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THE RISK OF GROUNDWATER POLLUTION BY ON-SITE SANITATION IN DEVELOPING COUNTRIES

A Literature Review

W. John Lewis
Stephen S.D. Foster
Bohumil S. Drasar

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Foreword

The development of groundwater resources for potable use has increased substantially over the last two decades. Many countries now rely on groundwater sources to supply a large fraction of their domestic demand for potable water. Still, in many developing countries a substantial portion of the population do not have access to an adequate water supply. Safe excreta facilities are even less common. In 1977 at the Mar Del Plata Conference, UN member nations proposed to make substantial efforts to improve the availability of adequate water supply and safe excreta facilities to all communities. Most governments have endorsed these UN proposals, and are intending to make further considerable investments in water supply and sanitation over the next decade to improve public health and wellbeing.

To provide adequate water supplies and sanitation facilities for developing countries will necessitate:

- increased use of groundwater;
- major construction programmes utilising on-site sanitation systems (particularly pour flush latrines and ventilated improved pit latrines).

The two solutions to the population's need may conflict, particularly under certain hydrogeological conditions. Without an integrated approach involving personnel in the fields of water engineering, hydrogeology, public health and sanitation, and without adequate design and construction, the extensive use of unsewered disposal systems may cause severe groundwater contamination (pathogenic micro-organisms and biodegradation products of human excreta, such as nitrates). This may expose people to the risk of disease, and thus reduce the anticipated health benefits of providing water supply and sanitation facilities.

In view of the proposed substantial investments in on-site sanitation, for both urban and rural areas, it is essential that the relationship between groundwater quality and on-site sanitation is investigated to ensure that improved sanitation does not cause excessive soil and groundwater pollution. An International Working Group concerned with aspects of environmental pollution due to low-cost sanitation proposed in 1980 that rigorous long term studies should be undertaken into the relationship between environmental pollution and on-site sanitation in different soils and hydrogeological conditions. As a first step the International Reference Centre for Waste Disposal (IRCWD) in collaboration with the World Bank Technology Advisory Group (TAG) conducted a study including a review and analysis of the existing literature and the preparation of a document on Groundwater Quality Monitoring Methodology.

This publication is the outcome of the literature review and was prepared by IRCWD consultants: W.J. LEWIS, S.S.D. FOSTER, Institute of Geological Sciences in Wallingford, England, and B. DRASER, Ross Institute of Tropical Hygiene in London. Final editing was done by D. Stuckey, IRCWD.

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Roland Schertenleib
Manager IRCWD

CHAPTER 1 : GENERAL INTRODUCTION

1.1 PURPOSE AND SCOPE OF LITERATURE REVIEW

Insufficient information is currently available on the relationship between groundwater quality and on-site sanitation. However, a considerable body of data exists on environmental pollution, pathogen dieaway, and the movement of anions, cations and pathogens in groundwater. In view of the proposed substantial investments in on-site sanitation, for both urban and rural areas, it is essential that this relationship is investigated to ensure that the improved sanitation does not cause excessive soil and groundwater pollution.

The purpose of this report is to:

(a) review applied field investigations of pollution from on-site sanitation;

(b) review the literature related to the survival and movement of micro-organisms and of certain inorganics (chiefly nitrates), through the unsaturated and saturated zones;

hence,

(c) identify factors which:

i) affect the movement of contaminants;

ii) can be used to assess the pollution risk to groundwater in the various hydrogeological environments likely to be encountered;

iii) require further research, with the aim of improving guidelines for risk assessment;

and,

(d) suggest methods of preventing or minimising groundwater pollution.

The literature reviewed in this report can be classified broadly into four categories:

(a) experimental field investigations on pollutant dispersion from on-site sanitation;

(b) laboratory investigations of the unsaturated zone;

(c) case histories of groundwater pollution;

(d) related studies, for example, the capacity of soils to remove pathogenic bacteria and viruses when used for land disposal of sewage effluent, and studies on bacterial survival in surface and groundwaters.

Most of the work reviewed has been published in the United States, and relates to disposal of septic tank effluent. An authoritative source of information on this topic is a US-EPA report (Kreissl, 1978), which is a comprehensive compilation of 6 years of laboratory and field investigations conducted at the University of Wisconsin.

There are few reported case histories of groundwater pollution in developing countries resulting from the use of on-site sanitation. This is due to a lack of research and reporting. Most of the literature emanates from the developed countries. Although the health and water supply problems in the developing countries are substantial, there is only limited funding available for investigation of groundwater pollution problems.

Although waterborne sewerage increases environmental health and is convenient for consumers, it is evident that conventional sewerage is not a realistic option for either urban or rural poor in developing countries (Kalbermatten et al., 1980). If targets for the UN Drinking Water Supply and Sanitation Decade are to be met, this would mean satisfying the basic needs of 900 million people in urban areas and 1,600 million people in rural areas by 1990 (World Bank, 1980),

1.2 SANITATION OPTIONS

The various options for excreta disposal using low cost technologies were reviewed and discussed in detail by several workers (Rybczynski et al., 1978; Kalbermatten et al., 1980). This report is mainly concerned with the risk of groundwater pollution from pit latrines and pour flush latrines.

1.2.1 *Ventilated improved pit latrines* (VIP)

The ventilated improved pit latrine overcomes the principal disadvantage of simple (unimproved) pit latrines - namely that they smell and create a serious fly nuisance (see Figure 1.1). The single pits are usually deep to permit a long operational life (up to 10 or more years) before they require desludging. Twin pit latrines are designed so that one pit is used until full, is then sealed and the second pit used. When this is full, the first is emptied, by which time pathogens should have died off, and the pit contents should be inoffensive to handle. Periodic emptying also keeps the infiltration surface more viable.

Twin pits also reduce the pollution hazard since:

(a) the potential for nitrate contamination is reduced because the nitrogenous matter is removed periodically;

(b) the pits are shallower and hence penetrate less into the unsaturated zone, thereby enhancing purification of the percolating effluent before it reaches the groundwater.

A typical fluid loading to a VIP is 7-13 l/d which is based on faecal fluids of 5 to 10 people daily[1]. Where water is used for anal cleansing (say 2 l/c/d), the loading would increase to 17-33 l/d. A typical base area of the pit would be approximately 0.8 m^2.

[1] See Feachem et al. (in press) for approximate quantity and composition of human excreta

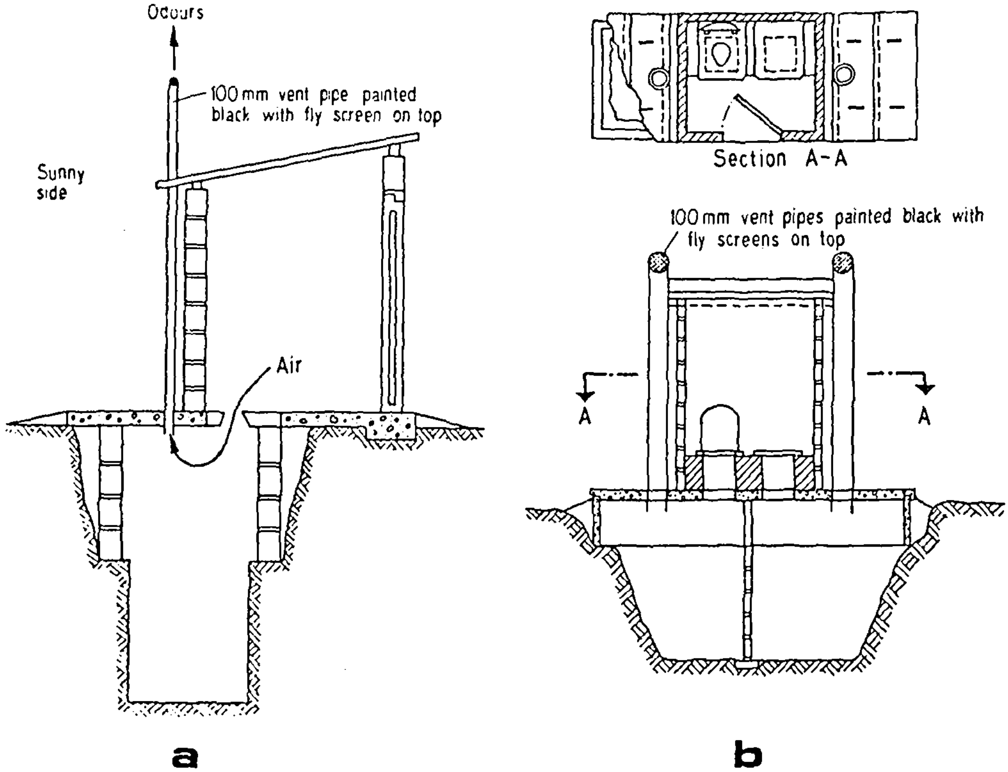


Fig. 1.1 Ventilated improved pit latrine (VIP): (a) single pit, and (b) twin pit version (Mara and Feacham, 1980)

1.2.2 *Pour flush latrines* (PF)

Pour flush latrines or toilets are very common in the Indian subcontinent and the Far East (Figure 1.2). They have 3 main advantages: low water requirements of 1-3 litres/flush as opposed to 9-20 litres/flush for most cistern toilets; complete odor elimination by the shallow water seal; also they can, if desired, be located inside the house (Mara and Feachem, 1980). They are particularly suited wherever water is used for anal cleansing. As in the case of the VIP latrine, the twin pit configuration is preferred.

A typical fluid loading to a pour flush latrine is 45-95 l/d. This includes faecal fluids (1.3 l), ablution water (2 l) and flush water (6 l) for 5 to 10 people daily. The base area of the soakaway is usually 0.8 m^2. Although designed primarily for faecal fluids, additional wastewater may be discharged to the latrine; this increases the hydraulic loading, and hence the likelihood of groundwater pollution.

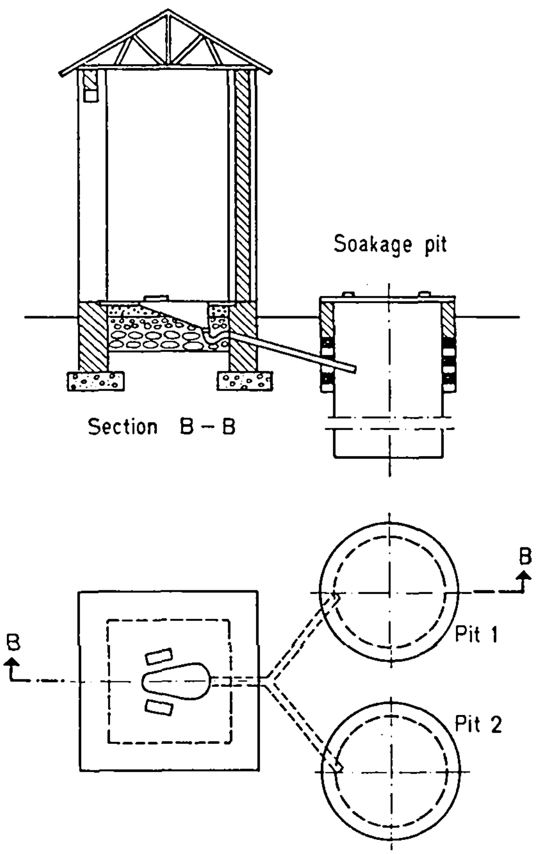


Fig. 1.2 Pour flush latrine (Mara and Feachem, 1980)

1.3. HEALTH HAZARDS OF CONTAMINATED GROUNDWATER

Diseases related to the use of contaminated groundwater may be divided into those which are caused by a biological agent (a pathogen), and those which are caused by chemical substances. However, these latter diseases are overshadowed in developing countries by the former which are the greatest cause of disease and death.

1.3.1 *Pathogen transmission*

Human excreta may contain four types of pathogens: eggs of helminths (worms), protozoa, bacteria and viruses. These may be excreted in vast numbers depending on the age and state of health of the individual. Faecal matter contains on average 10^9 bacteria/g (not necessarily pathogenic), and in the excreta of infected individuals as high as 10^6 viruses/g. Viruses differ fundamentally from other micro-organisms which occur in water. They consist of a nucleic acid core enclosed in a protective protein coat, and are transmitted as inert particles that are unable to replicate outside a living host, therefore in the environment their numbers decrease. These particles, or virons, have the potential to cause disease in people who ingest them with drinking water. A viral particle eventually loses its infectivity with time, and with exposure to the rigours of its environment (National Academy of Sciences, 1977).

Over 100 different types of viruses have been isolated from faecal material. These are referred to as enteric viruses, and include the true enteroviruses (polio-, echo- and coxsackieviruses), reoviruses, adenoviruses and rotaviruses as well as the agent of infectious hepatitis. Resistance to inactivation varies considerably between different types of viruses, and even different strains of the same type. The rate of inactivation depends on both the efficiency of removal and the numbers initially present. Bacteria and viruses may be transported with percolating effluent into the groundwater, and these organisms may be ingested causing infection. However, excreted viruses and bacteria are transmitted by many other routes, such as contaminated food, fingers or flies. Whether or not an individual will become infected will depend on the concentration and persistence of the pathogen in groundwater and the infectious dose required to initiate disease (see Feachem et al., in press). In general, excreted viruses have low infectious doses (< 100 organisms) whereas the median infectious dose for bacteria is typically 10,000 or more. Bacteria, however, unlike viruses, are able to multiply outside their host.

The diseases and their agents which might be spread by faecally contaminated groundwater include:

a) Viral diseases:	Pathogen
Infectious hepatitis	Hepatitis A virus
Poliomyelitis	Poliovirus
Diarrhoeal diseases	Rotavirus, Norwalk agent, other viruses
Varied symptoms and diseases	Echoviruses and Coxsackievirus

b) Bacterial diseases:	Pathogen
Cholera	*Vibrio cholerae*
Typhoid fever	*Salmonella typhi*
Paratyphoid fever	*Salmonella paratyphi*
Bacilliary dysentery	*Shigella* spp.
Diarrhoeal diseases	Enterotoxigenic *E. coli*
	Enteroinvasive *E. coli*
	Enteropathogenic *E. coli*
	Salmonella spp.
	Campylobacter petus ssp. *jejuni*

1.3.2 *Nitrate-related diseases*

The extensive use of on-site sanitation may lead to elevated concentrations of nitrate in the underlying groundwater. Two diseases have been associated with consumption of water containing high nitrate concentrations. These are:

(a) Methaemoglobinaemia (infantile cyanosis)

This is a disease primarily affecting young infants. In 1977 a WHO European Working Group on health hazards from drinking water proposed the adoption of 11.3 mg NO_3^--N/l[1)] (50 mg NO_3^-/l) as the maximum acceptable concentration for infants, and of 22.6 mg NO_3^--N/l (100 mg NO_3^-/l) as the maximum for the population as a whole (WHO, 1977), to safeguard against this disease. These recommendations are based largely on an analysis of relatively few reported cases. The potential health implications for infants ingesting excessive quantities of nitrate is a topic of continuing medical attention (Windle-Taylor, 1974; Shuval and Greuner, 1977; WHO, Geneva, 1978; Fraser and Chilvers, 1981). The acute toxicity of nitrate occurs as a result of its reduction to nitrite, a process that can occur under specific conditions in the stomach and saliva. The nitrite ion formed oxidizes iron in the haemoglobin molecule from ferrous (Fe^{2+}) to ferric (Fe^{3+}). The resultant methaemoglobin is incapable of reversibly binding oxygen, and consequently anoxia or death may ensue if the condition is left untreated.

(b) Carcinogenesis

Recently there has been increasing interest in the cancer risk associated with elevated quantities of nitrates in drinking water. Nitrites (and indirectly nitrates) can react with amines and amides to form nitrosamines and nitrosamides. Most N-nitroso compounds tested have proved to be carcinogenic in a wide range of animal species, and most were mutagenic. The epidemiological evidence suggests that high nitrate ingestion may be a contributing factor in gastric cancer. However, at present there is too little information available to draw any specific conclusions about the relationship between high nitrate ingestion and any other human cancer (Fraser et al., 1980).

[1)] Nitrate levels are commonly expressed either as nitrate or as the concentration of nitrogen in that nitrate. The former convention is followed throughout in this paper.

1.4 BACTERIAL INDICATORS OF FAECAL POLLUTION

Since the discovery that pathogenic organisms transmitted through contaminated drinking water caused various intestinal diseases, a great deal of effort has gone into developing bacteriological tests that indicate the presence of faecal contamination in water. It would be desirable to be able to test directly for pathogenic micro-organisms, however, because of the variable concentrations of pathogenic bacteria in faeces, it is not feasible to test for all disease-producing organisms. A more practical approach is to test for a particular group of bacteria common to the faeces of all warm-blooded animals, and hence can be used as an indicator of faecal pollution.

The characteristics required of an ideal indicator organism (Geldreich, 1978; Feachem et al., in press) restrict the selection to total coliforms, faecal coliforms, faecal streptococci, which are all aerobic bacteria, and to the anaerobic bacteria Clostridium perfringens, bacteroides and lactobacilli. The coliform group of bacteria has shown the most promise as an indicator organism.

Normal bacterial indicators do not always fulfill all the ideal characteristics, particularly in acute cases of diarrhoeal disorders, where pathogens may predominate. The indicator organisms currently employed serve only to indicate faecal pollution, and cannot be taken as a measure of the degree of faecal pollution or of the presence of pathogenic micro-organisms. Generally, if it is shown that faecal contamination of water has occurred, then pathogens may also be present.

However, values for faecal coliform counts in tropical waters should be interpreted with caution, since the standard enumeration procedures were developed in Europe and North America where the climates are temperate. Various studies in the tropics have detected a considerable proportion of coliforms of probable non-faecal origin which have the ability to ferment lactose at 44.5^{o}C (Feachem et al., in press), and recent tests in Gambia (Barrell and Rowland, 1979) have given high false positive results (up to 55 %).

CHAPTER 2 : PRINCIPLES OF POLLUTANT MOVEMENT AND ATTENUATION IN THE GROUND

2.1 PREAMBLE

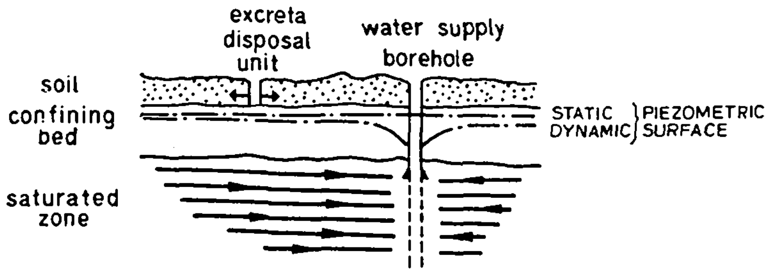
The natural soil profile has long been recognised as effective system for the disposal and purification of human wastes. The purification process normally includes the removal of faecal micro-organisms, and the breakdown of many chemical compounds. However, not all soil profiles are equally effective for this process. The term "soil" is used here in the engineering sense to mean sediment or unconsolidated strata.

Improper design, construction, operation or maintenance of on-site sanitation systems can lead to failure due to the loss of infiltration capacity, with consequent surfacing of effluent. Such failures are obvious, and are quite frequently reported. However, an equally important and more insidious failure is that of inadequate effluent purification. This can occur in certain hydrogeological environments and may result in serious pollution of groundwater, and thus to local water-supply, and under certain conditions, water reticulation mains subject to intermittent de-pressurisation.

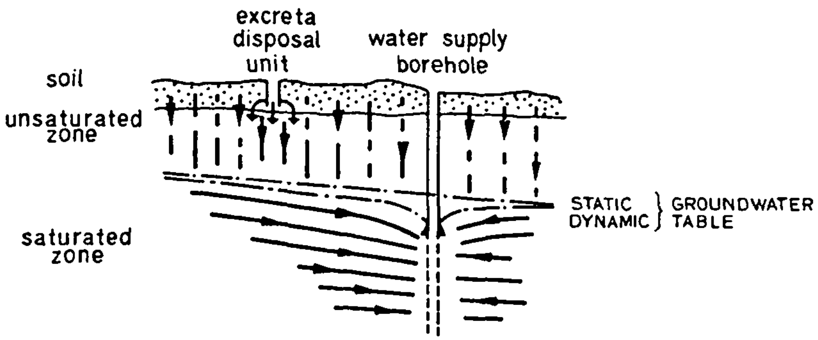
It is important to identify the main hydrogeological factors involved in the pollution of groundwater, and hence establish a classification of hydrogeological environments which can be used to evaluate on-site sanitation schemes. The assessment of pollution risk may be complicated by the intermittent ingress of pollutants from the land surface, since protection of groundwater-supply installations may not be effective.

Concern about groundwater pollution relates primarily to unconfined and, to a lesser degree, to semi-confined aquifers (Figure 2.1). Where groundwater supplies are drawn from deep and confined aquifers, unsewered sanitation is not a problem providing that the water-supply boreholes and wells are constructed to prevent the ingress of surface run-off or shallow groundwater and that reticulation mains are not subject to regular depressurisation.

