ENVIRONMENTAL IMPACT OF ON-SITE SANITATION A LITERATURE REVIEW WITH PARTICULAR APPLICATION TO SOUTH AFRICA

AB Fourie • MB van Reyneveld

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ENVIRONMENTAL IMPACT OF ON-SITE SANITATION A LITERATURE REVIEW WITH PARTICULAR APPLICATION TO SOUTH AFRICA

Prepared by:

University of the Witwatersrand

A B Fourie
M B van Reyneveld

Department of Civil Engineering
University of the Witwatersrand
Private Bag 3
2050 WITS

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EXECUTIVE SUMMARY

Why was the research done?

The provision of basic services to the rapidly growing population of South Africa is and will continue to occupy a high position on the country's agenda in the coming years. Crucial amongst these services is the provision of water supply and sanitation.

At the recent Water and Sanitation 2000 workshop it was shown that the cost of providing waterborne sewerage for all would be unaffordable for the country, and that provision of Ventilated Improved Pit Latrines for about half of the urban population (as at the year 2000) is the kind of strategy that the country needs to be looking at. The environmental impact of such a strategy is obviously of considerable concern.

Against the background of this concern, there appears to be limited understanding by the general public as well as by many professional personnel in South Africa of the pollution risk from on-site sanitation; more specifically limited understanding of the nature, extent and consequences of contamination from on-site sanitation. Allied to this is the probably erroneous assumption that there is no pollution risk associated with waterborne sewerage.

The lack of understanding of the pollution risk from on-site sanitation may be attributed to:

- The relative newness of the problem in South Africa (there are relatively few on-site sanitation systems currently in use, particularly in the urban areas)
- The limited amount of research carried out to date on the topic in South Africa
- The limited access to the results of research carried out on this topic world-wide

Additional factors are

- The complexity of the problem
- The lack of clear and unambiguous information on the problem

What were the project objectives and the methodology used?

The project objectives were as follows:

- Establish the current state of knowledge on this topic at both local and international level
- Appraise existing guidelines in this regard
- Clarify the objectives that guidelines for environmental impact of on-site sanitation needs to address, with particular reference to South Africa.
- Provide a means of disseminating this knowledge
- Identify further research work that may be required.

The methodology used to achieve the above objectives was as follows:

- Conduct an updated review of local and international literature on the environmental impact of on-site sanitation
- Detail case studies which are relevant to South Africa
- Review existing computer programs for the modelling of contaminant migration such as that which occurs from on-site sanitation, as a means of quantifying the pollution risk.

Who will benefit from the research?

The research report is aimed primarily at professionals with a technical background in the disciplines involved, although it is likely to be of interest to other sections of the community. (It is not aimed primarily at community groups. The key issues in the report will need to be communicated to such groups by a separate specifically written report).

Nevertheless all who have an interest in the wise use of our water resources will benefit from the results of this research in that it will enable more informed choices to be made with respect to the provision of water supply and sanitation.

Background to conclusions and recommendations

Use of on-site sanitation on a large scale in an area of scarce water resources is a significant departure from existing approaches.

Existing approaches appear to follow one of two routes. Both limit the use of on-site sanitation to certain densities of settlement:

- In developing countries, where waterborne sewerage is largely unaffordable, substantial improvements in health are obtained by the use of on-site sanitation. Possible pollution of the groundwater somehow appears less significant in the face of the alternatives (high standards for a few, and minimal coverage for the rest of the population). Consequently, contamination of the groundwater is not considered unless the water is to be used for drinking purposes.

- In developed countries, where waterborne sewerage is largely affordable, it is used primarily for reasons of convenience, but also on the assumption that there is no associated pollution risk.

  Where on-site sanitation has been used on a large scale (as in the case of septic tanks in the USA), it has been at relatively low densities.

The situation in South Africa straddles the above two scenarios. It does not have the financial resources of a highly developed country; at the same time it is a water-scarce country and is concerned not to pollute the water resources it does have.

In seeking to provide access to adequate sanitation for all its inhabitants, one of the options the country has is the widespread use of on-site sanitation at relatively high densities. It needs to be emphasised that the opportunity now exists for the actual <u>planning</u> of relatively high density use of on-site sanitation, particularly in the fast-growing urban areas.

Although the literature certainly does give certain indications on some aspects of relevance to the South African situation, there remain many areas where relevant information is scarce, and thus where further research is required.

Summary of conclusions and recommendations

One cannot give a categorical ruling in the line of: "Yes there is" or "No there isn't pollution from on-site sanitation".

The literature indicates emphatically that there is a *risk* of groundwater *contamination* from on-site sanitation. The risk of contamination from bacteria and viruses (as long as they are not washed out onto the surface) is very low. There is, however, likely to be contamination from nitrates. (Consideration of mass balances alone indicates that this is the case). The primary concern therefore is nitrate pollution.

It is not clear to what extent nitrate can be denitrified in the soil to produce nitrogen gas which will escape into the atmosphere (and hence not pollute the groundwater). It is also not clear to what extent this process can be induced to happen by artificial means.

The rate of movement of nitrate in the subsurface is heavily dependant on a number of local geological factors, which need to be determined for each particular site. An advance on current approaches suggested in this review is the calculation of a residence time in the unsaturated zone, instead of a blanket categorisation of allowable distances between the source of the contamination and groundwater withdrawal points.

Once the nutrients get into an impoundment, their effect on the ability of the water body to support life is also dependent on a number of factors (including whether the impoundment is nitrogen or phosphorus-limited).

In summary we are talking about long term effects that are too significant to ignore, but difficult to determine.

The approach recommended is a multi-facetted one:

- Determine appropriate compliance requirements; (if groundwater is to be used for drinking purposes, then there will be a particular compliance requirement, whereas if surface water resources are the primary concern, and protection of the groundwater is not a consideration, some other compliance requirement will prevail).
- Use the 'residence time' approach rather than the 'allowable distance' approach to determine possible microbiological pollution risk
- Use the mass balance approach to determine the nitrate pollution risk
- Institute field monitoring in order to provide early warning of contaminant build up

In order to address gaps in current knowledge, further research needs to be undertaken.

Recommendations for further research

Research topics that need to be addressed are:

- Quantification of the 'source' (ie the contaminant loading applied to the soakaway)
- Quantification of the movement of these contaminants in the subsurface
- Quantification of the effect of these contaminants on the groundwater and surface water resources (by both modelling and field monitoring)

- Remedial measures to contain and/or render harmless these contaminants
- An assessment of the cost of the pollution

Further topics which impact on the viability of on-site sanitation, and which need to be researched include:

- Treatment, disposal and re-use of sludge from on-site sanitation systems
- Installation and maintenance of on-site sanitation systems, including the management of these operations
- Communication of appropriate information to communities, particularly with regard to on-site sanitation, to enable them to make an informed choice of sanitation system

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1 INTRODUCTION

1.1 Statement of the problem

The increased awareness in South Africa of the need to provide an improved standard of living for a large percentage of the population has led to the identification of many problem areas that require immediate attention. One of the most pressing of these is the provision of basic services to the rapidly growing urban population of South Africa. Policies are being formulated and decisions are being made within a very short time frame which will have major long term consequences for the shape of South Africa's urban areas. It is therefore important that these decisions be as fully informed as possible.

At the recent Water and Sanitation 2000 workshop it was shown that the cost of providing waterborne sewerage for all would be unaffordable for the country, and that provision of Ventilated Improved Pit Latrines for about half of the urban population (as at the year 2000) is the kind of policy that the country needs to be looking at (Jackson 1991). The environmental impact of such a policy is obviously of considerable concern.

In most developing countries, waterborne sewerage is simply beyond the financial reach of most inhabitants, so that on-site sanitation is widely used. In most developed countries, on the other hand, full waterborne sewerage is the norm, although it is estimated that some 60 million people in the USA and Canada use on-site sanitation in the form of septic tank systems (which are waterborne but not sewered) for wastewater (Viraraghavan 1982).

South Africa falls somewhere in the middle: part developed and part underdeveloped. Its GNP per capita places it in the category of middle-income economies, on the border between the lower-middle and upper-middle income categories, along-side countries such as Malaysia, Brazil, Hungary and Argentina (World Bank 1990). It is also an unequal economy with large discrepancies in wealth between rich and poor. Some of its inhabitants have a high level of service; others have very little at all.

South Africa is also a water-scarce country and in response to this it has made a significant contribution to current knowledge in the field of wastewater treatment.

The combination of these factors has brought about resistance to the use of on-site sanitation in the country. This resistance has centred around issues such as:

- A perception that the use of on-site sanitation implies that the country is 'second-class'
- A perception that there is plenty of money in the country for a high level of service
- A disbelief that waterborne sewerage costs as much as it does
- A perception that full waterborne sewerage is a robust system, whereas it is in fact a fragile system that is sensitive to misuse and the use of inappropriate cleansing materials. Furthermore, there is a lack of appreciation of the consequences of failure of such systems.
- A perception that on-site sanitation is unhealthy; that it doesn't work as well as full waterborne off-site sanitation, and will cause disease
- Concern that on-site sanitation may pollute the country's scarce water resources

In order to address these objections, they need to be unravelled and grouped into more systematic categories. Categories which may be used are as follows (they are based on the World Bank Project Cycle (Baum and Tolbert 1985); it should also be noted that there is overlap between categories and that they are not watertight compartments):

- Financial and fiscal (what does it cost and how can these costs be met ?)
- Economic (how should resources best be used to produce growth and development in the country ?)
- Social (what factors affect the acceptability of sanitation systems by individuals and communities ?)
- Institutional (what institutions and educational/training programs need to be put in place to ensure that the sanitation systems work in practice on an on-going basis)
- Technical (how do the various sanitation systems actually work, and how effective are they ?)
- Environmental (what is the impact of these systems on human health and on the environment ?)

This report considers environmental aspects of the problem, together with some technical aspects.

A number of studies on this topic have been carried out in various parts of the world, but this experience does not seem to be common knowledge in South Africa. Current guidelines in this country are accordingly somewhat vague, e.g. a National Building Research Institute publication states that VIP latrines are acceptable for application in most low-density residential areas, (defined as a maximum of 250 people per hectare). No distinction is made on the basis of soil conditions, or the possible utilisation of local groundwater resources.

The first problem therefore is a lack of clarity as to what the problem actually is. If one questions whether on-site sanitation can cause pollution, the answer is an emphatic yes, as shown by Lewis, Farr and Foster (1980b) and Chairuca and Hassane (1991). A far more difficult question, and one that does not appear to have been addressed, is what the implications of this potential pollution are as regards the viability of on-site sanitation in a particular instance, and whether these dangers can be dealt with in the planning and design stages.

1.2 A proposed solution

The following work was carried out:

- A comprehensive updated literature review of the environmental impact of on-site sanitation, both on a national and international basis (coupled with:)
- Detailing of case studies which are of relevance to South Africa.
- A review of existing computer programs for the modelling of pollution migration such as that which occurs from on-site sanitation, in order to be able to quantify the pollution risk.

The aim of the work was:

- To clarify the objectives that guidelines for environmental impact of on-site sanitation need to address, taking particular cognisance of the needs of the relevant decision makers in South Africa.
- To appraise the existing guidelines and modelling methods, and
- To highlight areas where further research on this topic may be needed.

1.3 Key issues to be addressed: a proposed framework

To understand the risk of pollution from on-site sanitation, and to formulate strategies for dealing with it, it is first necessary to establish a framework which defines the key issues to be addressed. The framework which was established for this study is as follows:

WHAT IS ON-SITE SANITATION AND WHAT IS IT INTENDED TO DO ?

- Forms of on-site sanitation, in general and in South Africa in particular
- What is sanitation intended to do ?
- What is on-site sanitation in particular intended to do ?
- What is pollution ?

WHAT DO SANITARY WASTES CONSIST OF ?

- Constituents of sanitary wastes
- The use of indicators for contaminants
- Constituents of effluent from various sanitation systems

WHAT ARE THE HEALTH RISKS ASSOCIATED WITH SANITARY WASTES ?

