percolating STE. The development of a biomat improves the retention of pathogens (Hagedorn, 1984 and Sélas *et al*., 2002).

Bacteria	**Disease**	**Concentration in raw wastewater (MPN/100ml)**	**Infective dose**	**Survival time in soil at 20-30°C (days)**
Campylobacter spp.	Gastroenteritis	10^0-10^3		
Escherichia coli (enteropathogenic)	Gastroenteritis	10^6-10^8	1-10^{10}	<120 but usually <50
Legionella pneumophila	Legionnaires' disease			
Leptospira spp.	Leptospirosis			
Salmonella spp.				
S. typhimurium	Salmonellosis	$10^2 10^4$	10-10^8	<120 but usually <50
S. typhi	Typhoid fever	$10^2 10^4$	10-10^8	<120 but usually <50
Shigella spp.	Shigellosis	10^0-10^3	10-20	<120 but usually <50
Vibrio cholerae	Cholera	10^0-10^3		
Yersinia enterocolitica	Yersiniosis			

Table 4.2 Bacterial pathogenic organisms present in domestic wastewater (adapted from Gray (1999), Kiely (1997), Mason (2002), Metcalf and Eddy (2003) and Viessman and Hammer (1998)).

Microorganism	**Die off rate (day^{-1})**
Poliovirus 1	0.34
Coxsackievirus	0.19
Rotavirus SA-11	0.36
Coliphage T7	0.15
Esherichia coli	0.28
Salmonella	0.18

Table 4.3 Die off rate constants for visuses and bacteria in groundwater. (from, Gerba *et al.,* 1984).

Hagedorn (1984) outlines research that found that in a properly functioning percolation area the indicator bacteria were almost completely removed after 38cm. He also outlined other research that found that approximately 30-90cm of soil beneath the percolation areas monitored was adequate for complete bacterial removal provided that the subsoil had both a layer permeable to effluent flow and another region adequately restrictive to form a clogged zone. Reneau *et al.* (1989) report that while most treatment occurs in less than 3m of subsoil under unsaturated flow conditions bacteria can be adequately removed within 0.9 to 2.0m of effluent flow through soils.

The main factors affecting the growth and survival of bacteria and the survival of viruses (which require a living host for reproduction) in the subsoil are summarised in Table 4.4 (a) and (b). The type of pathogen and its physical state, type and characteristics of the subsurface soil, wastewater quality, temperature and hydrogeological conditions influence the fate of pathogens in the subsurface environment (Bitton and Harvey, 1992 and Scandura and Sobsey, 1997).

Factor	Comments
Moisture Content	Longer survival in moist soils.
Moisture holding capacity	Survival time is less in sandy soils with lower water-holding capacity.
Temperature	Longer survival at lower temperatures.
pH	Shorter survival time in acidic soils (pH 3-5) than in alkaline soils.
Organic matter	Increased survival and possible growth when sufficient amount of organic matter is present.
Biological interactions	Increased survival in sterile soil where there is a lack of competition from indigenous microflora. The presence of antibiotics also have a detrimental affect on their survival.

Table 4.4 (a) Factors affecting the growth and survival of bacteria (adapted from Abu-Ashour *et al*., 1994 and Gerba and Bitton, 1984).

Factor	Comments
Moisture Content	Increased virus reduction in drying soils.
Temperature	Viruses survive longer at lower temperatures.
pH	Most enteric viruses are stable over a pH range of 3-9; survival may be prolonged at near-neutral pH values.
Cations	Certain cations have a thermal stabilising effect on viruses. May also indirectly influence virus survival by increasing their adsorption to soil (viruses appear to survive better in sorbed state).
Biological factors	No clear trend with regard to the effect of soil microflora on viruses.

Table 4.4 (b) Factors affecting the survival of viruses in the subsoil (adapted from Abu-Ashour *et al*., 1994 and Gerba and Bitton, 1984).

For example, the average penetration rate of a motile strain of *E. coli* through sand has been shown to be four times faster than that of a nonmotile strain of the bacterium (Reynolds *et al*., (1989). Persistence of enteric bacteria in the subsoil is generally highest at low temperature, high soil moisture content and abundant organic matter that may allow the growth of certain bacteria (Bitton and Harvey, 1992), although the USEPA 1993 dispute this latter point. The USEPA (1993) cite the availability of nutrients and biological factors as most important for the survival of pathogenic bacteria with elimination faster at higher temperatures (37°C), at pH values of around 7, at low oxygen concentrations and at high levels of dissolved organic carbon, conditions that promote naturally occurring antagonistic bacteria. Soil temperature and moisture are the primary factors affecting virus survival in soils receiving STE with viruses surviving best in moist soils under low temperatures (Bitton and Harvey, 1992).