The primary cause for concern are the excreted pathogens. Of secondary concern are certain chemical constituents (principally nitrate), and this will be discussed later. The relatively large size of helminths and protozoa ($> 25\mu$) results in highly efficient removal by physical filtration in soils. It is unlikely that they would pollute groundwater, and therefore will not be considered further. Bacteria and viruses are very much smaller (see Figure 2.2), and may be transported with the effluent percolating from on-site sanitation systems into the groundwater.



a



b

Fig. 2.1 Schematic section of (a) confined and (b) unconfined aquifer

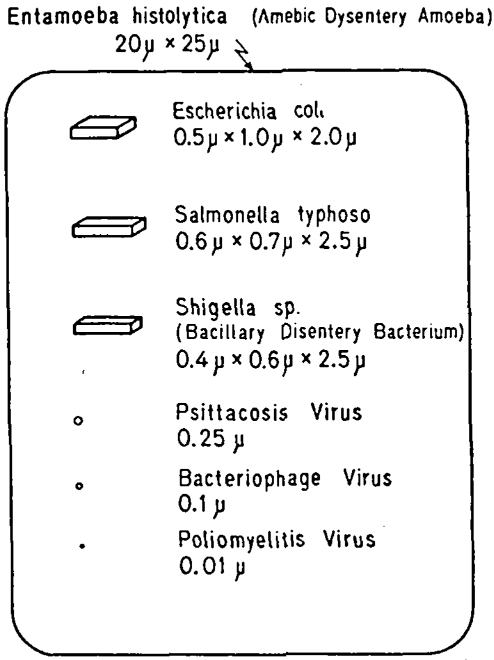


Fig. 2.2 Comparative size of selected micro-organisms (McNabb et al., 1977)

The performance of most on-site sanitation systems depends primarily upon the ability of the soils of the unsaturated zone (Figure 2.1) to accept and purify effluent; functions which may be in conflict under certain site conditions. Both functions relate, directly or indirectly, to the regime of groundwater movement which, in turn, is largely controlled by the hydraulic characteristics of the soil. It will be apparent from the ensuing sections that these hydraulic characteristics will determine the moisture content, flow path and residence time of pollutants.

2.2 GROUNDWATER MOVEMENT IN THE UNSATURATED ZONE

2.2.1 *Controlling factors*

The unsaturated zone is a complex arrangement of solid particles and pore space filled with ever-changing amounts of air and water. Water moves from points of higher to lower potential energy, the energy status of the water being referred to as moisture potential (Ψ). In the unsaturated zone, two types of moisture potential predominate - the gravitational potential and matrix potential; groundwater flow is normally, though not of necessity vertical.

Matrix potential is a consequence of the affinity of water for solid surfaces, resulting from the molecular forces of cohesion and adhesion, together with adsorption in drier clay soils. Capillary rise in glass tubes is a useful illustration of such forces. The phenomenon may operate to move water in any direction. Since the water is "sucked" into the tube, and is at less than atmospheric pressure, it is under tension. The matrix potential, or corresponding negative pressure in soil is, therefore, sometimes called soil tension or soil suction.

When the soil or rock is saturated, all the pores are filled with water, and at the groundwater table the matrix potential is zero. Drainage or drying is associated with increasing matrix potential, and progressively smaller pores empty because they do not have sufficient tension. The rate of decrease of moisture content with increasing tension is a function of the pore-size distribution, a fundamental characteristic of any given soil or rock (Figure 2.3). For example, sandy soils and certain sandstones have some relatively large pores that drain at quite low tensions, whereas clays, silts and silt-stones experience little drainage and have relatively large moisture contents over a wide range of tensions because most of their water is strongly retained in very fine pores. Unfortunately, the pore-size distribution is a difficult and costly characteristic to determine and, in practice, descriptive soil classifications have had to be based on the grain-size distribution.

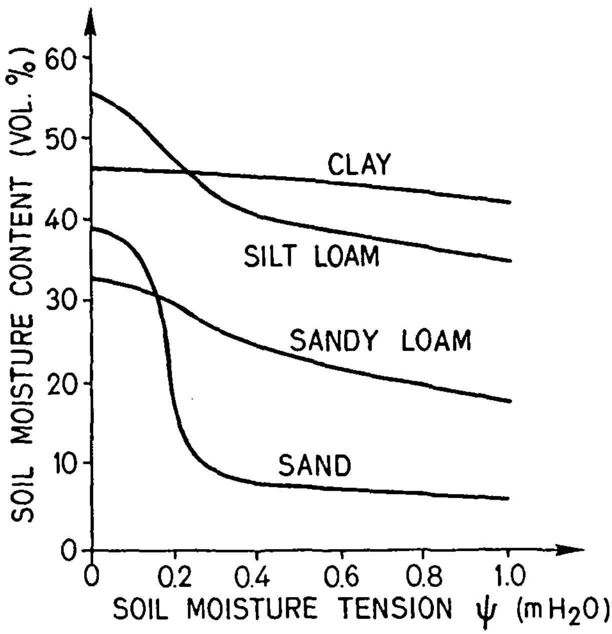


Fig. 2.3 Soil moisture retention curves for four different soil materials (Bouma et al., 1972)

In sands and sandy soils the pore-space is represented by the voids between individual grains, except in the rare instance when the grains themselves are significantly porous. Where significant amounts of clay and/or organic materials are present, aggregation of soil particles can occur, with the formation of planar voids. Plant roots may promote or accentuate this process. Since their aperture is normally relatively large compared to the size of the intergranular pores, such voids will only contain water at low tensions. At shallow depth all rocks contain comparable sub-planar voids, known as joints and fractures, or collectively as fissures.

The unsaturated vertical hydraulic conductivity, and thus the rate of unsaturated groundwater flow, is a complex function of the size, tortuosity and continuity of the pores and fissures. The hydraulic conductivity often changes dramatically with changes in the soil tension. At tensions approaching zero, the soil is saturated and all the pores conduct liquid, at higher tensions air is present in some of the pores and an unsaturated condition prevails. As the water content decreases (or tension increases), the flow path becomes more and more tortuous since water travels along surfaces and through pores sufficiently small to retain water at the prevailing moisture tension.

The hydraulic conductivity/soil tension relation is characteristic of a given soil texture and structure (Figure 2.4). Coarse soils with predominantly large pores have relatively high saturated hydraulic conductivity which reduces rapidly with increasing soil tension. Fine soils with predominantly small pores have relatively low saturated hydraulic conductivity decreasing more slowly with increasing tension.

Groundwater flows in unsaturated soils normally do not exceed 0.3 m/d. An important exception may occur, however, in fissured rocks and coarse gravels. At moisture tensions below 0.2 m the larger fissures can conduct water, and the hydraulic conductivity increases enormously (see Figure 2.4). Consequently, flow rates in excess of 5 m/d may occur, and the potential for groundwater contamination under these conditions is extremely high. Similarly, there may be rapid transport of pollutants through shrinkage cracks in dessicated clay soils with rain before swelling of the clay minerals seals cracks.

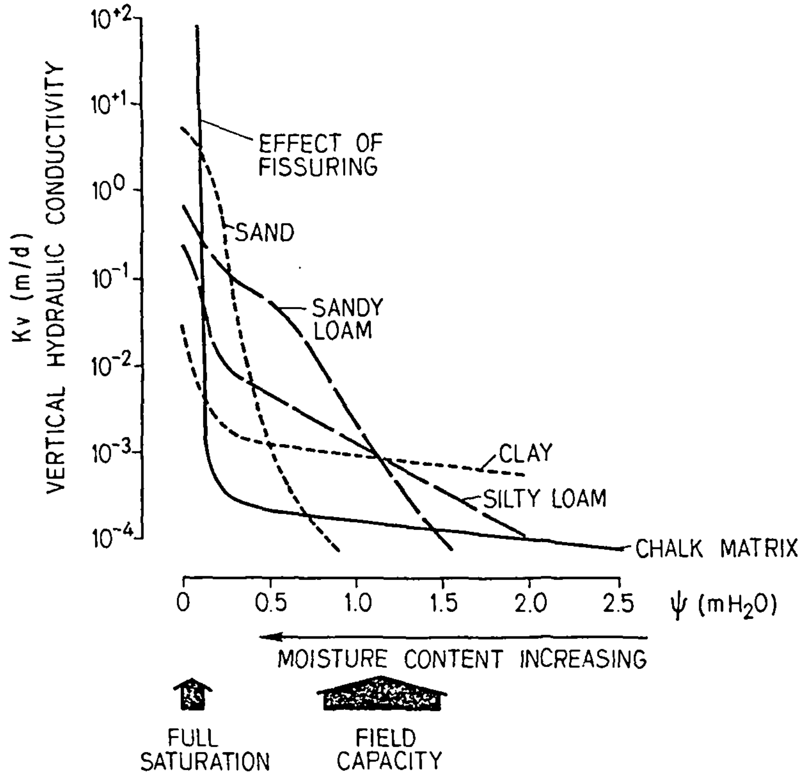


Fig. 2.4 In-situ unsaturated hydraulic conductivity as a function of moisture potential (tension)

2.2.2 *Hydraulic loading from on-site sanitation systems*

Estimation of the effective hydraulic loading on the unsaturated zone (in mm/d) associated with a design of an on-site sanitation system is not straightforward. Firstly, there are uncertainties about the actual daily effluent volume per capita, and the maximum and average number of people using an individual unit. Secondly, while the basal area of the pit is readily determined, the shape and cross-sectional area of the groundwater flow from any latrine will be complex. A particularly contentious issue is the scale of lateral movement of effluent out of the latrine (sidewall influence); this is a function of the ratio of horizontal to vertical hydraulic conductivity of the surrounding soil (over the appropriate range of saturation), and its prevailing natural moisture content, i.e. sidewall influence should be greatest in dry soils. For the purposes of this report, the hydraulic loadings have been expressed simply by dividing the probable range or daily effluent volume (assuming no sullage)[1] by the basal excavation area of the latrine concerned

[1] See sections entitled Ventilated improved pit latrines (VIP) and Pour flush latrines (PF) in Chapter 1 for estimates of daily effluent volume

(values of 25 $\pm$ 15 mm/d for "VIP"latrines to 90 $\pm$ 30 mm/d for "PF" latrines)[1].

2.2.3 *Process of pore clogging*

When effluent enters the unsaturated zone (soil), pore clogging eventually develops at the infiltration surface. Such clogging may reduce the rate of infiltration, cause ponding of liquid above the clogging layer (the crust or mat), and may lead to system failure due to surfacing of effluent.

Several phenomena contribute to the process of pore clogging and these include:

(a) blockage of pores by solids filtered directly from the effluent;

(b) accumulation of biomass from the growth of micro-organisms;

(c) excretion of slimes by some bacteria;

(d) deterioration and puddling of soil structure caused by saturation and swelling of clay minerals brought about by cation exchange;

(e) precipitation of insoluble metal sulphides under anaerobic conditions.

The soil structure may also be partially destroyed by compaction and smearing during construction of the on-site sanitation system.

The rate of development of the crust depends on many factors, and is believed to develop in three stages. Initially, aerobic bacteria decompose many of the organic solids filtered from the effluent, helping to keep soil pores open. However, they can only function when the infiltration surface drains, allowing the entry of air, and eventually will be unable to keep up with the influx of solids. Permanent ponding will result, leading to anaerobic conditions, where oxygen is no longer present to allow the rapid decomposition of organic matter. Clogging therefore proceeds more quickly; the reduction of sulphate by anaerobic bacteria binds up trace elements as insoluble sulphides, causing heavy black deposits. At this stage, the crust normally reaches an equilibrium state, and its hydraulic resistance stabilises.

The process of pore clogging has been investigated extensively for North American (Wisconsin) soils in relation to septic tank drainage systems (Kreissl, 1978), and in particular for the design of soakaway trenches and selection of optimum disposal and resting intervals. It has not been adequately investigated for the types of sanitation systems under consideration in this report, which may in some cases have significantly higher hydraulic loadings than these of septic tank soakaways (40-50 mm/d) and involve removal of greater thicknesses of the soil profile.

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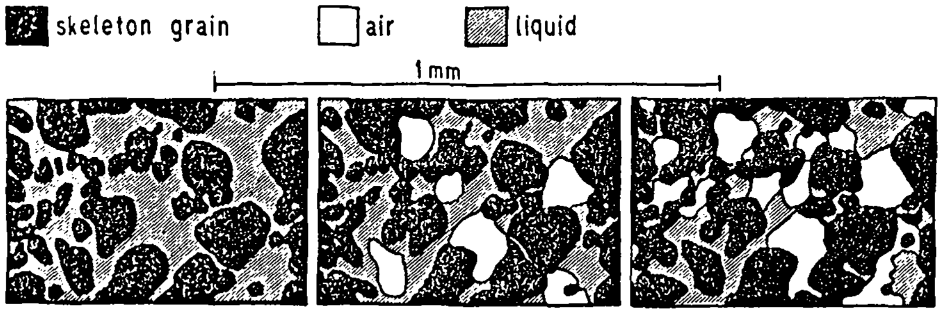
[1] An hydraulic loading of 1 mm/d is equivalent to 1 litre/m^2/d

2.2.4 *Significance of pore clogging*

Because of the partial barrier to flow created by soil clogging, the soil below the organic mat becomes unsaturated. This becomes significant when effluent is applied to the soil for disposal. Liquid flow in unsaturated soil proceeds at a much slower rate than in saturated soil because flow only occurs in the finer pores. This slows the rate of infiltration through the soil, but enhances purification. The effluent is purified by filtration, biological reactions, and adsorption processes which are more effective in unsaturated soils because of more intimate and prolonged contact between liquid and soil. This phenomenon can be illustrated by an example of a Wisconsin soil (Figure 2.5).



ψ (mH_2O)	0.02	0.30	0.80
m_c	0.33	0.29	0.19
k_v (m/d)	8×10^{-1}	7×10^{-2}	7×10^{-3}
t^* (d/m)	0.4	4.2	26.7

ψ moisture potential (tension)
m_c volumetric moisture content
k_v vertical hydraulic conductivity
t^* actual unit travel (residence) time for unity hydraulic gradient

Fig. 2.5 Groundwater movement in saturated and unsaturated sandy loam till (Bouma et al., 1972)

The flow velocity in the soil pores can be estimated knowing the percentage of liquid-filled pores at different soil tensions as given by its moisture retention curve (Figure 2.3).

$$\text{Flow velocity} = \frac{100\ \% \times \dfrac{\text{hydraulic conductivity at a given soil}}{\text{moisture tension}}}{\%\ \text{liquid-filled pores at a corresponding soil moisture tension}}$$

This velocity can be used to derive the time for effluent to travel 1 m, assuming a hydraulic gradient of 1 cm/cm (due only to gravity). For example, at saturation (zero soil tension) the sandy loam has 33 % of its volume filled with liquid (Figure 2.3) and the saturated hydraulic conductivity is 0.8 m/d (Figure 2.4). Thus the time taken for the effluent to travel 1 m is 0.41 d. Successively smaller pores empty at increasing tensions and the hydraulic conductivity decreases correspondingly (Figure 2.4). At a tension of 0.3 m the hydraulic conductivity of the sandy loam is only 0.07 m/d and 29 % of the pores are now filled with liquid. Hence, the calculated travel time through 1 m will be 4.1 d. Similarly, the travel time at 0.80 m moisture tension is 27 days.

2.3 POLLUTANT MOVEMENT IN SATURATED ZONE

In uniform aquifers, the lateral (horizontal) groundwater flow rate in the saturated zone is a function of the saturated horizontal hydraulic conductivity and the hydraulic gradient. In most hydrogeological conditions, the hydraulic gradient is small (less than 0.01), hence it might be expected that flow velocities, although considerably greater than in the unsaturated zone (because of much higher permeability) would invariably be relatively small (less than 2 m/d). Additional protection of water supply from on-site units could be accomplished by increasing their lateral separation from the normally accepted minimum of 15 m. This is the case in certain hydrogeological environments, but it is not generally a reliable method of protection against microbial contamination, and separations will have to be increased from 15 m to 25 m, or even up to as much as 50 m. This is due to a number of reasons.

Firstly, since the porous media are saturated, immobilisation of organisms will be less in all cases, except in finer-grained unconsolidated aquifers. In high-permeability fissured aquifers such processes may be almost negligible. Nevertheless, pore clogging will also develop in on-site sanitation systems, penetrating the groundwater table and entering the groundwater system, although anaerobic groundwater conditions may prevail.

Secondly, very few aquifers are uniform, and "permeability heterogeneity" will normally be present as is the case in some stratified alluvial sequences containing thin beds of well-sorted coarse gravel, and in many limestones where solution (karstification) along joints has occurred. In this case the presence of highly permeable flow paths of small cross-sectional area results in groundwater velocities often exceeding 10 m/d, and reaching 100 m/d or more in many fissured aquifers and 1 km/d or more in some karstic limestones. Also, to establish the variation in aquifer permeability field investigation techniques are required.

Average rates of abstraction of groundwater for local water supply in developing countries, when not providing water to a reticulation system, are relatively small (usually not exceeding 0.5 l/s). Generally, the natural (rather than the induced) hydraulic gradient will be the dominant factor in determining the spread of pollution from a point source, except close to pumping boreholes, or in regions of very low groundwater recharge and, therefore, negligible hydraulic gradient and aquifer throughflow.

Dilution of pollutant concentrations in groundwater systems is due to physical (hydraulic) dispersion, but its quantification requires carefully designed groundwater tracer experiments. Without such investigations dilution cannot be readily predicted, and thus cannot be used to determine the risk of microbial pollution to a water-supply borehole.

If water-supply boreholes are designed so that their intake is below the groundwater level, any contaminants in the water-table will have to move downward before polluting the water-supply. In the saturated zone the vertical hydraulic conductivity is almost always considerably less than the horizontal value, hence the induced rate of vertical groundwater flow will be lower, and the pollutant residence times will be substantially increased. This approach is generally a more practical method of protecting water-supply boreholes from pollution than increasing lateral separation from the on-site sanitation unit. It will be most effective where semi-confining clay layers are present above the intake, but reliable grouting of the solid lining tubes of the borehole will be necessary.

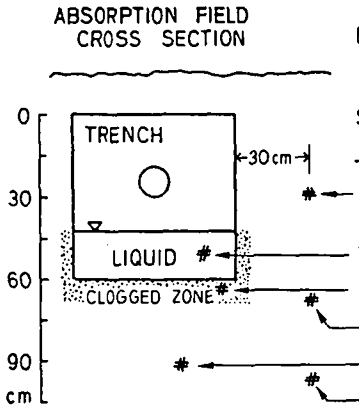
2.4 FACTORS AFFECTING PATHOGEN MOVEMENT

The unsaturated zone is the most important line of defence against faecal pollution of aquifers. Maximisation of effluent residence times in the unsaturated zone is, therefore, the key factor affecting the removal and elimination of bacteria and viruses.

2.4.1 *Filtration*

Filtration of bacteria at the infiltration surface appears to be the main mechanism limiting their movement through soil. It has been shown that filtration is most effective at the surface of the organic mat of the clogged zone. For instance, Ziebell et al. (1975b) found that the bacterial population below and to the side of a septic tank seepage bed was considerably reduced to about the level of the population in a control soil sample. This abrupt drop occurred within 30 cm of the clogged zone (Figure 2.6). Caldwell and Parr (1937) also noted that with a newly constructed latrine penetrating the water table, faecal coliforms were detected 10 m away. However, after clogging (3 months) pollutant dispersion was considerably curtailed. Krone et al. (1958) investigated E. coli removal in sand columns, and found that the effluent concentration of bacteria gradually rose and then declined which suggested that accumulating bacteria at the soil surface enhanced the straining mechanism. Butler et al. (1954) studied the penetration of coliform bacteria in sandy soils used to dispose of settled sewage. Measurements showed that there was a dramatic reduction in coliforms in the first 50 mm of soil, but that a subsequent build-up of bacteria occurred at lower levels.

The effect of temperature on the efficiency, drainage and maturation of slow filters suggested to Poynter and Slade (1977) that the removal of bacteria and viruses is essentially a biological process. Sterile sand was found not to remove viruses when operated at normal flow rates (4.8 m/day). It was postulated that a slow sand filter consists of a very large surface area populated by micro-organisms which remove other bacteria, small particles and chemicals dissolved in the filtering water, and that the sand merely acts as a support for the biological film. The process of maturation may simply be a measure of the time required for this situation to become established.



BACTERIA / 100 ml OR PER 100g OF SOIL

FECAL STREPTO-COCCI	FECAL COLI-FORMS	TOTAL COLI-FORMS	TOTAL BACTERIA × 10^7
<200	<200	<600	0.6
160,000	1,900,000	5,700,000	3.0
54,000	4,000,000	23,000,000	4,400
<200	17,000	23,000	6.7
<200	<200	<600	3.7
<200	700	1,800	2.8

Fig. 2.6 Cross-section of an absorption field in Plainfield loamy sand with typical bacterial counts at various locations (Ziebell et al., 1975b)

The data suggest that filtration is unlikely to be an important mechanism for the removal of bacteria in the saturated zone except possibly in very fine-grained strata where the pore diameters of the aquifer are smaller than the actual size of the organisms (i.e. less than 0.5 x 10^{-6}m).

