Process, performance and pollution potential: a review of septic tank-soil absorption systems

Author

Beal, CD, Gardner, EA, Menzies, NW

Published

2005

Journal Title

Australian Journal of Soil Research

DOI

10.1071/SR05018

Rights statement

Downloaded from

Griffith Research Online

Process, performance and pollution potential: A review of septic tank – soil absorption systems

C. D. Beal[A], *E. A. Gardner*[B], *N. W. Menzies*[C],

[A] *School of Land & Food Sciences, University of Queensland; and Coastal CRC, Indooroopilly, Qld*
[B] *Qld Department of Natural Resources, Mines & Energy; and Coastal CRC, Indooroopilly, Qld*
[C] *School of Land & Food Sciences, University of Queensland, St Lucia, Qld 4073*

Abstract

On-site wastewater treatment and dispersal systems (OWTS) are used in non-sewered populated areas in Australia to treat and dispose of household wastewater. The most common OWTS in Australia is the septic tank – soil absorption system (SAS) which relies on the soil to treat and disperse effluent. The mechanisms governing purification and hydraulic performance of a SAS are complex and have been shown to be highly influenced by the biological zone (biomat) which develops on the soil surface within the trench or bed. Studies suggest that removal mechanisms in the biomat zone, primarily adsorption and filtering, are important processes in the overall purification abilities of a SAS. There is growing concern that poorly functioning OWTS are impacting upon the environment. Though, to date, only a few investigations have been able to demonstrate pollution of waterways by on-site systems.

In this paper we review some key hydrological and biogeochemical mechanisms in SAS, and the processes leading to hydraulic failure. The nutrient and pathogen removal efficiencies in soil absorption systems are also reviewed, and a critical discussion of the evidence of failure and environmental and public health impacts arising from SAS operation is presented. Future research areas identified from the review include the interactions between hydraulic and treatment mechanisms, and the biomat and sub-biomat zone gas composition and its role in effluent treatment.

Additional keywords: biomat, LTAR, modelling, nutrients, pathogens, on-site systems, wastewater

Introduction

On-site wastewater treatment and dispersal systems (OWTS) are used in non-sewered populated areas in 20% of Australian households to treat and dispose of household wastewater. The most common OWTS in Australia is the septic tank – soil absorption system (SAS). Wastewater undergoes primary treatment in the septic tank via sedimentation and anaerobic digestion. Secondary treatment of the septic tank effluent occurs within the trenches or beds and surrounding soil (Figure 1). A SAS utilises the soil as the medium to treat and disperse effluent originating from the septic tank.

Fig 1.

Unlike alternative OWTS, the soil absorption system relies solely on the natural biogeochemical processes that occur in soil to assimilate various effluent pollutants. The advantage that this method has over other on-site systems is its relative simplicity, low cost and, if designed and constructed properly, treatment capability. The limitations of SAS relate to the inherent variability and heterogeneity of soil and soil biogeochemical processes. With aerated wastewater treatment systems (AWTS) the treatment processes (aeration, clarification, dispersion) can be directly controlled or regulated after system installation. Conversely, apart from septic tank inspections, the operation of a SAS is largely left to natural processes below the soil's surface. Consequently, appropriate design, construction, and maintenance, based on prior knowledge of the site and soil conditions, are crucial for the sustainable and successful operation of these systems. So it follows that an understanding of the processes that drive a SAS is essential for design and installation, and management in general. Nevertheless, success will not be achieved for every single system that has been designed, constructed and maintained properly; the factors required to achieve that success are too numerous and too varied. However, substantial improvement to the current sustainability of SAS is more likely to occur as a result of informed and well-considered management.

The provenance of the OWTS regulatory criteria has placed an emphasis on reducing health risks associated with OWTS. The progression of the on-site treatment industry, in terms of the use of new technologies and the introduction of performance-based criteria, has been held back by its "public health heritage" (Otis 1991). This has resulted in often conservative and prescriptive criteria set by the responsible regulatory body. In the United States, SAS are the predominant on-site system with approximately 20 million systems in use. The USEPA response to Congress on the long-term viability of on-site systems, neatly summarises their point-of-view: "*Adequately managed decentralized wastewater treatment systems are a cost-effective and long-term option for meeting public health and water quality goals, particularly in less densely populated areas*." (USEPA 1997). Despite this, SAS continue to have a mixed reputation in the industry and with the general public, and can be likened to "Dr Jekyll and Mr Hyde" in terms of its unpredictable and variable treatment efficiency and failure rates.

Some SAS can operate for many years with no discernable failure (either surcharge or groundwater impacts), while others can fail within weeks of installation, potentially resulting in adverse environmental and public health effects (and economic impacts to the owner). Yet, the evidence to support the argument that SAS cause widespread and serious pollution to surface and more commonly groundwaters, is by no means conclusive.

This paper begins with a review of some key hydrological and biogeochemical mechanisms in SAS including the processes leading to hydraulic failure. Following this, the treatment capabilities of the SAS are discussed drawing on previous investigations. A critical review of the contrasting findings of the contamination potential of septic trench effluent is presented. The paper concludes with a discussion of the key areas of further research in the sustainable management of soil-based wastewater treatment systems.

Hydraulic processes in soil absorption systems

The mechanisms governing purification and hydraulic performance of a SAS are complex and have been shown to be highly influenced by the biological zone (biomat) or 'clogging layer' which develops on the soil surface within the trench (Bouma 1975; Siegrist and Boyle 1987). The biomat generally has a low hydraulic conductivity. Bouma (1975) calculated values of approximately 0.6 mm/day for clay soils and 2 mm/day for sandy soils.

The hydraulic conductivity of the biomat reduces over time with a concomitant increase in biomat resistance. With this increase in resistance, flow through the biomat is reduced to an extent that effluent can build up above the biomat while the underlying soil remains unsaturated (Kristiansen 1981a). It is the unsaturated flow characteristics ($K(\Psi)$) of the soil and the resistance properties of the biomat that govern the long-term flow rates though the biomat and sub-biomat zone (Huntzinger Beach and McCray 2003).

A crust-capped soil, as is the case in a mature SAS, has been shown to behave as a "self-adjusting" system, where a steady-state infiltration rate and soil moisture profile develops over time (Hillel 1980). This steady-state is reached when sub-biomat soil matric potentials (Ψ) create a gradient across the biomat, and the unsaturated hydraulic conductivity below the biomat allows a state of equal flux through both the biomat and unsaturated zones (Hillel 1980). This condition can be expressed as:

$$Q_b = Q_u = K_b \left(\frac{dH}{dZ} \right)_b = K_u \left(\frac{dH}{dZ} \right)_u \qquad (1)$$

where Q_b is the steady-state flux through the biomat (m/day), Q_u is the steady-state flow through the unsaturated zone below the biomat (m/d), K_b is the biomat hydraulic conductivity (m/d), $(dH/dZ)_b$ the biomat hydraulic gradient, K_u the unsaturated hydraulic conductivity (m/d) and $(dH/dZ)_u$ the hydraulic gradient of the unsaturated sub-biomat zone.

Low flow rates in absorption systems are not exclusively a result of the interaction between unsaturated soil hydraulic conductivity and biomat resistance. Dispersion or swelling in sodic soils, resulting from low electrical conductivity and high sodium absorption ratio effluent applications, can substantially decrease soil hydraulic conductivity (Halliwell *et al.* 2001; Patterson 2001). In this case the steady-state flow would be less easily predicted as a factor solely of biomat zone hydraulic resistance.

The resultant *Q* in equation 1 is the long-term steady-state flux (m/d) at which, theoretically, a SAS can continue to accept effluent without hydraulic failure occurring. This flux value is also known as the long-term acceptance rate (LTAR), with units of mm/day or L/m^2/day. The LTAR is a key parameter used in the Australian and New Zealand Standard AS1547:2000 to calculate the soil surface area and length of trench required to ensure SAS are not overloaded. The LTAR curve was originally formulated from the work of Brouwer and Bujega (1983) who carried out a land capability assessment for septic systems in Victoria. The most significant aspect of the report was the development of a curve which plotted the relationship of soil hydraulic conductivity with LTAR of effluent into soil (Brouwer and Bugeja, 1983). This LTAR curve became the basis for sizing recommendations, e.g. trench length/equivalent person, in the Australian Standard publication AS1547:1994 (Standards Australia 1994) and Australian and New Zealand Standard - AS/NZS 1547: 2000 (Standards Australia and Standards New Zealand 2000). Unfortunately, there were a number of limitations with the study, including soil solution sampling difficulties, limited replications, questionable suitability of sites, and unusually dry weather. As a result of these difficulties, only one pair of septic absorption trenches was investigated (Brouwer and Bugeja 1983). The authors acknowledged these issues and emphasised that a more representative dataset would be required to substantiate the conclusions and recommendations made in their report.

Biomat zone genesis and development

Biomat zone genesis and development is a dynamic process which can be influenced by physical, biological and chemical processes. The dominance of any one process can be difficult to isolate as they often occur concurrently (Baveye *et al.* 1998). Biomat genesis is generally characterised by an initial physical clogging of the pores in the infiltrative surface of the native soil (Otis 1984; Siegrist *et al.* 1991). Anaerobic biological activity has commonly been identified as the main subsequent clogging process (Siegrist and Boyle 1987; Tyler and

Converse 1994). Clogging usually occurs within the first few months of full operation of a SAS (Kristiansen 1981a).

The main factors influencing the initial clogging and biomat zone genesis are the wastewater characteristics (suspended solids (SS), biochemical oxygen demand (BOD), and loading rate. Other factors include poor construction practices (smearing of the trench bottom), electrical conductivity, sodium absorption ratio and underlying soil structural and textural properties (Otis 1984). Stimulated by a favourable environment (eg. anaerobic conditions, high humidity and moisture) in the initially clogged soil, biological activity then plays a significant role in long-term growth of the biomat zone (Bouma 1979).

Based on marked reductions in infiltration rate over time, three distinct phases of soil clogging leading to reduction in infiltration rate have commonly been observed (Allison 1947; Jones and Lee 1979; Otis 1984; Siegrist and Boyle 1987). A general pattern of infiltration rate decline adapted from Otis (1984) is presented in Figure 2.

Fig. 2

Phase I of biomat zone development is characterised by a sharp reduction in infiltration rates following effluent application. The initial clogging mechanisms can be attributed to several factors. These include physical straining of organic materials and suspended particulates contained in the wastewater and entrapment of gases (Bouma 1979; Baveye *et al.* 1998; Siegrist *et al.* 2000). These processes act to block soil pores in the infiltrative surface, thus impeding pore connectivity and flow rate through to the unsaturated zone beneath the absorption system.

Following a sharp reduction in the infiltrative rate during phase I, a more gradual decrease in infiltration rate has been observed (Jones and Lee 1979; Otis 1984; Siegrist and Boyle 1987). Biological activity stimulated by changes to soil conditions created during phase I (eg. low aeration status, high humidity and moisture levels) is believed to be the predominant mechanism of clogging in this second phase (Frankenberger *et al.* 1979; Siegrist and Boyle 1987; Ronner and Wong 1998; van Cuyk *et al.* 2001b).

The third phase of biomat zone development is represented by low infiltration rates where some researchers have observed an equilibrium state (i.e. LTAR) to evolve. During this phase accumulation of clogging materials at the biomat's surface is balanced out by die-off and sloughing of clogging material at the bottom (McGauhney and Winneberger 1964; Bouma *et al.* 1974; Siegrist 1987; Jenssen and Krogstad 1988; Mamedov and Levy 2001). Once established, factors influencing *long-term* growth of the biomat zone include hydraulic ($L/m^2/day$) and organic loading rates (i. e. cumulative BOD and suspended solid), dosing regime (i.e pressure versus gravity) (Rice 1974; Siegrist and Boyle 1987; Stevik *et al.* 1999), aeration status of the infiltrative surface (De Vries 1972; Kristiansen 1981a; Siegrist and Boyle 1987) and the underlying soil biogeochemical properties (Bouma 1975; Kropf *et al.* 1977). Additionally, the temperature of the wastewater being applied to the soil is an influential factor on biomat development (Kristiansen 1981a; Ronner and Wong 1998), with cooler temperatures generally promoting greater biomat development (De Vries 1972; Ronner and Wong 1998).

Seasonal or infrequent use of SAS has also been shown to influence biomat development with only partial growth, or even the absence of a biomat, occurring from infrequent use (Postma *et al.* 1992).

Predicting Long-term Acceptance Rates of SAS

The hydraulic effects of the biomat zone on LTAR (Q) can be predicted if the resistance of the biomat (R_b) and the unsaturated hydraulic conductivity characteristics, K(Ψ) of the underlying soil are known. Bouma (1975) showed that the hydraulic conductivity of the biomat (K_b) is a function of both R_b and the biomat zone thickness (Z_b).

Biomat resistance is the product of the inverse of K_b and the effective thickness of the biomat (Z_b). Taking $K_b(dH/dZ)_b$ from Equation 1 and assuming a steady infiltrating soil profile where the hydraulic gradient approximates unity, we can write:

$$Q_u = K(\psi) = K_b \left(\frac{dH}{dZ} \right)_b$$

Fig. 3.

$$= K_b \left(\frac{H_o + \psi + Z_b}{Z_b} \right)$$

By rearranging this, the hydraulic resistance (R_b) of the biomat can be determined:

$$\frac{K(\psi)}{H_O + \psi + Z_b} = \frac{K_b}{Z_b} \equiv \frac{1}{R_b} \qquad (2)$$

where K(Ψ) is the unsaturated hydraulic conductivity of the sub-biomat zone as a function of soil moisture potential, and H_o is the positive hydraulic head on top of the biomat. A definition diagram of Equations 1 and 2 is presented in Figure 3.

Bouma (1975) calculated R_b from measured soil potentials below trenches and reported ranges from 5-7 days for sands, 150 days for silt loams and 45-65 days for clays and silty clays. The variation in values was attributed to differences in porosity, structural instability and biological activity between soils. The wetted perimeter or lower boundary of the saturated biomat is likely to be less abrupt in sandier soils compared with the finer-textured soils. This may have affected the tensiometer readings and hence the calculated R_b values reported in Bouma (1975), particularly if the position of the tensiometers in the soil profile was the same for each soil type.

Beal *et al*. (2004a) used Equation 2 to predict the effect of increasing *Rb* on flow rates for various Australian soils. Measured moisture retention characteristics (K-Ψ) taken from the literature (Forrest *et al.* 1985) were used to predict steady-state fluxes for various biomat resistances and soil textures. The predicted effect of R_b on LTAR is illustrated in Figure 4.

Fig. 4.