- Microbiological contaminants: How do you get sick ?
- Chemical contaminants: How do you get sick ?

WHAT ARE THE RISKS TO THE ENVIRONMENT ASSOCIATED WITH SANITARY WASTES ?

- What do we mean by the environment ?
- What is a healthy environment and what can go wrong with it?
- How might on-site sanitation contribute to an unhealthy environment ?
- How scarce is the resource ?

CONTAMINATION FROM OTHER SOURCES

SAFE CONTAMINANT LEVELS

THE FATE OF CONTAMINANTS IN THE SUB-SURFACE

- Relevance of transportation of pollutants
- Mechanisms of transportation
- Characteristics of pollutants affecting their transportation
- Hydro-geological factors affecting pollutant transportation
- Computer modelling of pollutant transport

GUIDELINES FOR ENVIRONMENTAL IMPACT OF ON-SITE SANITATION
- Existing guidelines
- Objectives that guidelines need to address
- A strategy for the evaluation of the environmental impact of on-site sanitation

1.4 What work has already been done ?

A considerable amount of work has been carried out on various aspects of pollution from on-site sanitation. However most of it consists of very specific studies rather than more general overviews. Three studies, which include specific literature reviews or give more general overviews, and which warrant specific mention are as follows:

1 Lewis, Foster and Drasar (1980b) *The Risk of Groundwater Pollution by On-Site Sanitation in Developing Countries: A Literature Review.*

The motivation for the work was the proposed substantial investments in on-site sanitation in the UN Drinking Water Supply and Sanitation Decade coupled with the lack of availability of information on the relationship between groundwater quality and on-site sanitation.

The report concentrated on the risk of pollution from pit latrines and pour flush latrines and gave an overview of the principles of pollutant movement and attenuation in the ground and of applied field investigations.

The study found very little information on groundwater pollution from developing countries. North American studies on groundwater pollution from septic tanks were used as a major source of information, although differences in performance between septic tanks and VIP's or pour flush latrines may be significant.

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. A tentative grouping of aquifer types in a risk array was used to summarise the current state of knowledge of microbial pollution from on-site sanitation, together with a simple algorithm for selecting separation between on-site sanitation and water supply.

For nitrate pollution, tentative indications were given, based on a mass balance approach. A theoretical graph relating infiltration from rainfall and population density to nitrate concentration in water infiltrating to the groundwater, was presented. The graph was based on a number of assumptions, in particular that only 10% of the total nitrogen excreted was leached from the pit (or soakaway). This graph was qualified by a further graph giving an indication of nitrogen removal in relation to infiltration rate and organic carbon, which showed that denitrification was promoted by a high carbon to nitrogen (C:N) ratio.

The report further proposed a number of fairly simple measures for alleviation of groundwater contamination and concluded that there was a great need for further research.

2 Palmer (1981) *Nitrates in the Groundwater Supply to Villages in Botswana*

This MEng paper consisted primarily of the results of a literature review rather than original fieldwork. The study was confined to the issue of nitrates, and dealt specifically with the case of Botswana (although the results could be more widely applied).

A particular contribution was the extension of previous work to include the effect of cattle waste as a source of nitrogen over and above the traditional source of pit latrines.

The study reviewed the effect of nitrate on health, in particular the link between nitrate intake and the development of cancer and commented on the limit for nitrate in drinking water. It looked at the effects of cattle waste as a source of nitrogen. It reviewed the mechanism by which nitrates are formed and leached into the groundwater. It then used the above data to carry out a theoretical mass balance for a typical village in Botswana, which gave an indication of where efforts to alleviate the nitrate problem should be directed. A significant conclusion was that although the nitrogen loading of cattle excreta to the ground surface was higher than that of human excreta, the nitrogen from human excreta reaching the groundwater was estimated to be significantly higher than that of the cattle excreta.

The study went on to recommend methods for alleviating the problem and reviewed methods of treatment to remove nitrates as well as estimating their costs.

3 Muller (1989) *The evaluation and health impact of on-site sanitation*

This MSc dissertation originated from work done while the author was employed by the Mozambican National Institute of Physical Planning (INPF) as co-ordinator of their UNDP-funded National Low-Cost Sanitation Programme from 1985-1987.

The objective of the study was to undertake a critical assessment of the effectiveness of on-site sanitation technologies in meeting the two technical objectives of:

- Removal of faecal pathogens from the domestic environment
- Conversion and disposal of the resulting products without damage to the broader environment

and thirdly an assessment of whether they do so in a cost-effective manner.

The work was carried out in the light of widespread adoption of on-site sanitation technologies, primarily in serving the needs of poorer communities in the developing world, but aimed to address the above questions in a Southern African context, with a subsidiary objective of assessing methodologies used for general application within the region and the establishment of guidelines for this purpose.

The study described a new experimental methodology of using Ascaris eggs as indicators of faecal pollution and presented results from a field study conducted in Maputo.

It reviewed the problem of pollution of groundwater by nitrates and outlined a systematic procedure, based on a mass balance approach, for assessing the potential pollution problem in a particular instance.

Uncertainties in existing knowledge point to the need for further research. Part of this is a restatement of basic principles, which follows within the framework previously established.

2 WHAT IS ON-SITE SANITATION AND WHAT IS IT INTENDED TO DO ?

2.1 Forms of on-site sanitation, in general and in South Africa in particular

There are various types of on-site sanitation, which are described and illustrated in a World Bank publication (Kalbermatten et al, 1982), see Figure 2.1. On-site systems refer to those where the sanitary wastes are not transported to an off-site location for primary treatment.

In South Africa three types are more generally used:

- VIP's
- LOFLOS (LOw FLush On Site sanitation systems, also commonly referred to as aquaprivies)
- Septic tanks

Both VIP's and LOFLOS are similar in that they generally receive only human excreta from a household (with occasional grey water addition); septic tanks on the other hand generally receive both grey water and human excreta.

A build-up of sludge in these systems inevitably means that they require periodic desludging. This sludge may be treated and re-used, or disposed of off-site. Problems associated with sludge treatment or disposal are not addressed in this study.

Of these three types, this study considers primarily VIP's and LOFLOS, which are more likely to be installed in large numbers at relatively high densities in the urban areas. Septic tanks are considered for comparative purposes, because it is well established as a technology and is widely used both in South Africa and around the world; however septic tank systems require a full water supply, whereas VIP's and LOFLOS do not. Consequently the cost of septic tanks is of the same order of magnitude as full waterborne sewerage.

2.2 What is sanitation intended to do ?

Sanitation is intended to be a means of disposal of sanitary wastes, which protects the health of individuals and communities as well as the integrity of the natural environment particularly the aquatic environment. This is done to the extent that there is an acceptable negative impact on the population and on the natural environment; either because the level of pollution is so small compared to naturally occurring levels or because the pollution is harmless to the environment or to man.

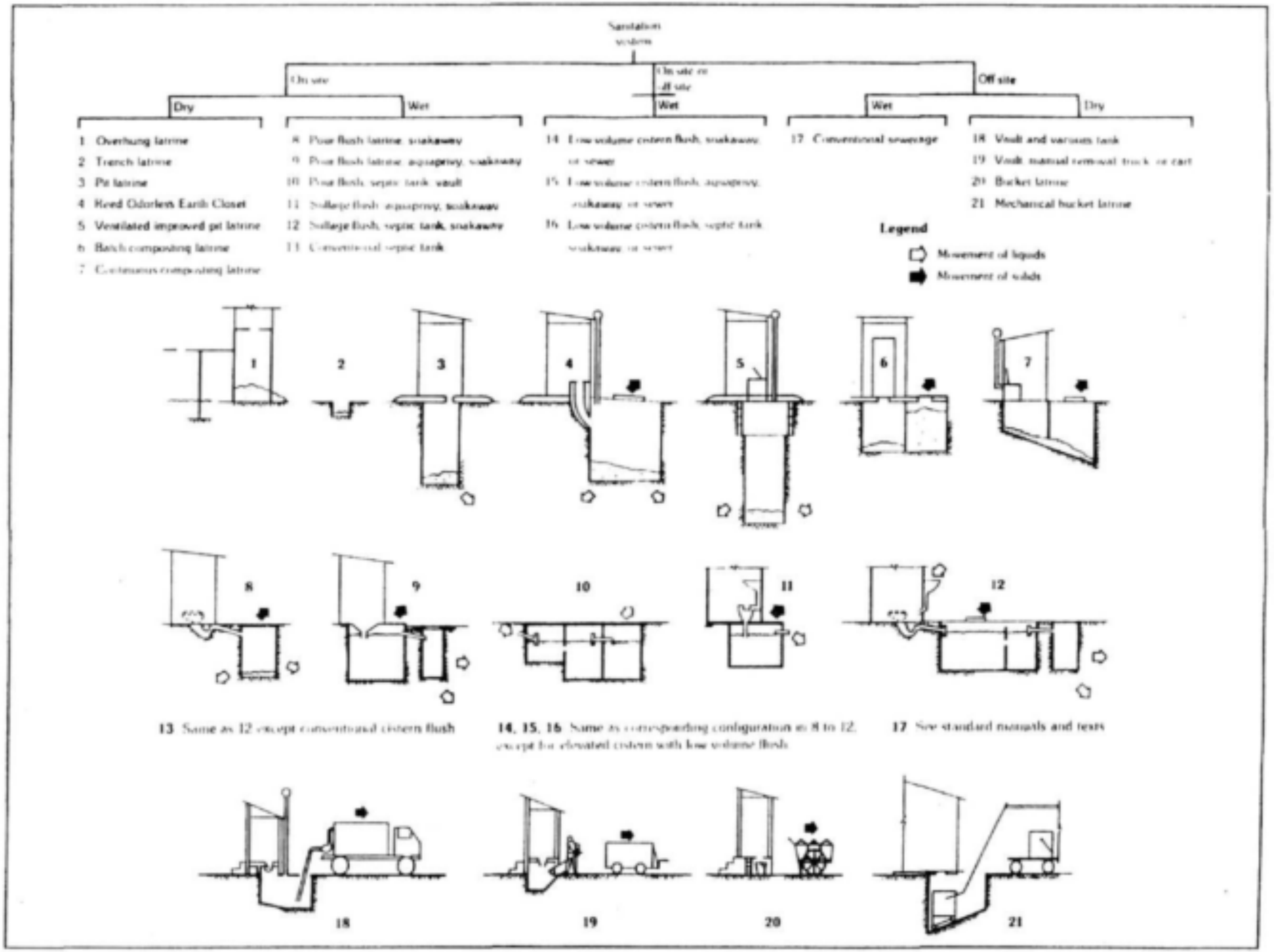


FIGURE 2.1: Generic Classification of Sanitation Systems (after Kalbermatten et al, 1982)

2.3 What is on-site sanitation in particular intended to do ?

Instead of transporting the sanitary wastes to some off-site treatment works, a considerable proportion of these wastes is disposed of on-site by a combination of:

- Transformation into other harmless products
- Dissipation

Residual wastes, in the form of sludge, require periodic removal (typically between one and three yearly intervals), followed by treatment and re-use or disposal off-site.

2.4 What is 'pollution' ?

The term 'pollution' is used where the concentrations exceed acceptable levels. Otherwise the term 'contaminant' is used.