2.4.2 *Adsorption*

Unlike bacteria, viruses are very small and removal appears to be dependent almost entirely on adsorption. Viruses are composed of a nucleic acid core encased in a protein sheath, and thus mimic the colloidal characteristics of proteins. It has been shown that adsorption of such hydrophilic colloids is strongly influenced by pH, the presence of cations, and the ionizable groups on the viruses (Stumm and Morgan, 1981). Viruses are strongly negative at high pHs, and strongly positive at low pHs. The iso-electric pH for enteric viruses is usually below pH 5; thus, in the pH range of most soils, enteroviruses have a net negative charge.

Burge and Enkiri (1978) studied the adsorption of bacteriophage Ø X-174 on five different soils in laboratory batch experiments. Good correlation was found between adsorption rates and cation exchange capacity, specific surface area and concentration of organic matter (r = 0.89, 0.85, 0.98 respectively). Thus, soils with increasing clay content will be more effective at adsorption than sandy soils. A negative correlation (r = -0.94) was found between the rate and pH. Hence, the lower the soil pH, the more positively charged the virus particles become, and the easier it is for them to be adsorbed.

A study by Green and Cliver (1975) with 60 cm soil columns showed that the Wisconsin soils tested removed all the viruses from septic tank effluent inoculated with polio virus type 1 (10^5 plaque-forming units (PFU) per litre). The columns were loaded at a rate of 50 mm/d applied in single doses over a period of more than a year. At a loading rate of 500 mm/d virus breakthrough occurred (Figure 2.7). Virus detention within the soil was found to be affected by the degree of saturation of the pores; the more saturated the pores, the less opportunity there was for contact with surfaces. Thus to enhance virus removal large hydraulic surges, or very uneven distribution of the waste should be avoided.

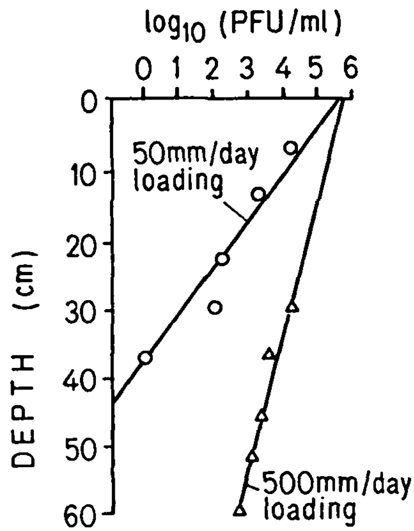


Fig. 2.7 Penetration of polio virus into packed sand columns at room temperature (Green and Cliver, 1975)

Aqueous bacterial suspensions have been classified as hydrophilic biocolloids since they have a negative zeta potential at pH 7, and are extensively solvated (Lamanna and Mallette, 1965). Thus, bacteria can also be removed in soils/unconsolidated strata by adsorption. The adsorption capacity of a soil usually increases with clay content, however, small particle size in clay soils also results in some filtration of bacteria at the surface. Hence, it is difficult to assess the relative importance of filtration and adsorption in removing bacteria from effluents.

Micro-organisms adsorbed onto soil particles are not necessarily permanently immobilized. Adsorption is a reversible phenomenon and microorganisms can become desorbed and thus penetrate deeper into the soil.

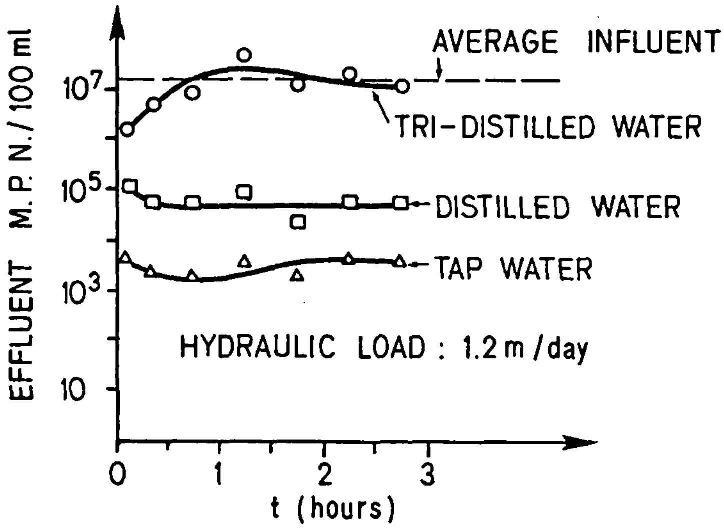


Fig. 2.8 Filtration of E.coli through sand in tap and distilled water (Goldshmid et al., 1973)

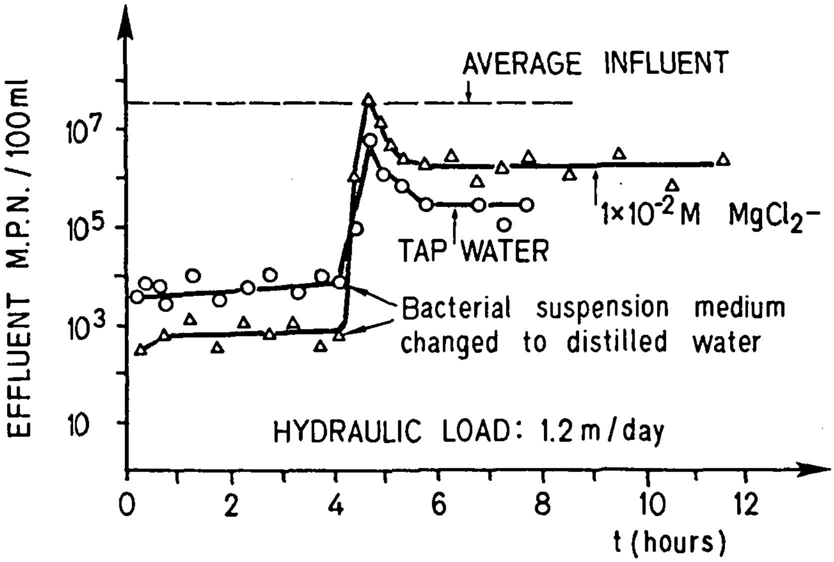


Fig. 2.9 Bacteria breakthrough in distilled water (Goldshmid et al., 1973)

Goldshmid et al. (1973) investigated the adsorptive behaviour of E. coli using sterile sand columns (effective size 0.12 mm) loaded at a constant rate of 1200 mm/d. They found that bacterial removal was greater with tapwater than with distilled water. When triple distilled water was used as the medium, virtually no bacterial removal occurred (Figures 2.8, 2.9), hence changes in ionic strength can reverse the process of adsorption. Also decreasing the pH from 9 to 4 or increasing the cation concentration or valence, was found to increase the bacterial removal capacity of the soil. Sand particles and bacteria are both negatively charged in low ionic strength media, and this causes repulsion. Addition of cations or protons (hydrogen ions) to bio-colloids decreases the zeta potential of hydration and may even reverse the polarity thus decreasing repulsion and increasing adsorption.

Similarly, Landry et al. (1979) have demonstrated desorption of viruses. They observed that flooding soil columns with de-ionized water caused virus desorption and increased their movement through the columns. Addition of calcium chloride to the de-ionized water followed by application of sewage effluent enabled the viruses to penetrate the bed, however, they were eventually readsorbed. These data suggest that large reductions (99.99 % or more) of viruses could be expected after passage of secondary effluent through 0.25 m of calcareous sand when loaded at rates up to 550 mm/d. Viruses would only move through this type of soil if heavy rains fell within 1 day after the application of sewage.

Landry et al. (1979) also observed that different strains of viruses have varying adsorptive properties. In laboratory studies on sand cores, they found that the number of viruses mobilized by simulated rainfall ranged from 24-66 %, and was dependent on the strain of virus. Goyal and Gerba (1979) also noted that virus adsorption was highly dependent on the strain, and concluded that no one virus or coliphage could be used as a model for determining viral adsorptive behaviour. More recent studies have found that there are even differences in adsorptive behaviour with a particular strain of virus. For instance, a study by Lance and Gerba (1980) on the factors affecting the rate and depth of virus penetration revealed that virus adsorption in soil is increased above some breakpoint velocity, whereas flow rate changes above and below the breakpoint do not affect virus adsorption. The breakthrough velocity was thought to correspond to the velocity where some water begins to move through the large soil pores, allowing little or no contact between viruses in the water and adsorptive surfaces. The breakthrough velocity for the coarse loamy sand tested was between 0.6 and 1.2 m/d but will, of course, be different for other soil types. Differences in the strength of the negative charge among members of a given viral population was thought to account for the adsorption of some of the viruses near the soil surface while others move further through the profile. It was postulated that the velocity of water movement through the soil may be the most important factor affecting the depth of virus penetration. This suggests that adsorption may not be an important factor of removal in the saturated zone, especially in formations where groundwater velocities are high.

The phenomenon of desorption with decrease in ionic strength has practical implications for groundwater pollution. Previously adsorbed bacteria and viruses could be desorbed by heavy rains. This has been shown at a Florida site irrigated with secondary sewage at a rate of 1-5 cm/day where viral penetration up to 6 m in sandy soil was attributed to heavy rains (711 mm) (Wellings et al., 1974). Viruses were also detected in 3 m deep observation wells beneath a cypress dome receiving septic tank effluent from a mobile home park 28 days after the last application because of heavy rains falling in the preceding period (Wellings et al., 1975). Martin and Noonan (1977) also observed that rainfalls of greater than 50 mm resulted in bacterial contamination of the groundwater under a sewage irrigation scheme in Burnham, New Zealand (Figure 2.10). No increase was noted in the control borehole located upstream of the irrigation scheme.

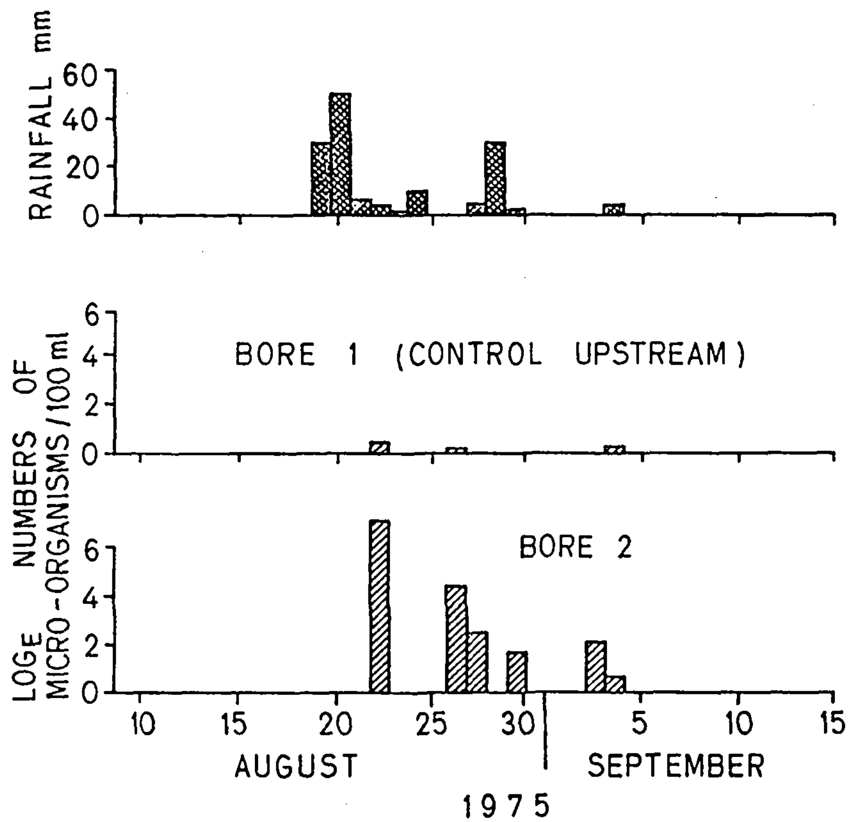


Fig. 2.10 Numbers of faecal coliform bacteria per 100 ml of groundwater found in investigation bores at Burnham after a period of heavy rainfall

Similar observations were made by Sinton (1980) using Bacillus stearothermophilus as a tracer for groundwater movement. B. stearothermophilus was found to be naturally present in Canterbury soils and groundwater systems. Concentrations of the species tended to increase following rainfall which limited the use of the organism as a tracer to periods of low rainfall. Also, Barrel and Rowland (1979) attributed the massive increase of faecal coliforms (5×10^5/100 ml) in Gambian village well waters to the onset of rains which flushed faecal material into the groundwater. Even wells with satisfactory construction methods (on a sanitary basis) showed an increase. A possible explanation of this may be that the heavy rains flushed adsorbed bacteria through the laterite soils causing an increase in the number of micro-organisms in the groundwater.

The adsorptive behaviour of micro-organisms can create difficulties in sampling for any pollution study investigating travel in the unsaturated zone. Porous ceramic cups have often been used in the past to collect samples of soil moisture; however, the results of these studies should be treated with caution. For instance, Dazzo and Rothwell (1974) conducted a study to determine the validity of obtaining soil water for faecal coliform analysis by a porcelain cup soil water sampler. Faecal coliforms were found to adsorb to the cups, and their numbers were considerably reduced compared with those obtained from the surface applied manure slurry, and 65 % of the cups yielded coliform free samples.

The factors influencing the movement of bacteria and viruses through soils are summarized in Table 2.1.

Table 2.1 Factors influencing movement of bacteria and viruses through soil (after Gerba et al., 1975)

Rainfall	Micro-organisms retained near the soil surface may be eluted after heavy rainfall because of the establishment of ionic gradients within the soil column.
pH	Low pH favours virus adsorption; high pH results in elution of adsorbed viruses.
Soil Composition	Bacteria and viruses are readily adsorbed to clays under appropriate conditions, and the higher the clay content of the soil, the greater the removal. Sandy loam soils and other soils containing organic matter are also favourable for removal.
Hydraulic Loading/ Flow Rate	As the flow rate increases, micro-organisms penetrate deeper. The hydraulic loading is naturally increased during periods of groundwater recharge by infiltrating rainfall.
Soluble Organics	Soluble organic matter has been shown to compete with organisms for adsorption sites on the soil particles, resulting in decreased adsorption or elution of already adsorbed viruses.
Cations	Cations, especially divalent ones, can act to neutralize or reduce repulsive forces between negatively charged micro-organisms and soil particles, allowing adsorption to proceed.

In conclusion it appears from the literature that the removal of bacteria and viruses by adsorption will be enhanced by maximising effluent residence times in the unsaturated zone, i.e. by permitting the greatest media-liquid contact. This can be achieved by maintaining a low hydraulic loading rate, or by restricting the infiltration rate, which will occur naturally after clogging of the infiltration surface. Soil type will also affect the movement of micro-organisms, with some soils being more effective in removal than others (Bitton et al., 1979). In general, sandy and organic soils are poor adsorbers, whilst soils with a clay content are better.

In the saturated zone adsorption is unlikely to greatly affect the movement of micro-organisms, with the possible exception of slow-moving groundwaters (say less than 1 m/d). Once bacteria and viruses penetrate the water table they can be transported over considerable distances. The rate and extent of movement will be controlled largely by hydrogeological factors.

2.5 FACTORS AFFECTING PATHOGEN SURVIVAL

Feachem et al. (in press) have reviewed the literature on the occurrence and survival of micro-organisms in various environments, e.g. surface, ground, drinking waters and soil.

2.5.1 *Survivals in soils*

(a) Viruses

Laboratory soil column studies (Gerba et al., 1975; Bitton et al., 1979) have shown that the nature of the soil can affect virus survival characteristics. The major factors appear to be moisture and temperature, with survival times of 175 days or more being possible. Hurst (1979) studied the persistence of viruses in soils used for land treatment of sludge and wastewater in Texas, USA, and found that virus survival increased with the degree of viral adsorption to the soil. Hence, soils which were most effective in removing viruses would also enable them to persist for the longest periods. Enterovirus survival in soil was found to be increased by low temperatures but was unaffected by ionic strength. The reduction of polioviruses held for 84 days in loamy sand was less than 90 % at 4°C, but 99.999 % at 20°C (Duboise et al., 1976). Also it was found that aerobic inactivation was more rapid under non sterile versus sterile conditions, and that anaerobic conditions let to a reduction in inactivation.

Lefler and Kott, (1974) studied the survival of poliovirus 1 in sand. When the saturated sand was kept at 4-8°C, 20 % of the viruses were active after 175 days. On dried sand at 4-8°C, 96 % inactivation occurred after 21 days, and viruses were still detectable after 77 days. Yeager and O'Brien (1979a) also found that poliovirus survival depended on temperature: viruses survived in saturated soils up to 12 days at 37°C, up to 92 days at 22°C, and up to 180 days at 4°C. Viruses were found to survive longer in sandy loam soils (90 % reduction in 6-21 days at 22°C) than in sand (90 % reduction in 4-8 days at 22°C). This may be due to the better moisture holding capacity of the sandy loam soil since drying the soil was found to be highly virucidal irrespective of soil type. Soil moistures of below 2.9 % appeared to be especially virucidal. In an accompanying study on the nature of virus inactivation (Yeager and

O'Brien, 1979b) the authors concluded that loss of infectivity was due to irreversible damage to the viruses.

Keswick and Gerba (1980) evaluated the factors controlling virus survival and found that inactivation was much more rapid near the surface. This is due to the detrimental effect of aerobic soil micro-organisms, evaporation, and higher temperatures close to the surface. Thus, virus survival is expected to increase with depth of penetration.

(b) Bacteria

Reported survival times of faecal bacteria vary widely, and the data are often complicated by the possibility of regrowth. The dominant factors controlling the survival of bacteria in soil are moisture and temperature.

Kligler (1921) investigated the survival of Salmonella typhi and Shigella dysenteriae in different soil types at room temperatures. He found that in moist soils some bacteria survived for 70 days, although 90 % died within 30 days. In dry soils no bacteria survived longer than 20 days, and with acid soil, irrespective of moisture content, this time was reduced to 10 days. Cold temperatures (below 4°C) favour the survival of most micro-organisms, and enteric bacteria are no exception. Mirzoev (1968) found that in areas with prolonged winters, e.g. the Russian Arctic, bacterial die-off is slowed down or suspended. He showed that low temperatures (down to -45°C) were favourable for the survival of Shigella dysenteriae, and was able to detect them 135 days after they had been added to the soil.

Kibbey et al. (1978) investigated the survival of Streptococcus faecalis in five Oregon soils. They found that die-off rates varied between the different soils, but were generally longest in soils maintained under cool, moist conditions. The longest survival times were obtained under saturated conditions, and this was attributed to the lack of antagonistic activity by soil micro-flora (Table 2.2).

Table 2.2 Average 95 % population reduction times (T_{95}) for S. faecalis in five Oregon soils (Kibbey et al., 1978)

Soil moisture equivalent	Moisture tension (bars)	T_{95} (days)			
		4°C	10°C	25°C	37°C
Saturation	0.0	94	80	53	29
Field capacity	0.3	60	43	38	16
50 % field capacity	7.5	35	29	22	8
Air dried	30.0	23	18	9	5

This finding was confirmed by Bouma et al. (1972) in field studies on pollution movement beneath septic tank disposal fields. They showed that within the first 30 cm of soil, actinomycetes and moulds began to appear, and were very numerous in the next 30 cm. These organisms produce antibiotics and hence, contribute to the die-off of enteric bacteria. Soil microflora also compete with enteric bacteria for available nutrients, and this may play a major role in their die-off. Survival may be prolonged in soils where nutrients are readily available, e.g. soils which receive sewage effluent or night soil. Dazzo et al. (1973) recorded T_{90} values for E. coli of 8.5 days in soil receiving 50 mm of cow manure slurry per week and 4 days in soil receiving no manure. Finally, Martin and Noonan (1977) found that faecal coliform and faecal streptococci numbers were reduced by 90 % (T_{90}) in 28 and 22 days respectively at depths of 0-100 mm, but required 182 and 25 days respectively at 100-200 mm depths in silt loam. This tendency to longer survival in the deeper soil layers may be due to decreasing antagonism from natural soil-microflora with increasing depth of soil.

Feachem et al. (in press) have reviewed the literature on the die-off of micro-organisms in various media. Conditions were found to be so variable that reported survival times and T_{90} values varied over a wide range. In general it appeared that faecal coliforms only survived for 10 weeks, with a 90 % reduction taking place within 2-3 weeks. However, under cool moist conditions some of the faecal coliforms can survive for many months. Where conditions are hot and arid it is probable that complete elimination of faecal indicator bacteria will occur within 2 weeks. For much the same reasons as were noted in virus survival in soils, bacteria that penetrate deeper into the soil will survive longer than those near the surface. The factors influencing the survival of bacteria and viruses in soil have been summarised in Table 2.3.