The results from Beal *et al* (2004a) were similar to those of other studies (e.g. Magdoff and Bouma 1974; Huntzinger Beach and McCray 2003) in that a 2 to 3 order of magnitude variation in saturated hydraulic conductivity between soils collapsed to a one order of magnitude variation in LTAR. Sidewall exfiltration of effluent above the ponded biomat was not included in the prediction, thus the LTAR were likely to be conservative.

The area of sidewall that has no impedance from a biomat zone can be referred to as the exfiltration zone and has been shown to be an important pathway for effluent under episodic peak loadings (Beal *et al.* 2004b). There are a limited number of studies modelling unsaturated flow in SAS (e.g Janni *et al.* 1980; Hansen and Mansell 1986; Huntzinger Beach and McCray 2003). Even less widely reported is the specific partitioning of biomat zone and non-biomat zone flow in SAS. A study by Brouwer *et al.* (1979) found flow through the sidewall to be greater than bottom flow in some texture-contrast soils (e.g Kurosols) in Victoria. This conclusion was drawn from field measurement of matric potentials below and adjacent to trenches and the ponded height of effluent in the trenches. The infiltration rate through the sidewalls was calculated at 35mm/day, however it is not clear if sidewall flow was through a biomat zone. Huntzinger Beach and McCray (2003) used HYDRUS-2D to predict unsaturated flow within SAS, and described a strong relationship between the biomat zone hydraulic properties, and the steady-state (long-term) infiltration rates within the unsaturated zone. The model assumed that all flow occurred through either the trench bottom or trench sidewall biomat layer, thus precluding the opportunity to predict flow dynamics for the remainder of the trench sidewall.

Predicting biomat zone development

Although factors influencing biomat development are generally well-known, predicting biomat development itself is difficult. Predicting biomat growth, and thus infiltration rate decline, would allow greater knowledge of the hydraulic capacity and likely treatment efficacy of SAS in various soil types. A clear relationship has been identified between organic loading rate into the trench (i.e. BOD and SS) and the rate and extent of biomat development (Laak 1986; Siegrist and Boyle 1987). Drawing from this, the concept of pretreating septic tank effluent prior to dispersal into trenches has been investigated, with researchers generally agreeing that this improves the longevity of SAS by inhibiting excessive biomat growth (Laak 1970; Siegrist and Boyle 1987; Converse *et al.* 1998; Harrison *et al.* 2000; Potts *et al.* 2004). Some models have been proposed to quantify biomat development as a function of organic loading rates (Laak 1986; Siegrist 1987). These models have provided a useful basis for predicting biomat development and hence LTAR of SAS. However, the applicability of these models is limited to the specific set of experimental design conditions and wastewater quality associated with the particular study. Consequently, they are unlikely to be reproducible for general estimations of biomat development. (Laak 1986) describes a cube root relationship between infiltration rate, trench (bottom) area and organic loading rate by:

$$A_A = A_s\left[\left(BOD_5 + TSS\right)/250\right]^{1/3} \tag{3}$$

where A_A is the adjusted area required (m^2), A_S is the area required for a standard septic system (m^2), BOD_5 is the five day biochemical oxygen demand (mg/L), and TSS the total suspended solids (mg/L). Soil absorption trench design may not be applicable using this relationship as Equation 3 was derived from soil column experiments where highly pre-treated effluent (i.e. low BOD and SS concentrations) was used. Additionally, only permeable soils were used in the soil column experiments.

Siegrist and Boyle (1987), using cumulative organic loading rates, also developed a logistic model formulated from a basic sigmoidal curve function of infiltration rate over time:

$$IR_t = 241 * \frac{\{\exp[2.63 - 5.70(tBOD) + 41.08(TSS) - 0.048(tBOD * TSS)]\}}{\{1 + \exp[2.63 - 5.70(tBOD) + 41.08(TSS) - 0.048(tBOD * TSS)]\}} \qquad (4)$$

where IR_t is the infiltration rate at time, t (cm/day), tBOD is the cumulative density loadings (kg/m^2) of total BOD (carbonaceous BOD and nitrogenous BOD), and TSS is total suspended solids (kg/m^2). This model suggests that, ultimately, the infiltration rate would approach zero with continual effluent loading (containing BOD and SS). Although Equation 4 considers cumulative density loadings of organic and inorganic fractions in septic tank effluent, it does not account for the dynamically bioactive nature of the biomat zone.

Weintraub *et al*. (2002) developed a biozone model that considered respiration, mortality and growth of bacteria in the biomat zone as a function of organic matter. The strength of this model is the incorporation of soil physical parameters such as water content and porosity, along with the biological components. However, the link between soil matric potential and unsaturated hydraulic conductivity below the biomat is not evident. Also absent is the key parameter of biomat resistance (thickness / hydraulic conductivity of the biomat). The absence of K(Ψ) as an input parameter weakens the model's predictions of biozone resistance effects.

Overview of treatment performance of SAS

The quality of septic tank effluent is highly variable (Table 1). The least variable of the key effluent pollutants is faecal coliforms which have been consistently reported at concentrations between 10^5 and 10^7 colony forming units (cfu)/mL (Brandes 1978; van Cuyk *et al.* 2001a; USEPA 2002). Virus concentrations in septic tank effluent are less widely reported but could be expected to be in the range 10^5 to 10^6 counts/100mL F-RNA bacteriophages (viral indicators) (Pang *et al.* 2003).

Table 1. Septic tank effluent quality for key pollutants

Studies suggest that nutrient and pathogen removal mechanisms in the biomat zone, primarily adsorption and filtering, are important processes in the overall purification abilities of a SAS. Table 2 provides a summary of some field and laboratory work on the removal efficiencies by soil with an emphasis on the biomat zone and the first 90 - 200 cm of unsaturated media. This list is by no means exhaustive but highlights some key work that has been done in this area.

The hydraulic and purification processes that occur when effluent passes through the biomat and underlying unsaturated zone are closely linked. Within the unsaturated zone, purification processes include sorption, chemical reactions, biotransformation, pathogen die-off and predation, and plant uptake (van Cuyk *et al.* 2001b). The biomat zone itself is an important component of the purification processes in SAS (Magdoff *et al.* 1974b; Siegrist 1987). The biomat zone can provide optimal conditions for treatment processes described above due to the increased humidity moisture levels and micro-organism populations present (Siegrist *et al.* 1991; Baveye *et al.* 1998).

Table 2

Role of unsaturated zone in effluent treatment

Data collated from Table 2 indicates the importance of the unsaturated depth of media in reducing contaminants in effluent, particularly pathogens. High removal rates of effluent pollutants (eg. pathogens, nutrients, organics and SS) in SAS are correlated with the presence of a well developed biomat zone (Magdoff and Bouma 1974; Kristiansen 1981b,c; Van Cuyk *et al.* 2001). Soil chemical characteristics, such as cation exchange capacity and organic matter content, are also important factors influencing the degree of effluent treatment in soil (McCardell and Davison 2003; Al-Shiek Khalil *et al.* 2004; Dawes and Goonetilleke 2004). However, the efficiency of sand media in effluent treatment, particularly the reduction of BOD, SS and pathogens, is also well documented (Crites and Tchobanoglous 1998; van Cuyk *et al.* 2001b; Davison *et al.* 2002), despite its high hydraulic conductivity and the relatively low physico-chemical activity of sand.

The treatment process in a SAS can be likened to the processes which occur in a single-pass sand filter. In a sand filter, effluent is applied intermittently at the top of the sand bed and percolates slowly and evenly throughout the bed. The removal of effluent contaminants occurs mainly in the upper few centimetres of the bed where a biologically active layer is formed (Adin 1998). In a properly functioning SAS, a biomat zone develops at the surface of the soil (bottom of trench) and effluent passes through this biomat and into the unsaturated zone. The presence of a mature biomat will reduce the flow to an extent where unsaturated conditions exist below the biomat, even when there is ponding above the biomat (Fig 5).

Fig. 5.

Unsaturated soil conditions create an environment much like a sand filter system, where the unsaturated zone promotes aerobic degradation of pathogens, prolonged retention time of effluent, and maximum contact with the soil media. Despite this analogy, studies suggest treatment efficiencies in SAS are generally inferior to those reported in sand-based treatment systems (Harrison *et al.* 2000; USEPA 2002). Much of the data presented in Table 2 has been generated from experiments or field work on sands or sandy soils. This is a clear limitation in the literature as many of the reported failing and / or poorly operating SAS occur in clay soils, rather than in sandy ones.

A key difference between an intermittently-dosed sand filter system and a conventional SAS is the inability of a gravity-fed SAS to evenly distribute effluent across the full length of trench bottom. Intermittent pressure dosing of effluent to soil absorption systems has been investigated as a method for improving the aeration status and therefore promoting longevity of a system (McGauhney and Winneberger 1964; Bouma *et al.* 1974; Uebler 1984; Jenssen and Krogstad 1988). Intermittent dosing can avoid saturated conditions where anaerobic bacteria thrive and accumulate at the infiltration surface (Siegrist *et al.* 1983; Bancole *et al.* 2003). Dosing also increases the uniformity of application, thereby reducing the 'creeping failure' that is associated with the localised accumulation of solids and organic materials. Oxygen diffusion can also occur more readily, allowing for the aerobic biodegradation of organic materials and improving the air-filled porosity of the infiltration surface (Bancole *et al.* 2003; Potts *et al.* 2004). The mechanisms within the biomat which are responsible for 'recovery' are thought to be associated with decomposition of accumulated organic compounds by oxidative respiration, and drying and contraction of microbial by-products (Bouma *et al.* 1974). These processes increase the macropore space within the biomat and allow for greater infiltration. Excessive and localised loading can lead to saturated flow conditions as only a portion of surface area available for effluent dispersal is utilised. In the United States, technologies such as pressure distribution systems are quite commonly incorporated into conventional SAS to improve the distribution of effluent into the trenches / beds (Siegrist *et al.* 2000). However, in Australia this method for improving system performance has not been so readily adopted - although pressure dosing trenches / beds is referred in the Australian and New Zealand Standard as a method to ensure uniform application (Standards Australia and Standards New Zealand 2000).

Gas composition in the vadose zone of a SAS is important in the treatment capacity of the system and in its long-term hydraulic functioning (Kristiansen 1981a; Wilhelm *et al.* 1994a; Potts *et al.* 2004). There are several investigations (Magdoff *et al.* 1974a; Magdoff *et al.* 1974b; Gerritse *et al.* 1995a; Cromer 2001) that demonstrate that aerobic and oxic conditions prevail under trench systems, despite the often saturated nature of the system and the anaerobic character of septic tank effluent. The presence of oxygen, carbon (C) and form of nitrogen (N) determines the composition of the microbial population (aerobic and anaerobic) responsible for effluent treatment. For example, nitrification of ammonium-N to nitrate will only occur in the presence of oxygen and aerobic bacteria. High rates of complete or near complete nitrification in the unsaturated sub-biomat zone in trenches or soil column experiments have often been reported (Walker *et al.* 1973a; Pell and Nyberg 1989b; van Cuyk *et al.* 2001b).

In soil column experiments, Magdoff *et al* (1974a, 1974b) demonstrated that columns with a developed biomat and perforated sides to allow air exchange, had similar N, C, and phosphorus (P) transformations as columns that had no perforations but had no biomat development. Conversely, columns that had biomat development but no perforations to allow air exchange had lower levels of N, P and C transformation, particularly N where almost 100% of the leachate was in the ammonium form (Magdoff *et al.* 1974b).

Organic matter and suspended solids

In addition to promoting an aerobic soil environment through the establishment of unsaturated conditions, the biomat acts as a filter in straining and trapping biodegradable organics, measured as BOD and SS. The biomat itself is composed partly of organics and SS originating from septic tank effluent (Bouma 1979; Laak 1986; van Cuyk *et al.* 2001b). High removal rates of BOD and SS, ranging from 70% to 95%, have been consistently reported in the literature (Daniel and Bouma 1974; Pell and Nyberg 1989a; van Cuyk *et al.* 2001b).

Pathogens

The fate of pathogens (eg. bacteria and viruses) from domestic wastewater has been widely investigated in both field and laboratory environments. The pathways for pathogen export from SAS are surface water run-off from surcharging (hydraulically failing) trenches or groundwater intrusion via saturated flow beneath a trench. Risk of human contact with pathogens from on-site systems generally increases with decreasing temperature, saturated and anaerobic soil conditions (Reneau *et al.* 1975; Yates and Yates 1988; Beavers and Gardner 1993; Powelson and Mills 2001). Pathogen populations are reduced dramatically by passage through 60 cm –

90 cm of unsaturated soil under the biomat zone (Kristiansen 1981c; Powelson *et al.* 1990; Stevik *et al.* 1999; Siegrist and Van Cuyk 2001; Pang *et al.* 2003; van Cuyk *et al.* 2004). Substantial reductions (≤ log 3-5 decreases) in bacteria concentrations within 90 cm depth of unsaturated media have been reported (Rahe *et al.* 1978; Anderson *et al.* 1994; Weiskel *et al.* 1996; van Cuyk *et al.* 2001b; Pang *et al.* 2003). Population reductions of >99% for viruses have been reported in 60 cm of unsaturated sand (Higgins *et al.* 1999; Oakley *et al.* 1999; Siegrist and Van Cuyk 2001). Weiskel *et al*. (1996) found that although faecal coliforms from SAS contributed the greatest potential load in the Buttermilk Bay catchment, significant subsurface attenuation (4-5 log removal) within the vadose zone 1 – 2 m down gradient of the SAS, resulted in an estimated 0.01% load into the Bay. Many of these studies were done in sandy soils where an unsaturated zone is more likely to occur under a trench due to a greater fraction of air-filled void space due to draining of large pores. However, good pathogen removal rates have been reported in finer-textured soils, for example Oakley *et al.* (1999) reported 100% removal of MS2-coliphage in 60 cm of clay loam.

The biomat zone itself plays a greater role in the removal of pathogens from the wastewater stream than it does in the removal of nutrients (van Cuyk and Siegrist 2001). While the anaerobic conditions in the biomat zone are not favourable for inactivation, pathogen removal is facilitated in the biomat zone by straining and adsorption processes. Whelan and Parker (1981) reported a marked reduction of faecal coliforms within approximately 30cm below the biomat zone of a SAS located in a in a sandy soil.