Bacterial transport through percolation areas is controlled by soil porosity and the degree of saturation with water (Bitton and Harvey, 1992). They may be transported over much greater distances under saturated conditions than under unsaturated conditions due macropore flow and to higher pore water velocities (Hagedorn *et al*., 1981). As the moisture content of the subsoil decreases the maximum water-conducting pore size decreases. As a result organisms larger than 8μm are unlikely to move through water-filled pores in unsaturated subsoils, and viruses, therefore, are most likely to be transported (Powelson and Gerba, 1995). Removal of pathogens from the percolating STE is mainly due to filtration for bacteria, although adsorption does also occur, and adsorption for viruses (Gerba and Bitton, 1984). As most bacteria and soil materials are negatively charged the electrostatic repulsion present between them must be overcome for adsorption to occur. The sorption of cations present in the percolating effluent onto subsoil particles can neutralise the negative charge and bacteria can be retained by van der Waals forces (Peavy, 1978 and Powelson and Gerba, 1995). Adsorption appears to be significant in soils having pore openings several times larger than typical sizes of bacteria (Canter and Knox, 1985). In flow systems bacterial transport through porous media has been largely described as a function of pore entrance size (Reynolds *et al*., 1989). While cell size and ionic strength of the effluent were shown to be important variables controlling bacterial transport through porous media grain size was seen to be the most important factor (Fontes *et al*., 1991). Physical straining occurs when bacteria are larger than the pore opening in the soil. Bacterial removal by straining is inversely proportional to particle size of soils. (Hagedorn *et al*., 1981 and Bitton and Harvey, 1992). When these bacteria accumulate within the subsoil the pore space is reduced which has the effect of removing smaller bacteria from the percolating effluent. Research (Gerba and Bitton, 1984) in which E. coli suspended in distilled water was allowed to percolate through sand columns suggest that once the capacity for removal by straining has been satisfied, sedimentation is the only effective process.

Straining as a removal mechanism is less affective for viruses than for bacteria due to their small size, 18 to 25nm as opposed to 750 nm for bacteria (Reneau *et al*., 1989; Bitton and Harvey, 1992 and Powelson and Gerba, 1995). Nicosia *et al*. (2001) report that soil characteristics affect virus removal and in general, virus sorption capacities increase as clay content, cation exchange capacity and specific surface area increase, and as organic content decreases. Adsorption of viruses onto soil particles is also dependent on the pH of the system. At low pH values below 7.4, virus adsorption by soils is rapid and effective but higher pH values greatly decrease the effectiveness of virus adsorption because of increased ionisation of the carboxyl group of the virus protein and the increasing negative charge on the soil particles (Canter and Knox, 1985). While the actual mechanism of viral adsorption is unknown, two general theories exist, both based on the net electronegativity of the interacting particles:

(i) that a clay-cation-virus bridge operates to link the two negatively charged particles, or
(ii) that the fixation of multivalent cations onto the ionisable groups on the virus particle is accompanied by a reduction of the net charge of the particle.

(Gerba, 1975)

In research carried out by Nicosia *et al*., (2001) in fine sand using the bacteriophage PRD1 as a human enteric virus surrogate it was found that the sorption of PRD1 was reversible as the phage particles slowly desorbed over time. Desorption was also reported when solutions of lower ionic strength were flushed through soils (Reneau *et al*., 1989). Brown *et al*. (1979) described a 2-year study where STE was monitored for coliphages in three undisturbed soils. Regardless of application rates the coliphages were rarely detected in soil leachates and 120cm of any of the soils tested appeared to be sufficient to minimise any possibility of groundwater contamination by coliphages.