Table 2.3 Factors influencing survival of bacteria and viruses in soils (Gerba, 1979)

Factor	Effects
Moisture content	Greater survival time in moist soils and during times of high rainfall
Moisture holding capacity	Survival time is less in sandy soils than in soils with greater water holding capacity
Temperature	Increased survival at lower temperatures
Adsorption	As virus adsorption to soil increases, viral survival is prolonged
pH	Shorter survival times in acid soils (pH 3-5) than in alkaline soils (bacteria)
Sunlight/evaporation	Shorter survival time at soil surface
Organic matter	Increased survival of bacteria and possible regrowth when sufficient amounts of organic matter are present
Antagonism from soil microflora	Increased survival time in sterile soil, soil microflora compete with bacteria for nutrients; aerobic soil micro-organisms adversely affect virus survival while anaerobic micro-organisms have no effect

2.5.2 *Survival in groundwater (saturated zone)*

(a) Viruses

Little is known about virus survival in groundwater, but rough estimates may be made from the data on their survival in surface waters. Field studies by Wellings et al. (1975) suggest enteroviruses can survive for at least 28 days in groundwater. Akin et al. (1971) reviewed the literature on the survival of enteric viruses in waters with varying degrees of pollution. They found that between 2 and 100 days are required for various members of the enteric family to lose 99.9 % of their initial infectivity when suspended in different surface waters at 20^{o}C. Survival was found to be largely determined by temperature and degree of contamination, being longer in very clean water and in heavily polluted water. Similar observations have been reported in more recent investigations (Niemi, 1976; O'Brien and Newman, 1977; Yeager and O'Brien, 1979a). A study with radioactively labelled poliovirus-1 and coxsackievirus B-1 indicated that inactivation at higher temperatures was due to damage to viral ribonucleic acid (O'Brien and Newman, 1977).

From these date it appears that temperature is the single most important factor in die-off, and 99 % reduction may be expected at 20^{o}C within about 10 days although a few enteroviruses may survive for many months (Feachem et al., in press).

Several workers (e.g. Cubbage et al., 1979; Katznelson, 1978; and Young and Sharp, 1977) have noted that the observed loss of infectivity of viruses in water may be due in part to genuine damage to the virus, and in part to an artefact caused by many viruses aggregating and simulating a single infectious particle. This aggregation may involve the adsorption of viruses onto organic or inorganic suspended particulate matter. Adsorption is enhanced at slightly acidic pH and in the presence of divalent cations, and is deterred by the presence of soluble proteins (Schaub et al., 1974 and 1975).

(b) Bacteria

Information on bacterial survival in groundwater is limited. It is generally accepted that survival is usually longer in groundwater than in surface water due to the absence of sunlight and because competition for available nutrients is not so great. Again temperature is important, with the bacteria surviving longer at lower temperatures. The chemical nature of the groundwater will also affect the survival capabilities of any bacteria present. Enteric bacteria are usually intolerant of acid conditions, and to varying degrees are also intolerant of saline groundwaters.

Enteric bacteria appear to survive in groundwater for considerable periods of time (100 days or more) depending on the temperature. In warmer countries the temperature of shallow groundwaters will be relatively high; for instance, $25^{o}C$ is common in shallow aquifers (< 12 m) in Botswana (Lewis et al., 1980), and temperatures of well waters in Gambia have been found to be in the range $28\text{-}30^{o}C$ (Barrell and Rowland, 1979). Hence, survival of enteric micro-organisms may be shorter in tropical groundwaters than in their temperate counterparts. However, the possibility of re-growth should be considered since it is known that many types of bacteria will grow in water containing mineral salts and an energy source

Kudryavtseva (1972) reported that coliforms introduced into a groundwater bed of fine-grained alluvial sand during the summer survived for up to 3.5 months. A pathogenic serotype of E. coli, similarly inoculated into the groundwater, survived for 3 months. In groundwater samples stored in the dark, coliforms survived for up to 5.5 months, while pathogenic E. coli survived for up to 4 months (temperatures unspecified). Mitchell and Chamberlain (1978) surveyed published data on the survival of indicator organisms in a variety of freshwater bodies. They found that bacterial die-off generally follows first order kinetics, although a significant increase in coliforms is often observed in the first few kilometers from the outfall. The median value for the rate of die-off for 28 studies was 0.040 hr^{-1}. McFeters et al. (1974) measured the comparative survival of various faecal indicator bacteria and enteric pathogens in well water using membrane chambers. T_{50} values (time required for a 50 % reduction) of the various cultures are given in Table 2.4.

Similar experiments were conducted in New Zealand (Martin and Noonan, 1977; and Pyle and Thorpe, 1979) to determine the time for a 90 % decrease in faecal coliforms and a hydrogen sulphide resistant strain of E. coli (E. coli H_2S^+). The E. coli (H_2S^+) survived for 4.0 days at $11^{o}C$ and 2.2 days at $15.5^{o}C$, whilst the faecal coliforms survived 6.2 days (temperature unspecified). Hagedorn et al. (1978) used antibiotic resistant E. coli and S. faecalis to monitor the degree of movement from a septic tank drainfield submerged in a perched water table. Their results indicate that both organisms survived in appreciable numbers even after 32 days. Ambient temperatures during the study period were cool ($2\text{-}15^{o}C$ with occasional frosts).

Table 2.4 Half time (time for 50 % reduction) of various bacterial cultures in well water, at 9-12°C, (McFeters et al., 1974)

Bacteria	Half time (hr)	Calculated die-off rate (hr^{-1})
Indicator bacteria		
coliforms (average)	17.0	0.040
Enterococci (average)	22.0	0.031
Streptococci (average)	19.5	0.035
Streptococcus equinis	10.0	0.067
St. bovis	4.3	0.149
Pathogenic dysenteria		
Shigella dysenteriae	22.4	0.030
S. sonnei	24.5	0.028
S. flexneri	26.8	0.026
Salmonella enteritidis ser. paratyphi A & D	16-19.2	0.042-0.035
S. enteritidis ser. Typhimurium	16.0	0.042
S. typhi	6.0	0.109
Vibrio cholerae	7.2	0.092
S. enteritidis ser. paratyphi B	2.4	0.251

Many investigations have found that faecal streptococci often persist longer than faecal coliforms (Geldreich et al., 1968; Figure 2.11). However, it is apparent from Table 2.4 that Streptococcus bovis and St. equinis die-off considerably faster than faecal coliforms and other species of faecal streptococci. St. bovis and St. equinis are the dominant streptococcal species in some animal faeces, however, they never occur in human faeces.

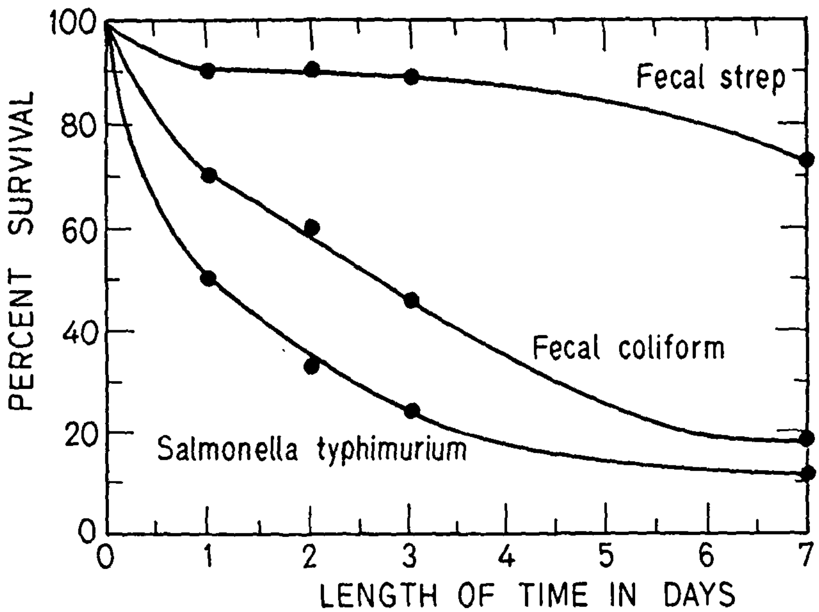


Fig. 2.11 Storage time versus "die-off" for selected enteric bacteria in samples of stormwater stored at 10°C (Geldreich et al., 1968). At higher temperatures the "die-off"rate is even steeper.

Faecal coliform to faecal streptococci (FC/FS) ratios have often been recommended (for example, see Millipore Field Manual) in stream pollution studies as an index of the origin of contamination, i.e. animal or human[1]. To obtain meaningful ratios bacteriological counts must be performed within 24 hours of sample collection; this is because of the difference in die-off rates. Sometimes, in samples containing mainly human faecal pollution, the FC/FS ratio falls with time, whereas when non-human pollution predominates the ratio may rise. However, the faecal coliform to faecal streptococci ratios in groundwater studies are totally meaningless because even if the sample is analysed immediately there is no way of determining how long the organisms took to reach the groundwater, and how long they have been in there.

[1] FC/FS ratio greater than 4.0 indicates pollution derived from human wastes, whilst FC/FS ratios of 0.7 or less indicate pollution derived from livestock or poultry wastes.

Chapter 3 : Applied Field Investigations of Pollutant Movement

In the preceding section the importance of the unsaturated zone in protecting aquifers against pollution was discussed, and how the maximisation of liquid residence time is the key factor in the elimination of pathogenic bacteria and viruses. However, early researchers such as Caldwell (1937) and Dyer and Bhaskaran (1943), concentrated on establishing the lateral separation necessary between wells/boreholes and on-site sanitation systems (chiefly pit latrines) to protect the groundwater.

3.1 BACTERIA IN THE SATURATED ZONE

Caldwell conducted a series of detailed studies measuring pollution travel from a variety of pit latrines. The first study (Caldwell and Parr, 1937) measured pollution travel from a 5.1 m bored hole latrine in a shallow (3.6m) perched water table located in a coarse sandy stratum. Faecal material from a family of six was added daily to the latrine. The natural groundwater velocity was found to be in the range 0.6-2.5 m/d. Initially faecal organisms travelled some 4.5 m in 3 days. After this time chemical contaminants preceded all bacterial recovery and were evident at 10.6 m after 9 days. The chemical stream was traced to 26 m but could not be detected at 31 m. After 2 months, faecal coliforms were present in 90 % of the samples at 4.5 m, and were occasionally detected at 10.6 m.

A conclusion of this study was that the clogging process was in important defence mechanism limiting the extent of bacterial penetration. After the onset of clogging in the latrine (3 months), the diffusion of organisms was inhibited and within 7 months bacterial pollution was limited practically to the latrine. However, chemical contamination of the groundwater still occurred, although this was somewhat diminished.

A parallel study (Caldwell, 1937) was conducted nearby using a dug pit latrine, with sampling boreholes sunk into a dense calcareous stratum underlying the permeable sands through which the groundwater flowed. The groundwater flow velocity at the site was 4 m/d, and it was found that the clogging process was not as effective with this type of latrine, possibly due to the greater volume per depth of penetration. Even after 16 months the outflow was never significantly inhibited by heavy material and sludge deposits (as was observed with the bored hole latrine). The higher groundwater flow velocity through these sands resulted in significant numbers of faecal coliforms being carried at least 24 m (to the most distant observation well), and gross contamination was evident at 18 m. The rate of groundwater flow was thought to be the dominant factor in determining the extent of travel of faecal organisms.

Another experiment was later conducted at the same test site (Caldwell, 1938b) in which groundwater contamination was reduced by construction of an envelope of fine sand (0.25 mm) around the pit latrine. No faecal coliforms were detected in any of the observation wells 3 m away, and this contrasts with the earlier study where gross contamination was evident at 18 m.

Another study by Caldwell (1938a) measured pollution travel from a pit latrine penetrating the water table in a medium-fine sand (0.5-0.1 mm) with a groundwater flow velocity of 0.5 m/d. Chemical pollution was traced as far as 106 m by odour and pH variation, and to 94 m by chemical analysis. As in previous studies it was observed that the bacterial plume was smaller than the chemical plume. Faecal coliforms initially penetrated 3 meters in 3-4 months before flow from the pit was restricted by clogging. At termination of the study, the apex of the bacterial plume barely reached 1.5 m (Figure 3.1).

Detailed studies, similar in nature to Caldwell and Parr's work, were conducted by Dyer and Bhaskaran (1943, 1945). The objective of these studies was to determine the practicability of bored hole latrines and shallow wells in rural communities in India. A bored hole latrine (0.4 m diameter) penetrating the groundwater was seeded with 9 litres per day of faecal matter for a period of 330 days. 196 observation wells (1.5 to 3.0 m deep) were sunk around the latrine at varying distances up to 19.5 m. The soil in the area was clayey silt with a decreasing clay content down to 4.9 m. Below this depth in the saturated zone the soil was a medium sized sand (0.5-0.25 mm). The groundwater flow velocity was found to be around 0.75 m/d, and to simulate the effect of village water supply wells. A total of 2700 l/d was pumped from six wells located 6 m away. Addition of faecal material had to be temporarily halted when clogging caused the effluent level to rise to within 0.6 m of the ground surface.

Data from this study showed that bacteria travelled up to 3 m in the direction of groundwater flow, but later diminished and were virtually absent during the final period. Chemical contaminants were found to travel a distance of 4.5 m.

The study concluded that in sandy soils (≤ 0.2 mm) bored latrines could be placed as close as 6 m to a water supply well. With coarser soils (0.3 mm) this distance should be 15 m.

Subrahmanyan and Bhaskaran (1950) later reviewed the Indian and American studies and concluded that:

(a) Bacterial travel appears to depend mainly on the velocity of groundwater flow.

(b) The penetration of bacteria into the saturated zone is the distance covered by the groundwater in 4-7 days, which is the probable survival time for coliform organisms in the anaerobic groundwater environment[1].

(c) The spread of pollution is reduced when a gelatinous membrane is established on soil particles, since this acts as a physical barrier to the penetration of bacteria. In this condition the soil becomes a real biological filter comparable to a slow sand filter in water treatment.

(d) The safe distance between a borehole latrine or leaching cesspit and a well may be taken to be the distance represented by about 8 days' travel of the groundwater.

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[1] Many groundwaters are in fact slightly aerobic

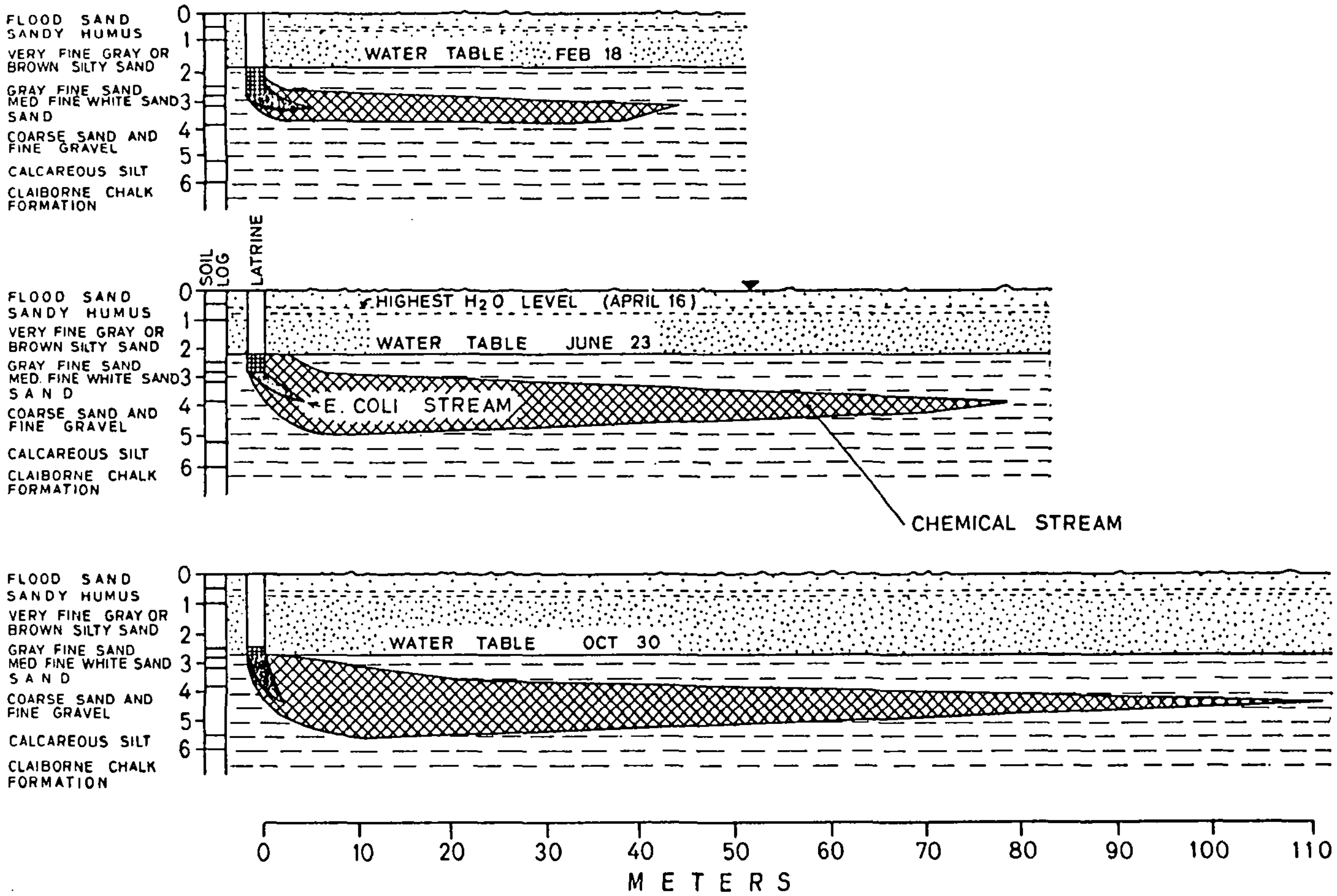

Fig. 3.1 Pollution travel from a pit latrine penetrating the water table in a medium-fine sand (Caldwell, 1938a)

(e) In the study areas in India where the hydraulic gradient is less than 0.01, and the soil is sandy (effective size less than 0.25 mm), the groundwater velocity is unlikely to exceed 0.9 m/d, and a horizontal distance of 7.5 m will provide an ample margin of safety against bacterial pollution.

From the findings of these early researchers, a general rule of 15 m (50 feet) between a pit privy and a well became widely accepted. Unfortunately, this guideline has been applied indiscriminately, with little thought to its applicability to the particular site conditions under consideration. Even in non-fissured formation, where the hydraulic gradient is artificially induced by man or where the gradient is naturally steep as, for example, in coarse gravels, this 15 m rule will not be valid. Dappert (1932) traced a stream containing faecal bacteria 120 m away from sewerage seepage beds. Butler et al. (1954) injected primary settled sewage diluted with water into a confined aquifer at the rate of 2 l/sec for 41 days, and found it reached wells 30 m away in only 33 hours. The longest distance of bacterial travel in non-fissured formations recorded in the literature is by Pyle et al. (1979) of 920 m in coarse alluvial gravels. However, groundwater velocities as high as 350 m/d have been measured in this aquifer.

The results of the studies concerning bacterial pollution travel in the saturated zone are summarized in Table 3.1. This shows that the linear travel of pollution is governed primarily by the groundwater flow velocity, and the viability of the organisms. Maximum linear diffusion appears to be the distance the groundwater flows in a period of around 10 days. This implied survival time contrasts sharply with laboratory studies and controlled field studies which suggest a possibility of survival of 100 days or more. The distance over which enteric bacteria can be traced will depend not only on the groundwater velocity and their death rate, but also on their initial concentration, dispersion within the groundwater body, the sample volume tested and the sensitivity of the method used to detect them.

In an aquifer of uniform permeability it would be a relatively simple matter to estimate the groundwater velocity and hence calculate a safe distance of separation between a groundwater source and an excreta disposal system. Unfortunately, one rarely finds a uniform aquifer material where "permeability heterogeneity" will normally be present. This will make the procedure of predicting safe distances much more difficult. Pollutants may be transported along preferential paths at velocities very much in excess of the average groundwater flow velocity. For instance, Allen and Morrison (1973) injected bacterial tracer organisms (Bacillus steorothermophilus) into a borehole penetrating the water table in fissured bedrock. The tracer organism was detected in a well 29 m away within 24 hours, although it could not be detected in two closer wells (6 m and 16 m).

Lewis et al. (1980) injected a chemical tracer (lithium chloride) into a borehole penetrating the water table in weathered and fissured bedrock. The presence of lithium was monitored in water pumped from aborehole 20 m distant. During the first 200 minutes the lithium concentration remained at the background level (0.08 mg/l); it peaked at 1.05 mg/l after 210 minutes, and returned to the background level after 230 minutes. These data suggest that the flow of the tracer ouf of the borehole had occurred only at isolated fissure-flow horizons.