Alhajjar *et al.* (1988) reported that filtering in the biomat zone effectively removed bacteria, but not the much smaller poliovirus; although the biomat zone appeared to at least retard the passage of poliovirus into the ground water. Both Bouma *et al*. (1972) and Van Cuyk *et al*. (2001) observed high densities of coliform bacteria in the biomat zone (approximately 15-20 mm thick). Lysimeter experiments have demonstrated an association between the stage of biomat development and pathogen removal percentage, where a proportional increase in removal rates occurred as the biomat matures (van Cuyk *et al.* 2001b). However, it is unclear whether this relationship was tested for statistical significance. In van Cuyk *et al.*'s study, viral die-off through the unsaturated zone was calculated by a one-dimensional model where microbial die-off is described as:

$$C_t = C_o \exp(-kt) \tag{5}$$

where C_t is the concentration of viruses at time, t (days), C_o is the initial concentration of viruses (pfu/mL) and k is the first-order rate co-efficient for net die-off rate of viruses.

The retention time in the vadose zone was described as:

$$t = \frac{SoilDepthxEffectivePorosityxEIS}{EffluentApplicationRate} \tag{6}$$

where EIS is the effective infiltration surface which is 1 for a mature trench and 0 for a new trench (van Cuyk *et al.* 2001b). Van Cuyk *et al*. (2001) used Equations 5 and 6 to investigate whether the type of infiltrative surface (e.g. gravel versus gravel-free sand lysimeters) was important in the treatment efficiency. The authors found comparable treatment efficiencies between the two surfaces, and suggested that hydraulic retention time and proportion of surface utilised was more important than the type of surface. This paper drew a credible link between hydraulic processes and treatment efficiency, and illustrated the importance of the first few centimetres, including the biomat zone, in SAS performance.

Pathogen removal in SAS is well-studied and generally understood in terms of the need for an unsaturated zone for effective treatment of effluent. Without this vadose zone, the chances of pathogens contaminating groundwaters are much higher. Pathogen pollution into surface waters (via run-off) in non-sewered areas has also been explored (Geary 1992; Heisig 2000; Graves *et al.* 2002; Ahmed *et al.* 2005). It can be difficult to unambiguously determine the contribution from on-site systems to pathogen pollution due to difficulties in identifying other sources of faecal pollution (e.g. birds and animals). However, there is an increasing number of water quality studies using methods to differentiate between human and non-human sources of faecal pollution, as discussed in more detail later.

Nitrogen

Several factors influence how effectively N is assimilated in the soil. The redox status, soil microbial composition and labile C source are the key factors which determine the degree of total N removal in a SAS (Wilhelm *et al.* 1994a; EPRI 2000). The main form of N in septic tank effluent is ammonium (Whelan and Titamnis 1982) and this commonly undergoes rapid nitrification once the effluent leaches into the unsaturated, aerobic zone underlying the biomat zone (Aravena *et al.* 1993; Wilhelm *et al.* 1994b; Aravena and Robertson 1998). Removal of nitrate, the product of nitrification, is not efficient in the aerobic, low C soil conditions of the unsaturated zone as these conditions inhibit denitrification and subsequent gaseous loss of N. Whelan and Barrow (1984a) found that nitrification was inhibited by the anaerobic state of the biomat layer, but once the effluent passed through the biomat, nitrification was almost complete within half a metre. Other researchers have described similar results with virtually complete nitrification occurring within a few centimetres of the bottom of the biomat zone (Walker *et al.* 1973b; Kristiansen 1981b; Cogger *et al.* 1988; Pell and Nyberg 1989b; Gerritse *et al.* 1995b). Nitrogen can also be physically removed from the effluent via filtration through the biomat as observed by Magdoff and Bouma (1974). They found high concentrations of organic N at, and just below, the infiltrative surface layer of clogged soil columns where a biomat was present compared with columns where there was no biomat present (Magdoff and Bouma 1974). Under anaerobic conditions (i. e. no opportunity for conversion to nitrate), ammonium adsorption can occur (Harrison *et al.* 2000) particularly for soils with a high cation exchange capacity e.g. montmorillonitic soils (McCardell and Davison 2003).

Although removal of total N in SAS is generally not efficient, due to the absence of a significant denitrification phase, there is some evidence to suggest N removal by SAS can occur. Dawes and Goonetilleke (2003) reported that the greatest improvement in water quality occurred within 1m of absorption trenches (sample size 16 SAS) with negligible further removal between 1-3m from trench. Data was not given on the removal rates between the trench and the piezometers located 1m from the trench, though this data would have been useful. Cromer (2001) reported markedly reduced nitrate concentrations within 10 m to 20 m from the trench located in a sandy soil in Hobart, Tasmania. Unfortunately only one SAS was investigated in this study, thus limiting the comparison of these results with similar coastal environments. Gerritse *et al*. (1995b) using a mass balance of inorganic N and bromide, reported ≈ 80% of N was lost within 10 m of travel in sandy soil in Peth, Western Australia. They concluded that N additions to catchment waterways were originating to a much greater extent from agricultural areas compared to non-sewered areas (Gerritse *et al*., 1995b).

In general though, loss of N from SAS is usually poor and elevated nitrate concentrations in groundwater associated with SAS have been well-documented (EPRI 2000). Tracer experiments have revealed that nitrate can travel in aquifers underlying SAS in relatively well-defined, narrow plumes which have been recorded to be up to 130 m in length (Robertson *et al.* 1991) but may extend up to 200 m (Valiela *et al.* 1997).

Denitrification

In the absence of oxygen, nitrate can be lost from the system by denitrification, providing a supply of labile C is available. Optimal conditions for denitrification are generally limited to small anaerobic pockets in the vadose zone or the biomat itself, therefore N removal is generally low in SAS (Reneau *et al.* 1975; Wilhelm *et al.* 1994a). However, with their source of labile C and often anaerobic soil conditions, riparian zones, can help to reduce nitrate concentrations in (shallow ≤2m) groundwater that intercepts this zone (Robertson *et al.* 1991; Hill 1996; Anderson 1999; Gold *et al.* 1999).

Denitrification in riparian zones has been reported to be both spatially and temporally variable (Gold *et al.* 1999). Robertson et al. (1997) studied plumes from SAS in sandy soils and found that over time, SAS-derived nitrate concentrations in receiving waters were much lower than expected, and concluded that denitrification in the riparian zone was a key removal mechanism for nitrate. Geary (2004) demonstrated the importance of downstream riparian zones in his reporting of substantial nitrate losses between an absorption trench and an adjacent surface water body. Groundwater nitrate concentrations decreased from a mean of 75 mg/L to a mean of 2.8 mg/L over a distance of approximately 3 m (Geary 2004). A riparian strip was situated between the two monitoring points, and Geary (2004) suggested that nitrate loss was from plant uptake, denitrification and possibly dilution from converging groundwater flows. Chemical denitrification can also occur in certain soils, for example the oxidation of pyrite, using nitrate as an electron acceptor, will produce nitrogen gas (Postma *et al.* 1991).

Phosphorus

The main form of P in septic tank effluent is orthophosphate. The geochemical mechanisms influencing the availability of P in and below SAS are adsorption/desorption, precipitation/dissolution and biological immobilisation (EPRI 2000). Most studies indicate that P removal in SAS is generally effective (Brouwer and Bugeja 1983; Gerritse 1993; Wilhelm *et al.* 1996; Robertson and Harman 1999). In general, soils (including many sandy soils) have the capacity to retain P (Jones and Lee 1979; Weiskel and Howes 1992).

While the role of the biomat on P removal is poorly understood, clearly some retention of P would occur during the development phase of the biomat zone. Phosphate removal is certainly indirectly influenced by the biomat, if not directly. Sub-biomat soil conditions such as redox and pH, which are important in P retention processes, are influenced substantially by the extent of biomat development. Magdoff *et al*. (1974b) found greater P removal in soil columns with well developed biomats compared to columns with poorly developed or absent biomats. Changes to pH conditions in and just below the biomat were suggested as one factor contributing to this difference.

Phosphate is more strongly retained by the soil matrix under conditions of high pH and oxidation status (Ponnamperuma 1972), conditions which are more likely to prevail in an aerobic, unsaturated environment that is promoted by the presence of an overlying biomat zone. Localised P enrichment can lead to slow release of P into groundwaters over the long-term, though has only been reported for sandy soils (Whelan and Barrow 1984b; Gerritse *et al.* 1995a). Calcareous sands however, have been reported to have a good ability to retain P via rapid sorption then slow precipitation of phosphate with calcium to form amorphous crystalline minerals (Wilhelm *et al.* 1994a). Soils containing high sesquioxide (iron and aluminium hydrous oxides) fractions can strongly adsorb P (Menzies *et al.* 1999; Redding *et al.* 2002). Precipitation of calcium and iron phosphates have also been observed below SAS (Whelan and Barrow 1984b). Saturation of soils, or soils exposed to alternate wetting and drying cycles, of which both scenarios may occur in SAS, will also influence P retention. Ptacek (1998) and Zanini *et al*. (1998) describe the influence of suboxic and oxic regions on P within the subsoil and aquifer zones below SAS. For example, Ptacek (1998) describes the elevated presence of vivianite, an amorphous ferric phosphate, which has been formed following dissolution and subsequent precipitation in the more reduced lower zones of the aquifer.

The influence of micro-scale characteristics in the biomat, vadose and aquifer zone on nutrient assimilation is really only superficially understood in terms of the long term treatment capacity of SAS. Gold and Sims (2000) describe several areas of nutrient dynamics in SAS which require further research, particularly in the context of developing a risk-based model for SAS design and siting.

Environmental and human health impacts associated with on-site systems

Both hydraulic and treatment failures can occur from a poorly operating SAS. *Hydraulic* failure (surcharging) occurs when the infiltration rates through the biomat are exceeded by the loading rate of effluent into the trench, resulting in effluent discharge onto the soil surface. Field studies have identified this as a relatively common occurrence for older and / or poorly designed SAS (Brouwer *et al.* 1979; Geary 1994; Dawes and Goonetilleke 2001). *Treatment* failure is less obvious and is linked closely with the soil biogeochemical processes governing the hydraulic behaviour in the soil system (Siegrist and Van Cuyk 2001). A shallow water table and / or saturated subsoil can result in inadequately treated effluent entering the groundwater. The shortened hydraulic retention time and reduced aerobic conditions that occur in these circumstances preclude adequate treatment of effluent prior to contact with groundwater. Both forms of failure may ultimately result in effluent pollutants (nutrients and pathogens) being exported from the application area and entering surface and / or groundwaters (see selected studies in Table 3).

Table 3

Environmental impacts

The pollutants of key environmental concern are N and P. Nutrient enrichment of a freshwater body can lead to accelerated eutrophication. Land use practices such as agriculture and horticulture, are well studied sources of contaminant loads into catchments (Kookana *et al.* 1998). However, non-sewered areas have also been shown to contribute to catchment nutrient loads (Valiela and Costa 1988; Weiskel *et al.* 1996).

Clear hydraulic links have been made between on-site wastewater dispersal practices and nutrient contamination of ground and surface waters (Table 3) with the use of tracers such as bromide (eg. Robertson *et al.* 1991), chloride (e.g. Stewart and Reneau 1988), ^{15}N stable isotopes (eg. Chen and Harkin 1998), sterol biomarkers (eg. Leeming *et al.* 1996), and viral tracers (eg. Paul *et al.* 1995). All of these studies have found some causal link between elevated effluent-derived pollutant in groundwaters and OWTS, specifically septic systems. It is the *degree* and *impact* of contribution to ground and surface water contamination that remains ambiguous in these studies. The extent of contamination is extremely variable and can be a function of soil type and unsaturated soil depth, existing quality of water body, OWTS density and distance to receptor, groundwater flow velocity, climate and seasonal factors, and age and design of system.

Human health impacts

Concern over microbiological contamination from SAS, particularly in groundwater, has been rising over the years, but as Van Cuyk and Siegrist (2001) suggest, this appears to be based on the widespread distribution of on-site wastewater systems, rather than documented evidence.

Where there have been reports of SAS-related waterborne disease outbreaks, these have usually been localised and associated with a single poorly performing system (Yates and Yates 1988; Cliver 2000; Bopp *et al.* 2003) (Table 3). Cumulative impacts and catchment-wide pathogen loads in non-sewered areas have not been adequately investigated and / or reported. Obviously, there are economic and logistical difficulties in quantifying the origins of a specific diffuse-source pollutant, although catchment water quality models have proved a useful tool in this regard (Yates and Yates 1988; Tong and Chen 2002; Weintraub *et al.* 2002; Whitehead *et al.* 2003).

Several studies have considered cesspools, leaching pools and injection wells as sources of pollution (Table 3) as well as the conventional SAS (Vaughn *et al.* 1983; Chen 1988; Dillon *et al.* 2000). Unlike SAS, cesspools and injection wells are not specifically designed to interact with the soil as part of the key treatment process and they are therefore more likely to provide a substandard level of treatment (USEPA 2002). Despite this, they are commonly grouped in with SAS statistics on failure or poor performance of ‘septic systems’ (Craun 1985; Dillon *et al.* 1999; USEPA 2002).

The number of drinking water-associated disease outbreaks in the United States decreased from 39 during 1999-2000 to 31 during 2001—2002 (Blackburn *et al.* 2004). These 31 outbreaks caused illness among an estimated 1,020 persons, with 52 people (5.5%) becoming ill from drinking groundwater in ‘individual’ homes which were presumably non-sewered (Blackburn *et al.* 2004). Assuming these 52 cases were in non-sewered areas, this represents about 7×10^{-5}% of the non-sewered population in the United States (on the basis that 23% of the population lives in non-sewered areas (USEPA 2002)). Outbreaks associated with private wells remained relatively stable, although there was an increase in reporting of outbreaks involving private, treated wells during 2001—2002 (Blackburn *et al.* 2004).

The contaminant that is repeatedly found in elevated concentrations in groundwaters down gradient from SAS is nitrate (Robertson *et al.* 1991; Harman *et al.* 1996; Robertson *et al*. 2000; Steffy and Kilham 2004). Despite this, it is interesting to note that very few reports of nitrate toxicity associated with SAS are present in the literature. L'hirondel and L'hirondel (2002) critically examined the evidence for well-water induced methaemoglobinaemia. They concluded that there was a poor correlation between high concentrations of nitrates in well-water and infant methaemoglobinaemia (blue-baby syndrome), with the cases of infant methaemoglobinaemia almost completely absent in the US since 1960 (L'hirondel and L'hirondel 2002). Elevated nitrates in well-water have been linked to privies and cesspools but there is scant mention of the conventional SAS being linked to well-water methaemoglobinaemia. Obviously there is much evidence reporting elevated groundwater nitrate concentrations from SAS, however the assumption that these elevated nitrate levels from SAS are a human health problem should not be automatic and there appears to be little recent scientific evidence to support this assumption.