3 WHAT DO SANITARY WASTES CONSIST OF AND IN WHAT FORMS ARE THEY FOUND ?

3.1 Constituents of sanitary wastes and their quantification

Sanitary wastes consist of:

- Human excreta (human faeces and urine)
- Sullage or grey water (discharge from kitchen sink, bathroom excluding the toilet, and laundry)

These in turn consist of the following:

TABLE 3.1: Contaminants from sanitary wastes
(Metcalf and Eddy 1979)

Contaminant	Reason for importance
Suspended solids	Suspended solids can lead to the development of sludge deposits and anaerobic conditions when untreated wastewater is discharged in the aquatic environment
Biodegradable organics	Composed principally of proteins, carbohydrates, and fats, biodegradable organics are measured most commonly in terms of BOD (Biochemical Oxygen Demand) and COD (Chemical Oxygen Demand). If discharged untreated to the environment, their biological stabilization can lead to the depletion of natural oxygen resources and to the development of septic conditions
Pathogenic micro-organisms	Communicable diseases can be transmitted by the pathogenic micro-organisms in wastewater

Contaminant	Reason for importance
Nutrients	Both nitrogen and phosphorus, along with carbon, are essential nutrients for growth. When discharged to the aquatic environment, these nutrients can lead to the growth of undesirable aquatic life
Refractory organics	These organics tend to resist conventional methods of wastewater treatment. Typical examples include surfactants, phenols, and agricultural pesticides
Heavy metals	Heavy metals are usually added to wastewater from commercial and industrial activities
Dissolved inorganic solids	Inorganic constituents such as calcium, sodium, and sulphate are added to the original domestic water supply as a result of water use

Constituents of primary concern are:

- Biodegradable organics (carbon)
- Nutrients (nitrogen and phosphorus)
- Pathogenic micro-organisms (helminths, protozoa, bacteria and viruses)

These are found in both human excreta and grey water to a greater or lesser degree.

Figures indicating the constituents of human excreta are given in Tables 3.2 and 3.3:

TABLE 3.2: Composition of human faeces and urine
(Mara 1976)

	Faeces	Urine
Quantity (wet) per person per day	135-270g	1.0-1.3kg
Quantity (dry solids) per person per day	35-70g	50-70g
Approximate composition (%)		
Moisture	66-80	93-96
Organic matter	88-97	65-85
Nitrogen	5.0-7.0	15-19
Phosphorus (as P_2O_5)	3.0-5.4	2.5-5.0
Potassium (as K_2O)	1.0-2.5	3.0-4.5
Carbon	44-55	11-17
Calcium (as CaO)	4.5	4.5-6.0

The above table is also quoted by Feachem et al (1983), who comment that the approximate composition refers to percentage of *dry* weight.

Of particular interest is that the carbon:nitrogen ratio (C:N) for feces is around 8 whereas for urine it is under 1.

Feachem et al (1983) give the following speculative table for the possible BOD content of excreta and night soil:

TABLE 3.3: Possible standard biochemical oxygen demand (BOD_5) content of excreta and night soil (Feachem et al 1983)

	Europe and North America	Developing country: Urban	Developing country: Rural
Assumed adult fecal weight (grams daily)	150	250	350
Assumed adult urine weight (kilograms daily)	1.2	1.2	1.2
Estimated water in feces (percent)	75	80	85
BOD_5 content:			
In wet feces (grams daily)	96	77	58
Per adult in feces (grams daily)	14.4	19.3	20.3
Per adult in urine (grams daily)	10.3	10.3	10.3
Total per adult in excreta (grams daily)	24.7	29.6	30.6
Per adult in anal-cleansing materials (grams daily)	3.5	3.0	2.0
Strength of night soil (mg/l)	18 800	21 700	21 700

For untreated domestic sewage, the *approximate* relationship of $COD = 2xBOD_5$ may be used.

Grey water or sullage consists of a wide variety of chemicals - detergents, soaps, fats, greases; anything in fact that goes down the kitchen sink or whatever is equivalent.

The results of a survey of 5 households in the United States by Laak (1974) is quoted by Feachem et al (1983), and given in Table 3.4:

TABLE 3.4: Pollution loads of wastewater from various plumbing fixtures in the USA (mg/capita daily; values given in brackets are the percentage contribution of the source)

Wastewater source	BOD	COD	NO_3-N	NH_3-N	PO_4
Bathroom sink	1 860 (4)	3 250 (2)	2 (3)	9 (0.3)	386 (3)
Bathtub	6 180 (13)	9 080 (8)	12 (16)	43 (1)	30 (0.3)
Kitchen sink	9 200 (19)	18 800 (16)	8 (10)	74 (2)	173 (2)
Laundry machine	7 900 (16)	20 300 (17)	35 (49)	316 (10)	4 790 (40)
Toilet	23 540 (48)	67 780 (57)	16 (22)	2 782 (87)	6 473 (55)
Total	48 690 (100)	119 410 (100)	73 (100)	3 224 (100)	11 862 (100)

Palmer Development Group/UCT Water Research Group (1992) provide figures for both human excreta (to VIP's) and grey water (disposed of to the ground surface) as follows (some figures taken from other sources):

Human excreta:

COD	100g/person/day
TKN	10g/person/day (COD x 10%)
Total P	2.5g/person/day (COD x 2.5%)
water consumption	30l/person/day

Grey water:

COD	200mg/l
Total N	20mgN/l
Total P	31mgP/l
Flow	140l/day

Solids are of concern in as much as they affect the working of the sanitation system (particularly sludge build-up), but are not considered in this study.

Pathogenic micro-organisms consist of 4 groups:

- Helminths (or parasitic worms)
- Protozoa
- Bacteria
- Viruses

The sizes of the different groups are distinctly different, with helminth eggs being the largest and viruses being the smallest.

Although it refers to sewage rather than human excreta, the following hypothetical table by Feachem et al (1983) gives a good indication of the levels of pathogens to be expected in sanitary wastes:

TABLE 3.5: Possible output of selected pathogens in the feces and sewage of a tropical community of 50 000 in a developing country (Feachem et al 1983)

Pathogen	Prevalence of infection in country (percent)[a]	Average number of organisms per gram of feces[b]	Total excreted daily per infected person[c]	Total excreted daily by town	Concentration per litre in town sewage[b]
Viruses					
Enteroviruses[d]	5	10^6	10^8	2.5×10^{11}	5 000
Bacteria					
Pathogenic *E. Coli*[e]	?	10^8	10^{10}	?	?
Salmonella spp.	7	10^6	10^8	3.5×10^{11}	7 000
Shigella spp	7	10^6	10^8	3.5×10^{11}	7 000
Vibrio Cholerae	1	10^6	10^8	5×10^{10}	1 000
Protozoa					
Entamoeba histolytica	30	15×10^4	15×10^6	2.25×10^{11}	4 500
Helminths					
Ascaris Lumbricoides	60	10^4[f]	10^6	3×10^{10}	600
Hookworms[g]	40	800[f]	8×10^4	1.6×10^9	32
Schistosoma mansoni	25	40[f]	4×10^3	5×10^7	1
Taenia saginata	1	10^4	10^6	5×10^8	10
Trichuris trichiura	60	2×10^3[f]	2×10^5	6×10^9	120

? Uncertain

Note: This table is hypothetical, and the data are not taken from any actual, single town. For each pathogen, however, the figures are reasonable and congruous with those found in the literature. The concentrations derived for each pathogen in sewage are in line with higher figures in the literature, but it is unlikely that all these infections at such relatively high prevalences would occur in any one community.

a The prevalences given in this column refer to infection and *not* to morbidity.

b It must be recognised that the pathogens listed have different abilities to survive outside the host and that the concentrations of some of them will rapidly decline after the feces have been passed. The concentrations of pathogens per liter in the sewage of the town were calculated by assuming that 100 liters of sewage are produced daily per capita and that 90 percent of the pathogens do not enter the sewers or are inactivated in the first few minutes after the excretion.

c To calculate this figure it is necessary to estimate a mean fecal weight for those people infected. This must necessarily be the roughest of estimates because of the age-specific fecal weights and the age distribution of infected people in the community. It was assumed that people over 15 years old excrete 150 grams daily and that people under 15 excrete, on average, 75 grams daily. It was also assumed that two-thirds of all infected people are under 15. This gives a mean fecal weight for infected individuals of 100 grams.

d Includes polio-, echo-, and coxsackieviruses.

e Includes enterotoxigenic, enteroinvasive, and enteropathogenic *E. Coli*

f The distribution of egg output from people infected by these helminths is extremely skewed; a few people excrete very high egg concentrations.

g *Ancylostoma duodenale* and *Necator americanus*

There is little information available on the microbiological quality of grey water. Although data are lacking it may be assumed that the grey water from bathrooms and laundries will contain small numbers of any pathogens being excreted by the people who use those facilities. Bacterial counts (for each of the categories of total coliforms, fecal coliforms and fecal streptococci) of the order of 10^2 to 10^4 are quoted (Feachem et al 1983)

3.2 Characteristics of effluent from sanitation systems and their quantification

On-site sanitation systems such as VIP's and LOFLOS treat effluent to a certain degree. As there is no separate soakaway for VIP's (the pit itself forming the soakaway), the characteristics of the effluent may be taken as that for raw human excreta. For LOFLOS, the effluent reaching the soakaway may differ significantly in quality from the raw sewage (depending on the characteristics of the LOFLOS), but little data are currently available. Table 3.6 gives figures for the effluent from septic tanks, which give a very rough indication of what may be expected from a LOFLOS:

TABLE 3.6: Comparison of effluent characteristics from septic tanks; data from various studies in Australia, USA, Canada and New Zealand (quoted by Whelan and Titmanis, 1982)

	pH	expressed in mg/l						
		TSS	BOD_5	Total N	NH_4^+ -N	NO_3^- -N	Total P	Dissolved inorganic P
a	6.6-7.4	22-47	52-316	74-237	63-201	0,01-0,03	12-26	12-26
b				44-124	33-100	0.4-0.7	17-90	7-40
c	7.4	136		83	65	<0.02	14.8	13.0
d	6.5-7.5	176	280		97	0.03	11.6	
e	7.9	81	143		131	0.16	189.6	16.7
f				42.3	21.8	0.04	11.8	10.1

Cases:

a Perth, Western Australia; range for 5 households, 14 days (Whelan and Titmanis 1982)
b Stevens Point, Wisconsin; range for 7 households, 1 day (Bouma et al 1972)
c Bolton Landing, New York; mean for 1 household, 5 months (Clesceri 1977)
d Ottawa, Canada; mean for 1 household, 2 months (Viraraghavan and Warnock, 1976)
e Hawkestone, Ontario; mean for 1 household, 12 months (Brandes 1976)
f Lauke Taupo, New Zealand, 1 household (Gibbs 1977)

(It should be noted that the figure of 189.6 mg/l Total P, quoted from Brandes (1976), is very high, and may be incorrect).

4 WHAT ARE THE HEALTH RISKS ASSOCIATED WITH SANITARY WASTES ?

Health risks are of two kinds: from pathogenic micro-organisms and from chemical contaminants. For each, there are two questions to be asked:

- What are the illnesses associated with the different contaminants ?
- How does one become ill from these contaminants ?

4.1 Pathogenic micro-organisms

A variety of bacteria and the illnesses associated with them are detailed in Table 4.1 below; similar data for viruses are given in Table 4.2 below:

TABLE 4.1: Bacteria and their associated illnesses

Microorganism	Associated illness
Vibrio cholerae	Cholera
Salmonella typhi	Typhoid fever
Salmonella paratyphi	Paratyphoid fever
Shigella spp.	Bacillary dysentery
Enterotoxigenic *E.coli* Enteroinvasive *E.coli* Enteropathogenic *E.coli* *Salmonella* spp. *Campylobacter petus* ssp. *jejuni*	Diarrhoeal diseases

TABLE 4.2: Viruses and their associated illnesses

Microorganism	Associated illnesses
Echovirus	Meningitis, diarrhoea, respiratory disease, fever
Poliovirus	Paralysis, meningitis, fever
Coxsackievirus A	Meningitis, herpetic angina, respiratory disease, pleurodynia
Coxsackievirus B	Myocarditis, meningitis, fever, respiratory disease, pleurodynia
Hepatitis A	Infectious hepatitis
Norwalk virus	Vomiting, diarrhoea, fever
Rotavirus	Vomiting, diarrhoea
Adenovirus	Respiratory disease, eye infections, gastroenteritis
Renovirus	Not clearly established
Astrovirus	Gastroenteritis

As regards becoming ill, the first point to be made is that there is a distinction between being ill and being infected. Being ill involves showing symptoms of the disease. The distinction is that one can be infected without being ill. The most important segment of the population involved in transmitting an infection frequently shows few or no signs of the disease; on the other hand, individuals in advanced state of disease may be of little or no importance in transmission. Although illness is what affects a person's life and productivity and so is the major concern, infection is what permits a disease to be transmitted (and cause further infection and illness). Infection will therefore be used in this study rather than illness as the indicator of a problem.