Table 3.1 Summary of bacterial travel in the saturated zone (includes travel through unsaturated zone)

Location	Water bearing zone	Lateral travel (m)	Groundwater velocity (m/d)	Computed residence time (d)	Temp. (°C)	Remarks	Investigator
Singapore Malaya	Medium sand	21	3	7	-	Bored hole latrine penetrating groundwater	Yeager (1929)
Alabama, USA	Medium sand	10.6	0.9-2.5	4-11	21	Bored hole latrine penetrating groundwater	Caldwell and Parr (1937)
Alabama, USA	Coarse sand	24	4	6	17-22	Pit latrine penetrating groundwater	Caldwell (1937)
Alabama, USA	Coarse sand	3	4	-	17-22	Envelope of fine silty sand constructed around pit latrine	Caldwell (1938)
Alabama, USA	Medium sand	3	0.5	6	17-22	Pit latrine penetrating groundwater	Caldwell (1938a)
West Bengal, India	Medium sand	3	0.74	4	26.5	Bored hole latrine penetrating groundwater	Dyer and Bhaskaran (1945)
Long Island, USA	Fine sand	122	-	-	-	Effect of induced hydraulic gradient - SEWAGE INFILTRATION	Dappert (1932)
California, USA	Sand	30-68	22	3	-	Artificial recharge with treated sewage directly to groundwater	Butler et al. (1954)
Ontario, Canada	Silty sand	3	0.1*	30	-	Septic tank effluent, shallow water table	Brandes (1974)
Ontario, Canada	Fine sand	16	0.6*	30	-	Septic tank effluent, imported fill material	Brandes (1974)
Ottawa, Canada	Sandy clay	15	-	-	-	Septic tank drainfield 0.15 m above groundwater	Viraraghavan (1978)
Burnham, New Zealand	Alluvial gravels	920	350	3	11	Injection of tracer organisms at land disposal site	Pyle et al. (1979)
Hawkes Bay, New Zealand	Alluvial gravels	125	168	-	-	Simulated leaking sewer pipe	Thorpe (1979)
Colorado, USA	Fractured rock	28+	25	-	-	Septic tank effluent in mountainous terrain	Allen and Morrison (1973)
Mochudi, Botswana	Fractured rock	25+	120	-	25	Pit latrine dug down to weathered and fractured bedrock	Lewis et al. (1980)
Oregon, USA	Silty clay loam	15+	360	-	9-13	Septic tank effluent, rapid flow through macropores	Rahe et al. (1978)
Oregon, USA	Silty clay loam	15+	30	-	9-13	Septic tank drainfield submerged in perched water table	Rahe et al. (1978)

These two studies clearly demonstrate that fissures in consolidated rock formations permit rapid groundwater movement. Similarly macropores in soils can also influence the direction and rate of movement of groundwater. Rahe et al. (1978) conducted field experiments using strains of antibiotic resistant E. coli to evaluate the events which would occur when a septic tank drain-field became submerged in a perched water table and faecal bacteria were subsequently introduced into the groundwater. At one of the sites it was concluded that the rapid water movement rates in the soil (colluvium weathered from sedimentary bedrock) was caused by flow through old root channels, rodent burrows, etc. Movement through these macropores was demonstrated when E. coli were recovered at the 15 m well before the 10 m well. At peak recovery rates in the 15 m well (110 cm depth) approximately 10^5/100 ml organisms were detected (5.6×10^{12}/100 ml cells injected).

These examples serve to emphasise the danger of relying on a fixed distance of separation between a groundwater source and on-site sanitation to protect the water supply against faecal contamination. Unless the aquifer is uniform, uncertainties over the degree of permeability heterogeneity will make the procedure of predicting safe distances a risky affair.

3.2 BACTERIA IN THE UNSATURATED ZONE

Kligler (1921) was one of the earliest researchers to investigate the relationship between pit latrines and the spread of waterborne infectious diseases. Field and laboratory studies were conducted to determine the viability of pathogenic bacteria, and their extent of penetration in the soil underneath a pit latrine. The field studies were conducted in a variety of soils with pit latrines that had been in use for 1-3 years. He concluded that pit latrines and septic privies, if properly constructed, are unlikely to cause the spread of bacterial intestinal infections. The pathogenic organisms were found to die off rapidly in faeces, and bacteria were only transported 0.9-1.5 m in the soil types studies. The pit latrine appeared to involve minimal pollution risk in sandy or clay type soils provided the groundwater level did not rise higher than 3-4 m below ground level, i.e. 1.5-2.5 m beneath the base of the pit latrine.

Caldwell (1938c) investigated the penetration of faecal coliforms in a permeable sandy soil below a pit latrine seeded with the faecal material from a family of six. A similar quantity of faecal material was added to two further pits. One was left open to determine the effect of rainfall and 380 litres of water were added daily to the remaining pit to simulate a cesspool. Bacterial dispersion from the latrine receiving faecal fluids only, was less than 0.3 m, either laterally or vertically. In the pit subjected to seepage from rains, lateral penetration was confined to 0.3 m horizontally, and 0.9 m vertically. Maximum travel was observed in the pit which was artificially dosed with 380 litres of water daily, and distances of 1.8 m vertically and 0.6 m laterally were measured.

Baars (1957) investigated dispersion from pit latrines at a camping site in the Netherlands. The soil at the site was sandy (effective size 0.17 mm), and the groundwater table was 3.5 m below ground level. Faecal coliform concentrations in the subsoil were determined 7 months after the end of the camping season, and it was found that no microorganisms were present at depths greater than 1.3 m. Comparison of these counts with those obtained at the end of the camping season indicate that bacteria may penetrate some distance into the soil. However, no figure was given for the depth of maximum penetration.

The conclusion that can be drawn from these early studies is that at least 2 m of sandy soil is required beneath a pit latrine to prevent pollution of any underlying groundwater (see Table 3.2, Part A). In recent years, because of fears of pollution, the fate of septic tank effluent has been the subject of increasing attention, especially in North America. Almost 20 million housing units, representing 29 % of the United States' population, dispose of domestic waste through individual on-site sanitation systems, discharging approximately 3 x 10^{12} m^3/yr of water to the soil. A survey by officials, consultants, water well drillers and other water resource officials in 35 states, disclosed that septic tanks and cesspools rank highest in the total volume of wastewater discharged directly to the ground, and were the most frequently reported source of groundwater contamination (Miller and Scalf, 1974).

A more recent survey (Scalf et al., 1977) concluded that soils in many areas are not suitable for conventional septic tank soil absorption systems. The areas which were found to be unsuitable contained: thin soil over creviced bedrock, or a high groundwater table, or both.

For example, most of the reported instances of microbial contamination of groundwater resulting from the use of on-site sanitation are associated with areas of thin soil cover over fissured bedrock (Neefe and Stokes, 1945:, Vogt, 1961; Doehring and Butler, 1973; Van der Velde, 1973; Waltz, 1972; Scalf et al., 1977; Lewis et al., 1980), or areas of high or seasonally high groundwater conditions (generally less than 3 m) Sridhar and Pillai, 1973; Brandes, 1974; Binnie and Partners, 1975; Reneau and Pettry, 1975; Viraraghavan and Warnock, 1976; Scalf et al., 1977; Rahe et al., 1978). The unsuitability of these two soil types as areas for on-site sanitation can be illustrated by the following studies.

3.2.1 *Thin soil over fractured bedrock*

Allen and Morrison (1973) noted that a large percentage of water samples from mountainous areas of Colorado contained high coliform counts indicating possible contamination from faecal sources such as septic tank disposal systems. They conducted a study to determine the fate of septic tank effluents in this mountainous terrain typically lacking well developed soil profiles and underlaid by fractured crystalline rock. Inoculated waters containing B. stearothermophilus were injected into boreholes and wells at two geologically different sites (granitic and metamorphic) to evaluate microbial "filtration" along bedrock fractures. The results of these field studies show that bedrock fractures can readily convey polluted waters to shallow groundwater supplies with little microbial removal (only 12 %).

3.2.2 *High or seasonally high groundwater conditions*

Viraraghavan (1978) conducted a study in Ontario, Canada, to monitor the horizontal movement of faecal bacteria from the end of a septic tank disposal field located in a sandy clay soil. The groundwater level at the site

Table 3.2 Penetration and removal of bacteria in the unsaturated zone - Part A: On-site excreta disposal

Site Location	Soil type	Hydraulic loading (mm/day)	Vertical penetration (m)	Bacterial populations (No/100 ml) Influent	Effluent	% removal	Investigator
Columbia, USA	Sand-clayey sand	40-300	0.9	-	-	-	Kligler (1921)[1]
Alabama, USA	Fine sand, effective size: 0.06-0.14	Ponded	0.6	-	-	-	Caldwell (1938c)[1]
Hilversum, Holland	Medium sand, effective size: 0.17	-	1.5	-	-	-	Baars (1957)[1]
Wisconsin, USA	Sand + silt loam	80	0.6 + 0.3	1.7×10^5	0	100	Magdoff et al. (1974)[2]
Wisconsin, USA	Loamy sand	50	0.6	5.1×10^6	30	99.999	Ziebell et al. (1975a)[2]
Wisconsin, USA	Sand loam	24	0.6	2.5×10^5	2-500	100-99.8	Bouma et al. (1974)[3]
North Carolina, USA	Loamy sand	33	0.3	6.9×10^4	0	100	Stewart et al. (1979)[2]
Texas, USA	Sandy loam	82	1.2	1.1×10^6	0	100	Brown et al. (1979)[4]
Texas, USA	Sandy clay	33	1.2	1.1×10^6	0	100	Brown et al. (1979)[4]
Texas, USA	Clay	16	1.2	1.1×10^6	0	100	Brown et al. (1979)
Colorado, USA	Fractured rock	7000	3.5	7.7×10^5	6.8×10^5	11.7	Allen and Morrison (1973)[1]
Mochudi, Botswana	Weathered bedrock	20	2.5 + 5 m horizontal	-	350	-	Lewis et al.[1] (1980)

1 Field data

2 Laboratory column experiments

3 Mound disposal system

4 Lysimiter studies

fluctuated between near ground level during the spring snow-melt to a depth of 2.5-3.0 m during late summer. At the time of the investigation the water table was only 0.15 m below the tile drain (0.6 m below ground level). The septic tank effluent contained 1.6 x 10^5 faecal coliforms/100 ml. Because the depth of the unsaturated zone available for purification was limited, relatively high levels of organisms (100/100 ml) were found in the groundwater even in the most distant observation well (15.25 m). Another Canadian study (Brandes, 1974) reported a reduction in total coliform bacteria from 8 x 10^6/100 ml in the septic tank effluent to 4 x 10^3/100 ml in the groundwater 7.5 m from the tile drain. It was thought that the fill material (stones and boulders) used for the absorption field allowed the effluent to penetrate the groundwater (0.5-2.1 m).

Much of the present research on the topic of groundwater pollution and on-site sanitation is directed toward developing modifications to conventional septic tank systems to make them less likely to cause problems (Kreissl et al., 1978). For example, Ziebell et al. (1975) investigated septic tank effluent purification by two Wisconsin soils in 60 cm columns, subjected to different hydraulic loading rates and temperatures. They found that:

(a) 60 cm of a loam sand or low permeability silt loam was sufficient to remove 95-100 % of the faecal bacteria;

(b) the initial period (first 100 days) was critical until the bacterial film built up on the sand surfaces sufficiently to provide the retentive power necessary;

(c) low temperatures affected the removal process by inducing early soil clogging;

(d) short circuiting through natural soil voids occurred at a loading of 10 mm/day in intact cores of silt loam soil;

(e) low dosing rates gave better removal in the unclogged loam sand.

With low permeability soils, ponding is likely to occur, and this increases bacterial removal because unsaturated flow takes place below the infiltrative surface.

The findings of these and other laboratory studies (Magdoff et al., 1974) have led to the development of a mound system with the septic tank adsorption field in a sand fill medium above the natural soil (Figure 3.2).

Experimental mounds have also been constructed and tested in the field (Bouma et al., 1974 and 1975). One in particular was designed primarily to provide purification because of its location over a shallow creviced bedrock. The fill material used was a sandy loam with a base area of 235 m^2 and a seepage area of 28 m^2. This had a designed maximum loading rate of 1700 l/day (60 mm/day) but during the investigation was loaded at an average rate of 660 l/day (24 mm per day). Samples were taken of the septic tank effluent applied, the percolant, and soil samples beneath the mound (Bouma et al., 1974). Results of these analyses indicate that there was virtually complete removal of faecal bacteria (Table 3.3), and oxidation of nitrogen to nitrate after passage through the mound.

Table 3.3 Monitoring data of a mound system (Bouma et al., 1974)

Liquid samples	NH_4-N (mgN/L)	NO_3-N + NO_2-N (mgN/L)	ORG N (MgN/L)	Total P (mgP/L)	BOD (mg/L)	COD (mg/L)	Total[1] coliforms (no/ml)	Faecal[1] coliforms (no/ml)	Faecal[1] strep. (no/ml)	Total[2] bacteria (no/ml) x 10^5
Septic tank effluent	56 ± 9	<1	6 ± 2	15	90 ± 35	256 ± 80	37,000(15)	2,500(14)	100(13)	100(13)
Fill soil interface[2]	<1	66	<1	14	-	-				
12 cm into topsoil[2]	<1	62	2	13	-	-				
20 cm into topsoil[2]	<1	54	5	8	-	-				
Collection pipe	2 ± 1	54 ± 6[3]	<1[3]	8	0	42	54 (13)	5 (7)	1.8 (9) <0.02(4)	2.2 (11) <0.02(3)

1 Medium values were obtained from log-probability graphs. Numbers in parentheses indicate the number of samples.

2 Sampling 3/8/73 of porous cups inserted in the soil (unsuitable for bacterial analyses due to filtering action).

3 Significant differences were observed between measurements in December (30 ppm NO_3-N; 8 ppm NH_4; 10 ppm Org N) and May (57 ppm NO_4-N; 3 ppm NH_4-N; 0.3 Org N) indicating a relatively low biological activity in the winter period.

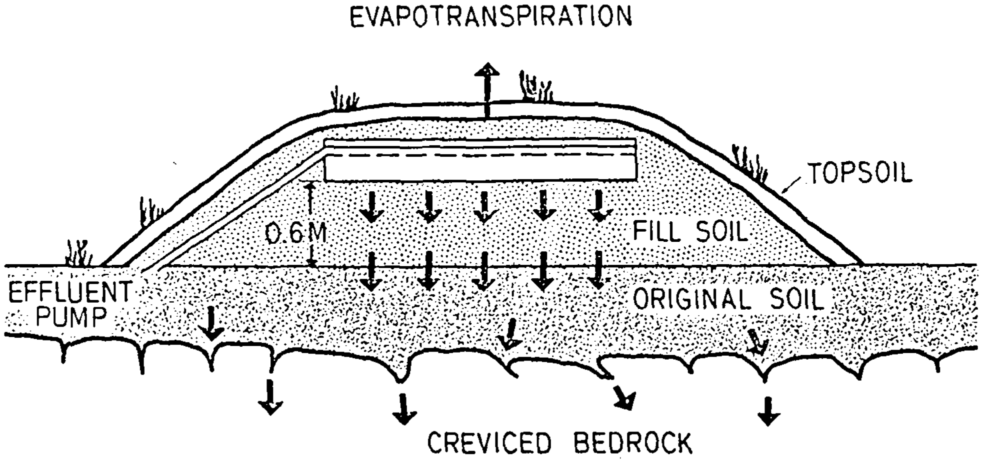

Fig. 3.2 Mound over creviced bedrock (Scalf et al., 1977)

These mound systems have also proved successful for use in areas of thin soils over fissured bedrock, and in conditions of high groundwater table. In the former case, the mound accomplishes most of the required treatment of the septic tank effluent prior to its introduction to the natural soil, thereby protecting the groundwater. In the latter case, a similar treatment justification exists because of the enhanced travel of pollutants under saturated flow conditions. In this circumstance the mound merely lifts the disposal system above the saturated zone during high groundwater periods to ensure the protection afforded by a sufficiently thick zone of unsaturated flow. They have also been found to alleviate problems with the disposal of effluent caused by highly impermeable clay soils (Simons and Magdoff, 1979). These mound systems will not, however, alleviate the problem of nitrate contamination of groundwater.

The data obtained in the US septic tank studies suggest that in permeable soils (but not coarse gravels) without structure, i.e. clayey aggregated soils, there is little likelihood of bacteriological groundwater pollution from an on-site sanitation system, provided there is at least 2 m of continuous soil/unconsolidated strata beneath the system and low hydraulic loadings can be maintained (less than 50 mm/d). It is difficult to generalize on what thickness of unsaturated zone is necessary to remove faecal bacteria at higher hydraulic loadings. The literature on land disposal of sewage effluent reports varying degrees of bacteriological purification (Table 3.4).

For instance, Gilbert et al. (1976) reported a 99.96 % removal of faecal coliform (93/100 ml) after passage through 6 m of a fine loamy sand when treated at 250 mm/d, whilst Schaub and Sorber (1977) observed that wastewater still contained 2000 faecal streptococci organisms/100 ml (99 % removal) after

Table 3.4 Isolation of bacteria beneath land disposal sites

Location	Soil type	Hydraulic loading (mm/day)	Vertical penetration (m)	Bacterial counts (No/100 ml) Influent	Effluent	% removal	Investigator
Flushing Meadows, Arizona	Fine loamy sand	250	6	2.4×10^5	93	99.96	Gilbert et al. (1976) [1]
Fort Devins, Massachusetts	Silty sand & gravel	390-580	18.3	2×10^5	2.0×10^3	99	Schaub and Sorber (1977)[1]
East Meadows, New York USA	Coarse sand, fine gravel & 2% silt	-	11.3	-	$3\text{-}2.4 \times 10^6$	-	Vaughn et al. (1978)[1]
Holbrook, New York USA	Coarse sand, fine gravel & 2% silt	-	6.1 + 45.7 (lateral)	-	$3\text{-}9.3 \times 10^4$	-	Vaughn et al. (1978)[1]
Stony Brook, New York USA	Coarse sand, fine gravel & 2% silt	-	25.6	-	$3\text{-}4.3 \times 10^5$	-	Vaughn et al. (1978)[1]
Bedfordshire, USA	Fissured chalk	400	15	2×10^6	1.3×10^5	93.5	Edworthy et al. (1978)[1]
Lake George, New York USA	Sand	-	1.5	1.3×10^3	20	98.5	Aulenbach (1974)[1]
California, USA	Fine sandy loam	-	1.2	2×10^5- 5×10^6	0-2	100	Butler et al. (1954)[2]
Burnham, New Zealand	Alluvial gravels	-	15	2.5×10^4	55	99.8	Martin and Noonan (1977)[1]

[1] Land disposal of sewage

[2] Lysimiter studies

passage through 18.3 m of silty sand and gravel treated at 390-580 mm/d. Vaughn et al. (1978) monitored human enteroviruses and faecal bacteria in groundwater samples collected beneath three sewage recharge installations over a period of 1 year. The soils in the recharge basins were composed of coarse sands and fine gravels, and high rate disposal was practised. The number of coliforms in the samples collected varied from 0 to 4.3 x 10^5/100 ml even at the site where there was 25.6 m of unsaturated material present. This variability could not be explained, and illustrates the importance of long term monitoring.

3.3 VIRUSES

Little data exist on virus contamination of groundwater supplies associated with the extensive use of on-site sanitation. Virus determinations are expensive, require specialized laboratory facilities and highly trained personnel. Hence, few laboratories monitor viruses in water supplies on a routine basis. Furthermore, methods are only available for less than half of all the viruses known to be present in human wastes (Keswick and Gerba, 1980). For instance, it is still not possible to detect hepatitis A and many agents of viral gastroenteritis. Also surveys are hampered by the lack of a standardized method for the detection of viruses. Current recommendations require drinking water to be free of human enteric viruses in 100 l to 1000 l samples tested (WHO, 1979). It is improbable that untreated groundwater underlying on-site sanitation schemes will conform to such a standard, and it is debatable whether such a stringent standard is justifiable for untreated groundwater supplies.

3.3.1 *Case histories of groundwater pollution by viruses*

In the past the demonstration of viruses in potable groundwater supplies was essentially confined to those sources where an outbreak of illness had occurred. For example, Neefe and Stokes (1945) described an extensive outbreak of infectious hepatitis at a summer camp in the USA. Over a 13-week period 350 out of 572 campers became infected. Transmission studies indicated that the disease was waterborne, being derived from a covered well which was contaminated by nearby cesspools. The cesspools were approximately 2 m deep and were located 23-55 m distant from the well (6.7 m deep). The soil depth in the camp varied from a few cm to 1.8 m and was underlaid by fissured red shale and limestone.

An epidemic of infectious hepatitis was attributed to contamination of well water by septic tank effluent in Posen, Michigan, USA (Vogt, 1961). The wells were sunk in highly fissured limestone, and it was thought that the high transmissibility allowed rapid recharge and lateral movement of septic tank effluent to the well.

Van der Velde (1973) isolated poliovirus from a well responsible for a gastroenteritis outbreak in Michigan. The well passed through 2.5 m of clay, a limestone shale formation, and finished in limestone at a depth of 35 m. It was cased to a depth of 7.5 m, and an attempt had been made to grout the casing. The pollution source was a septic tank drain field located 43 m from the well. Coliform levels in the well ranged from 0-16/100 ml but no Salmonella or Shigella were found. Although poliovirus was isolated, the outbreak was probably attributable to some other virus. Also, Wellings et al. (1975) detected poliovirus in water collected 3 m below a cypress dome receiving secondary sewage effluent.