Australian studies

In 1997 over 400 people contracted *Hepatitis* A from eating suspected sewage-contaminated oysters from Wallis Lake (Ryan v Great Lakes Council 1999), triggering the formation of the SepticSafe programme by NSW Department of Local Government. There have been several studies in Australia investigating the contamination of surface and groundwaters from septic systems before and after the Wallis Lake incident (Geary 1994; Hoxley and Dudding 1994;

Jelliffe 1998; Whitehead and Geary 2000). A common problem with many investigations into SAS impacts in Australia are the poor datasets and / or evidence of clear links to SAS as the source of contamination.

A study of two rural towns in Victoria reported variable concentrations of nitrate and faecal coliforms in shallow groundwater below areas of high SAS densities (Hoxley and Dudding 1994). However, most of the data which was discussed was not presented in the paper. The authors concluded that "other towns or cities that have septic tanks in densities similar.. are most likely causing nitrate and bacteriological contamination of the local groundwater…" (Hoxley and Dudding, 1994). This conclusion is highly generalised and very likely to be inaccurate given the wealth of literature that demonstrates treatment in SAS is determined by a number of factors including site and soil conditions, system type, loading rates and depth to groundwater (length of vadose zone).

Sampling and monitoring of surface water, downstream of non-sewered communities on Scotland Island, NSW, showed high concentrations of faecal coliform and enterococci (Martens and Geary 1999). These results, together with sampling and testing of effluent in absorption trenches, led the authors to conclude that the transportation of poorly treated septic tank effluent into surface waters resulted in excessively high bacterial concentrations (Martens and Geary 1999). However, the report did not clearly state the number of water quality samples taken, or if other land use practices and sources of bacterial pollution were considered. Additionally, sampling methods for the drainfield water quality samples were unclear. The results provided appear to indicate that the effluent is of primary treated quality, which suggests that the samples were taken (prematurely) in the saturated zone within the trench itself, rather than the underlying vadose zone where treatment predominantly occurs.

In southeast Queensland the majority of septic systems use separate greywater and blackwater treatment and dispersal (Beal *et al.* 2005). Beal *et al.* (2005) identified the key management issues in the region as the frequency of greywater failure and inappropriate greywater discharge, and suggested that the common practice of discharging effluent into streams, gutters and drains (Jelliffe *et al.* 1995; Beal *et al.* 2005) may be making the greatest contribution to OWTS-related water quality impacts.

To date, only a few Australian investigations have been able to demonstrate conclusive evidence of pollution of groundwater by on-site systems. The contribution of on-site systems to poor surface water quality is even more ambiguous. This can, in part, be attributed to the difficulties associated with quantifying diffuse pollution sources. It can also be difficult to clearly determine the contribution from on-site systems to pathogen pollution due to difficulties in separating human-sources from other sources of faecal pollution (e.g. birds and animals). However, there is an increasing number of studies using methods to differentiate between human and non-human sources of faecal pollution. Recent work using 'biomarkers' as a method of distinguishing human faecal contamination from that of other animals is gaining impetus as a useful tool for assessing water quality impacts to catchment water quality, particularly from diffuse sources such as on-site wastewater systems and agricultural land use (Apte and Bately 1992; Leeming 1998; Geary and Davies 2004). Data obtained from such studies will provide more certainty of the role on-site systems play in catchment water quality degradation.

In areas where shellfish such as oysters are farmed for human consumption, the risks of human health impacts from faecal contamination are increased. Oysters, being filter feeders, are particularly vulnerable to bioaccumulation of contaminants due to their highly efficient filtering capacity compared with humans, for example Pacific oysters can filter up to 50L of water per day (Bayne *et al.* 1999). Geary and Davies (2004) used bacterial source tracking in the shellfish farming region of Port Stephens, NSW, to determine the source of faecal streptococci isolates. They reported that although there was no one source that was statistically significant, SAS were considered one human source of faecal streptococci.

Ahmed *et al*. (2005) used biochemical fingerprinting to identify unique *E. coli* and enterococci phenotypes in a south-east Queensland waterway. A total of 151 unique biochemical phenotypes, that were identified from 48 septic tanks, were found in Eudlo Creek downstream of a non-sewered area. Of these septic tanks, 23 had absorption trenches that were classified as 'soggy' (surface surcharging) and 32 required desludging during the study (Ahmed *et al.* 2005). Unfortunately, the testing is not quantitative so it is impossible to ascertain whether these phenotypes were substantial contributors to the total faecal contamination load in the creek.

The actual *impacts* (e.g. disease, eutrophication, nitrate toxicity) from on-site effluent-polluted waterways are poorly documented. For example, there are few reported cases of outbreaks of endemic illness (Table 3) in areas of high system density and permeable soils; though this would be expected if serious groundwater contamination was present; as was purported to be the case by Hoxley and Dudding (1994). It is acknowledged however, that reporting depends on whether the illness is notifiable or reportable. Absence of evidence is obviously not evidence of absence, and the need for well-considered, long-term water quality monitoring in non-sewered areas is essential to build up an accurate picture of the location, number, and degree of failing on-site systems. This is becoming particularly pressing in areas of high density non-sewered development located near sensitive water receptors and / or on shallow aquifers. Water quality monitoring programmes which generate useful data are difficult to undertake. Event-based monitoring, in addition to baseflow monitoring, is essential as significant surface export of effluent contaminants is only likely to occur during storm events. As discussed, human-sourced faecal pollutants need to be separated from other animals, and SAS-derived nitrate also needs to be discriminated from that arising from other land uses. In this respect, using δN^{15} may be applicable (Steffy and Kilham 2004).

Density of on-site systems

A key factor in sustainable OWTS management is providing sufficient lot area for pollution reduction. As the density of on-site systems increases, there is greater potential for adverse impacts such as cumulative water quality contamination (nitrates) and groundwater mounding (Siegrist *et al.* 2000). Allotment density is determined largely by the horizontal (and vertical) setback distances required in each lot in order for sufficient assimilation of effluent pollutants to occur. Research generally suggests that as OWTS density increases, pollutant loads (e.g nitrates and pathogens) in ground waters have also increased (Perkins 1984; Yates 1985;

Tuthill *et al.* 1998; Lipp *et al.* 2001). Hence, on-site system density is becoming a critical issue in rapidly developing non-sewered areas, and there is a need to determine sustainable lot densities, although this is an inherently difficult task to undertake. A major problem in determining minimum setback distances from on-site systems is estimating the proportion of off-site export of contaminants due to pollutant concentration reduction and reinfiltration, which will vary from site to site.

Despite the difficulties there have been some attempts to quantify minimum lot densities (Perkins 1984; Yates 1985; Geary and Gardner 1998; Jelliffe 1999). A density of greater than 15 systems /km^2 is quoted as being unsustainable, although as Whitehead *et al*. (Whitehead *et al.* 2001) point out, this is based on very limited data. Jelliffe (1998) suggested that setback distances to contain surface exports should vary with soil type and the target receiving water quality objectives, and proposed a relatively simple biophysical model to calculate sufficient nutrient assimilation area. A summary of some minimum lot sizes and densities recommended in the literature is provided in Table 4.

Table 4

There is considerable variation between what is considered a sustainable lot size (Table 4). This is partly due to the variety of factors determining the assimilative capability of a site, partly due to the environmental sensitivity of the area, and partly a result of inadequate knowledge of treatment processes / removal efficiencies of on-site systems.

Areas of further research

The 'National Research Needs' Conference held in the United States in 2000 aimed to identify and discuss essential research areas for on-site wastewater management (EPRI 2000). Many of the identified research needs also apply in Australia, including the development of sound risk-based decision making tools, greater understanding of the interactions between hydraulic and treatment mechanisms in SAS, pathogen pathways and removal processes, and the minimisation of nitrate export to groundwater. Some key research areas which can be identified from this literature review are summarised in Table 5.

Table 5.

Many of the studies presented in Tables 2 and 3 were undertaken in sandy media, with medium to low clay content. Obviously soils are highly variable and more research across a range of soils types would benefit our understanding of hydrology and treatment processes occurring, not only in SAS, but also on soils being irrigated with secondary effluent from alternative on-site systems.

Greater knowledge of the biomat zone development, and its interaction with oxygen availability and subsequent N and pathogen transformations, would be a useful advance in predicting the long-term performance of SAS. Research into improvements in nitrate reduction in the sub-biomat zone, is essential for the sustainable future of SAS.

Identifying and evaluating the potential risks to receiving water bodies from on-site systems is becoming an increasingly popular management approach (Jones *et al.* 2000; Whitehead *et al.*

2003; Day 2004). Risk assessment concepts have been implicitly included in regulatory guidelines for many years; for example, the imposition of setback distance criteria of an OWTS to a boundary or sensitive receptor. However, setback distances are site-specific and a risk-based management approach could help to determine suitable setback distances and sustainable OWTS densities at a site and catchment scale. A good risk assessment would also consider cumulative impacts from on-site systems; something which has received little attention in the literature, or from regulators. Research into risk assessment models is being undertaken to investigate their viability and applicability as a tool in onsite system risk assessment. These include the On-site Sewage Assessment Risk Assessment System (OSRAS) (Whitehead *et al.* 2003), the Development Assessment Module (DAM) (McGuinness and Martens 2003) and a Quantitative Microbial Risk Assessment (Charles and Ashbolt 2004).

Summary

The mechanisms governing purification and hydraulic performance of a SAS are complex and have been shown to be highly influenced by the biomat zone which develops on the soil infiltrative surface within a SAS. The physical, chemical and biological processes that occur in the biomat zone and underlying unsaturated zone are complex and dynamically interactive. Hydraulic and purification processes do not occur separately from each other but interact in a complex, yet potentially predictable way. Equations defining the theoretical movement of effluent through barriers and unsaturated soil zones have been described (eg. Bouma 1975), and more recently, attempts have been made to predict soil clogging development (Siegrist and Boyle 1987; Weintraub *et al.* 2002) and to model the clogging zone impact to flow (Huntzinger Beach and McCray 2003; Beal *et al.* 2004b).

Laboratory studies involving soil columns and lysimeters, designed with an increasing degree of complexity, have yielded many valuable insights and illuminated our understanding of soil-effluent relationships in various media. Field investigations have, in many cases, validated the results and theories from laboratory studies. Field surveys have identified commonalities in hydraulic failure (poor design, low permeable soil, severe clogging) and purification failure (shallow water table, highly permeable soil, shallow depth of unsaturated zone) between SAS. Although there has been considerable research in the area of on-site wastewater management and soil-based treatment systems, some key processes remain unclear.

Key treatment processes, occurring in the unsaturated zone beneath a SAS are oxidation, adsorption, pathogen die-off, and ion exchange. Studies of effluent plumes under SAS, and water quality monitoring in non-sewered areas, suggest that on-site systems are contributing to poor water quality. Nitrate and faecal pollution are the two major effluent contaminants. Groundwater nitrate concentrations have been widely reported in non-sewered areas, particularly in high allotment densities on permeable soils and / or shallow water tables. The contribution of on-site systems to catchment pollutant loads and cumulative water quality effects is less clear. In this respect, targeted water quality monitoring, identified from a risk assessment process, is recommended to improve our knowledge of pollution potential of on-site systems.

In terms of future research, a greater understanding the interactions between hydraulic and treatment mechanisms, and the biomat and sub-biomat zone gas composition including its role in effluent treatment is needed. Improved N removal in SAS remains a priority. Further work on the role of the riparian zone in the reduction of nitrates and the design of systems that incorporate C enriched layers (e.g. Bedessem *et al.* 2005) is necessary. The critical lot density in non-sewered areas is not well understood and consequently setback distances are often unjustified and conservative. Improving our knowledge of micro-scale and site scale processes will lead to a greater understanding of catchment scale impacts from on-site systems.

References

Adin A (1998) Physicochemical mechanisms in treatment processes for water reuse. In 'Wastewater reclamation and reuse'. (Ed. T Asano). (Technomic Publishing Co. Inc.: USA)

Ahmed W, Neller R, Katouli M (2005) Evidence of septic system failure determined by a bacterial biochemical fingerprinting method. *Journal of Applied Microbiology* **98**, 910-920.

Alhajjar BJ, Stramer SL, Cliver DO, Harkin JM (1988) Transport modelling of biological tracers from septic systems. *Water Research* **22**, 907-915.

Allison LE (1947) Effect of microorganisms on permeability of soil under prolonged submergence. *Soil Science* **63**, 439-450.

Al-Shiek Khalil WR, Goonetilleke A, Rigby PA (2004) Use of undisturbed soil columns to evaluate soil capability to renovate on-site sewage treatment system effluent. In 'Tenth National Symposium on Individual and Small Community Sewage Systems'. March 21-24, Sacramento, California pp. 201-209. (ASAE, St Joseph, USA)

Anderson DL (1999) Natural denitrification in shallow groundwater systems. In 'Proceedings 10th Northwest On-site Wastewater Shortcourse, Univ. Washington, Seattle WA' pp. 201-210

Anderson DL, Otis RJ, McNellie JJ, Apfel RA (1994) In-situ lysimeter investigation of pollutant attenuation in the vadose zone of a fine sand. In 'Proceedings of the Seventh National Symposium on Individual and Small Community Sewage Systems'. Atlanta, Georgia pp. 209-218. (ASAE; St Joseph; USA)

Apte SC, Bately GE (1992) Rapid detection of sewage contamination in natural waters. *Chemistry in Australia* **59**, 397-399.

Aravena R, Evans ML, Cherry JA (1993) Stable isotopes of oxygen and nitrogen in source identification of nitrate from septic systems. *Ground Water* **31**, 180-186.

Aravena R, Robertson WD (1998) Use of multiple isotope tracers to evaluate denitrification in ground water: Study of nitrate from a large-flux septic system plume. *Ground Water* **36**, 975-982.

Bancole A, Brissaud F, Gnagne T (2003) Oxidation processes and clogging in intermittent unsaturated infiltration. *Water Science and Technology* **48**, 139-146.

Baveye P, Vandevivere P, Hoyle BL, DeLeo PC, de Lozada DS (1998) Environmental impact and mechanisms of the biological clogging of saturated soils and aquifer materials. *Critical Reviews in Environmental Science and Technology* **28**, 123-191.

Bayne BL, Hedgecock D, McGoldrick D, Rees R (1999) Feeding behaviour and metabolic efficiency contribute to growth heterosis in Pacific oysters [*Crassostrea gigas* (Thunberg)]. *Journal of Experimental Marine Biology and Ecology* **233**, 115-130.

Beal CD, Gardner EA, Christiansen C, Beavers P (2005) A review of on-site wastewater management practices in south-east Queensland. *Water (Australia)* **32 (4)**, 69-72.