What constitutes an infective dose is therefore a concept of prime importance. The magnitude of an infective dose depends on

- The characteristics of the micro-organism as well as
- The response of the host (person being infected).

Data on infective doses are very hard to acquire as it entails administering a known dose to a human volunteer, and observing the consequences. In general viruses require low infective doses (typically <100 organisms); similarly a single egg or larva can infect a person in helminthic infections; bacteria on the other hand generally large infective doses (of the order of 10 000 or more) (Lewis et al, 1980b). The response of the host also varies from person to person and community to community. It depends on immunity as well general health (and hence resistance to infection). Infective dose is therefore usually expressed as the dose required to infect half of those exposed to the particular organism (ID_{50}).

Pathogens are generally difficult to detect and quantify. As there are some 100 pathogenic bacteria/viruses that are commonly found in water, it is not feasible to test for each individually. Use is therefore commonly made of indicator organisms. *E. Coli* is one such organism which is often used as an indicator of faecal contamination.

The ideal faecal indicator bacterium should be:

- Normally resident in the intestinal flora of healthy people
- Exclusively intestinal in habitat, and hence exclusively intestinal in origin when found in the environment
- Absent from non-human animals (a requirement not met by any of the indicator bacteria currently used)
- Present whenever faecal pathogens are present, and present only when faecal pathogens might reasonably be expected ~~to be~~ present
- Present in higher numbers than faecal pathogens
- Unable to grow outside the intestine, with a die-off rate slightly less than that of faecal pathogens
- Resistant to natural antagonistic factors and to water and waste treatment processes to a degree equal to or greater than that of faecal pathogens
- Easy to detect and count
- Non-pathogenic

Although no one bacterial species completely fulfils all these requirements, a few come close.

A key point to be recognised is that while the presence of the indicator organism indicates the presence of faecal contamination, the absence of the indicator organism does not necessarily prove the absence of faecal contamination.

For an excreted infection to be transmitted, an infective dose of the disease agent has to pass from the excreta of a patient, carrier, or reservoir of the infection to the mouth or some other entryway of a susceptible person. The risks of an infective dose being transmitted depend on the following factors:

- Excreted load of an infected individual
- Latency (the interval between the excretion of a pathogen and its becoming infective to a new host)
- Persistence (viability of the pathogen in the environment, or how quickly it dies after leaving the human body)
- Multiplication (certain pathogens will multiply under favourable conditions; originally low numbers can thus produce a potentially infective dose. Important to note is that while bacteria are able to multiply outside their host, viruses and protozoa do not)

Biological classifications of pathogenic micro-organisms are not particularly helpful in determining the health interventions necessary to control infection. Bradley (1977) proposed a classification system based on a number of disease vectors (namely water-borne, water-washed, water-based, water-related insect vectors). These classifications are useful for putting disease from on-site sanitation into perspective (compared to other water-related disease). However, an environmental classification of excreted infections is more useful. Feachem et al (1983) has provided such a classification, which is as follows:

TABLE 4.3: Environmental classification of excreted infections.

Category	Description	Infection
I	Non-latent; low infective dose	Amoebiasis Enterobiasis Enteroviral infection Giardiasis Infectious hepatitis Rotavirus infection
II	Non-latent; medium or high infective dose; moderately persistent; able to multiply	*Campylobacter* infection Cholera Pathogenic *Escherichia coli* infection Salmonellosis Shigellosis Typhoid
III	Latent and persistent; no intermediate host	Ascariasis Hookworm infection Trichuriasis

Category	Description	Infection
IV	Latent and persistent; cow or pig as intermediate host	Taeniasis
V	Latent and persistent; aquatic intermediate host(s)	Clonorchiosis Schistosomiasis
VI	Spread by excreta-related insects	Bancroftian filariasis All infections in I-V able to be transmitted mechanically by flies and cockroaches

4.2 Chemical contaminants

For chemical substances, the approach is split between

- Non-carcinogenic (acute or chronic toxicity) and
- Carcinogenic substances (cancer-causing).

Dose thresholds such as the infective dose for micro-biological contaminants do not exist for all chemical contaminants; in particular there is a probably unverifiable assumption that dose thresholds *do* exist for non-carcinogenic effects, while there is an unverifiable assumption that dose thresholds *do not* exist for carcinogenic effects (Cotruvo 1989).

It should also be noted that chemicals can have acutely toxic effects at one dose level and carcinogenic effects at another dose level. A further point to be made with respect to cancer is that although the proportion of individuals who get cancer is related to the degree of exposure to the carcinogenic chemical, the severity of ther cancer (ie the extent of the tumour spread) is independent of exposure (Kamrin 1988).

Chemical toxicity can vary widely from chemical to chemical with the toxicity of one chemical being millions, billions or even more times greater than the toxicity of another chemical. Put differently, it takes much more of one chemical to cause the same effect as another chemical.

The chemical contaminant of primary concern in sanitary wastes as encountered in on-site sanitation is nitrate.

High nitrate levels in drinking water may cause methaemoglobinaemia (also called infantile cyanosis) (which is the conversion of haemoglobin to methaemoglobin, and results in the inability of the bloodstream to transport oxygen) in infants. This condition is also known as 'blue baby' syndrome. Methaemaglobinaemia is non-carcinogenic.

High nitrate levels have also been linked to stomach cancer in adults, although the link is at present still tenuous. A recent European study on the subject concluded that recent epidemiological research provided no evidence that nitrate induces cancer in man (ECETOC 1988).

4.3 Routes of transmission for contaminants impacting on human health

There are a number of different possible points of contact with human excreta and greywater. In considering the health impact of on-site sanitation, the routes of transmission for both micro-biological and chemical contaminants, that we are considering are:

- By means of contact with fresh human excreta and grey water
- By contact with soakaway effluent (say, from a failed soakaway, where the effluent has surfaced)
- From contact with soil in the vicinity of the soakaway
- From ingesting groundwater, which has been contaminated by the effluent from the on-site sanitation.

The first two may be controlled by ensuring good hygiene practices and properly designed and constructed on-site sanitation system. Both are beyond the scope of this study. Here, we will consider the last two: hazards from a well-operating system.

To put the problem of health impact of on-site sanitation into perspective, effluent from conventional sewage treatment plants contains significant concentrations of viruses, bacteria, protozoa and helminth ova, even if it meets quality standards of oxygen demand (organics) and suspended solids. Effluents are certainly not suitable for re-use without additional treatment, and may often be unsuitable for discharge to freshwater bodies where those water bodies are used for domestic water supplies by downstream populations. Wastewater treatment also does not remove dissolved inorganic or organic chemicals. Dangerous chemicals are controlled at source.

Reliance is therefore placed on a combination of

- Dilution
- The ability of the natural watercourses (particularly reed beds) to continue the cleansing process
- The water treatment process

to ensure acceptable health impact of this sanitation system.

5 WHAT ARE THE RISKS TO THE ENVIRONMENT ASSOCIATED WITH SANITARY WASTES ?

Effluent from sewage treatment works (unless specifically treated) contains high levels of nutrients: particularly high levels of phosphates, but lower levels of nitrogen-containing compounds.

The enrichment of water bodies with plant nutrients causes a change in the ability of the water body to support life. The ability of the water body to support life is termed the 'trophic state' ('trophic' = concerned with nutrition or the food chain); and this change in the trophic state of the water body is termed 'eutrophication'.

Eutrophication follows a progression from oligotrophic to eutrophic state, the various trophic states being as follows:

- *Oligotrophic* (low concentrations of essential nutrients such as nitrogen, phosphorus and iron, hence life forms are present in small numbers)
- *Mesotrophic* (abundance and diversity of life forms at all food chain levels)
- *Eutrophic* (fewer species present, but the concentration of algae is particularly high and dissolved oxygen levels fluctuate widely between day and night)

There are other terms as well, but these tend to refer to lake systems which we would regard as very severely degraded:

- *Hypertrophic* (excessive productivity leading to the dying off of life forms; synonymous with an advanced stage of eutrophic state)
- *Dystrophic* (literally 'ill-fed'; the entire lake is deoxygenated; anaerobic breakdown produces highly acid 'brown water', typical of many swamps

Beyond the dystrophic state, a lake will fill up, progressing through the stages of marsh, bog forest to highly productive climax forest.

Department of Water Affairs (1986) gives an overview of problems caused by eutrophication (referring to deoxygenation and excessive algal growth), which include the following:

- Increased water purification costs
- Increased expertise required for operation and control
- Toxins produced by algae may result in fish and stock losses
- Deoxygenation may cause disturbances in biological activity and water chemistry
- Algal growths on canal linings which result in loss of hydraulic capacity
- Water surfaces aesthetically degraded
- Recreational use of water surfaces affected
- Values of lakeside properties decreased
- Trihalomethanes (THM's) may be formed when eutrophied water is chlorinated during treatment

Eutrophication is in fact a natural process. The progression does not always follow identical paths and the time scales may vary considerably: A small lake may pass from oligotrophy through eutrophy through dystrophy to climax forest in a few thousand years. The huge Lake Baikal in what was the USSR is still oligotrophic after nearly 100 million years. As it occurs naturally, eutrophication is caused by the lake filling up with sediment. (Clapham 1983)

This natural process of eutrophication, however, may be greatly accelerated by nutrient addition, which tends to be primarily of human origin, with nutrients being in the form of Nitrogen or Phosphorus (N or P).

There is assumed to be a natural ratio between Nitrogen and Phosphorus in living organisms. Although the natural ratio can vary considerably, an average value of 7:1 (ie 7N to 1P) is quoted for South African impoundments (Grobler and Silberbauer 1985). Some authors quote a figure of 10:1 (Forsberg 1977).

The response of a water body to N or P addition depends on:

- The N:P ratio in the water body compared to the natural ratio of living organisms in that water body.
- The nutrient being added

The actual N:P ratio in water bodies varies from water body to water body. If the ratio is say 4:1, then further growth will be N-limited; if it is say 12:1, then further growth will be P-limited. If an impoundment is N-limited, then additional P will have relatively little impact on its trophic state; if it is P-limited, then additional P is likely to have a direct and significant impact on its trophic state. This will obviously have implications for control measures that are instituted. In fact, it has been found that limiting either does have *some* impact on further growth (Grobler and Silberbauer 1985).

To give an indication of the effect of control measures, the special sewage effluent standard of 1mg P/l which was introduced for sensitive catchments (including several major catchments of the PWV) in 1985 was predicted to result in 80 to 90 % reduction in the phosphate load from sewage works, which in turn were estimated to contribute 60 to 80 % of the total phosphate load on the water environment. For the Hartbeespoort dam, severe nuisance conditions (due to algal growth) occurred in the dam 100% of the time in 1985 (ie % time SNC = 100%). The new P effluent standard is expected to reduce the % time SNC for the dam to only 35% by the year 2000.

A further point that should be noted is that the impact of nutrient addition on the trophic state of a water body is also dependent on whether it is a continuous flow or whether it is intermittent. A continuous low flow such as effluent from a sewage treatment works has a much greater impact that does high intermittent flow such as polluted storm water run-off. This is a particular concern in relation to pollution from on-site sanitation (Ashton and Grobler 1988).

The sub-surface transport of phosphorus from on-site sanitation systems does not appear to be a major concern. All previous studies have shown virtual complete removal of phosphorus by the soil within relatively small distances from the source, (Jones and Lee, 1979, and Sawhney and Starr, 1977). This topic is therefore not further pursued in the report.

6 CONTAMINATION FROM OTHER SOURCES

Where a large number of on-site sanitation systems are installed, these 'multiple point sources' become in effect a non-point source of pollution.

In looking at the effect of a large number of sanitation systems, it is important to be able to do two things:

- To isolate pollution from this source from other sources of pollution
- To compare the magnitude of pollution from this source against that of other sources.