A recent study in Israel (Marzouk et al., 1979) indicated that 20 % of 99 shallow groundwater samples (3 m) analysed contained enteric viruses; the source of these viruses was thought to be septic tank effluents. Viruses were isolated from 12 samples (20-400 l) which contained no detectable faecal bacteria in the 100 ml sample volumes tested. Hence, enteric viruses may be present in water that shows little or no sign of bacterial pollution.

3.3.2 *Land disposal studies*

Most of the studies concerning possible contamination of groundwater with viral pathogens were in connection with land disposal of sewage effluent. Viruses are much smaller than bacteria and removal is dependent almost entirely on adsorption, thus, of all the pathogens present in sewage, viruses are the most likely to find their way into groundwater during land application (Gerba, 1979).

Wellings et al. (1974) recovered viruses from groundwater after spray irrigation of secondary sewage effluent onto a sandy soil. Penetration through 6 m of the soil was attributed to heavy rainfall. In contrast, Gilbert et al. (1976) did not recover any viruses in groundwater samples collected 6 m beneath sewage spreading basins composed of fine loamy sand underlaid by coarser sand. Schaub and Sorber (1977) studied the migration of coliphage F2 and enteroviruses through soils underlying a rapid infiltration wastewater application site. The site was composed of unconsolidated silty sand and gravel, and the groundwater table was approximately 18.3 m below ground level. Tracer F2 virus was applied, and was detected in the groundwater directly beneath the disposal site after 2 days. The concentration remained constant at approximately 47 % of the average applied load after 3 days. The tracer and indigenous enteroviruses were also detected sporadically in wells 183 m deep at concentrations of 4-8 % of the applied effluent.

Vaughn et al. (1978) conducted a survey of human virus occurrence in groundwater recharged with sewage effluent at Long Island, New York. The recharge sites contained coarse sandy gravel soils with 2-4 % silt. Viruses were recovered at depths up to 11.3 m, and at distances up to 45.7 m from the recharge site. Virus concentrations between 0-2.8 pfu/l (plaque forming units) were reported in about 20-33 % of the 40 litre samples collected. Edworthy et al. (1978) recovered viruses in groundwater 15 m beneath a sewage effluent and disposal site located on an outcrop of chalk. Virus concentrations were 63 pfu/l at the groundwater table, but zero in boreholes 100 m away. No viruses were found to penetrate 8 m of hunter sandstone at a site where polluted river water was used for recharge of groundwater (Edworthy et al., 1978).

It is difficult to draw conclusions from the land disposal data since the hydraulic loadings to the soil may be many times greater than that of a properly designed excreta disposal system. In general, viral concentrations in the effluent reaching the water table were very much reduced (>95 % removal) although considerable thicknesses of the unsaturated zone were needed to achieve this degree of purification (Table 3.5). Once they enter the saturated zone viruses can be transported with the groundwater flow for considerable distances. For example, an investigation in New Zealand (Noonan and McNabb, 1979) detected tracer organisms (T4 phage) 920 m distant from the point of injection into fast-moving groundwater in alluvial gravels. The rate of movement of the virus was approximately 300 m/d, which was similar to that reported for another study using bacterial tracers at the same site (Pyle et al., 1979).

Table 3.5 Isolation of viruses beneath land disposal sites

Site location	Soil type	Influent PFU/l	Effluent PFU/l	% removal	Vertical travel (m)	Horizontal travel (m)	Hydraulic loading rate (mm/d)	Remarks	Investigator
St Petersburg, Florida USA	Sand	2.98	-		6	-	10-50	Spray irrigation deep penetration attributed to heavy rains	Wellings et al. (1974)
Cypress Dome, Florida USA	Sand + 19% clay	161	0-0.005	99.99	3	7	14*	*discharge from septic tanks, heavy rains	Wellings et al. (1975)
Flushing Meadows, Arizona USA	Fine loamy sand	1.6-75	0	100	< 6	< 3	250	Land disposal viruses not detected	Gilbert et al. (1976)
Fort Devins, Massachusetts USA	Silty sand and gravel	276	4.2-10.6	96.2	18.3	183	-	High rate land disposal	Schaub and Sorber (1977)
East Meadows, New York USA	Coarse sand, fine gravel, 4.2% silt	0-26	0-1	96.15	11.3	3	-	High rate land disposal	Vaughn et al. (1978)
Holbrook, New York USA	Coarse sand, fine gravel, 4.2% silt	0-282	0-2.8	99	6.1	45.7	-	High rate land disposal	Vaughn et al. (1978)
Stonybrook, New York USA	Coarse sand, fine gravel, 2% silt	0-97	0	100	25.6	5.5	-	High rate land disposal	Vaughn et al. (1978)
Bedfordshire UK	Fissured chalk	5500	63.4	98.85	15	<200	400	Viruses absent in well 200 m away	Edworthy et al. (1978)
Nottinghamshire UK	Sandstone	-	0	100	<8	-	360	Recharge of polluted river water	Edworthy and Downing (1979)
Burnham New Zealand	Alluvial gravels	59000[1]	10	>99.98	15	920[2]	0-60	Horizontal rate 300 m d^{-1}	Noonan and McNabb (1979)

[1] Coliphage M12 (PB15)

[2] Coliphage T4 tracer tests

CHAPTER 4 : NITRATE POLLUTION OF GROUNDWATER

4.1 NATURAL OCCURRENCE OF NITRATES

Nitrates in soil and groundwater result from the microbial degradation of organic nitrogenous material such as protein to ammonium ions (NH_4^+) which are then biologically oxidised to nitrite and nitrate, in a two step process.

$$2NH_4^+ + 2OH^- + 3O_2 = 2NO_2^- + 2H^+ + 4H_2O \quad (1)$$

$$2NO_2^- + O_2 = 2NO_3^- \quad (2)$$

These two reactions are carried out by different bacteria: Reaction (1) by Nitrosomonas, and reaction (2) by Nitrobacter; and both organisms are aerobic chemolithotrophs. Higher plants assimilate nitrite from the soil after reducing the nitrate to nitrite, and this reaction is catalysed by the enzyme nitrate reductase. Bacteria can also reduce nitrate to nitrite. However, because nitrite is easily oxidised to nitrate the concentration of nitrites in surface water is usually very low (generally less than 0.3 mg NO_2-N/l).

4.2 SOURCES RELATED TO MAN'S ACTIVITIES

One major source of nitrates are farm animal wastes and certain human excreta disposal practices. Animal wastes are rich in nitrogenous materials that may be converted into nitrates, and the problem is particularly acute where farming is carried out intensively, e.g. in feedlots (Gilbertson et al., 1970; Adriano et al., 1971). The amount of nitrogen in human wastes is estimated to be about 5 kg per person per year (Committee on Nitrate Accumulation, 1972). Ammonium ions in the effluent of septic tanks may be rapidly converted to nitrate which can then penetrate some distance into the subsoil beneath the septic tank absorption field.

Also the increased use of fertilizers in farming, from 15.8 million tonnes as N in 1962-1963 to 42.3 million tonnes in 1975 (United Nations, 1976), has resulted in elevated concentrations of nitrate in the groundwater below these fertilized areas (Schmidt, 1972; Nightingale, 1972; Pratt et al.,1972; and Foster, 1976)[1)].

Woodward et al. (1961) attributed the cause of groundwater nitrate contamination in unsewered areas of Minnesota to the widespread use of septic tanks and seepage pits. In a detailed literature review of septic tanks and their public health and environmental impact (Patterson et al., 1971), the consistently poor performance of septic tanks led to recommendations that other waste disposal methods be used in densely populated areas if extensive groundwater contamination problems were to be avoided. Brooks and Cech (1979) found that nitrate contamination of groundwater was widespread in rural areas of Texas. Background nitrate concentrations in the unsaturated zone were found to be in the range of 0-1 mg NO_3-N/kg. Nitrates higher than this were of animal and of human origin, the principal source being septic tanks.

[1)] Grassland leaks very little NO_3-N, even when heavily fertilized

Walker et al. (1973b) calculated that in Wisconsin, USA, the average nitrogen input reaching the groundwater per year was 7.5 kg for a family of four people discharging septic tank effluent into sandy soils. His data suggested that the only active mechanism of lowering the nitrate content was by dilution with uncontaminated groundwater. Relatively large areas were needed before concentrations in the top layer of groundwater were lower than 10 mg NO_3-N/l. Nitrate contamination of groundwater was also found to be a particularly severe problem in a densely populated low-income residential area in Delaware (Robertson, 1980). The area was not sewered, and relied entirely on septic tanks; 28 % of the supplies tested in the area had a nitrate concentration exceeding 17 mg NO_3-N/l. Recharge in the sandy, well drained soils was estimated to be approximately 535 mm/year.

Hutton et al. (1976) attributed widespread and severe nitrate contamination of shallow village groundwater supplies in eastern Botswana to pollution emanating from pit latrines[1]. Lewis et al. (1980) conducted a hydrogeological study in the vicinity of a severely polluted village water supply borehole which had a nitrate concentration in excess of 135 mg NO_3-N/l. The results of this study (Figure 4.1) show that pit latrines caused a major build-up of nitrogenous material in the surrounding soil and weathered rock, from where nitrate is leached intermittently by infiltrating rainfall.

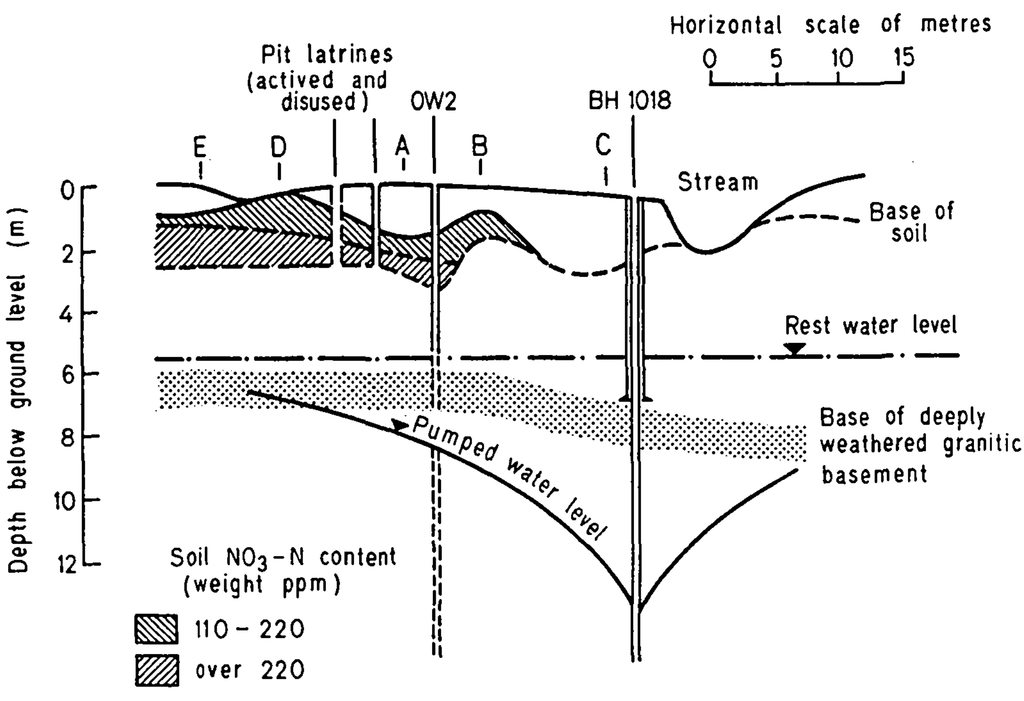

Fig. 4.1 Hydrogeological section of study area showing build-up of nitrates in soil around a pit latrine (Lewis et al., 1980)

[1] Nitrate concentrations of 50 mg NO_3-N/l and higher commonly observed in groundwater supplies located within village limits

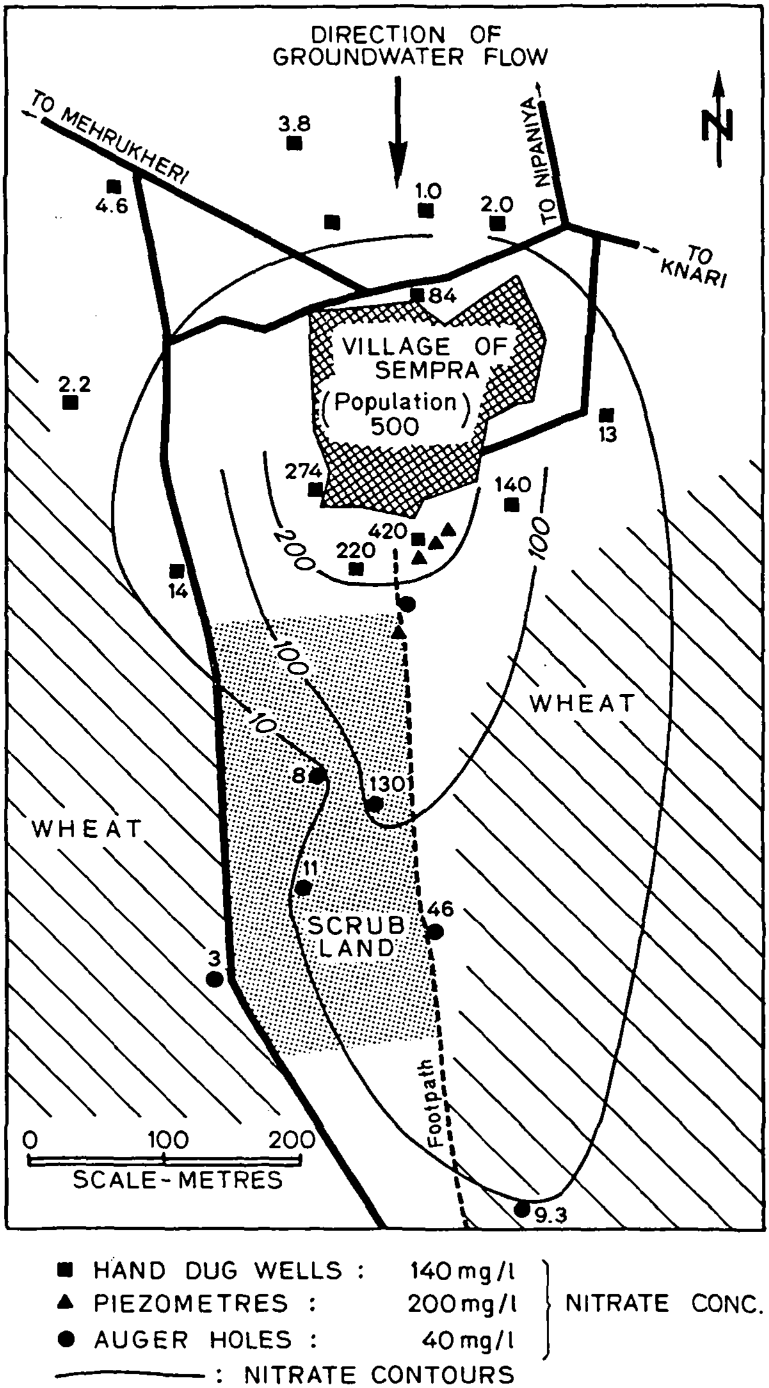

Fig. 4.2 Nitrate pollution plume (Cook and Das,-1980)

The authors calculated that the total mass of readily oxidizable nitrogen in a column of soil from the surface to the bedrock for the sites in the immediate vicinity of the pit latrine (auger holes A-E, Figure 4.1) was 0.1-0.5 kg N/m^2.

Data collected by Cook and Das (1980) in a case study of groundwater pollution in Central India clearly shows a nitrate plume emanating from the village (Figure 4.2). The normal direction of groundwater flow was north-south, and it was concluded that wells south of the village intercept polluted groundwater (30-100 mg NO_3-N/l) resulting from human and animal activities within the village. The village lay in a lowland plain of black cotton soil, and although the rainfall in the area was high (1000-1500 mm annually) it was estimated that only 50 mm was available for recharge.

Thus it is apparent that nitrate contamination of shallow groundwater is likely to be a problem where the density of on-site sanitation facilities is high, and where nitrogen removal and groundwater recharge is moderate to low.

Lewis et al. (1980) observed that nitrate-rich polluted groundwaters also had elevated concentrations of calcium and magnesium, i.e. increased hardness. A similar phenomenon was evident in the data collected by Cook and Das (1980). This is thought to be caused by the process of nitrification which produces hydrogen ions which are able to dissolve more carbonate material present in the soil (Andreoli et al., 1979).

Increasing attention is being paid to soils as disposal systems for secondary sewage effluents and other liquid wastes because the natural chemical and biological processes can be used for purification. The removal of nitrogen may be as high as 90 % in secondary sewage effluent (Table 4.1), and the nitrogen removed may be stored in the soil or diffuse into the atmosphere (Figure 4.3).

Nitrogen not removed by the soil will eventually reach the groundwater as either nitrate or ammonium ion, depending on the amount of oxygen available. The more important nitrogen removal processes are biological denitrification, volatilization of ammonia by aeration, adsorption of ammonium ions, fixation by organic matter, and incorporation into microbial protoplasm. In areas where natural nitrate concentrations in groundwater are low (<2 mg NO_3-N/l), and where agricultural fertilizers are not used, an increase in the nitrate concentration may indicate faecal contamination.

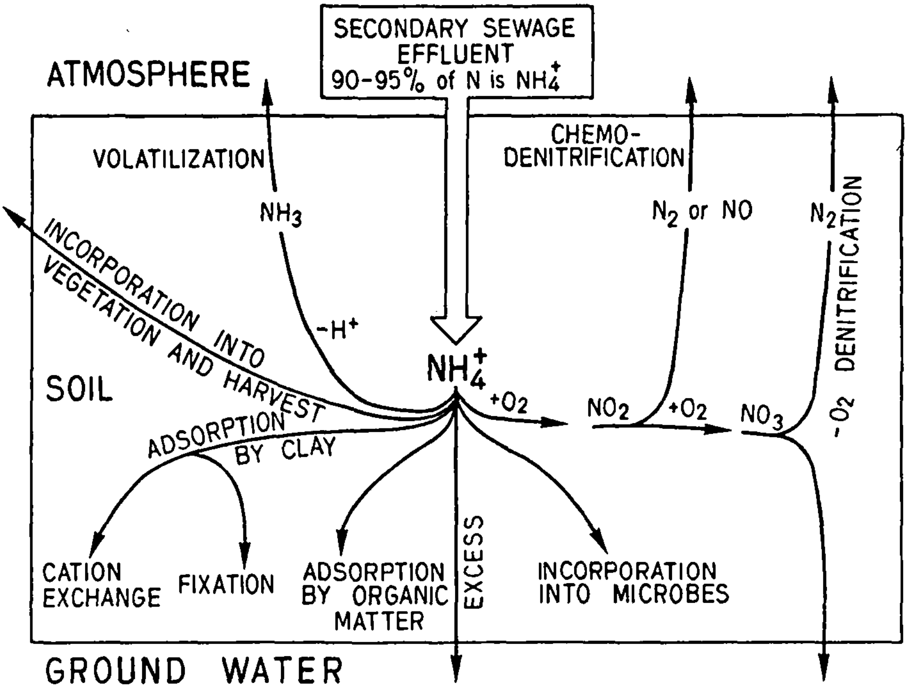

Fig. 4.3 Nitrogen transformations during land disposal of wastewater (from Lance, 1972)

Table 4.1 Summary of nitrogen removal by land disposal - A: Studies related to disposal of septic tank effluent

Location	Soil type	Depth sampled	Loading rate mm/day	Influent nitrogen content mg/l	Calculated nitrogen removal %	Remarks	Investigator
Virginia USA	Sand, silt and clay (50%, 20% and 30%)	1.0	20	12-36	76-77	Changes in chemical content of septic tank effluent during travel in perched water table	Reneau (1979)
Connecticut USA	Coarse sand	0.9	-	102	75	Drainfield alternately dosed and rested for six months	Starr and Sawkney (1980)
North Carolina USA	Loamy sand (83%, 13% and 4%) over saturated organic soil	1.8	3.3	30-55	22-93	Column studies simulating a sound disposal system	Stewart et al. (1979)
Wisconsin USA	Loamy sand	0.6	10-600	78-85	20-80	5 disposal systems studied, greatest removal observed in system submerged in groundwater	Walker et al. (1973a)
Wisconsin USA	Glacial lake deposits	3-6	80	75-85	<20	Estimated input to groundwater 2 Kg N/c/y	Walker et al. (1973b)
Ontario Canada	Clayey sand	0-0.8	50	77-11	15-90	Fluctuating water table; tile submerged for part of study period	Viraraghavan and Warnock (1979)
Wisconsin USA	Sand over anaerobic silt loam	0.9	80	42	32	Laboratory column studies simulating a mound disposal system	Magdoff et al. (1974)
Wisconsin USA	Sand over clayey topsoil	0.6	50	40-58	55	Mound disposal system - low permeability soils and seasonally high water table	Bouma et al. (1975)

Table 4.1 Summary of nitrogen removal by land disposal - B: Studies related to land disposal of secondary sewage effluent

Location	Soil type	Depth sampled	Loading rate mm/day	Influent nitrogen content mg/l	Calculated nitrogen removal %	Remarks	Investigator
Arizona USA	Loamy sand	2.5	150	32	80	Soil column experiments to study denitrification of applied effluent at various infiltration rates	Lance et al. (1976b)
Arizona USA	Loamy sand (89%, 8% and 3%)	2.5	100-400	27	85-20	Losses related to infiltration rates; high losses and rates possible by additions of organic C	Gilbert et al. (1979)
Massachusetts USA	Wet organic soil	-	-	45	90-95	Laboratory study measuring denitrification of applied nitrate solution	Bartlett et al. (1979)
Florida USA	Wet organic soil	-	-	50	80	Flooding periodically an organic soil can control N content of drainage water	Terry and Tate (1979)
Frederiks Denmark	Glacial sands and gravel	1.5	50	46	81	Land disposal of sewage effluent	Sørensen (1979) (unpublished)
Nottingham UK	Sandstone	8	360	42	0	Recharge of polluted river water	Edworthy and Downing (1979)
Lake George, New York, USA	Sand	3	-	19	56	Discharge of treated sewage effluent onto natural delta and seepage beds 7.5 m deep	Aulenbach et al. (1974)

CHAPTER 5 : SUMMARY OF THE LITERATURE, AND RECOMMENDATIONS FOR FURTHER WORK

In order to draw together all the information reviewed in the literature, the important conclusions will be reiterated, together with recommendations on suitable topics for further work.