Beal CD, Gardner EA, Menzies NW, Rassam DW, Vieritz AM (2004b) Prediction of steady-state flux through variably saturated zones within a septic absorption trench. In 'Proceedings of the International Soil Science Conference, Dec. 2004, Sydney, Australia'

Beal CD, Gardner EA, Vieritz AM, Menzies NW (2004a) The role of the biomat in the sustainable performance of soil absorption systems in Australia: A review. In 'Proceedings of the Tenth National Symposium on Individual and Small Community Sewage Systems'. Sacramento, CA pp. 241-248. (ASAE, St Joseph, USA)

Beavers PD, Gardner EA (1993) Prediction of virus transport through soils. In '15th AWWA Federal Convention, Gold Coast QLD,' pp. 530-535. (AWWA Canberra ACT)

Bedessem ME, Edgar TV, Roll R (2005) Nitrogen removal in laboratory model leachfields with organic rich layers. *Journal of Environmental Quality* **34**, 936-942.

Blackburn MD, Craun G, Yoder JS, Hill V, Calderon R, Chen N, Lee S, Levy D, Beach M (2004) 'Surveillance for waterborne-disease outbreaks - United States, 2001 - 2002.' In: Surveillance Summaries, October 22, 2004. *MMWR* 2004; 53 (No. SS-8):23-45.

Bopp DJ, Sauders BD, *et al.* (2003) Detection, isolation and molecular subtyping of *Escherichia coli* O157:H7 and *Campylobacter jejuni* associated with a large waterborne outbreak. *Journal of Clinical Microbiology* **41**, 174-180.

Borchardt MA, Chyou P, De Vries E, O, Belongia EA (2003) Septic system density and infectious diarrhea in a defined population of children. *Environmental Health Perspectives* **111**, 742-747.

Bouma J (1975) Unsaturated flow during soil treatment of septic tank effluent. *Journal of Environmental Engineering*, 967-981.

Bouma J (1979) Subsurface applications of sewage effluent septic tanks, soil factors, hydrology, pathogens. *Agronomy Advances Series Monograph, American Society of Agronomists*, 665-703.

Bouma J, Converse JC, Magdoff FR (1974) Dosing and resting to improve soil absorption beds. *Transactions of the ASAE* **17**, 295-298.

Bouma J, Ziebell WA, Walker PG, Olcott PG, McCoy EL, Hole FD (1972) 'Soil absorption of septic tank effluent: a field study of some major soils in Wisconsin.' (University of Wisconsin Extension Circular no. 20, Soil Survey Division, Wisconsin. 235 p)

Brandes M (1978) Characteristics of effluents from gray and black water septic tanks. *Journal of Water Pollution Control Federation* **50**, 2547-2559.

Brouwer J, Bugeja RM (1983) 'Land capability for septic tank effluent absorption fields -part A: project report.' Australian Water Resources Council Research Project No 79/118, Canberra.

Brouwer J, Willatt ST, van der Graaff R (1979) The hydrology of on-site septic tank effluent disposal on a yellow duplex soil. In 'Hydrology and Water Resources Symposium, 10-12 Sept 1979, Perth'. (IEAust, ACT)

Carroll SP, Goonetilleke A, Hargreaves M (2004) Assessment of environmental and public health risk of on-site wastewater treatment systems. In 'Proceedings of the Tenth National Symposium on Individual and Small Community Sewage Systems'. Sacramento, CA pp. 368-376. (ASAE, St Joseph, USA)

Charles K, Ashbolt N (2004) Quantitative Microbial Risk Assessment: a catchment management tool to delineate setback distances for septic systems. In 'Water and Environmental Management Series (Young Researchers Conference 2004)'. (Eds P Lens and R Stuetz). (IWA: Wageningen, The Netherlands)

Charles KJ, Ashbolt NJ, Roser DJ, McGuinness R, Deere DA (2004) Effluent quality from 200 on-site sewage systems: Design values for guidelines. In '6th Specialist Conference on Small Water & Wastewater Systems and 1st International Conference on Onsite Wastewater Treatment & Recycling, Fremantle, WA, Australia, International Water Association (IWA).'

Chen C, Harkin JM (1998) Transformations and transport of ^{15}N-based fixed nitrogen from septic tanks in soil absorption systems and underlying aquifers. In 'Proceedings of the Eighth National Symposium on Individual and Small Community Sewage Systems'. Orlando, Florida pp. 293-305. (ASAE; St Joseph; USA)

Chen M (1988) Pollution of groundwater by nutrients and fecal coliforms from lakeshore septic tank systems. *Water Air and Soil Pollution* **73**, 407-417.

Cliver DO (2000) Research needs in decentralized wastewater treatment and management: Fate and transport of pathogens. *National Research Needs Conference Proceedings: Risk Based Decision Making for Onsite Wastewater Treatment.* EPRI Palo Alto, CA, U.S. Environmental Protection Agency and National Decentralized Water Resources Capacity Development Project: 2001 1101446 p. 31

Cogger CG (1988) On-site septic systems: The risk of groundwater contamination. *Journal of Environmental Health* **51**, 12-16.

Cogger CG, Hajjar LM, Moe CL, Sobsey MD (1988) Septic system performance on a coastal barrier island. *Journal of Environmental Quality* **17**, 401-407.

Converse JC, Tyler EJ, Sievers DM (1998) Soil treatment of aerobically treated domestic waste water with emphasis on modified mounds. In 'On site wastewater treatment. Proceedings of the Eighth National Symposium on Individual and Small Community Sewage Systems, Orlando, Florida, USA,'. ((ASAE); St Joseph; USA)

Craun GF (1985) A summary of waterborne illnesses transmitted through contaminated groundwater. *Journal of Environmental Health* **48**, 122-127.

Crites RW, Tchobanoglous G (1998) 'Small and decentralized wastewater management systems.' (WCB/McGraw-Hill)

Cromer WC (2001) Treating domestic wastewater in a shallow coastal sand aquifer near Hobart. In ' Proceedings of On-site '01 Conference'. University of New England, Armidale, NSW pp. 113-120. (Lanfax Laboratories)

Daniel TC, Bouma J (1974) Column studies of soil clogging in a slowly permeable soil as a function of effluent quality. *Journal of Environmental Quality* **3**, 321-326.

Davison L, Bayley M, Kohlenberg T, Craven J (2002) 'Performance of reed beds and single pass sand filters with characterisation of domestic effluent: NSW North Coast.' Research Report for NSW Department of Local Government's SepticSafe scheme.

Dawes L, Goonetilleke A (2001) Importance of site characteristics in designing effluent disposal areas. In 'Proceedings of On-site '01 Conference'. University of New England, Armidale, NSW pp. 133-140. (Lanfax Laboratories)

Dawes L, Goonetilleke A (2003) An investigation into the role of site and soil characteristics in onsite sewage treatment. *Environmental Geology* **44**, 467-477.

Dawes L, Goonetilleke A (2004) Assessing changes in soil physical and chemical properties under long term effluent disposal. In 'Tenth National Symposium on Individual and Small Community Sewage Systems'. March 21-24, Sacramento, California. (Ed. KR Mankin) pp. 349-357. (ASAE, St Joseph, Michigan)

Day L (2004) Septic systems as potential pollution sources in the Cannonsville Reservoir watershed, New York. *Journal of Environmental Quality* **33**, 1989-1996.

De Vries J (1972) Soil filtration of wastewater effluent and the mechanism of pore clogging. *Journal of Water Pollution Control Federation* **44**, 565-573.

DeBorde D, C., Woessner W, W., Lauerman B, Ball PN (1998) Virus occurrence and transport in a school septic system and unconfined aquifer. *Ground Water* **36**, 825-834.

Dillon KS, Corbett DR, Chanton JP, Burnett WC, Furbish DJ (1999) The use of sulfur hexafluoride (SF_6) as a tracer of septic tank effluent in the Florida Keys. *Journal of Hydrology* **220**, 129-140.

Dillon KS, Corbett DR, Chanton JP, Burnett WC, Kump L (2000) Bimodal transport of a waste water plume injected into saline ground water of the Florida Keys. *Ground Water* **38**, 624-634.

Edgar NB (1999) Land use in the Taupo catchment, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **33**, 375-383.

EPRI (2000) *National Research Needs Conference Proceedings: Risk Based Decision Making for Onsite Wastewater Treatment.* EPRI Palo Alto, CA, U.S. Environmental Protection Agency and National Decentralized Water Resources Capacity Development Project: 2001 1101446

Forrest JA, Beatty J, Hignett CT, Pickering J, Williams RGP (1985) 'A survey of the physical properties of wheatland soils in Eastern Australia.' Divisional Report No. 78, CSIRO Division of Soils.

Frankenberger WT, Troeh FR, Dumenil LC (1979) Bacteria effects on hydraulic conductivity of soils. *Soil Science Society of America Journal* **43**, 333-338.

Frenzel S, Couvillion C (2002) Fecal-indicator bacteria in streams along a gradient of residential development. *Journal of the American Water Resources Association* **38**, 265-273.

Gardner T, Geary P, Gordon I (1997) Ecological sustainability and on-site effluent treatment systems. *Australian Journal of Environmental Management* **4**, 144-156.

Geary P (1992) Diffuse pollution from wastewater disposal in small unsewered communities. *Australian Journal of Soil and Water Conservation* **5**, 28-33.

Geary P (1994) Soil survey and the design of wastewater disposal systems. *Australian Journal of Soil and Water Conservation* **7**, 16-23.

Geary PM (2004) On-site domestic system effluent tracing in a coastal catchment. In 'Tenth National Symposium on Individual and Small Community Sewage Systems'. March 21-24, Sacramento, California. (Ed. KR Mankin) pp. 722-732. (ASAE, St Joseph, Michigan)

Geary PM, Davies CM (2004) Bacterial source tracking and shellfish contamination in a coastal catchment. *Water Science and Technology* **47**, 95-100.

Geary PM, Gardner EA (1998) Sustainable on-site treatment systems. In 'On site wastewater treatment. Proceedings of the Eighth National Symposium on Individual and Small Community Sewage Systems, Orlando, Florida, USA'. ((ASAE); St Joseph; USA)

Gerritse RG (1993) Mobility of phosphate from wastewater in calcareous sands of Rottnest Island (WA). *Australian Journal of Soil Research* **31**, 235-244.

Gerritse RG (2002) 'Movement of nutrients from onsite wastewater systems in soils.' Geoprocc Pty Ltd prepared for Department of Health, Western Australia, May 28, 2002.

Gerritse RG, Adeney JA, Dimmock GM, Oliver YM (1995a) Retention of nitrate and phosphate in soils of the Darling Plateau in Western Australia: Implications for domestic septic tank systems. *Australian Journal of Soil Research* **33**, 353-367.

Gerritse RG, Adeney JA, Hosking J (1995b) Nitrogen losses from a domestic septic tank system on the Darling Plateau in Western Australia. *Water Research* **29**, 2055-2058.

Gibbs MM (1991) Nutrient concentration changes in the ground water beneath Taupo township following sewage reticulation. *New Zealand Journal of Marine and Freshwater Research* **25**, 153-161.

Gilliom RJ, Patmont CR (1983) Lake phosphorus loading from septic systems by seasonally perched groundwater. *Journal of Water Pollution Control Federation* **55**, 1297-1305.

Gold A, Sims J (2000) Research needs in decentralized wastewater treatment and management: A risk-based approach to nutrient contamination. *National Research Needs Conference Proceedings: Risk Based Decision Making for Onsite Wastewater Treatment.* EPRI Palo Alto, CA, U.S. EPA and National Decentralized Water Resources Capacity Development Project: 2001 1101446 p. 48

Gold AJ, Addy K, Kellogg DQ, Rosenblatt AE (1999) Nitrate removal in shallow groundwater. In 'Proceedings 10th Northwest On-site Wastewater Shortcourse, Univ. Washington, Seattle WA' pp. 211-231

Graves AK, Hagedorn C, Teetor A, Mahal M, Booth AM, Reneau RB (2002) Antibiotic resistance profiles to determine sources of fecal contamination in a rural Virginia watershed. *Journal of Environmental Quality* **31**, 1300-1308.

Halliwell DJ, Barlow KM, Nash DM (2001) A review of the effects of wastewater sodium on soil physical properties and their implications for irrigation systems. *Australian Journal of Soil Research* **39**, 1259-1267.

Hansen EC, Mansell RS (1986) Simulated unsaturated-saturated water flow from a septic tank drain to groundwater. *Soil and Crop Science Society of Florida* **45**, 21-29.

Harman J, Robertson WD, Cherry JA, Zanini L (1996) Impacts on a sand aquifer from an old septic system: nitrate and phosphate. *Ground Water* **34**, 1105-1114.

Harrison RB, Turner NS, Hoyle JA, Krejsl J, Tone DD, Henry CL, Isaksen PJ, Xue D (2000) Treatment of septic effluent for fecal coliform and nitrogen in coarse-textured soils: use of soil-only and sand filter systems. *Water Air and Soil Pollution* **124**, 205-215.

Heisig P (2000) 'Effects of residential and agricultural land uses on the chemical quality of baseflow of small streams in the Croton Watershed, southeastern New York.' Water-Resources Investigations Report 99-4173, U.S Geological Survey: Reston, Virginia.

Higgins JJ, Heufelder G, Foss S (1999) Removal efficiency of standard septic tank and leach trench septic systems for MS2 Coliphage. In 'Proceedings 10th Northwest On-site Wastewater Shortcourse, Univ. Washington, Seattle WA'

Hill AR (1996) Nitrate removal in stream riparian zones. *Journal of Environmental Quality* **25**, 743-755.

Hillel D (1980) 'Applications of soil physics.' (Academic Press, Inc.)

Hoxley G, Dudding M (1994) Groundwater contamination by septic tank effluent: two case studies in Victoria, Australia. In 'Water Down Under 94, 21-25 Nov 1994, Adelaide SA,' pp. 145-152. (Institution of Engineers Australia)

Huntzinger Beach DN, McCray JE (2003) Numerical modeling of unsaturated flow in wastewater soil absorption systems. *Ground Water Monitoring and Remediation* **23**, 64-72.

Jacoby JM, Anderson CW, Welch EB (1997) Pine Lake response to diversion of wetland inflow. *Lake and Reservoir Management* **13**, 302-314.

Janni KA, Nye JC, Jones DD, Krutz GW, Yahner JE (1980) Finite element analysis of effluent flow from subsurface sewage disposal fields. *Transactions of the ASAE* **23**, 336-342.

Jelliffe PA (1998) Predicting stormwater quality from unsewered development. *Waterfall (Journal of the Stormwater Industry Association)* **9**, 19-24.