Stormwater is not a primary source of pollution; rather it should be viewed as a mechanism of mobilisation and conveyance, which may pick up pollution from other sources and the wider environment, including:

- Soil and air
- Grey water and human excreta (outlined above)
- Domestic refuse or solid waste
- Natural detritus and products of aquatic decay

7 SAFE CONTAMINANT LEVELS

The critical question to be asked here is "How safe is safe enough ?". Risk assessment is a means of answering this question. Although risk assessment may be applied to impact on the environment, it is restricted here to impact on health.

7.1 Risk and risk assessment

The word risk has different meanings for different groups of people. From a scientific and engineering point of view, it is regarded as an expression of the probability (or likelihood) that something unpleasant will happen owing to exposure to hazardous circumstances that could have harmful consequences. It is expressed as a product of the probability of occurrence of the particular event and the effect of that event (financial loss, loss of life etc)

This is very different from the general perception of risk which focuses on the possibility of an unpleasant event and takes no cognisance of its probability. (Wium 1988)

Risk assessment is simply the identification and quantification of risk. There are basically 4 steps involved (Pieterse 1989):

- Hazard identification
- Exposure assessment
- Hazard or dose-response assessment
- Characterisation of the risk associated with human exposure

7.2 Application of risk assessment to drinking water quality

Cotruvo (1989) gives a good overview of the application of risk assessment to drinking water quality, on which this section is based.

In the case of drinking water, the conclusion of a risk analysis would be expressed in terms of the probability (within specified limits of uncertainty) of cases of adverse effects (eg fatalities) in the reference population group.

For risks from pathogenic organisms, the whole issue of risk assessment has been side-stepped by the use of water treatment processes (in particular disinfection and filtration) which are able to achieve the goal of 'virtually no risk for all practical purposes'.

For chemical substances, the approach is split between non-carcinogenic (i.e. acute or chronic toxicity) and carcinogenic substances. In general, the conclusion of a risk analysis is limited in their reliability and credibility by:

- Lack of exposure and toxicology data
- The mathematical expressions used
- The lack of scientific understanding of the mechanisms of carcinogenesis operative at low environmental doses in genetically diverse humans as opposed to the high doses to which test animals are exposed.

The other factor that one needs to appreciate is that the contribution of contaminants in drinking water may be low in comparison to the intake of those contaminants via other exposure routes, particularly food (Pieterse 1989)

For non-carcinogenic effects, a 'safe' dose is set called the Acceptable Daily Intake (ADI) which is the dose which is anticipated to be without lifetime risk to humans when taken daily. It is calculated by applying a safety factor to the no-observed adverse effect level (NOAEL) which is determined from experimental data. The ADI is <u>not</u> an estimate of risk.

For carcinogenic effects, having no dose threshold, there is no completely 'safe' dose. Goals are therefore set at zero dose. This is not always attainable. Standards (or criteria) are therefore set for a certain level of risk which is deemed to be acceptable for public health. In the USA the procedure for setting drinking water standards (criteria) has been as follows:

- Substances are assigned to 5 groups which are then split into 3 categories (I to III): I: carcinogenic, II: equivocal and III: non-carcinogenic substances.
- Maximum Contaminant Level Goals (MCLGs) are then set for each category (the level in drinking water that would result in no known or anticipated adverse effect on health with a margin of safety): An 'aspirational' MCLG of zero for 'probable' carcinogens i.e. Category I and non-zero MCLGs for Categories II and III
- Legally enforceable drinking water standards - Maximum Contaminant Levels (MCLs) - are set as close to the MCLGs as is technically and economically feasible.
- For 'non-carcinogen' and 'equivocal' evidence substances the MCL is usually the same as the MCLG. For 'probable carcinogens' the MCL is set based on a variety of technological performance/cost factors but also a risk targeted between 10^{-4}-10^{-6}.

What is important to recognise is that the setting of standards for carcinogenic substances in the USA is based on a combination of what is achievable and what it will cost; the standards are *not* set to achieve zero risk.

7.3 Communication of risk to the public

Risk assessment is the use of a base of scientific research to define the probability of some harm coming to an individual or population as a result of an exposure. Risk management is the public process of deciding what actions to take when risk has been determined to exist. These are two very different things. The British Health and Safety Executive, in a study of a number of cases, point to the complexity of risk and of the decision-making process (HSE 1989).

When WHO published their drinking water standards, they made it clear that they were to be used as a guide and that each country would have to decide for itself what was appropriate. One of the reasons raised for this was the differences between countries in matters like temperature. A second reason was put by Briscoe (in another context): that because water supply and sanitation programs have economic and social as well as health implications, these decisions are not and should not be made solely on the basis of health considerations (Briscoe et al 1986).

7.4 Safe contaminant levels for drinking water in South Africa.

There are no legally enforceable drinking water standards in South Africa. The SABS Specification No. 241 of 1984 lays down the minimum physical, chemical and bacteriological requirements for 26 determinands for the purity of water (as delivered to the consumer) for domestic supplies. A new three-tier system of drinking water quality guidelines based on recommendations by the CSIR has been proposed more recently (Pieterse 1989). The values for this guideline (excluding the micro-elements) is given in Table 7.1.

The first level is the recommended or working limit, which is the goal or ideal. This limit closely follows the recommended levels set by the USEPA, EEC, WHO and SABS. Water conforming to these levels are considered to be safe for lifetime consumption. The second level is the maximum permissible level or maximum allowable level. The range between the first and second limits is considered to be the insignificant risk range. The third limit is called the crisis limit, the limit at which extreme action must be taken. The range between the second and third limits is considered to be the low risk range.

TABLE 7.1: Proposed three-tier set of drinking water criteria for application in South Africa (excluding micro-elements)(after Pieterse 1989)

Determinand	Unit	A	B	C
Physical and organoleptical				
Colour	mg/l Pt	20	-	-
Conductivity	mS/m (25°C)	70	300	400
DOC	mg/l C	5	10	20
Dissolved oxygen	% Sat.	70	30	10
Odour	TON	1	5	10
pH	pH unit	6.0-9.0	5.5-9.5	<4.0 or >11.0
Taste	TTN	1	5	10
Temperature	°C	<25	<30	<40
Turbidity	NTU	1	5	10

Determinand	Unit	A	B	C
Microbiological				
Standard plate count	per 1ml	< 100	1 000	10 000
Total coliform	per 100ml	0	5	100
Faecal coliform	per 100ml	0	1	10
Clostridium perfringens	per 100ml	0	10	100
Coliphages	per 100ml	0	10	100
Enteric viruses	per 10l	0	1	10
Macro-elements (all units mg/l)				
Aluminium	Al	0.15	0.5	1.0
Ammonia	N	1.0	2.0	4.0
Barium	Ba	0.5	1.0	2.0
Boron	B	0.5	2.0	4.0
Bromide	Br	1.0	3.0	6.0
Calcium	Ca	150	200	400
Cerium	Ce	1.0	2.0	4.0
Chloride	Cl	250	600	1 200
Copper	Cu	0.5	1.0	2.0
Fluoride	F	1.0	1.5	3.0
Hardness	$CaCO_3$	20-300	650	1 300
Iodide	I	0.5	1.0	2.0
Iron	Fe	0.1	1.0	2.0
Lithium	Li	2.5	5.0	10.0
Magnesium	Mg	70	100	200
Manganese	Mn	0.05	1.0	2.0
Nitrate	N	6.0	10.0	20.0
Potassium	K	200	400	800
Sodium	Na	100	400	800
Sulphate	SO_4	200	600	1 200
Uranium	U	1	4	8
Zinc	Zn	1	5	10

A Recommended limit (maximum limit for no risk)
B Maximum permissible limit (Maximum limit for insignificant risk)
C Crisis limit (Maximum limit for low risk)
TON Threshold odour number
TTN Threshold taste number
NTU Nephtelometric turbidity units

A key feature of these proposals is the introduction of the crisis limit, set at a level which is higher than the maximum permissible limit. This indicates that the transition from a 'safe' concentration to a 'poisonous' concentration is a gradual transition rather than a sharp cut-off limit as suggested by previous criteria.

8 THE FATE OF CONTAMINANTS IN THE SUB-SURFACE

8.1 Mobility of contaminants

It is difficult to be categorical about which of the contaminants mentioned earlier in the report are the most mobile, and thus likely to travel the furtherest in a particular sub-surface environment, although for bacteria it can be said that their mobility will be inversely proportional to their physical size. Unfortunately, in the literature most studies of contamination due to on-site sanitation have been restricted to only one or two contaminants, e.g. tests may be done for bacteria, but not for nitrates, or vice versa. In two reported comparative studies, nitrates have been shown to be significantly more mobile than the other contaminants discussed above. Yamaura et al (1986) conducted experiments using laboratory soil columns and lysimeters, as well as full-scale field tests. Septic effluent was added to the soil in these apparatuses on a daily basis, and the quality of the effluent monitored. In general, removal efficiencies of around 90% were obtained for BOD, Total Phosphorous, and coliforms, whereas for Total Nitrogen the removal efficiency was less than 50%. The field experiments were continued for 10 years, and no apparent change in removal efficiency was detected. In a field study reported by Robertson et al (1989), contaminant migration from a $100m^2$ weeping tile bed was monitored at a number of locations down-gradient from the tile bed. Residence time in the vadose zone was monitored using a Sodium Bromide tracer, and was found to be approximately 10 days. They found an extremely long, thin plume of impacted groundwater, with nitrate levels of about 50% of their source concentration occurring at distances of 100m from the tile bed. No trace of other contaminants was detected at these distances.

8.2 Mechanisms of transportation

Implicit in this study is the assumption that one is concerned about the transport to, and accumulation in the groundwater of contaminants. To reach the groundwater, travel through the unsaturated (also referred to as the vadose) zone will usually be necessary. During the movement of contaminated water through this zone a number of processes may occur that alter the concentration and composition of the contamination. Retardation processes such as complexation, sorption, precipitation, and solution should be accounted for, where appropriate, when trying to quantify the amount of ground water pollution that may occur. The sub-surface transport of contaminants is therefore not merely an advection/diffusion process. However, it is apparent from the literature that the magnitude of the effect of these retardation processes has not been quantified to any significant degree. Preliminary evaluation of the suitability of a particular site for the use of on-site sanitation could therefore be based on the assumption that viruses are transported at the same rate as moisture through the unsaturated zone. This would definitely be a worst-case scenario.

Once a particular contaminant has reached the ground water, the rate of transport will be much greater (than in the unsaturated zone), and movement will be in the direction of the regional ground water movement. Micro-organisms may be carried up to several hundred meters in this zone. Vertical restriction of drainage by an impermeable soil layer or bedrock may lead to saturated flow conditions in a zone just above the contact between the permeable soil and the impermeable layer. In a situation similar to this, McCoy and Hagendorn (1980) recorded bacteria movement at rates of between 1 and 10m/hour in a strongly sloping silt loam.

An interesting observation made by Romero (1972), was that the contamination travelled primarily in a thin sheet at the surface of the saturated zone. He found no evidence of dispersion, and highlighted the benefit of drawing water from great depth in screened wells, where much lower potential for contamination existed.

The modelling of ground water (i.e. in the saturated zone) is a relatively straightforward matter (see section 8.6). By comparison, modelling moisture movement in the unsaturated zone is a far more difficult problem. As discussed in section 8.6, techniques for analysing this type of problem are now available, although they require a great many more parameters than for the modelling of saturated flow. Implicit in the above procedures, however, is the assumption that the soil conditions are homogeneous and continuous. A problem that has been identified in certain soil profiles is that of so-called macro-pore flow. This refers to flow that may occur along channels formed by decomposed roots, or along fissures in rock or residual soil profiles, amongst others. Once again advection is the primary means of transport, although flow is now along discreet 'channels', rather than through a porous continuum.