5.1 SUMMARY

5.1.1 *Unsaturated zone*

(a) Soil (and unconsolidated strata) is a very effective wastewater purification system, having the ability to remove faecal micro-organisms and break down many chemical compounds. The unsaturated zone is the most important line of defence against pollution of underlying aquifers.

(b) Maximisation of effluent residence times in the unsaturated zone is the key factor in the removal and elimination of bacteria and viruses since the degree of purification will be determined by their retention and survival on soil particles.

(c) After the onset of clogging around the latrine pit, bacterial and virus removal processes (physical filtration and adsorption) will be enhanced because the rate of effluent infiltration will be restricted, and liquid movement will only occur in the smaller pores of the soil or rock with greatest media-liquid contact. Thus it is important to avoid high hydraulic loading rates during the initial period of usage.

(d) Generally, the risk of faecal groundwater pollution is minimal when the thickness of relatively fine (< 1 mm), continuous unsaturated soil (unconsolidated strata) beneath the base of the latrine is greater than 2 m, and provided the hydraulic loading does not exceed 50 mm/d.

(e) Groundwater pollution is likely in areas where the water table is shallow, or seasonally shallow, and in areas where fissured bedrock is overlaid by shallow soils. Standard latrine designs dictate that 1-1.5 m of soil is removed, and this increases the chance of pollution in these high risk areas.

(f) Fissures in consolidated rock formations permit rapid effluent movement to underlying groundwater with little or no microbial removal. Similarly, macropores in soils can also behave like fissures. When subjected to high hydraulic loading rates, these discontinuities will allow short-circuiting, and consequently poor bacterial filtration. Under certain circumstances, high intensity rainfall may promote such short-circuiting.

(g) Bacteria and viruses that do penetrate the clogged interface of the latrine may be immobilized by adsorption and can survive in moist soils for long periods. Heavy rainfall may cause desorption resulting in a sudden increase in numbers of micro-organisms entering the groundwater.

5.1.2 *Saturated zone*

(a) Bacteria and viruses in the saturated zone have been observed to travel several hundred meters with the groundwater. The maximum extent of their travel is governed principally by the groundwater flow velocity, and data from the literature indicate that this is the distance the groundwater flows in a period of about 10 days. The distance over which they can be traced depends on other factors such as their survival rate, initial concentration, dilution and dispersion within the groundwater, and the sensitivity of the method used to detect them.

(b) Decrease of bacterial numbers in the saturated zone is due primarily to die-off. The extent of micro-organism travel is limited by adsorption in slow-moving groundwaters (< 1 m/d), and by physical filtration in very fine-grained formations (with pore diameters of less than 0.5×10^{-6} m). Mechanisms responsible for the loss of viral infectivity are not well understood. Possible explanations include degradation by bacteria, or physical damage to the viral ribonucleic acid caused by environmental stress.

(c) It is very difficult to establish a safe minimum lateral spacing between a water supply and an on-site sanitation unit. This is because of the complexity of factors such as permeability and hydraulic gradients which control saturated flow rates.

(d) The extensive use of on-site sanitation schemes will almost inevitably lead to increased nitrate levels in the underlying aquifers. Nitrate concentrations in groundwater in excess of 50 mg NO_3-N/l are undesirable as they may cause methaemoglobinaemia in neonates.

5.2 INFORMATION GAPS AND RECOMMENDATIONS FOR FURTHER WORK

(a) There is very little information on groundwater pollution from developing countries, and no published information available regarding the risk of contamination to water supply mains where intermittent flow is likely. Practical experience indicates that there is a very real risk of contamination where these mains are located below the groundwater table and intermittently depressurised.

(b) There are no published guidelines for the design of on-site sanitation schemes to prevent groundwater pollution. Also, there is a need for a classification of hydrogeological environments in relation to pollution risk. This would be of great value in the appraisal and implementation of on-site sanitation schemes.

(c) The majority of the field studies have been confined, in the main, to fine-grained sediments which are of low risk, and the most suitable for on-site sanitation. There is a need to obtain more information on other soil types.

(d) The North American studies on groundwater pollution from septic tank effluent ave been a major source of information. However, differences in the design and construction of septic tank disposal systems, and the proposed sanitation systems (VIP and PF latrines) may be significant. The most important differences are:

- Septic tank disposal systems generally rely on shallow excavations (frequently less than 0.6 m). The soil at this depth is usually more permeable than at deeper levels, and would normally be expected to have some organic content and to retain some natrual microbiological activity. Enhanced bacterial (and virus) removal has been reported with soils possessing an organic fraction. The proposed excreta disposal systems employ deeper excavations which typically remove at least 1 m of soil/unconsolidated strata.

- The nature of the effluent discharged to these systems will also be different from septic effluent. Raw sewage contains a much higher bacteria and virus concentration than the septic effluent. Thus even a 99.9 % bacterial removal efficiency by the unsaturated zone will still leave a considerable number of coliform bacteria in the percolating effluent. However, this may be counteracted by a thicker and less permeable clogging mat in the latrine which will restrict the rate of liquid flow, and hence increase the degree of purification by the unsaturated zone.

- the limited range of hydraulic loadings used. For septic tanks the hydraulic loading is usually less than 50 mm/d, based on the bottom area of the disposal system only. It is anticipated that pour-flush latrines will receive between 50-100 l/d in 1 meter diameter soakaways, i.e. hydraulic loadings of 90 $\pm$ 30 mm/d (calculated for base area only); this figure will be considerably increased if any domestic waste water is also added, for instance the familiar practice of bathing in the privacy of the latrine. Studies on the land disposal of sewage effluent indicate that there is a serious risk of groundwater pollution in areas of shallow water table when high loading rates are used.

(e) Insufficient is known about the potential risk of viral contamination of groundwater associated with on-site sanitation. Enteric viruses may be present in water that shows little or no sign of bacterial pollution (especially anaerobic groundwaters). Viruses are potentially more dangerous than bacteria because of the low infectious doses necessary to cause disease.

(f) Bacterial studies suggest a possibility of survival for 100 days or more, whereas field monitoring studies imply a much shorter survival time in the groundwater. It has not been established whether this is due to the elimination of the micro-organisms, or to other factors such as the low initial concentration, and dispersion within the groundwater body itself.

CHAPTER 6 : DISCUSSION

6.1 ENVIRONMENTAL CLASSIFICATION OF POLLUTION RISK

No comprehensive guidelines for the safe separation between a water supply well and an on-site sanitation unit could be established from the literature review. Such guidelines must identify in which soil profiles and hydrogeological environments the "traditional" separation of 15 m (a) can be reduced, (b) is acceptable, (c) should be increased to reduce risk and, (d) involves high risk and specialist advice should be sought. In view of the difficulties in formulating guidelines monitoring of groundwater is essential at least during the initial stages of on-site sanitation schemes.

In order to formulate guidelines, classification of hydrogeological environments will be a prerequisite. Because of the complex factors involved, and the importance of rather detailed hydrogeological considerations, it is reasonable to ask the question: why attempt to classify? It would be more logical to treat each settlement or site on individual merit when assessing the faecal pollution risk associated with unsewered sanitation. However, the economics and logistics of low-cost sanitation schemes are such as to preclude the routine use of (costly) hydrogeological field investigations.

It is further apparent that any classification must be workable with data available to public health engineers and planners from records normally held locally in government offices/agencies, or that can be collected on site following a simple manual. Such data will usually be limited to the following:

(a) details of existing water supply arrangements, especially where groundwater supplies (boreholes, wells, springs) are involved, or where reticulation mains are buried below the local groundwater table;

(b) the general character of the aquifer, and the degree of confinement of its groundwater (that is, the thickness and nature of the strata between groundwater level and the borehole intake);

(c) details of proposed on-site sanitation units, including depth of excavation, characteristics of effluent and maximum hydraulic loading;

(d) the depth to the groundwater table at times of highest groundwater levels;

(e) nature of the strata forming the unsaturated zone profile, with detailed description, including grain-size distributions, clay mineral characteristics and natural moisture contents, of those strata occurring at and immediately below the base of excavation (that is to 3-4 m depth);

(f) topographic character of the surrounding area;

(g) average annual rainfall, estimated average annual "excess" rainfall, where available, and approximate maximum daily rainfall intensity.

The appraisal of the principles of pollution movement in groundwater systems, coupled with the evidence from the literature review of applied investigations, enabled the following key parameters involved in assessing microbial pollution to be identified:

(a) the thickness of the (permanently) unsaturated zone (below the base of the on-site sanitation unit) and the nature of the soils and rocks of which this is composed; and

(b) the degree of confinement and character of the aquifer horizons from which groundwater supplies are drawn.

The problem of nitrate pollution is discussed in a later section.

The situation is expressed diagrammatically in Figure 6.1: the safe separation between a water-supply borehole and an on-site sanitation unit (x) being a function of z_1 and z_2, coupled with the physico-chemical character of the strata present in these intervals. Other important parameters are the latrine hydraulic loading and borehole pumping rate, and any guidelines would have to be qualified by these parameters. The aquifer hydraulic gradient can also be significant, but in many cases it is not possible to easily determine its precise direction and magnitude. Hence this,parameter should not be included in any general guidelines. Finally, in naturally anaerobic groundwater systems,viral pathogens may behave differently to indicator bacteria, but it is not considered practical to include groundwater oxygen levels in the guidelines.

A preliminary guideline in the form of an algorithm (Figure 6.2) has been drawn up, covering only shallow boreholes in unconfined aquifers. This is presented for discussion purposes only since more information is required for its consolidation and extension. Such information will have to be collected by detailed field research, and by routine monitoring of pilot on-site sanitation schemes.

Included in the key parameters above were the character of the soil and rocks forming the unsaturated zone, saturated zone and confining beds of aquifers. The pertinent characteristics of these strata (pore-size distribution, vertical hydraulic conductivity-tension relationship, saturated hydraulic conductivity, etc.) are difficult and/or costly to determine. Therefore, in the formulation of guidelines, the very wide range of naturally occurring soils and rocks will have to be grouped on a relatively simple basis.

A tentative grouping, within a simplified framework of relative pollution vulnerability, is given in Figure 6.3. However, no such grouping can be fully comprehensive and free from ambiguity. The vertical sub-division in the array, made on the basis of degree of consolidation, is fundamental. The vast majority of all consolidated deposits (rocks) are traversed by discontinuities (fissures) at some scale or other which increases pollution vulnerability. Such fissuring is likely to be associated with higher hydraulic conductivity in carbonate and some basic volcanic rocks, than in siliceous and acid rock types. Either a descriptive or a genetic classification of soils, sediments and rocks could be used as a basis. The former, involving grain size and mineral character, may appear, at first sight, easier to apply but since a genetic classification better reflects such factors as stratification and

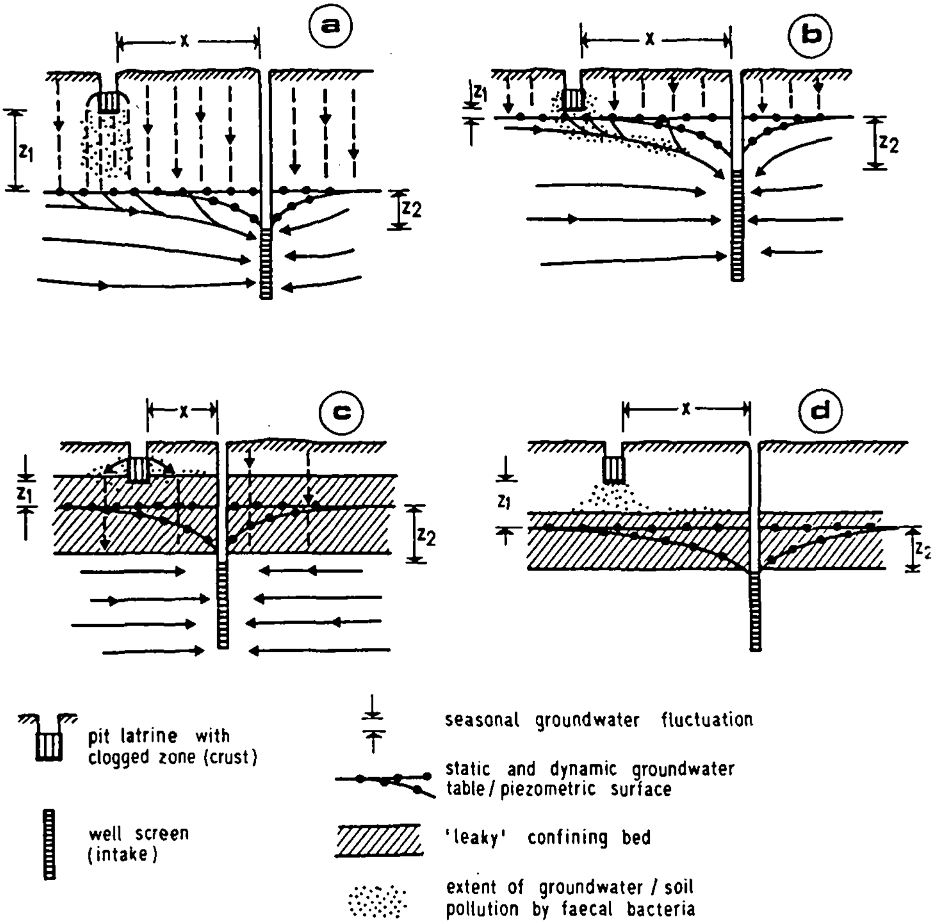

Fig. 6.1 Sketch sections illustrating typical pollution regimes: unconfined aquifer with (a) deep and (b) seasonally shallow groundwater table; (c) semi-confined and (d) fully confined aquifer

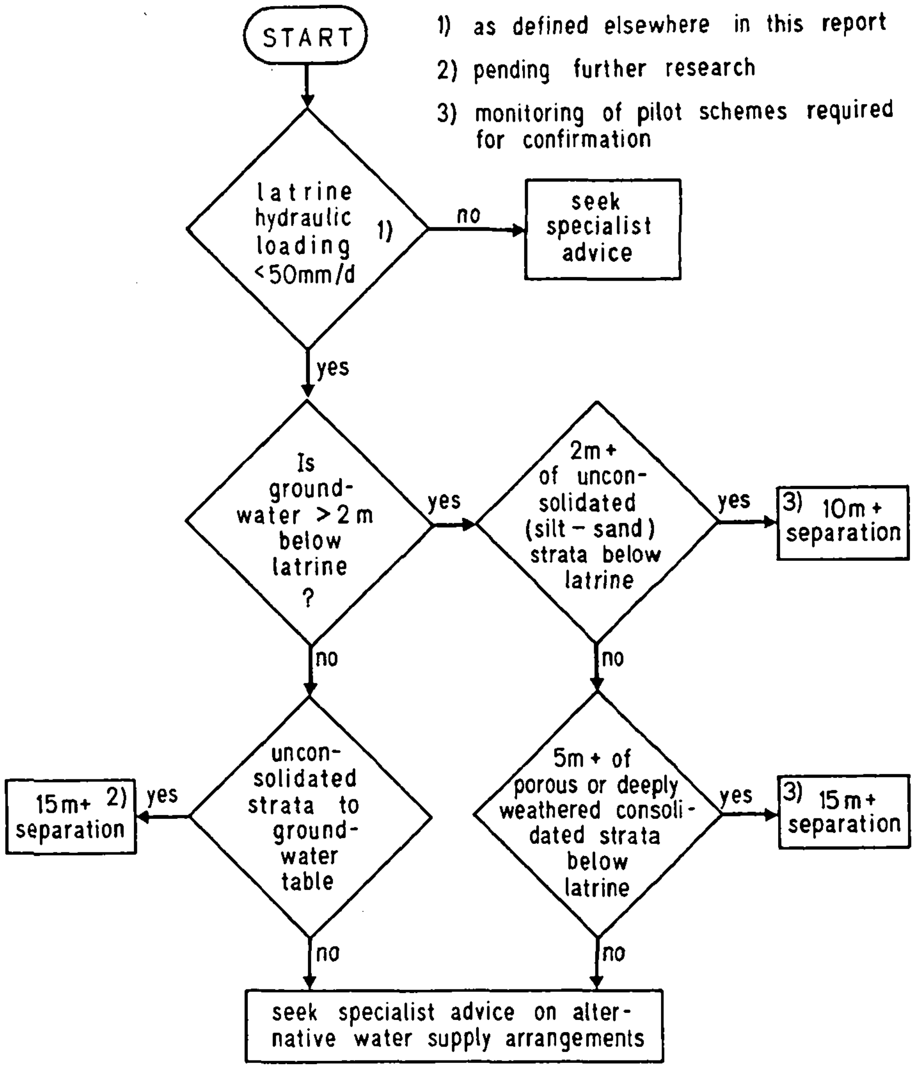

Fig. 6.2 Preliminary algorithm for selecting separation between on-site excreta disposal units and water supply installations in a range of geohydrological environments

structure, which are important in the present context, a hybrid system has been adopted here. It was decided to eliminate most agricultural soils from the grouping, since their thickness is not normally sufficient for them to be present below the excavation base of on-site sanitation units. Only two common soil categories are retained; all other groupings are transported (geological) sediments. These categories are the desert calcretes, and the deep residual soils. The latter form the land surface over a large area of the tropical belt, and because of their inherent variability, and tendency towards deep cracking in the dry season, they are difficult to place with confidence in the array.

This tentative grouping of aquifer types in a risk array can, and has been (Figure 6.3) used to summarise the current state of knowledge of the microbial pollution risk resulting from the close association of on-site sanitation units and water-supply boreholes.

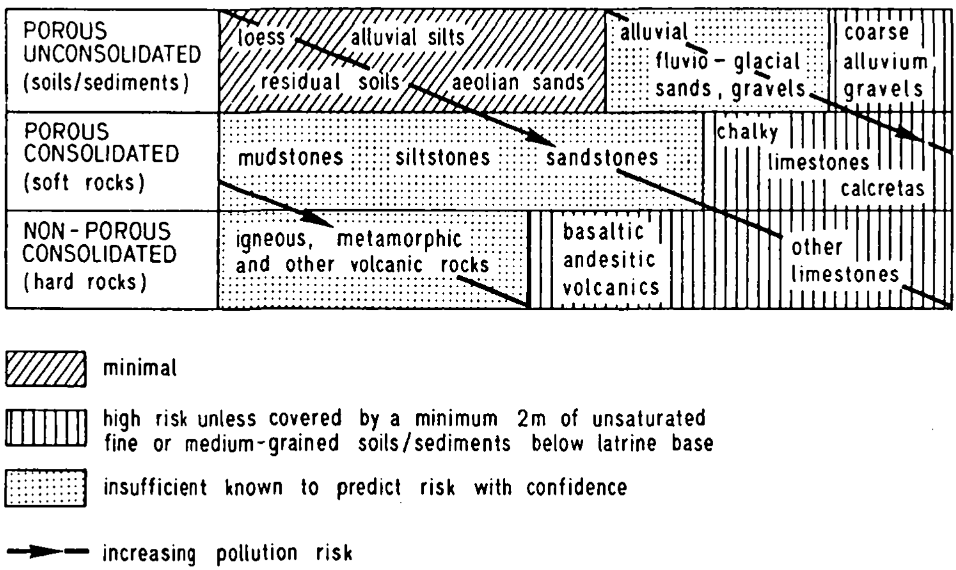

Fig. 6.3 Classification of soils and rocks in an array of relative pollution risk

6.2 FACTORS CONTROLLING THE SEVERITY OF NITRATE CONTAMINATION IN GROUNDWATER BODIES

Nitrates, once they enter a groundwater body, will remain there for a very long time. The overall factors controlling the severity of nitrate contamination are:

(a) efficiency of the nitrogen removal processes beneath the latrine. This will depend on many factors, such as the soil's hydraulic conductivity, the hydraulic loading of the latrine, whether anaerobic conditions favourable for denitrification are established, and the clay/organic content of the sub-soil,

(b) the population using on-site sanitation systems and the density of the units,

(c) dilution by local recharge and regional aquifer through flow, where this has a lower nitrate concentration,

(d) denitrification in the saturated zone. However, groundwater conditions giving rise to denitrification may well be associated with other problems, for example, high concentration of iron, manganese and other metals. Although not a serious health hazard, this may give the water an objectionable taste,and hence cause people to use alternative surface water sources.