Jelliffe PA (1999) Developments in determining critical lot density for the protection of water quality. In 'On-site '99. Proceedings of On-site '99 Conference: Making on-site wastewater systems work'. Univeristy of New England,Armidale, NSW. (Ed. A Patterson Robert) pp. 191-198. (Lanfax Laboratories)

Jelliffe PA, Sabburg G, Wolff J (1995) Key factors in minimising water pollution from unsewered areas. In 'Australian Water and Wastewater Association Federal Convention, 16th, 2- 6 Apr 1995, Sydney NSW, Proceedings' pp. 85-90. (Australian Water and Wastewater Association, Artarmon NSW)

Jenssen PD, Krogstad (1988) Particles found in clogging layers of wastewater infiltration systems may cause reduction in infiltration rate and enhance phosphorus absorption. *Water Science and Technology* **20**, 251-253.

Jones A, Lee F (1979) Septic tank wastewater disposal systems as phosphorus sources for surface waters. *Journal of Water Pollution Control Federation* **51**, 2764-2775.

Jones DS, Armstrong AQ, Mulheim MD, Sorensen BV (2000) Integrated risk assessment / risk management as applied to decentralized wastewater treatment: A high level framework. In '*National Research Needs Conference Proceedings: Risk Based Decision Making for Onsite Wastewater Treatment.* EPRI Palo Alto, CA, U.S EPA and National Decentralized Water Resources Capacity Development Project: 2001 1101446'

Katz BG, Lindner JB, Ragone SE (1980) A comparison of nitrogen in shallow groundwater from sewered and unsewered areas. Nassau County, New York from 1952 through 1976. *Ground Water* **18**, 607-616.

Kookana RS, Baskaran S, Naidu R (1998) Pesticide fate and behaviour in Australian soils in relation to contamination and management of soil and water: a review. *Australian Journal of Soil Research* **36**, 715 - 764.

Kristiansen R (1981a) Sand-filter trenches for purification of septic tank effluent: I. The clogging mechanism and soil physical environment. *Journal of Environmental Quality* **10**, 353-357.

Kristiansen R (1981b) Sand-filter trenches for purification of septic tank effluent: II. The fate of nitrogen. *Journal of Environmental Quality* **10**, 358-361.

Kristiansen R (1981c) Sand-filter trenches for purification of septic tank effluent: III. The microflora. *J Environ Qual* **10**, 361-364.

Kropf FW, Laak R, Healy KA (1977) Equilibrium operation of subsurface absorption systems. *Journal of Water Pollution Control Federation* **49**, 2007-2016.

Laak R (1970) Influence of domestic wastewater pretreatment on soil clogging. *Journal of Water Pollution Control Federation* **42**, 1495-1500.

Laak R (Ed.) (1986) 'Wastewater engineering design for unsewered areas (2nd edn).' (Technomic Publishing Co. Inc.)

Lapointe BE, Clark W (1992) Nutrient inputs from the watershed and coastal eutrophication in the Florida Keys. *Estuaries* **15**, 465-476.

Lapointe BE, O'Connell JD, Garrett GS (1990) Nutrient couplings between on-site sewage disposal systems, groundwaters, and nearshore surface waters of the Florida Keys. *Biogeochemistry* **10**, 289-307.

Leeming R (1998) Distinguishing sources of faecal pollution in Australian inland and coastal waters. In 'Microbial indicators of river health, 31 July-1Aug 1997, Macquarie University, Sydney NSW, Proceedings' pp. 29-31. (Land and Water Resources Research and Development Corporation Canberra ACT)

Leeming R, Ball A, Ashbolt NJ, Nichols PD (1996) Using faecal sterols from humans and animals to distinguish faecal pollution in receiving waters. *Water Research* **30**, 2893-2900.

L'hirondel J, L'hirondel J-L (2002) 'Nitrate and man.' (CABI Publishing, UK)

Lipp EK, Farrah S, Rose JB (2001) Assessment and impact of microbial fecal pollution and human enteric pathogens in a coastal community. *Marine Pollution Bulletin* **42**, 286-293.

Magdoff FR, Bouma J (1974) The development of soil clogging in sands leached with septic tank effluent. In ' Proceedings of the National Homesewage Disposal Symposium, 1974 .' pp. 37-47. (ASAE; St Joseph; USA)

Magdoff FR, Bouma J, Keeney DR (1974a) Columns representing mound-type disposal systems for septic tank effluent: I. Soil-water and gas relations. *Journal of Environmental Quality* **3**, 223-228.

Magdoff FR, Keeney DR, Bouma J (1974b) Columns representing mound-type disposal systems for septic tank effluent: II. Nutrient transformations and bacterial populations. *Journal of Environmental Quality* **3**, 228-233.

Mamedov AI, Levy GJ (2001) Clay dispersivity and aggregate stability effects on seal formation and erosion in effluent-irrigated soils. *Soil Science* **166**, 631-639.

Martens DM, Geary PM (1999) Australian on-site wastewater strategies: A case study of Scotland Island, Sydney, Australia. In 'Proceedings of On-site '99 Conference'. University of New England, Armidale, NSW pp. 255-263. (Lanfax Laboratories)

McCardell A, Davison L (2003) Greywater attenuation in a basaltic soil. In 'Proceedings of On-site '03 Conference'. Armidale NSW. (Lanfax Laboratories)

McGauhney PH, Winneberger JH (1964) Studies of the failure of septic tank percolation systems. *Journal of Water Pollution Control Federation* **36**, 593-606.

McGuinness RI, Martens DM (2003) GIS-based model to assess potential risk of individual on-site effluent management systems - development assessment module (DAM). In 'Proceedings of On-site '03 Conference'. University of New England, Armidale, NSW. (Lanfax Laboratories)

Menzies NW, Skilton JA, Guppy CN (1999) Phosphorus storage on effluent irrigated land. *Journal of Environmental Quality* **28**, 750-754.

Moore JW, Scheuerell MD, Frodge J (2003) Lake eutrophication at the urban fringe, Seattle region, USA. *Ambio* **32**, 13-18.

Oakley S, Greenwood WP, Lee M (1999) Monitoring nitrogen and virus removal in the vadose zone with suction lysimeters. In 'Proceedings 10th Northwest On-site Wastewater Shortcourse, Univ. Washington, Seattle WA'

Otis RJ (1984) Soil clogging:mechanisms and control. In 'On site wastewater treatment. Proceedings of the Fourth National Symposium on Individual and Small Community Sewage Systems, New Orleans, Louisiana, USA' pp. 238-251. ((ASAE); St Joseph; USA)

Otis RJ (1991) Demythologizing the septic tank. In 'On site wastewater treatment. Proceedings of the Sixth National Symposium on Individual and Small Community Sewage Systems, Chicago, Illinios, USA' pp. 1-6. ((ASAE); St Joseph; USA)

Pang L, Close M, Goltz M, Sinton L, Davies H, Hall C, Stanton G (2003) Estimation of septic tank setback distances based on transport of *E. coli* and F-RNA phages. *Environment International* **29**, 907-921.

Patterson RA (2001) Wastewater quality relationships with reuse options. *Water Science and Technology* **43**, 147-154.

Paul JH, Rose JB, Brown J, Shinn E, Miller S, Farrah S (1995) Viral tracer studies indicate contamination of marine waters by sewage disposal practices in Key Largo, Florida. *Applied and Environmental Microbiology* **61**, 2230-2234.

Pell M, Nyberg F (1989a) Infiltration of wastewater in a newly started pilot sand-filter system: I. Reduction of organic matter and phosphorus. *Journal of Environmental Quality* **18**, 451-457.

Pell M, Nyberg F (1989b) Infiltration of wastewater in a newly started pilot sand-filter system: III. Transformation of nitrogen. *Journal of Environmental Quality* **18**, 463-467.

Perkins RJ (1984) Septic tanks, lot size and pollution of water table aquifers. *Journal of Environmental Health* **46**, 298-304.

Ponnamperuma FN (1972) The chemistry of submerged soils. *Advances in Agronomy* **24**, 29 - 96.

Postma D, Boesen C, Kristiansen H, Larsen F (1991) Nitrate reduction in an unconfined aquifer: water chemistry, reduction processes, and geochemical modelling. *Water Resources Research* **27**, 2027-2045.

Postma FB, Gold AJ, Loomis GW (1992) Nutrient and microbial movement from seasonally-used septic systems. *Journal of Environmental Health* **55**, 5-10.

Potts DA, Gorres JH, Nicosia EL, Amador JA (2004) Effects of aeration on water quality from septic system leachfields. *Journal of Environmental Quality* **33**, 1828-1838.

Powelson DK, Mills AL (2001) Transport of *Escherichia coli* in sand columns with constant and changing water contents. *Journal of Environmental Quality* **30**, 238-245.

Powelson DK, Simpson JR, Gerba CP (1990) Virus transport and survival in saturated and unsaturated flow through soil columns. *Journal of Environmental Quality* **19**, 396-401.

Ptacek CJ (1998) Geochemistry of a septic-system plume in a coastal barrier bar, Point Pelee, Ontario, Canada. *Journal of Contaminant Hydrology* **33**, 293-312.

Rahe TM, Hagedorn C, McCoy EL, Kling GF (1978) Transport of antibiotic resistant Escherichia coli through western Orgeon hillslope soils under saturated flow. *Journal of Environmental Quality* **7**, 487-494.

Redding MR, A B, Gardner T, Duperouzel D (2002) An overview of land application of pig effluent-P using soil P chemistry and mass balance calculations. *Australian Journal of Soil Research* **40**, 81-91.

Reneau RB, Elder JH, Pettry DE, Weston CW (1975) Influence of soils on bacterial contamination of a watershed from septic sources. *Journal of Environmental Quality* **4**, 249-252.

Reneau RB, Pettry DE (1975) Movement of coliform bacteria from septic tank effluent through selected coastal plain soils of Virginia. *Journal of Environmental Quality* **4**, 41-44.

Rice RC (1974) Soil clogging during infiltration of secondary effluent. *Journal of Water Pollution Control Federation* **46**, 708-716.

Robertson WD, Blowes DW, Ptacek CJ, Cherry JA (2000) Long-term performance of in situ reactive barriers for nitrate remediation. *Ground Water* **38**, 689-695.

Robertson WD, Cherry JA, Sudicky EA (1991) Ground-water contamination from two small septic systems on sand aquifers. *Ground Water* **29**, 82-92.

Robertson WD, Harman J (1999) Phosphate plume persistence at two decommissioned septic system sites. *Ground Water* **37**, 228-236.

Ronner AB, Wong ACL (1998) Characterization of microbial clogging in wastewater infiltration systems. In 'Proceedings of the Eighth National Symposium on Individual and Small Community Sewage Systems'. Orlando, Florida pp. 37-42. (ASAE; St Joseph; USA)

Ryan v Great Lakes Council (1999) Federal Court of Australia, 177 (5 March 1999).

Siegrist R (1987) Soil clogging during subsurface wastewater infiltration as affected by effluent composition and loading rate. *Journal of Environmental Quality* **16**, 181-187.

Siegrist R, Boyle WC (1987) Wastewater-induced soil clogging development. *Journal of Environmental Engineering* **113**, 550-566.

Siegrist R, Hargett DL, Anderson DL (1983) Vadose zone effects on the performance of large subsurface wastewater absorption systems. In 'Proceedings on Characterization and Monitoring of the Vadose (unsaturated zone)'. Las Vegas NV, National Water Well Association, Worthington OH, pp 223-242

Siegrist R, Smed-Hildmann R, Filip ZK, Jenssen PD (1991) Humic substances formation during wastewater infiltration. In 'Proceedings of the Sixth National Symposium on Individual and Small Community Sewage Systems'. Chicago, IL pp. 223-232. (ASAE; St Joseph)

Siegrist R, Tyler EJ, Jenssen PD (2000) Design and performance of onsite wastewater soil absorption systems. *National Research Needs Conference Proceedings: Risk Based Decision Making for Onsite Wastewater Treatment.* EPRI Palo Alto, CA, U.S. Environmental Protection Agency and National Decentralized Water Resources Capacity Development Project: 2001 1101446 p. 48

Siegrist R, Van Cuyk S (2001) Wastewater soil absorption systems: The performance effects of process and environmental conditions. In 'Proceedings of the Ninth National Symposium on Individual and Small Community Sewage Systems'. Fort Worth, Texas pp. 41-51. (ASAE; St Joseph)

Standards Australia (1994) 'AS 1547: 1994 - Disposal systems for effluent from domestic premises.' Standards Association of Australia, NSW.

Standards Australia, Standards New Zealand (2000) 'AS/NZS 1547:2000 On-site domestic wastewater management.' (Jointly published by Standards Australia International Ltd, NSW and Standards New Zealand, Wellington)

Starr JL, Sawhney BL (1980) Movement of nitrogen and carbon from a septic system drainfield. *Water Air and Soil Pollution* **13**, 113-123.

Steffy LY, Kilham S (2004) Elevated $\delta 15^{N}$ in stream biota in areas with septic tank systems in an urban watershed. *Ecological Applications* **14**, 637-641.

Stevik TK, Ausland G, Jenssen PD, Siegrist R (1999) Removal of *E.coli* during intermittent filtration of wastewater effluent as affected by dosing rate and media type. *Water Research* **33**, 2088-2098.

Stewart LW, Reneau RB, Jr. (1988) Shallowly placed, low pressure distribution system to treat domestic wastewater in soils with fluctuating high water tables. *Journal of Environmental Quality* **17**, 499-504.

Tong ST, Chen W (2002) Modeling the relationship between land use and surface water quality. *Journal of Environmental Management* **66**, 377-393.

Tuthill A, Meikle D, Alavanja M (1998) Coliform bacteria and nitrate contamination of wells in major soils of Frederick, Maryland. *Journal of Environmental Health* **60**, 16-20.

Tyler EJ, Converse JC (1994) Soil acceptance of onsite wastewater as affected by soil morphology and wastewater quality. In 'Proceedings of the Seventh National Symposium on Individual and Small Community Sewage Systems'. Atlanta, GA pp. 185-194. (ASAE; St Joseph)

Uebler RL (1984) Effect of loading rate and soil amendments on inorganic nitrogen and phosphorus leached from a wastewater soil absorption system. *Journal of Environmental Quality* **13**, 475-479.

USEPA (1977) 'Waste disposal practices and their effects on ground water. Report to Congress, January, 1977, Washington DC, USA, 512pp.'

USEPA (1997) 'Response to Congress On Use of Decentralized Wastewater Treatment Systems. USEPA publication EPA/832/R-97/001b. U.S. Environmental Protection Agency, Office of Water, Washington, DC.'