Protozoa are a topical issue in relation to the use and treatment of surface water for human consumption although little has been written on their persistence in the subsoil or the threat they present to contamination of groundwater resources. Thy are of lesser concern than bacteria and viruses however since they are relatively large and therefore are removed more efficiently by subsoil filtration (Reneau *et al*., 1989). Certain protozoa, excreted from mammals into the environment, form protective “oocysts” which allow them to survive for long periods (generally several months) in damp cool situations whilst waiting to be ingested by another host before continuing its life cycle (Gray, 1994). The two pathogenic protozoa oocysts most frequently found in surface drinking water sources are cryptosporidium (4-7 µm in diameter) and Giardia lambia (8-14 µm long and 7-10 µm wide). While the transport of these specific protozoa in subsoil systems has not been studied directly, the transport of an indigenous strain of flagellated protozoa in a sandy aquifer was studied by Harvey *et al.* (1995). A tenfold decrease in the numbers was observed over the first metre of flowpath and numbers of protozoa became undetectable 3.6 metres down gradient from the injection point. This suggests that transport of protozoa through the unsaturated subsoil would be inhibited.

4.5 Irish Experience of Groundwater Contamination by STE

Septic tank systems, together with farmyards, are considered to be the two main point sources of groundwater pollution in rural areas (Daly *et al.*, 1993; Daly, 1993; Dames and Moore, 2000). Groundwater is an important national asset that accounts for about 15% of total water volume supplied by public authorities and about 25% of all water supplies (Environmental Research Unit, 1993). The degree of microbial contamination of groundwater in Ireland is very high with, in many areas, at least 30% of private domestic and farm wells polluted by faecal bacteria and in some highly vulnerable areas more then 50% polluted. It is likely that there are areas in Ireland where more than 70% of private wells contain faecal bacteria at some time during their use (Daly, 2003). Groundwater contamination from septic tank treatment systems is mainly a problem where dwellings are not serviced by public sewerage and where water supplies are obtained from poorly constructed wells in vulnerable aquifers. In 1993 there were over 300,000 septic tank systems in Ireland serving a population of about 1.2 million people and discharging 80 million m^3 of effluent into the ground annually (Daly *et al*., 1993); this figure increased to approximately 400,000 dwellings in 1999 (DoELG *et al*., 2000). Of the estimated 200,000 wells and springs in use at present in Ireland those in unserviced areas are therefore often at risk of STE contamination because of their location.

E. coli, an enteric bacteria, when discovered in groundwater is an indicator of faecal contamination but unfortunately its presence is not specific to STE contamination as it is also present in the gut of warm blooded animals. The absence of farmyards in the locality, therefore, could be indicative of contamination from septic tanks. In the 1980s, Daly (1987) argued that septic tanks, rather than farmyards, were the more likely cause of groundwater contamination because:

- they are more numerous than farmyards,
- contaminated wells are more often located closer to septic tanks than to farmyards, and
- the use of soakage pits allows the STE to enter groundwater more readily than farmyard effluent.

Daly (1987) also reported that of samples taken from 146 groundwater sources in one local authority area 84 (58%) contained E. coli. Out of 39 high yielding wells and springs in the same county 29 (74%) were contaminated. In another county surveyed 22 of 41 group schemes contained *E. coli*. Thorn *et al*. (1986) examined the groundwater quality in south Co. Sligo and found that of the 42 sources examined, 28 (67%) were contaminated with faecal coliform and/or faecal streptococci. They concluded that septic tanks were the main source of this contamination. The presence of *Cryptospiridium* in Lough Owel, the water supply for Mullingar, in 2003 caused 26 people to become ill. As the lake is groundwater fed and as the vulnerability of the catchment is mostly ‘extreme’ and ‘high’ groundwater was considered to be a potentially significant pathway for *Cryptospiridium* to enter the lake from septic tanks and/or farmyards (Moran, 2003).

A study sponsored by the Department of Environment and Local Government examining the methods available for the reduction of nutrient inputs to Lough Leane from all sources included a section on the possible impacts of STE on water quality in the Lough Leane catchment (DoELG, 2000). It was found that the location of domestic wastewater treatment systems in areas of shallow overburden resulted in contamination of surface streams by STE. However, a survey of part of the catchment area revealed the most representative type of wastewater treatment system to be a one-chamber, blockwork septic tank greater than 20 years old with effluent discharge going to the soakaway area. It could be hoped, therefore, that the situation with respect to STE contamination will improve, both in the Lough Leane catchment and nationwide, with the implementation of the EPA guidelines (2000).

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APPENDIX A:

SCANDINAVIAN SAND FILTER EXPERIENCES

Green Innovation