Human wastes contain about 5 kg N/ca/annum in the form of ammonium and complex organic compounds, both of which can be readily converted to (highly mobile) nitrate under aerobic conditions. Not all of this nitrogen will reach the groundwater table because some denitrification may occur. Also, urine accounts for some 80 % of the nitrogen excreted, and not all of this will end up in the latrine due to varying urinating habits. It is difficult to estimate the amount of nitrogen lost; however, from biochemical considerations there is little possibility of loss of nitrogen from the pit back to the atmosphere. Figure 6.4 attempts to show the theoretical relationship of nitrogen input (assuming only 10 % actually leached from pit) from on-site sanitation and the amount of local recharge, to the likely nitrogen concentration in water infiltrating to the groundwater body. This is an over-simplification of the problem because there may be a dilution effect when large volumes of waste waters (e.g. bath water, etc.) are discharged to the soil, but it does give some idea of the magnitude of the problem. Figure 6.4 can best be illustrated by considering actual field data. For example in a village in Botswana, Lewis et al. (1980) reported 200 people used 30 pit latrines in an area of 3.2 hectares, i.e. 63 people/hectare. Assuming only 10 % of the total nitrogen excreted by this number of people is leached to the groundwater table, then the total input is:

$$63/\text{ha} \times 5\ \text{kg/annum} \times 0.1 = 31.5\ \text{kg N/annum/ha}$$

The annual average rainfall in south eastern Botswana is approximately 500 mm, with an estimated 50 mm of this infiltrating the groundwater[1]). Thus it is

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[1])The fraction of the annual precipitation that actually results in diffuse groundwater recharge will vary enormously depending on the prevailing climate, local soils and vegetation

evident from Figure 6.4 that with 63 people/ha, and only 50 mm infiltration, a severe localised nitrate problem will exist. Nitrate concentrations in excess of 90 mg NO_3-N/l, i.e. four times greater than the permitted WHO maximum acceptable concentration are not uncommon in village groundwater supplies in Botswana (Hutton, Lewis and Skinner, 1976). Furthermore, 63 people/ha is not considered to be a densely populated area, particularly if one compares it to the extremely high densities found in many parts of the world, e.g. India.

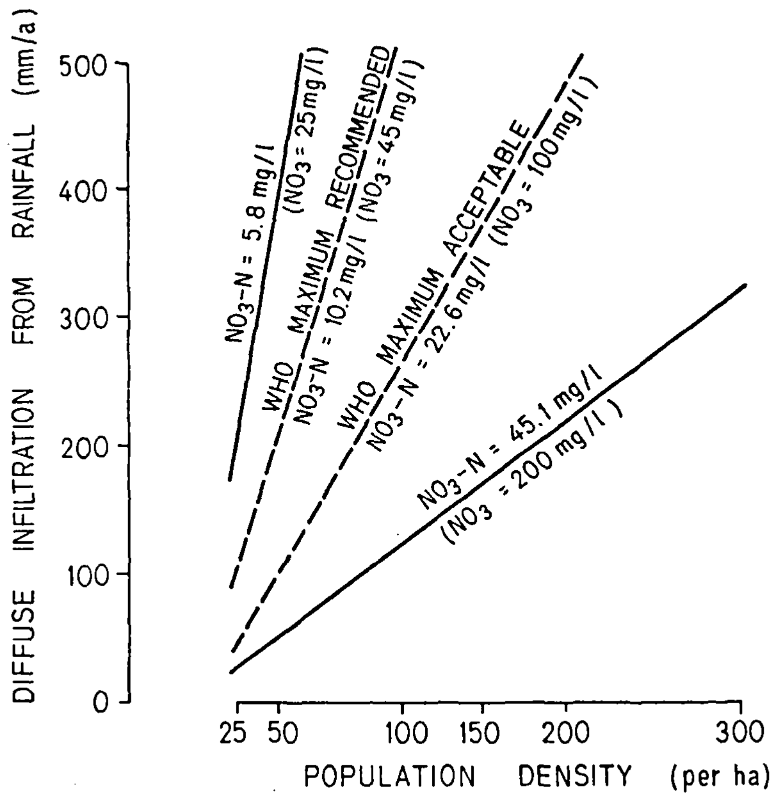

Fig. 6.4 Estimation of the effect of unsewered sanitation schemes on the NO_3-N concentrations in local groundwaters (assumptions: (i) generation of 5 kg N/capita/annum and effluent volume of 1 to 10 l/capita/d; (ii) 90 % of N lost directly to atmosphere from excreta disposal unit; (iii) dilution from diffuse infiltration and effluent liquid only.)

In order to minimise the severity of nitrate contamination it may be worthwhile considering methods of maximising natural denitrification. Some recent studies on the land disposal of sewage effluent suggest how this might be achieved. For instance, Gilbert et al. (1979) and Lance et al. (1976b) investigated the effect of infiltration rate of sewage effluent on the rate of denitrification. Nitrogen removal was increased exponentially by decreasing the infiltration rate from 350 to 150 mm/d which allowed nitrates formed during dry periods to mix with sewage effluent during the succeeding flooding periods (Figure 6.5). Anaerobic conditions will then prevail, and in the absence of oxygen, denitrifying bacteria in the sewage and soil will utilize the nitrates as an electron acceptor (denitrification).

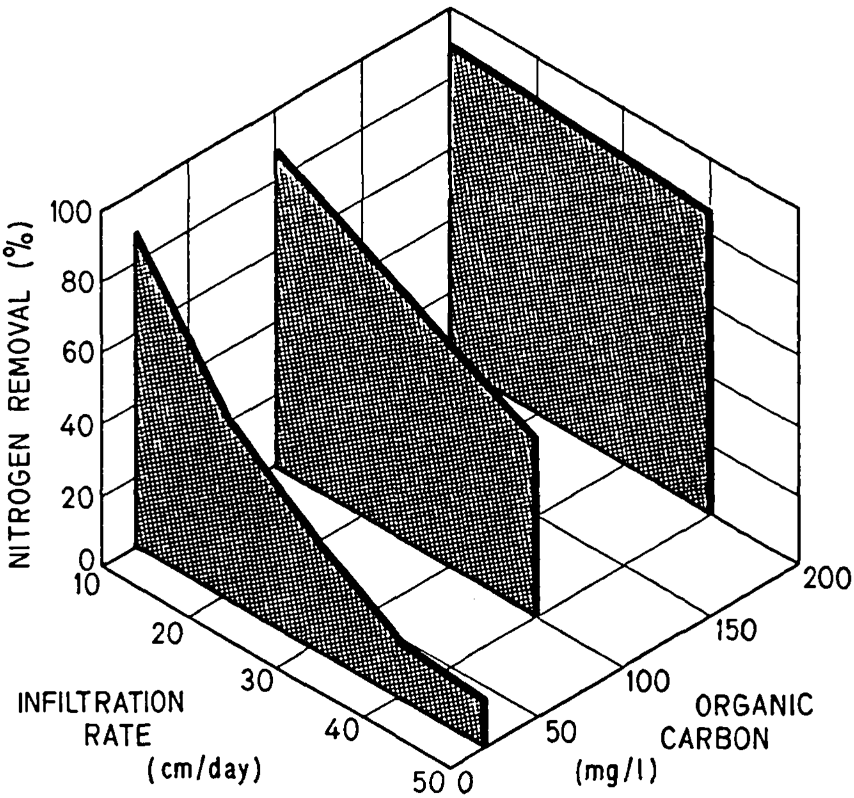

Fig. 6.5 Influence of infiltration rates and continuous C treatments on percent N removal from soil columns intermittently flooded (9 days) with secondary sewage effluent and dried (5 days) (Gilbert et al., 1979)

Gilbert et al. (1979) established that adding a carbon source (glucose, 200 mg C/l) resulted in 80 % N removal at infiltration rates as high as 40 cm/d (see Figure 6.5). A high carbon to nitrogen ratio (C:N) is required in order to promote denitrification, and an extra carbon source was necessary because secondary sewage effluent has an unfavourable ratio. Thus it may be feasible to induce denitrification in twin pit versions of the VIP by simple changes in latrine usage, e.g. by alternating the chambers used. The higher C:N ratio in raw effluent should be favourable for denitrification. The major disadvantage of this method is that constantly alternating the pit does not allow long term drying of the pit contents in order to make them less offensive to handle when emptying, and to allow time for pathogens to die off.

6.3 IMPLEMENTATION STRATEGY FOR ON-SITE SANITATION

It is of importance to discuss how any guidelines for the separation of water supply installations and on-site sanitation units will be utilised in practice. There exists a very strong case, on social grounds, for the provision of sanitation at the individual family or dwelling level. However, in all but the lowest density settlements, plot sizes are relatively small. Thus, it must be recognised that adoption of guidelines is often likely to exclude the possibility of retaining or constructing individual private groundwater supplies (handpump tubewells, "protected" dug wells, etc.). Hence, closure of existing private water supplies and provision or improvement of communal supplies will often be required in parallel with the introduction of on-site sanitation.

Other than where groundwater conditions preclude it, communal handpump tubewells, regularly distributed through the settlement, will generally be the most reliable form of water supply. In hydrogeological environments allowing deep penetration and lateral travel of microbial pollution, the siting of such water supply installations will have to be considered carefully. In environments where the two technologies are, as a result of high pollution risk, incompatible, it will generally be preferable to provide an alternative form of water supply rahter than of sanitation. However, alternative water supplies will almost invariably involve some reticulation mains in which flow is generated by mechanical plant. Reliability and ease of maintenance of this plant, and associated continuity of water supplies, will be an important consideration.

A closely integrated view of sanitation and water supply is thus highly desirable, with an overall reappraisal of water supply arrangements for entire settlements during the planning of on-site sanitation schemes. A piecemeal approach to the provision of sanitation will make the protection of groundwater supplies all the more difficult. Discussion of the relative benefits to be derived from protection of groundwater quality in comparison with investments in other aspects of public health is outside the scope of this report.

6.4 SUGGESTIONS FOR ALLEVIATION OF MICROBIAL GROUNDWATER POLLUTION PROBLEMS

Where the hydrogeological environment should prove to be unsuitable for unsewered sanitation, there are a number of modifications to latrines and/or water supply designs which may minimise the risk of microbiological pollution problems. These include:

(a) minimising the latrine's hydraulic loading by increasing the soakaway area and excluding additions of any household waste water;

(b) maximising the thickness of the unsaturated zone available for purification by constructing a raised latrine (Firgure 6.6 and 6.7);

(c) inducing longer saturated flow times by deepening (with grouting) of water supply borehole solid lining tubes; this will only be feasible in certain aquifer materials.

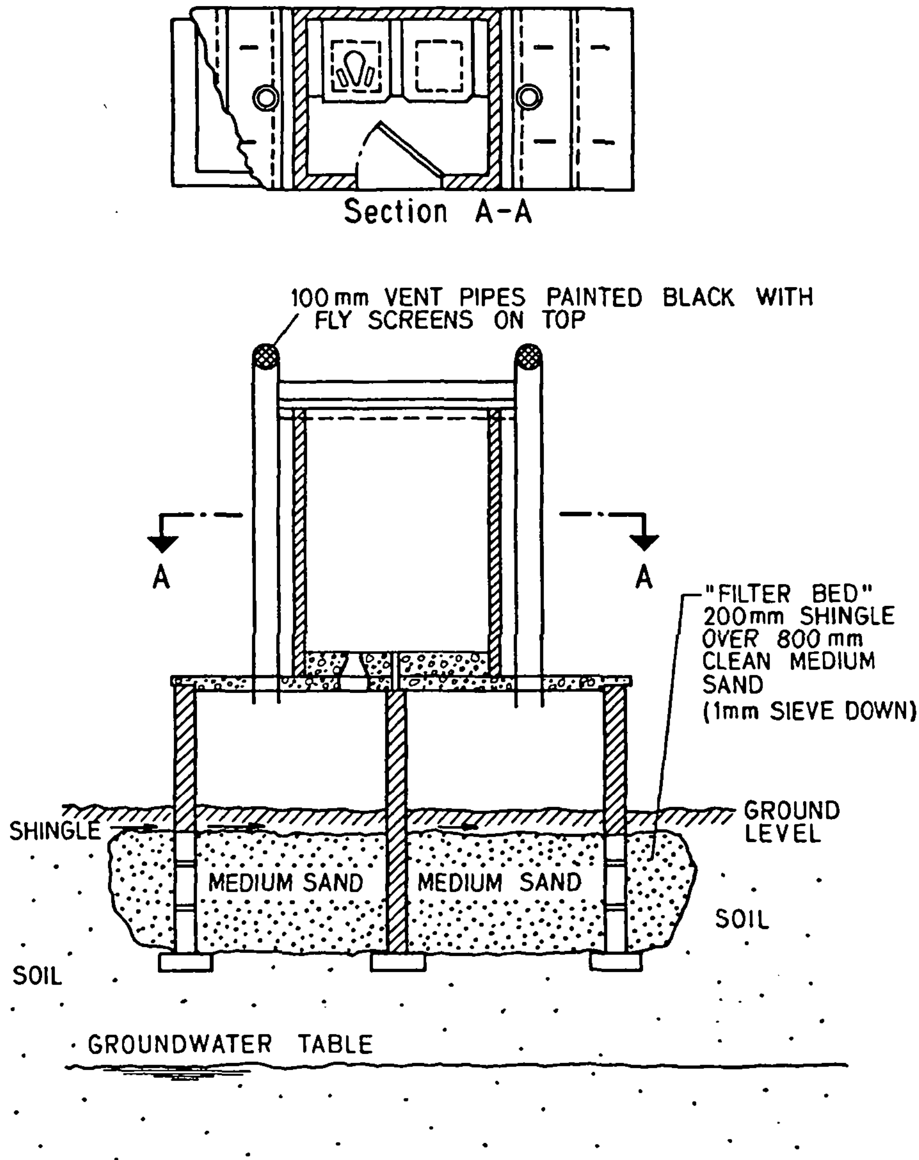

Fig. 6.6 Raised improved pit latrine (RIP) for use in areas where risk of groundwater pollution is high

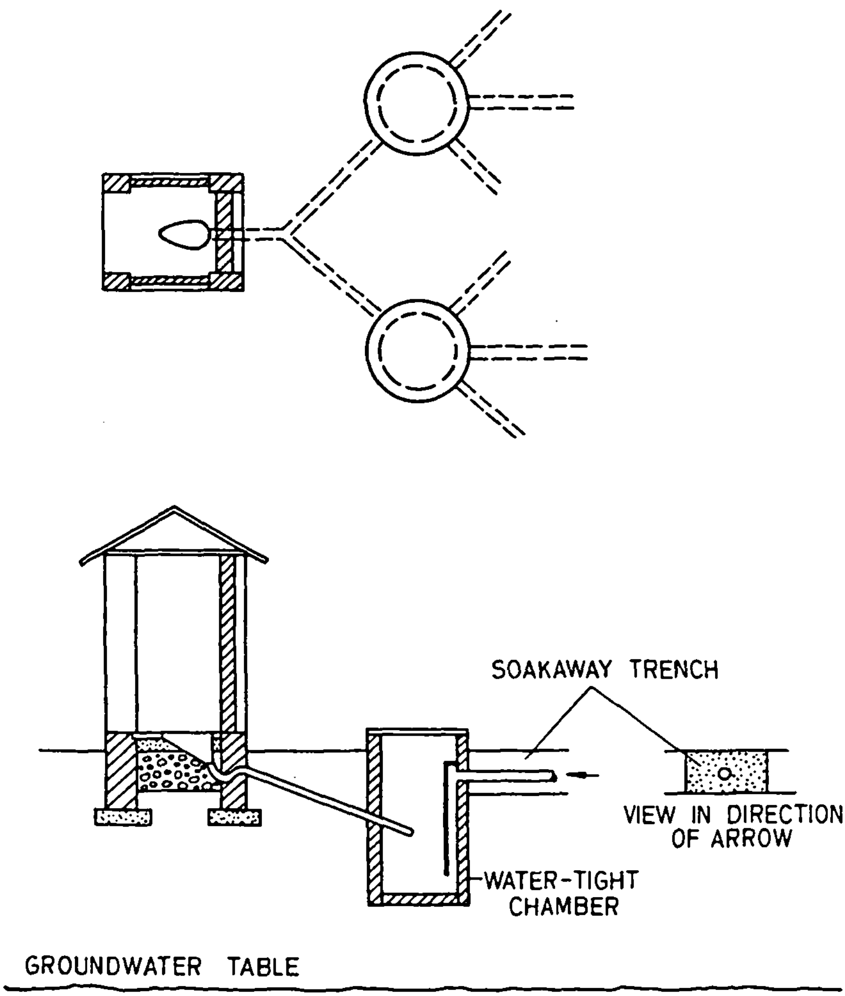

Fig. 6.7 Modified pour flush latrine for use in areas where risk of groundwater pollution is high

Chapter 7 : Conclusions

(a) There is a great need for more research into the extent of groundwater pollution from the type of on-site sanitation systems proposed for use in developing countries.

(b) The research carried out should be on factors which will lead to an improvement of guidelines for future pollution risk assessment.

(c) Priority should be given to the study of pollutant removal and elimination in the unsaturated zone below and around selected excreta disposal units in a variety of soil-hydrogeological profiles.

(d) The research needs to refer to a wide variety of hydrogeological environments. The applied investigation needs are summarized in Table 7.1.

(e) The effect of a range of hydraulic loadings between 25-150 mm/d should be investigated, if necessary by artificial additions of fluids and/or increasing the area of soakaways where these loadings are outside the operational range for local latrines. The hydraulic loadings are calculated solely on the base area of the latrine and ignores the influence of side walls. There is considerable divergence of opinion on this, and any research should take the opportunity of exploring this aspect more fully.

(f) The use of unsewered disposal systems has been shown to cause nitrate contamination of groundwater in many hydrogeological environments. High nitrate concentrations in drinking water have been implicated in the deaths of infants, though the evidence is scanty and often confusing. A vigorous medical investigation is needed to clarify the position. The safe limits of nitrate concentrations in drinking water must be established.

(g) Investigate methods of artificially inducing increased rates of denitrification (possibly by simple changes to latrine usage pattern, i.e. alternate dosing technique).

Table 7.1 Investigations needs

Geological environment	Residual weathering depth and material	TYPE OF INVESTIGATION WORK — Groundwater table depth generally 1-3 m	generally 3-10 m
Fine-grained soils/ sediments (d_{50} size < 0.2 mm)	n/a	$2/3^{BC}$	$2/3^{ABC}$
Coarse-grained soils/ sediments (d_{50} size 2-10 mm	n/a	$2/3^{C}$	$1\ 2\ 3^{C}$
Non-porous siliceous bedrock	< 3 m sandy/silty soil	HR	1 2 3
	3-6 m sandy/silty soil	$2/3^{A}$	$1\ 2\ 3^{A}$
Non-porous non-siliceous bedrock	< 3 m clayey soil	HR	1 2 3
	3-6 m clayey soil	2/3	$1\ 2\ 3^{C}$
Medium-grained consolidated porous rock (sandstone/d_{50} size 0.2-2 mm)	< 3 m sandy soil	HR	1 2 3
	3-6 m sandy soil	2/3	1 2 3
Calcrete/semi-desert soils	n/a	HR	2/3

KEY

1. Detailed investigations at 4-6 individual latrine sites at a specific location, follow-up detailed monitoring on saturated zone.
2. Medium-term monitoring throughout an area with limited local investigation; where population and latrine/well densities are high, sampling could be restricted to existing wells, but in other locations installation of samplers in the groundwater will be required.
3. Medium-term monitoring throughout an area, as above, but in areas without sanitation.

A: Condition not expected to occur widely (work of lower priority or possibly omitted).

B: Deeper semi-confined (leaky) aquifers frequently present, which will be tapped by boreholes or tubewells; necessary to avoid such conditions or ensure aquifer unconfined, unless dug wells are predominant water-supply installations.

C: Characterised by naturally anaerobic groundwater conditions in saturated zone (therefore NO_3^- problem unlikely, but virus migration may be a problem and Fe/Mn quality problems may be present).

HR: Normally high pollution risk, therefore applied investigations not justified.

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