USEPA (2002) 'Onsite wastewater treatment systems manual.' Office of Research and Development, US Environmental Protection Agency EPA/625/R-00/008, February.

Valiela I, Collins G, Kremer J, Lajtha K, Geist M, Seely B, Brawely J, Sham CH (1997) Nitrogen loading from coastal watersheds to receiving estuaries: New method and application. *Ecological Applications* **7**, 358-380.

Valiela I, Costa JE (1988) Eutrophication of Buttermillk Bay, a Cape Cod Coastal embayment: Concentrations of nutrients and watershed nutrient budgets. *Environmental Management* **12**, 539-553.

van Cuyk S, Siegrist R (2001) Pathogen fate in wastewater soil absorption systems as affected by effluent quality and soil clogging genesis. In ' Proceedings of the Ninth National Symposium on Individual and Small Community Sewage Systems'. Fort Worth, Texas pp. 125-135. (ASAE; St Joseph; USA)

van Cuyk S, Siegrist R, Logan A (2001a) Evaluation of virus and microbial purification in wastewater soil absorption systems using multicomponent surrogate and tracer additions. In ' Proceedings of the Ninth National Symposium Individual and Small Community Sewage Systems' pp. 30-40. (ASAE; St Joseph; USA)

van Cuyk S, Siegrist R, Logan A, Masson S, Fischer E, Figueroa L (2001b) Hydraulic and purification behaviours and their interactions during wastewater treatment in soil infiltration systems. *Water Research* **35**, 953-964.

van Cuyk S, Siegrist RL, Lowe K, Harvey R (2004) Evaluating microbial purification during soil treatment of wastewater with multicomponent tracer and surrogate tests. *Journal of Environmental Quality* **33**, 316-329.

Vaughn JM, Landry EF, McHarrell TZ (1983) Entrainment of viruses from septic tank leach fields through a shallow, sandy soil aquifer. *Applied and Environmental Microbiology* **1983**, 1474-1480.

Walker WG, Bouma J, Keeney DR, Magdoff FR (1973a) Nitrogen transformations during subsurface disposal of septic tank effluent in sands. i. soil transformations. *Journal of Environmental Quality* **2**, 475-480.

Walker WG, Bouma J, Keeney DR, Olcott PG (1973b) Nitrogen transformations during subsurface disposal of septic tank effluent in sands. ii. ground water quality. *Journal of Environmental Quality* **2**, 521-525.

Weintraub L, Chen C, Tsai W, Herr J, Goldstein R, Siegrist R (2002) Modifications of WARMF to assess the efficacy of onsite wastewater systems on public health. In 'WEFTEC 2002, Chicago, IL, September 28 - October 2, 2002'

Weiskel PK, Howes BL (1992) Differential transport of sewage-derived nitrogen and phosphorus through a coastal watershed. *Environmental Science and Technology* **26**, 352-360.

Weiskel PK, Howes BL, Heufelder G (1996) Coliform contamination of a coastal embayment: sources and transport pathways. *Environmental Science and Technology* **30**, 1872-1881.

Whelan BR, Barrow NJ (1984a) The movement of septic tank effluent through sandy soils near Perth: I. Movement of nitrogen. *Australian Journal of Soil Research* **22**, 283-292.

Whelan BR, Barrow NJ (1984b) The movement of septic tank effluent through sandy soils near Perth: II. Movement of phosphorus. *Australian Journal of Soil Research* **22**, 293-302.

Whelan BR, Parker WF (1981) Bacterial and chemical transmission through sand: a field study in groundwater pollution form a septic tank in Perth, Western Australia. In 'Groundwater resources of the Swan Coastal Plain'. (Ed. BR Whelan) pp. 313-333. (CSIRO Aust. Perth)

Whelan BR, Titamnis ZV (1982) Daily chemical variability of domestic septic tank effluent. *Water Air and Soil Pollution* **17**, 131-139.

Whitehead JH, Geary PM (2000) Geotechnical aspects of domestic on-site effluent management systems. *Australian Journal of Earth Sciences* **47**, 75-82.

Whitehead JH, Geary PM, Saunders M (2001) Towards a better understanding of sustainable lot density - evidence from five Australian case studies. In 'On-site '01: Proceedings of On-site '01 Conference'. University of New England, Armidale, NSW pp. 383-390. (Lanfax Laboratories)

Whitehead JH, Geary PM, Saunders M, Robertson G, Barry M, Syme W (2003) GIS-based risk assessment for catchment scale on-site wastewater management. In 'Proceedings of On-site '03 Conference:'. University of New England, Armidale, NSW. (Lanfax Laboratories)

Wilhelm SR, Schiff SL, Cherry JA (1994b) Biogeochemical evolution of domestic waste water in septic systems: 1. Conceptual model. *Ground Water*, 905-916.

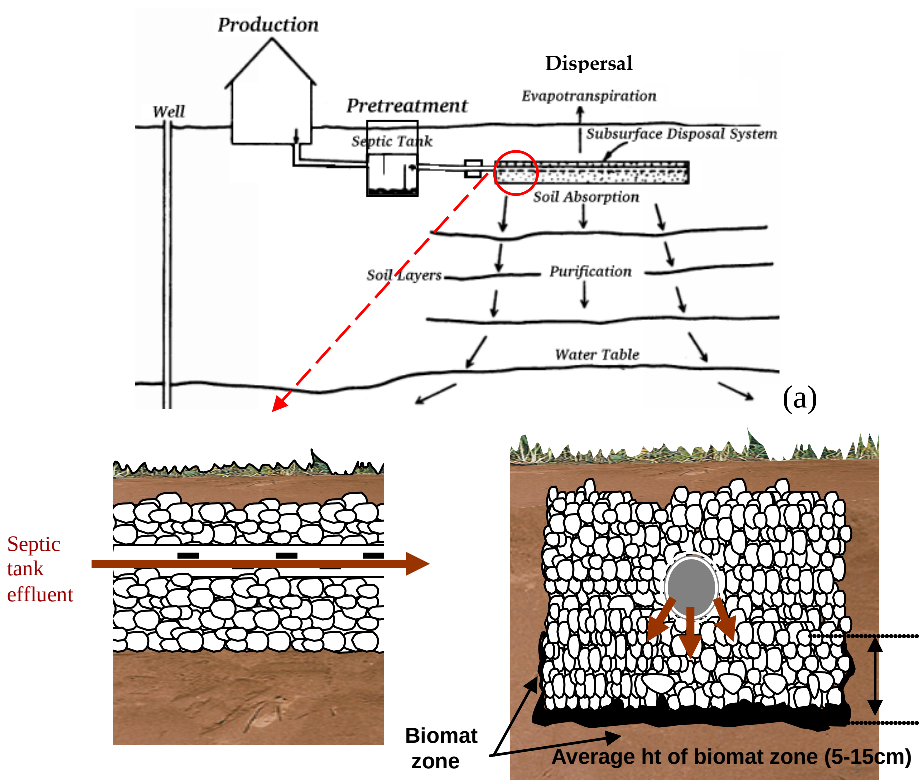
Wilhelm SR, Schiff SL, Robertson WD (1994a) Chemical fate and transport in a domestic septic system: unsaturated and saturated zone geochemistry. *Environmental Toxicology and Chemistry* **13**, 193-23.

Wilhelm SR, Schiff SL, Robertson WD (1996) Biogeochemical evolution of domestic waste water in septic systems: 2. application of conceptual model in sandy aquifers. *Ground Water* **34**, 853-864.

Yates MV (1985) Septic tank density and ground-water contamination. *Ground Water* **23**, 586-591.

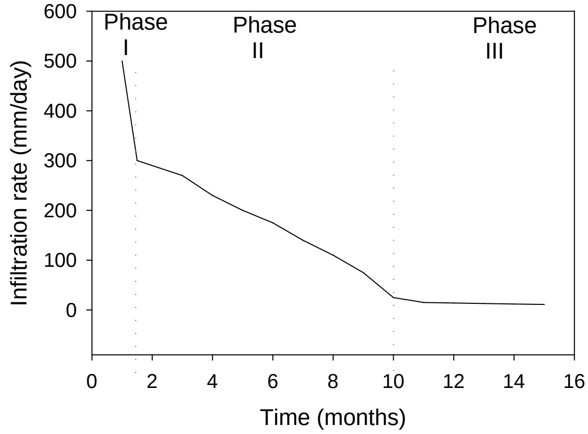
Yates MV, Yates SR (1988) Modeling microbial fate in the subsurface environment. *Critical Reviews in Environmental Science and Technology* **17**, 307-345.

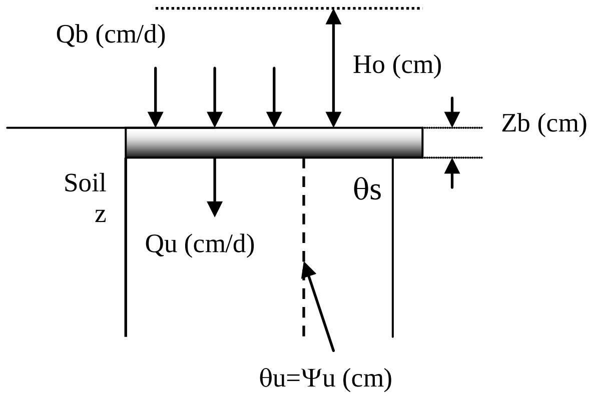
Zanini L, Robertson WD, Ptacek CJ, Schiff SL, Mayer T (1998) Phosphorus characterization in sediments impacted by septic effluent at four sites in central Canada. *Journal of Contaminant Hydrology* **33**, 405-429.

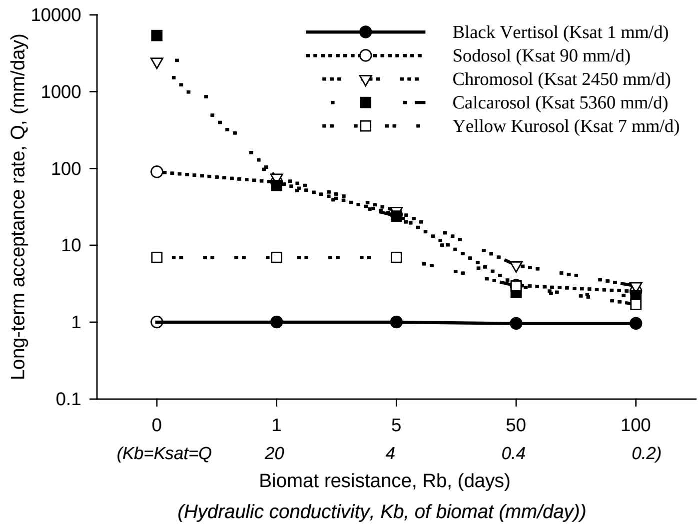


(b) Plan view of distribution pipe in gravel trench

(c) Cross section of distribution pipe in gravel trench


Phase I
Phase II
Phase III
Infiltration rate (mm/day)
600
500
400
300
200
100
0
0
2
4
6
8
10
12
14
16
Time (months)


Qb (cm/d)
Ho (cm)
Zb (cm)
Soil
z
θs
Qu (cm/d)
θu=Ψu (cm)



10000
1000
100
10
1
0.1
Long-term acceptance rate, Q, (mm/day)
Black Vertisol (Ksat 1 mm/d)
Sodosol (Ksat 90 mm/d)
Chromosol (Ksat 2450 mm/d)
Calcarosol (Ksat 5360 mm/d)
Yellow Kurosol (Ksat 7 mm/d)
0
1
5
50
100
(Kb=Ksat=Q
20
4
0.4
0.2)
Biomat resistance, Rb, (days)
(Hydraulic conductivity, Kb, of biomat (mm/day))

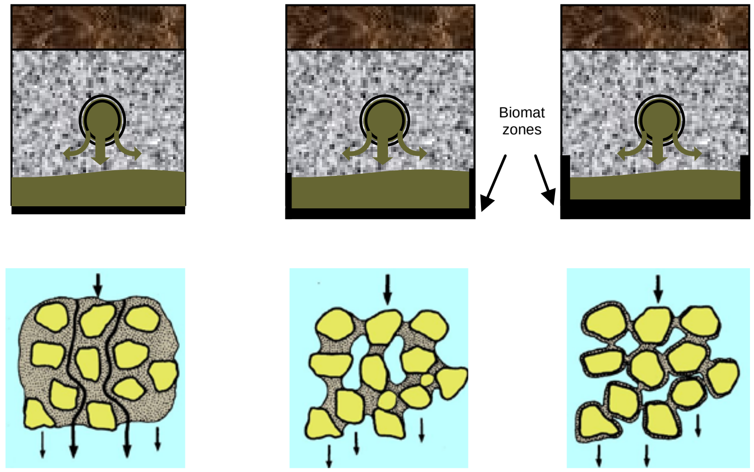

Biomat
zones

Table 1. Septic tank effluent quality for key pollutants

Effluent Parameter	(Whelan and Titamnis 1982)	Beavers and Gardner (1993)	Gardner *et al*. (1997)	Charles *et al*. (2004)[A]
BOD_5 (mg/L)	52 - 316	150 – 180	120 - 180	224
Suspended solids (mg/L	22 - 47	100 – 180	40 - 190	379
Total N (mg/L)	74 - 237	50 – 60	40 - 50	160
Total P (mg/L)	12.3 - 26	10 – 15	10 - 15	21
Faecal coliforms (cfu/100mL)	-	10^5 - 10^7	10^5 - 10^7	10^6 [B]

A Mean

B Thermotolerant coliforms

Table 2 Summary of selected research investigating the removal of nutrients and pathogens in soil absorption systems