Attempts have also been made to predict the sub-surface movement of nitrates. The most common (and most conservative) approach has been to treat nitrate as a conservative species, i.e. no denitrification is accounted for, (Oakes et al, 1980). Nitrogen removal mechanisms such as volatilization, adsorption, fixation, biological denitrification, incorporation into cell tissue, removal by vegetation, and leaching, are ignored. The paucity of experimental data in the literature on these topics means that modelling of the complete sub-surface denitrification process is unlikely to be attainable in the near future.

8.3 Characteristics of pollutants affecting their transportation

As has already been mentioned, the sub-surface movement of contaminants will not exceed the rate of movement of the contaminated water, whether in the unsaturated or saturated zone, except in extremely exceptional circumstances. However, it is not sufficient to merely know the rate of sub-surface water movement. Knowing that an influent (that may be contaminated) from a particular location will eventually reach a drinking water well does not necessarily mean that the well will become contaminated. There are many processes, both physical and chemical, that serve to remove contaminants from the water during its movement through the sub-surface. This section discusses the more important characteristics of contaminants that govern the degree to which this attenuation will take place.

8.3.1 Viral and bacterial contamination

Physical size

The large size of helminths and protozoa (typically $>25\mu m$), which is the same size as silt particles, will usually result in highly efficient physical filtration of these contaminants by the soil below and adjacent to the source. Exceptions could be coarse sand deposits, or fissured soil conditions. The effectiveness of a particular soil in filtering these contaminants can be evaluated by using the conventional Terzaghi and Peck filter criteria (1967). Bacteria and viruses, however, are of significantly smaller size, with some examples given below.

Escherichia col.	$0.5\mu m$x$1.0\mu m$x$2.0\mu m$
Salmonella typhoso	$0.6\mu m$x$0.7\mu m$x$2.5\mu m$
Shigella sp.	$0.4\mu m$x$0.6\mu m$x$2.5\mu m$
Psittacosis virus	$0.25\mu m$ diameter
Bacteriophage virus	$0.1\mu m$ diameter
Poliomyelitis virus	$0.01\mu m$ diameter

The bacteria in the above list may be filtered out by soil that is predominantly clayey, i.e. clay sized particles would be small enough to filter out most bacteria. The question remains as to how much of the soil must be clay sized in order for the filtration process to be completely effective. This can only really be answered by appropriate laboratory experimentation.

As an indication of the relatively longer survival time of bacteria, the table below summarises the finding of Patterson et al (1971).

The contaminant of most concern in terms of transportability through soil is then clearly the range of water borne viruses that may be derived from on-site sanitation. Viruses are too small for even the finest grained clays to have anything but a slight filtration effect. One must look to other processes, such as adsorption, for effects that will attenuate virus concentrations in the sub-surface zone. A mitigating factor is that viruses cannot survive outside a host, and therefore generally have a very low survival time, particularly when compared with the table below.

TABLE 8.1: Time of survival of faecal bacteria (after Patterson et al, 1971)

Type of organism	Survival time			Reference
	Septic tank	Soil	Other	
Salmonella typhosa			52 days	Caldwell, 1938
Salmonella typhosa			165 days	Warrick and Muegge, 1930
Salmonella typhosa	27 days	25-41 days		Beard, 1938
Salmonella typhosa	24 days			Green and Beard, 1938
E. coli			2yr 8mon	Warrick and Muegge, 1930
E. coli		2 years		Mom and Schaafsma, 1933
Coliform bacteria		3 months		Malin and Snelgrove, 1958
Coliform bacteria		4-7 days		Subrahmanyan and Bhaskaran, 1950

Chemical and other processes

It appears that the removal of viruses in the sub-surface region depends almost entirely on the process of adsorption (the process whereby foreign atoms or molecules become attached to the surface of e.g. a soil particle, thus lowering the free energy of the surface).

The actual mechanism of viral adsorption to solids does not appear to be fully understood as yet. Two general theories have been proposed to explain the process, both of which are based on the net electronegativity of the interacting particles. The essential differences in the theories, that relate to how the virus particles actually become fixated to the solid particles is not of relevance to the present study, and for more information reference should be made to Carlson et al (1968) and Schaub et al (1974). What is of relevance is that virus adsorption cannot be considered a process of complete immobilisation of the virus from the carrier liquid. Therefore although adsorption is effective in decreasing the concentration of a virus, it is essentially a reversible process, and adsorbed viruses may be flushed out of a soil by heavy rains, or similar events. As an example, Wellings et al (1974), recorded a burst of virus levels in groundwater obtained from both 3 and 6m deep wells at a wastewater land disposal site following a heavy rainfall. Whilst recognising this problem, it should be emphasised that viruses cannot reproduce in soil and have a finite survival time, and simply by fixing the virus for a certain time the concentration of the virus will have been decreased. Since virus adsorption is likely to be negligible in the saturated zone, maximisation of effluent residence times in the unsaturated zone is the key factor affecting the removal and elimination of viruses.

Factors that have been identified by numerous researchers as being particularly important in the removal of viruses from soil are summarised in Table 8.2 below.

TABLE 8.2: Factors affecting virus removal in soils
(after Gerba et al, 1975)

Factors	Remarks
Flow rate	Low flow rates ($<$1 litre/day/m^2) result in very efficient removal of viruses ($>$99%) in clean waters
Cations	Cations, particularly divalent cations can act to neutralize repulsive electrostatic potential between negatively charged virus and soil particles, allowing adsorption to occur
Clays	High virus retention by clays results from their high ion exchange capacity and large surface area per unit volume
Soluble organics	These have been shown to compete with viruses for adsorption sites on the soil particles
pH	Generally, a low pH favours virus adsorption, while a high pH results in elution of an adsorbed virus
Isoelectric (i.e. zero net charge) point of virus	The optimum pH for virus adsorption will likely occur at or below its isoelectric point.
Chemical composition of soil	Certain metal complexes such as magnetic iron oxide have been found to readily adsorb viruses

Reference to the literature confirms the generality of these conclusions. In general then, adsorption of viruses is best in clayey soils, and for soils with a low pH. Carlson et al (1968) found up to 99% adsorption of added bacteriophage T2 to common clays such as kaolinite, montmorillonite and illite in concentrations of cations that occur in natural waters at pH of 7. In laboratory tests of a range of soil types, Bitton et al (1979) found that Echovirus 1 was least adsorbed to a 'sandy soil'. The rate at which adsorption occurs varies with the type of virus, and for a particular type of virus will vary with the rate of water movement. A final point worth noting, and that is of importance when considering ways of modelling sub-surface virus movement, is that all viruses are not biochemically identical and adsorptive behaviour may not be similar under identical environmental conditions with all viruses.

It was noted above that, in general, bacteria are larger than viruses, and are much more efficiently filtered by all soils. This does not mean that emphasis should be placed on preventing only the movement of viruses, since e.g. bacteria have the ability to multiply outside the host, which viruses do not have. In addition, their survival time in a soil may be significant, as shown in Table 8.1. In general it appears that 2 to 3 months is sufficient time for reduction of pathogenic bacteria to negligible numbers once they have been applied to the soil, although exceptions have been reported (Rudolfs et al, 1950).

In addition to physical filtration, bacteria can also be removed by the process of adsorption, in the same way as discussed above for viruses. The primary factors that affect the survival time of enteric bacteria in soils has been summarised in Table 8.3 below (after Gerba et al, 1975).

TABLE 8.3: Factors Affecting Survival of Enteric Bacteria in Soil

Factors	Remarks
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 retention capacities
Temperature	Longer survival times at low temperatures; longer survival in winter than in summer
pH	Shorter survival time in acid soils (pH 3-5) than in alkaline soils
Sunlight	Shorter survival time at soil surface
Organic matter	Increased survival time and possible regrowth when sufficient amounts of organic matter are present

The above should only be regarded as general guidelines. In specific cases different conclusions have been drawn by some workers, e.g. Romero (1972) found that survival times of some bacteria increased in tropical climates.

The question of developing a predictive capacity for determining the likely movement of bacteria is obviously extremely difficult. Not only will it be necessary to characterise the soil profile and its hydraulic conductivity parameters, but some account of the above factors (where relevant) will be necessary.

8.3.2 Nitrate contamination

Discussions on the suitability of on-site sanitation are inevitably linked to the possibility of nitrate contamination of the groundwater, and the possibility of the occurrence of methaemoglobinaemia. In view of this widespread concern, it is surprising that so little has in fact been done to characterise and quantify the amount of nitrate contamination that may result from on-site sanitation facilities. Information in the literature is rather scant and incomplete.

The attenuation of nitrates in the sub-surface does not depend on physical processes such as filtration, but rather on microbiological processes. Similarly to other mobile contaminants such as chloride, nitrates also have a very low affinity for sorption to solid particles. Denitrification occurs under anaerobic conditions, as bacteria couple oxidation of organic carbon with reduction of nitrogen. According to Kinzelbach et al (1989), in aerobic aquifers nitrate behaves, to a good approximation as an ideal tracer.

The required conditions for denitrification are the presence of denitrifying bacteria, existence of reducing conditions, and the availability of an adequate supply of readily biodegradable organic carbon. Denitrification only occurs in anaerobic conditions. This does not, however, necessarily mean that it is restricted to the saturated zone. It can occur in the vadose zone, since isolated, local 'pockets' of saturation may occur. In a study by Starr and Gillham (1989), they showed that under certain conditions, if there was sufficient labile (i.e. able to participate in very fast reactions) organic content present in the subsoil, the dissolved oxygen would be reduced, causing localised anaerobic conditions. Based on their findings, these authors stress the importance of maximising the residence time in the relatively organic-rich vadose zone. The organic content of any soil is, however, finite. In a study by Rödelsperger (1989), decreasing denitrification processes in an aquifer were attributed to exhaustion of the natural supplies of organic substances. Another problem associated with nitrates in the subsurface is that a significant rainfall after an extended dry period may flush nitrates out of the soil that have accumulated during the dry period.

8.4 Extent of contaminant transport reported in the literature

Instances of where the movement of contaminants have been recorded in the literature are discussed below. The movement of viruses, bacteria, and nitrates are dealt with separately. Although the discussion is not exhaustive, it gives a good indication of the order of magnitude of how much the various contaminants may travel, depending on other factors, e.g. hydrogeology.

8.4.1 Reported movement of viruses

Most of the reported studies of viruses indicate that virtually complete die-off occurs within 3m of the source. Laboratory studies have shown that this is true even of movement though relatively permeable soil, e.g. studies at University of California showed that complete virus die-off occurred within 0.75-1.0m in a 'sandy soil' (Romero, 1972). Most field studies have tended to confirm these observations, as shown by the data summarised in the table below.

TABLE 8.4: Summary of virus removal by soils

Nature of fluid	Virus	Nature of media	Flow rate (lpm/m^2)	Travel (cm)	Percentage removal(%)
Spring water	cox-sackie	garden soil	unknown	90	50
Spring water	T4	garden soil	unknown	56	22
Sewage effluent	polio 1	soil high in iron oxide	unknown	116	unknown
effluent	polio 1	0.65mm ϕ sand	112.5	20	82-99.8
tapwater	cox-sackie	sand	unknown	70	0_>90
tapwater	polio 1	unsaturated sand	1000-2000	60	99.5-99.9
oxidation tank effluent	polio 3	sand, sandy loam	unknown	32	unknown
tapwater	polio 1	coarse, fine sand	0.4	60	1_>98
distilled water with added salts	T1, T2 and f2	Arkansas soil	0.078 to 0.295 ml per min	40-50	>99
distilled water	polio 2	latersol	5-7000	4-15	96-99.3
distilled water	T4	latersol	5-7000	4-15	100
distilled water	polio 2	cinder	1.7 million	15-40	22-35
distilled water	T4	cinder	1.7 million	15-40	100
distilled water with added salts	polio 1	dune sand	1-2 ml per min.	20	44-27
distilled water with added salts	polio 1	dune sand	1-2 ml per min.	20	99.8 - 99.9
distilled water	polio 1	sandy soil	unknown	20	97
distilled water	T7	sandy soil	unknown	20	88
treated sewage	polio 1	sandy soil	unknown	20	98.6
treated sewage	T7	sandy soil	unknown	20	99.6

It can be seen from the above table that, except for one reported instance, the travel of viruses did not exceed 1m. It is not clear from the paper by Gerba et al (1975) which of the cases cited above are for saturated soil and which are for unsaturated soil. However, in soil column tests on both saturated and unsaturated soil reported by Powelson et al (1990), it was found that little removal of an inoculated bacterium (MS-2 bacteriophage) occurred in the saturated column, whereas the unsaturated column was extremely effective. After the passage of as many as 18 pore volumes, the virus concentration maintained an exponentially declining profile with depth. At the end of the tests on unsaturated soil, the soil was removed, and number balances of the virus carried out. Approximately 61% of the input viruses were accounted for, and the reduction was attributed to inactivation of the virus (in contrast, all the input viruses were accounted for in the saturated column tests). Whatever the degree of saturation in the above table, it is clear that the sub-surface movement of viruses is usually very limited, and the rate of removal by soil is high. Although the information given in Table 8.3 on the soil particle size distribution is very limited, it is apparent that poorer attenuation is achieved in coarser soils (e.g. <50% for polio 2 removal by cinder, and for polio 1 removal by dune sand), which is consistent with the discussion in Section 8.2.

Removal of viruses by soil appears to occur primarily by adsorption. However, it should be borne in mind that retention of viruses by soils does not necessarily result in their permanent immobilisation, and that changes in water quality (e.g. infiltration of a large amount of rainwater), can result in their de-adsorption, and subsequent further travel. Ironically, soils that are most effective in removing viruses would also enable them to persist for the longest periods of time.

Although it appears from the foregoing that the potential for viral contamination of groundwater as a consequence of on-site sanitation is negligible, as reported by Lewis et al (1980b), there are cases where outbreaks of illness have been associated with such facilities. According to Vogt (1961), an epidemic of infectious hepatitis was attributed to contamination of well water by septic tank effluent in Posen, Michigan. In another instance, poliovirus was isolated from a well responsible for a gastroenteritis outbreak in Michigan (Van der Velde, 1973). The pollution source was identified as a septic tank drain field located 43m from the well. It is important to note that in both cases the wells were finished in limestone, and therefore the potential for rapid lateral movement of water could have been predicted. With the large areas in South Africa that are underlain by dolomites, the above reports serve to reinforce the potential dangers of building on such formations.

8.4.2 Reported movement of bacteria

TABLE 8.5 : Distance of travel of microorganisms (from Patterson et al, 1971)

Type of organism	Distance transported (m)		Reference
	Vertical	Horizontal	
E. coli		70	Warrick and Muegge, 1930
E. coli	3-9		Mom and Schaafsma, 1933
E. coli		24	Caldwell and Parr, 1937
E. coli		122	Dappert, 1932
Coliform bacteria		33-122	Miller et al, 1957
Coliform bacteria	0.6-0.9		Malin and Snelgrove, 1958
Coliform bacteria		55	Randall, 1970
Coliform bacteria	46		Hickey and Duncan, 1966
Clostridium welchii	2.1-2.4		Hickey and Duncan, 1966
'Lactose Fermenters'	0.8	0.6	Giovanardi, 1938
'Bacteria'	1.8	0.5	Szoplik and Milkowska, 1961
'Bacteria'		610	Walker, 1969

The above studies do not give information concerning the source of the pollution, and it is difficult to extrapolate this information to other sites. In the paper by Romero (1972), however, details of this kind are given, and it is instructional to review some of these studies. Baars (1957) studied the bacterial pollution associated with 1m deep pit latrines at a campground in the Netherlands. The ground water level varied between 3 and 4m below surface, and drinking water was obtained from 40m deep wells located within the campground. Loadings of 10 000kg of faecal matter and 400m^3 of urine were deposited each season. The underlying soil was a fine sand. In bacterial analyses of cores obtained directly adjacent to the tested pit latrines, E.coli could not be detected at depths exceeding 1.5m. An important point to note is that the pollutants were washed below the bottom of the pit latrines by urine and rain water only, i.e. no water flushing was used.

Caldwell (1937, 1938) has reported on a number of studies of ground-water pollution caused by pit latrines extending below the water table. In one study, a 4.5m deep latrine, located in sand and sandy clays, was accepting the wastes from a family of six. The latrine penetrated about 1.5m into the ground water, and was underlain by a stratum of clay. This factor, coupled with the water table gradient of 35m per km, would have resulted in primarily lateral flow of the groundwater. In observation wells located at 1.5m intervals up to 10m from the latrine, B.coli were detected at 3m after 5 weeks of the start of the experiment. After 2 months B.coli were detected in 90% of the samples at 8m, and an occasional organism was detected at 10m. After about 7 months an interesting phenomenon occurred. A retreat of the bacterial stream practically all the way back to the latrine was observed.

The above phenomenon was attributed to so-called soil 'self-defense' mechanisms. It is more likely a manifestation of the observation of Romero (1972), that the nature of the soil immediately in contact with the pollution largely governs how far it will travel. As the soil immediately in contact with the source becomes clogged due to filtration of solid particles, the grading of the soil will effectively be altered, perhaps finally resulting in complete filtration of bacteria. This could also be what Lewis et al (1980b) meant when they observed that a sterile soil first has to 'mature' before it becomes a very effective bacteria filter.

In a second study, reported by Caldwell (loc.cit.) a latrine overlying beds of fine sand with some coarse sand and fine gravel, extended to a depth of 2.5m. These sediments were also underlain by a clay stratum. The water table was between 1.5 and 2m deep, and the water table gradient approximately 6m per km. The pollution source was once again the faecal matter from a family of six. Observation wells were located at 3m intervals up to a distance of 24m from the latrine. Significant numbers of B.coli were initially carried the full 24m, but at the termination of the experiment (the time period was not reported) the length of the B.coli stream had been reduced to some 16m, where only an occasional B.coli was detected.

In a more recent study, Viraraghavan (1978), found high concentrations of indicator organisms at horizontal distances of 15m from a septic tank tile, the bottom of which was located at a depth of 0.6m. The soil at this site consisted of sandy clay for about 0.6m, underlain by clay with a small sand content (i.e. a decreasing permeability with depth). At the time of the test, the ground water level was 0.15m below the bottom of the trench. The volume of source material was not given. Although in this instance the latrine did not penetrate the ground water, the horizontal extent of the contamination was similar to the above two cases, probably because of the additional water associated with a septic tank as opposed to a pit latrine.

Further evidence that movement of bacteria from pit latrines that are finished above the water table can be expected to be minimal, is provided by the results of a study in Maseru, Lesotho, reported by Lister and Stewart (1984). A large number of test pits were dug adjacent to, and immediately downgradient of two pit latrines. The soil profile consisted of a permeable silty sand upper layer of 0.3-0.45m thickness, overlying an impermeable clayey lower horizon. Despite the fact that some of the downgradient test pits were only 1m from the pit latrines, tests on soil samples recovered from the test pits produced zero readings for Faecal Coliforms, Salmonella, and Streptococci Faecalis bacteria. Provided that additional water (e.g. sullage) is not poured directly into the pit, it appears that this type of system may be well suited to minimising transport of contaminants within the sub-surface.

Another unusual observation related to movement of bacteria was reported by Goldshmid (1974). In this study, which was carried out in Israel, surface water was pumped underground during the rainy season to recharge an aquifer. During the dry season, water was withdrawn from the well. Although the water pumped underground contained less than 2 coliforms/100ml, the repumped water contained counts as high as 10^5-10^6/100ml, and these counts persisted for long periods after the commencement of pumping. This was attributed to re-growth of the coliforms being stimulated by the organic matter accumulated around the well casing.

In general, it appears that as long as there is a sufficient depth of intact, unsaturated soil between the source of the pollution and the ground-water, bacterial contamination should not be a major problem. Exactly what constitutes 'sufficient depth' will be addressed in Section 10, which discusses some guidelines for the use of on-site sanitation.

8.4.3 Reported movement of nitrates

A particular problem with quantifying the degree of nitrate contamination due to on-site sanitation is evaluating the background levels. As an example, a study by Schmidt (1972) showed widely fluctuating nitrate levels, with some values as high as 155mg/l (compared with World Health Organisation standards of 45mg/l). These elevated, in-situ nitrate levels were attributed to farming activities. Even higher levels were reported for soil under livestock feedlots (Smith (1967)), who found amounts of nitrate of 150-200mg/l of soil (and therefore much greater concentrations per litre of soil moisture). In a more recent study in Venda (Taussig and Connelly, 1991), nitrate levels of >150mg/l were measured up-gradient of the village under study, and were located in large cultivated lands. High nitrate levels are not necessarily therefore such good indicators of pollution from on-site sanitation as, for example, are bacteriological indicators, and care must be taken to obtain representative background readings.

There are, however, instances of nitrate contamination that have been reported in the literature that are directly attributable to on-site sanitation systems. In a densely populated low-income residential area of Delaware (U.S.A.), where only septic tanks were in use, 28% of the groundwater supplies were found to have nitrate concentrations exceeding 17 mg NO_3-N/l. In a study in Botswana by Lewis et al (1980a), concentrations in excess of 135 mg/l were found in a water supply borehole that was in close proximity to a village with a population of 20 000 people. Of particular interest in this study was their finding that immediately adjacent to pit latrines, nitrate concentrations were as high as 220 mg NO_3-N/l. Pollution associated with a much smaller village, with a population of about 500, in Central India was reported by Cook and Das (1980). A very distinct plume of nitrate pollution extending down-gradient of the village was measured, with concentrations of 200 mg NO_3-N/l being found in the subsoil at a distance of about 30m from the edge of the village.

Results from a series of shallow wells in the Maputo aquifer have clearly linked the nitrate concentrations within the groundwater to the density of settlements making use of on-site sanitation, (Muller, 1989). Although it is not entirely clear in any of the above studies, it appears that the depth to ground-water below the sanitation facilities was small (2-3m).

Furthermore, in the first two cases discussed above, rapid migration to the groundwater was possible, with the sub-surface material consisting of either sandy, well-drained sand or fractured, weathered rock. Based on these and other findings, Lewis et al (1980b) have proposed an allowable density of population using on-site sanitation that accounts for infiltration due to precipitation.

8.5 Hydro-geological factors affecting pollutant transportation

The permeability (or hydraulic conductivity) of the soil between a privy and the groundwater level is obviously of paramount importance in determining the possible extent of pollution due to such facilities. Soil permeability is, however, notoriously difficult to measure, and varies over many orders of magnitude for different soil types. There are also many other problems, e.g. most natural soil profiles are to some extent heterogeneous, and may also have different hydraulic conductivities in the vertical and horizontal directions. Furthermore, it is usually difficult to obtain undisturbed, representative soil samples for carrying out laboratory tests, and recourse must often be made to field permeability testing. Unfortunately there are many different types of field permeability tests in current use, and they do not necessarily all give the same value. It has already been mentioned that the risk of groundwater contamination can be decreased by maximising the residence time in the unsaturated (or vadose) zone. Measuring the permeability of soil in the vadose zone presents an additional set of difficulties. This is because the permeability varies with the degree of soil saturation (i.e. the moisture content). The highest permeability occurs when the soil is fully saturated, and the permeability decreases as the moisture content decreases. There is no universally accepted technique for measuring this property, and it is in fact currently the subject of investigation worldwide.

The above discussion only pertains to soil profiles that are continuous, i.e. not interrupted by discontinuities. A major problem in predicting moisture migration occurs when the possibility of macro-pore flow exists. This effect may be due to anything from root holes to fracturing of the bedrock, or soil fissures. Whatever the cause, the sub-surface flow regime will be significantly altered by the presence of these macro-pores. As discussed later on, some surprising results may occur when flow of a contaminant occurs through these macro-pores.