Reference	Summary of research	Removal efficiencies and comments
Nutrients		
(Walker *et al.* 1973a)	Five field SAS studied. Organic N retained in biomat zone.	All $NH_4 \rightarrow NO_3$ within 2cm of biomat in unsaturated soil.
(Starr and Sawhney 1980)	Monitored movement of N and C from SAS in a coarse sand.	Rainfall variability influenced NH_4 / NO_3 speciation. NH_4 at depths >90cm in wet periods (anaerobic soil).
(Kristiansen 1981b)	Pilot sand filters tested N removal and biomat zone influence.	Low N loss due to lack of labile C and anaerobic conditions.
(Whelan and Barrow 1984a, 1984b)	Movement of effluent in 7 SAS investigated. Measured nutrients in GW[A] in sandy soil	Organic N accumulated in biomat zone. NO_3 mobile in GW. Poor P adsorption in sandy subsoil.
(Stewart and Reneau 1988)	Fluctuating GW monitored near SAS in sandy loam.	>99% P removal. Denitrification observed during rising WT[B].
(Cogger 1988)	Effects of STE loading rate and WT depth on treatment in sandy soil. Examined shallow and deep wells.	Saturated, anaerobic conditions ↑ NH_4 in GW (30cm). Unsaturated conditions N removal high due to aerobic then anaerobic (GW) pathway of effluent - aiding denitrification.
(Pell and Nyberg 1989a, 1989b)	Measured nutrient and organic matter removal rates in newly operating sand filter and columns.	83% P removal in~75cm unsaturated sand. 91% COD[C] removal in 15cm unsaturated sand. Total nitrification in <15cm of unsaturated sand
(Gerritse *et al.* 1995a)	GW monitoring near SAS in sandy soil. Bromide tracer used. One system studied.	80% of inorganic N from STE removed within 10m. Fluctuating WT promoted denitrification. P movement to GW slow.
(Geary 2004)	Instrumented SAS using piezometers and suction lysimeters to monitor GW. Tracer used to determine GW flow pathways.	Found low N removal rates near SAS but substantial reduction 75 to 3.8mg/L of nitrate after flow through riparian strip.
(Bedessem *et al.* 2005)	Soil columns with organic layers to ↑ N removal via denitrification	Average total N removal ↑ from 31% to 67% with organic layer present
Pathogens		
(Bouma *et al.* 1972)	Major research effort including instrumentation of 19 SAS for hydraulic and purification monitoring in a range of soil types.	3-log reduction of pathogens within 38cm below trench. Largest reduction in biomat zone.
(Magdoff *et al.* 1974b)	Laboratory columns. Layered soils (representing mounds).	Complete removal of FC[D] and FS[E] within 90cm of sand underlain by silt.
(Kristiansen 1981c)	Pilot sand filters tested bacterial removal and biomat zone influence.	2 to 4-log FC reduction. Clogging significant in FC removal.
(Stewart and Reneau 1988)	Fluctuating GW monitored near SAS in sandy loam.	5-log reduction in FC <10m distance and 1.5m depth to SAS
(Cogger 1988)	Effects of STE loading rate and WT depth on treatment in sandy soil.	5-log FC; 5-log FS ↓ in 60cm of unsaturated soil
(Alhajjar *et al.* 1988)	Down gradient GW monitored from 17 SAS. Newly developed biomat zone reported to be effective in filtering bacteria but not virus	4-log FC; 3-log FS ↓ in <1m depth, <6m distance of SAS. Poliovirus was not effectively removed.
(Postma *et al.* 1992)	GW monitoring from 3 seasonally-used SAS (gravely fill material). Biomat zone not well developed in these systems. No 'control' (ie permanent SAS) used, so site factors cannot be discounted in results.	Estimated 4-log FC ↓ But 1.5m of unsaturated sand still high. Absence of proper biomat limited adequate removal. *Clostridium perfringens* poorly inactivated in unsaturated zone.
(Stevik *et al.* 1999)	Column experiments investigating E. *coli* removal in filter media.	Highest removal in upper column – dosing rate important.
(van Cuyk *et al.* 2001b)	3-D lysimeter studies in medium sand.	>90% removal of FC within 30cm of unsaturated sand.
(van Cuyk *et al.* 2004)	Field and laboratory studies of microbial purification in effluent.	2 to 3-log reduction in bacteriophages within 30-60cm unsaturated depth (field). Nearly complete removal FC.

[A]GW - groundwater, [B]WT - water table, [C]COD – chemical oxygen demand, , [D]FC - faecal coliforms; [E]FS - faecal streptococci

Table 3 Summary of conclusions from various studies investigating contamination of waters from onsite systems

Reference	Summary of investigation	Hydraulic link to SAS established[A]	Reported SAS-related impacts[B]
Nutrients			
(Walker *et al.* 1973b)	Wisconsin, USA. GW sampling in loamy sand. Elevated nitrate (NO_3^-) levels in GW. NO_3^- concentrations >10mg/L 30m down gradient from SAS	No tracers but NO_3^- levels suggests a strong link. No control data.	No. Authors suggest that nitrate from SAS minor compared with other land uses.
(Jones and Lee 1979)	Wisconsin, USA. 4 yr GW monitoring. No evidence for PO_4 movement from SAS in sandy soil. Nutrients in SW from GW low.	Yes. Used EC and chloride (Cl^-) data as indicator (tracer)	No. Authors suggest that SAS nutrients minor compared with other land uses
(Katz *et al.* 1980)	Compared GW NO_3^- in sewered and non-sewered areas. No significant differences found in 25 yrs of data.	No, other land uses also contributing to nitrate loads throughout period of study	No. Authors suggest a range of land uses would have contributed to nitrate in GW
(Gilliom and Patmont 1983)	Puget Sound, USA. 23 monitoring wells downstream of 8 SAS adjacent to Lake Pine. Movement of >1% of STE P rare.	Yes, used Cl^- to infer SAS link.	Nothing reported in study. Later studies have determined lake is eutrophic (Jacoby *et al.* 1997).
(Valiela *et al.* 1988)	Buttermilk Bay, USA. Calculated N, P loadings from SAS into bay and interception rates by denitrification etc	Water sampling and modelling suggests link.	Insufficient data to be certain, as high nutrient interception rates within catchment.
(Gibbs 1991)	Lake Taupo, New Zealand. GW monitoring in retro-sewered town. Possible storage of N in unsaturated zone, N processes not discussed.	No. Nutrient concentrations in GW declining since town sewered.	Later studies indicate agricultural land use to be severe WQ impact on lake (Edgar 1999).
(Robertson *et al.* 1991)	Ontario, Canada. GW monitoring of 2 SAS. NO_3 mobile in GW but good denitrification in enriched anaerobic sediments.	Yes, bromide (Br^-) tracers used.	Study did not indicate previous or existing eutrophication in region.
(Weiskel and Howes 1992)	Buttermilk Bay, USA. GW monitoring of nutrient movement from 2 SAS and 2 cesspools.	WQ monitoring suggests link. Inorganic N higher in GW than P	No. Authors suggest that P retention in groundwater is effective and that SAS "contribute little to eutrophication".
(Lapointe *et al.* 1990; Lapointe and Clark 1992)	Florida Keys, USA. 30 water sampling stations, twice/yr. High P and N attributed to OWTS. Sewage outfall located in area.	No. No tracers used. Specific SAS link undetermined.	Yes eutrophication studied in area. OWTS considered part of overall contribution but no estimated loads from non-sewered areas.
(Paul *et al.* 1995)	Florida Keys, USA. Tracers used to seed septic tanks and injection wells. Found tracers in septic tank moved to marine waters in <24 hrs. Tidal movement a factor.	Yes with use of tracers.	Yes eutrophication studied in area. OWTS considered part of overall contribution but no estimated loads from non-sewered areas.
(Gerriste *et al.* 1995b)	GW monitoring near SAS in sandy soil. High N reduction. Fluctuating WT promoted denitrification.	Yes, Br^- tracer used..	Study did not indicate any reported impacts. Agricultural uses ↑ N in GW.
(Harman *et al.* 1996)	Ontario, Canada, GW monitoring below 1.6m vadose zone under SAS. High NO_3^-. Slow PO_4 release.	Yes, Br^- tracers.	Study did not indicate previous or existing eutrophication in region.
(Dillon *et al.* 1999; Dillon *et al.* 2000)	Florida Keys, USA. Tracers in sewage injection well. Found rapid horizontal & vertical plume path. Dilution up to 7x by GW and SW.	No. Injection well plume was studied only. No soil-based systems studied.	Yes eutrophication studied in area. OWTS considered part of overall contribution but no estimated loads from non-sewered areas.
(Moore *et al.* 2003)	Seattle, USA. Compared eutrophication in lakes draining developed and undeveloped land. Non-sewered lakes had significantly higher P and chlorophyll-*a* than undeveloped lakes but less P than sewered lakes.	WQ monitoring suggests link.	Non-sewered lakes believed to be more eutrophic than sewered or undeveloped. No N measured.
(Steffy and Kilham 2004)	Penns., USA. $\delta^{15}N$ stable-isotope analysis to detect sewage - derived N. Compared sewered and non-sewered areas.	Not directly though $\delta^{15}N$ used as an indicator of anthropogenic N inputs	Non-sewered areas had elevated $\delta^{15}N$ in food web compared with sewered areas.

Pathogens			
(Reneau and Pettry 1975)	Virginia, USA. 3 SAS instrumented. Few FC observed at depth. Concluded GW not contaminated from vertical movement of FC.	Yes - monitoring above and below drainfield.	Study did not discuss / identify previous or existing waterborne-diseases in region.
(Reneau *et al.* 1975)	SW and GW bacteriological sampling. Variation in results. Surcharging observed in unsuitable soils.	No tracers but WQ monitoring suggests a strong link.	No. Area sewered during study.
(Rahe *et al.* 1978)	Oregon, USA. E. coli movement in simulated saturated conditions. Tracer injected into lines in soil trenches.	Unsure, operating SAS not studied only constructed soil ‘trenches’.	N/A.
(Vaughn *et al.* 1983)	Long Island, USA. Viruses detected 67m and 18m depths from large septic tank-leaching pool system in sandy GW.	Unsure. Leaching ‘pools’ not soil-based absorption system used.	Unsure. Ongoing studies carried out on WQ in Long Island Sound.
(Alhajjar *et al.* 1988)	Wisconsin, USA. Down gradient GW monitored from 17 SAS. No tracer bacteria but poliovirus detected in GW	Yes, tracers (poliovirus and Cl^-) used	No although later studies in region have suggested illness links to contaminated wells (see Borchardt *et al* 2003 below)
(Postma *et al.* 1992)	Rhode Island, USA. GW monitored from 3 seasonally-used SAS. Poor FC and nutrient removal	Yes, NO_3^- as tracer and WQ monitoring suggests a strong link.	Study did not identify previous or existing waterborne-diseases in region.
(Hoxley and Dudding 1994)	Victoria, Australia. GW monitoring in 2 non-sewered towns. NO_3^- and E. *coli* reported in GW in Benalla. E.*coli* reported in GW in Venus Bay. No data given.	No. GW monitoring suggests a possible link.	Study did not identify previous or existing waterborne-diseases in region.
(DeBorde *et al.* 1998)	Montana, USA. Instrumentation of school SAS and GW monitoring for viruses. Low levels detected only.	Yes Br^- and coliphage tracers used.	Study did not identify previous or existing waterborne-diseases in region.
(Whitehead and Geary 2000)	Tasmania, Australia. GW monitoring non-sewered area. High nitrate concentrations measured in areas of residential clusters. FC almost undetectable at all sites.	Yes, NO_3^- as tracer and WQ monitoring suggests a strongly link. Limited sample size.	Study did not identify previous or existing waterborne-diseases in region.
(Frenzel and Couvillion 2002)	Alaska, USA. SW samples from areas of varying development density. Sewered areas had significantly higher faecal-indicator bacteria than non-sewered areas	No. SW data compared for different catchment land uses. Storm sewers linked to high faecal-indicator bacteria.	Study did not identify previous or existing waterborne-diseases in region.
(Bopp *et al.* 2003)	New York, USA. Investigation of largest reported outbreak of waterborne *E. coli*. GW supply located near school SAS. *E.coli* source may be animal origin.	Yes, dye seepage tests used and link detected in one well.	Yes. Largest reported outbreak of waterborne *E. coli* in USA. Direct cause by SAS not fully determined but likely.
(Borchardt *et al.* 2003)	Wisconsin, USA. Assessed association between septic system density and illness in children. Viral diarrhoea associated mainly with holding tanks not SAS	Unknown. Well water quality not actually tested in non-sewered areas that were studied.	Unknown, although authors suggest 11% of unexplained acute illness caused by ingesting faecal - contaminated well water
(Carroll *et al.* 2004)	Brisbane, Australia. GW and SW monitoring in high density non-sewered area. Elevated NO_3^- and FC. Agricultural practices may contribute to NO_3^-.	No. Found increase in nutrient concentrations in GW directly below township.	Study did not identify previous or existing waterborne-diseases in region. GW not a drinking water source.

[A] A hydraulic link / pathway / connection has been established between the onsite system investigated and contaminant(s) investigated in receiving ground / surface waters.

[B] Environmental and / or public health impact associated with wastewater contaminants reported in the area of investigation.

Table 4. Summary of some reported minimum sustainable lot sizes and OWTS densities

Lot size (m^2)	System density (per km^2)	Comments	Reference
65,000	15	Septic system density considered potential for ground water contamination	(USEPA 1977)
2000 – 4000	500 - 250	Simple modelling showed increased N concentration below SAS as lot size decreased	(Perkins 1984)
1000 – 12,000	1000 - 85	Range of densities where ground water contamination had been reported	(Yates 1985)
10,000	100	Example density for NSW coastal town based on minimum assimilative buffer area for pollutants (N, P, pathogens)	(Jelliffe 1999)
2000 - 4000	500 - 250	Based on average nutrient and hydraulic loadings from OWTS and minimum setback distances for pathogens	(Geary and Gardner 1998)
50,000 – 100,000	10-20	Recommended in environmentally sensitive areas based on N and P assimilation	(Gerritse 2002)

Table 5. Summary of some key areas of further OWTS research

Site-scale	Catchment-scale
Effects of pressure dosing on LTAR for in Australian soils	Risk-based decision models e.g further validation of OSRAS, DAM
Oxygen dynamics under biomat zone	Quantifying contaminant loads from on-site systems
Denitrification in SAS and technologies for removal of N	Sustainable on-site system densities and separation distances
Treatment capacities of a greater range of soil types in Australia, e.g . colder climates	Monitoring sewered v non-sewered catchments – looking for the “smoking gun”
Effects of pre-treated effluent and innovative soil-based designs	Role of riparian zone and fluctuating water tables in nitrate removal and P sorption in sediments
Predicting biomat zone development and its role in long term effluent treatment / hydraulics	Use of tracers to identify contaminant pathways in surface and groundwaters

Fig. 1. Schematic representation of system layout and main processes (a); plan and cross section view of the trenches in a soil absorption system (b & c).

Fig. 2 A commonly observed schematic pattern of wastewater infiltration rate decline through soil infiltrative surface over time, adapted from Otis (1984).

Fig. 3. Definition diagram of Equation 2. θu is the volumetric water content of the underlying soil (cm^3/cm^3), θs is the saturated volumetric water content of the underlying soil (cm^3/cm^3), z is soil depth (cm), and Ψu is the soil moisture potential of the underlying soil

Fig. 4. Predicted effect of increasing R_b on steady-state flow rates for a range of Australian soils (Beal et al. 2004a) (Note: X axis not to scale)

Fig. 5. Cross section of trench with increasingly resistant biomat zone (top) resulting in increasingly unsaturated flow (below).