An important factor in the prevention of pollution migration, particularly bacterial pollution, is the particle size distribution of the soil. A well-graded soil will be a much more effective filter than a similar, but uniformly or gap-graded soil. The importance of this filtration mechanism has been recognised in many studies, and it has often been noted that very efficient removal of bacteria occur within the first few cm of the soil. As an example, a study in Romania (see Romero, 1972) showed that 92-97% of the coliforms were retained in the uppermost 1cm of the soil tested, and Gerba et al (1975) suggest that 'straining' of bacteria occurs right at the soil surface, and that a finer filter is gradually built up by accumulation of retained bacteria. This process has sometimes been termed 'clogging', and it has even been claimed that, 'until the clogged zone has been established, the pit latrine poses a potential source of pollution', (Taussig and Connelly, 1991). However, as pointed out by Lewis et al (1980b), pore clogging may result in system failure in the form of surfacing of effluent. In a comprehensive study by Kropf et al (1975), this was found to be improbable as long as the effluent contained minimal solid matter and grease. They found that the infiltration interfaces never become irretrievably clogged, but that an equilibrium between the growth and die-off of the biomass filter reaches an equilibrium with the nutrients available in the liquid.

What then constitutes the ideal soil for minimising the potential for pollution as a consequence of on-site sanitation? Based on studies at the University of California, Richmond, Romero (1972) concluded that passage through 1-2m of coarse sand would provide acceptable quality drinking water. Similar emphasis on suitability of coarse grained soils for use of on-site sanitation systems has been put forward by Fekpe et al (1992). They discuss a number of field percolation tests that were undertaken in Kumasi, Ghana, to determine the infiltration capacities of a range of soils. The more free draining (and thus higher permeability) soils were classified as most suitable for on-site sanitation systems. This, of course, conflicts with the requirement to maximise the residence time in the vadose zone. Other workers (e.g. Taussig and Connelly, 1991) have expressed a preference for fine, 'graded' soil (probably a fine sand) below a pit latrine, with suggested thicknesses of 2-3m. Still another view is that fine grained soils such as clays are most suitable. Stolzy and Fluhler (1978) found a correlation between clay content and the disappearance of nitrate due to denitrification, which is of course highly desirable. Muller (1989) substantiates this finding, and concludes that clayey soil is better for inhibiting nitrate pollution than is free-draining soil. There is thus obviously a conflict of viewpoints, with some workers attempting to satisfy one set of criteria (perhaps minimising bacteria movement), whilst others attempt to satisfy another (e.g. maximising denitrification). In Section 10, an attempt has been made to address the entire problem of contamination related to on-site sanitation, and not to focus on only one issue. Guidelines are discussed which set out to minimise the overall health risk due to this form of sanitation.

The above discussion has focused on the migration of contamination through a soil profile that is essentially continuous. A situation which has proved to be potentially hazardous from the point of view of rapid travel times of pollution is when macro-pore flow may occur. As an example, in an unfractured shale, after fifty years hydrocarbons had penetrated no more than 3m into the unsaturated shale (Barcelona, 1990). If the shale were fractured, however, flow could be virtually unrestrained and thus very rapid. An additional difficulty is that it is extremely difficult to locate monitoring wells to intersect potentially contaminated flow because of the myriad of potential flow paths. As reported by Taussig and Connelly (1991), latrines founded on highly jointed Sibasa formation lava, 8m above the water table caused extensive pollution, resulting in nitrate levels up to 310mg/l. In their study in Botswana, Lewis et al (1980a) carried out tracer tests, and found that movement took place very quickly, with the tracer travelling through the 4m of fissured bedrock between the pit latrine and the water table in only 25 minutes. There was thus obviously very little time for natural elimination of faecal bacteria to occur.

In another study, Cherry (1989) reports on an investigation centred on the contamination of an aquifer overlain by fractured clayey till. The vertical, or near vertical fractures resulted in a bulk vertical hydraulic conductivity that was two orders of magnitude higher than that of the intact clayey till. A point that emerges from this study is that the orientation (if any exists) of fractures may be extremely important in determining whether or not rapid migration of pollution is likely. As an example, if fracturing were predominantly horizontal or near-horizontal, the pollution plume would still have to travel through intact material in order to reach the groundwater. In the above study, contaminant migration through the unfractured till was found to be due to diffusion only, and the breakthrough time was of the order of hundreds to thousands of years.

Another unusual mode of contamination migration is described by Germain and Frind (1989). Although their study was analytical, the model they developed solved for advection and diffusion along a fracture, together with diffusion through the intact 'porous' blocks. Their analysis showed that under the assumed conditions, migration of a single pulse of contaminant along fractures is significantly retarded by diffusion into the porous material. Thereafter, diffusion out of the blocks is extremely slow. Another point made in this study is that if the fractures are not fully saturated, the flow rate may be reduced, as has been noted for flow through unsaturated soil.

The above discussion has once again served to illustrate the confusing nature of the influence of hydrogeological parameters on the sub-surface movement of contamination. A consistent approach to dealing with this problem is suggested in Section 10.

8.6 Computer modelling of pollutant transport

Groundwater modelling provides a potentially useful tool for understanding the mechanisms of groundwater systems and the processes that influence their response to various inputs (e.g. a point source of contamination) or outputs (such as withdrawal of water from a well). Modelling provides a technique for assisting in the interpretation of field data, and for planning additional field monitoring.

The term 'groundwater modelling' is very general, and covers many possible techniques and approaches. In the past, much attention was focused on the modelling of saturated flow in the sub-surface. Recently there has been a great deal of work on modelling more complex flow regimes, such as variably saturated flow, and flow through fractured media. All these models concern themselves with the movement of water, i.e. advection. An additional development has been the coupling of some of these models to techniques for simulating the movement of contaminants within the water, which may include allowance for diffusion, retardation or even growth (i.e. an increase in concentration of a particular contaminant). It is these latter models that potentially provide aids towards evaluating risks associated with on-site sanitation.

8.6.1 Applicability and verification of groundwater models

Before discussing the availability of models, and the input requirements for these models, it is perhaps appropriate to discuss some of the difficulties that may be encountered with the modelling of groundwater flow. One often-reported problem is so-called 'numerical dispersion', in which the actual physical dispersion mechanism of the contaminant transport cannot be distinguished from dispersion resulting from the numerical approximations. Another general problem is associated with the use of mass transport models in conjunction with flow models. Although a pollution problem is typically three-dimensional, a technique of vertical averaging is generally used, resulting in the utilisation of a two-dimensional, horizontal mass transport model that is connected to a hydraulic flow model. These models tend to underestimate peak values of contaminants, and could therefore fail to predict dangerously high levels of a particular substance.

Associated with concerns about numerical accuracy is the question of testing and verification of models for simulating a particular problem. As discussed below, there is a plethora of computer models currently available to modellers, many of which appear to be ideal for the envisaged application. In the documentation of many of these models, however, there is no information or data given on how the model was verified. The user is therefore expected to take the accuracy of the program at face value. This issue has been examined in a document produced by the International Ground Water Modelling Centre (I.G.W.M.C.) (van der Heijde, 1985). To fully verify a model, three levels of testing are recommended.

1 Comparison of predictions from the model with analytical solutions. This level of testing is in some ways the most rigorous, because exact solutions are available, but is also limited because analytical solutions are available for only relatively simple, well defined problems.

2 The solution of hypothetical, irregular problems using the model under study. This requires that the problems to be studied have already been analysed using previously verified models. It is thus an on-going process, and this level of testing will become more rigorous as more models are tested.

3 The analysis of field problems, for which data have been collected. This provides perhaps the most severe level of testing, in that extremely complex problems may be tackled, but suffers from the problem of accurately defining all the input parameters (e.g. soil hydraulic conductivity), as well as errors or inaccuracies in the field data.

It is clearly very important that some form of verification of a particular computer code be carried out, and that the verification be appropriate to the planned application, e.g. it is not sufficient to verify that a code produces correct solutions to a particular problem of fully saturated flow if it is planned to use the code to analyse contaminant migration in a partially saturated aquifer. Another question that needs to be addressed is the question of public versus private domain software. In the United States, the Environmental Protection Agency (EPA) have reported on problems of scrutiny and incentive, and questions of legality relating to the use of non-public domain software. Their approach is increasingly to insist that any software to be used on an EPA-funded project must be available in the public domain, and have gone through a process of verification. With the potentially politically sensitive nature of the widespread use of on-site sanitation in South Africa, it may be advisable to consider adopting a similar approach here.

8.6.2 Available computer codes for the analysis of groundwater flow

This section summarises some of the computer codes that are available for the analysis of groundwater problems. For completeness, fully saturated as well as variably saturated codes are included, as are codes for the modelling of contaminant transport. A listing of commonly used codes in the United States is given in EPA document #66/2-89/028 (van der Heijde et al, 1988), which details 97 codes for fully saturated flow, and 73 codes for solute transport. Furthermore, it should be borne in mind that the list details primarily USA derived codes. A large amount of work has been carried out at places such as the University of Karlsruhe, in Germany, which is not reflected in the list.

Evaluation of specific codes

It would clearly be a very large undertaking to evaluate all the models listed in EPA document #66/2-89/028. Summary evaluations of a few of these codes by the EPA indicate that the vast majority require of the order of 2-3 months of training (per program). In the following section therefore, a limited number of codes have been considered with respect to their likely applicability to on-site sanitation systems, as well as availability.

Codes that deal with the percolation of agricultural chemicals into the sub-surface provide a reasonable first estimate for simulating the influence of on-site sanitation on the groundwater environment. A review of seven of the most widely used codes for this purpose was conducted by Jones et al (1988), and is briefly summarised here. The codes that were evaluated were, PESTAN, MOUSE, PRZM, CMLS, GLEAMS, LEACHMP, and RZWQ. It is notable that not all of these codes are included in the above EPA document, again indicating the very large number of codes that are available. Most of the codes use a simple water balance approach, and require daily rainfall data as input. They tend to assume that no movement of water (or solute) occurs until the field capacity of the soil is reached, which may underestimate diffusion of contaminants. The code LEACHMP appears to be the most robust, and to be the most applicable to movement of contamination from on-site sanitation out of the seven reviewed. In solving the water balance equation it makes use of the Richards equation for unsaturated soil. However, it suffers from the same limitation as the other codes that are discussed, which is that they are unable to simulate non-point source pollution. Furthermore, all the codes appear to suffer to some extent from problems of numerical dispersion, a topic that was briefly touched on earlier.

In the discussion below, codes that appear to hold most promise for application to modelling on-site sanitation systems are described. It should be noted that these codes were developed for applications other than those discussed in this report (e.g. primarily for modelling the movement of hazardous waste).

UNSAT-H

This is apparently a fairly widely used program in the USA, and was thus chosen for evaluation. It simulates variably saturated flow, but is limited to one-dimensional flow. It can also incorporate infiltration, drainage, evaporation and plant water uptake. It is very cumbersome to use, and a great deal of effort is necessary in input preparation. It was also designed primarily for use in simulating sites in arid areas, and has not been verified for other climatic conditions. It is unlikely to be of major use in the analysis of the kind of problems discussed in this report.

CHEMFLO

The code was developed by the EPA, and is intended to simulate the movement of water and chemicals in the unsaturated zone. Water movement and chemical transport are modeled using the Richards equation and convection-dispersion equations respectively. Flow is limited to one dimension, the soil is assumed to be homogeneous, and hysteresis in the wetting and drying of a soil is neglected. Although the program had a dialogue input facility, it was found to be very inflexible, and data input was very slow. The latest version apparently addresses these issues, and includes a menu for data input. This version was not available for evaluation, but if it facilitates data input the code could be useful for the intended application. The code allows the user to choose from a variety of pre-defined moisture retention characteristic curves (e.g. the Brooks-Corey or the van Genuchten models). However, during the evaluation process, an error in the coding of some of the above characteristics was discovered. The error was communicated to the suppliers, but to date no response has been received.

VIROTRANS

From the paper by Tim and Mostaghimi (1991), this model looked to be very promising for the envisaged application. It was designed to simulate the movement of viruses through soils, and the split between advection and adsorption is accounted for. Unfortunately all attempts to contact the authors, with a view to obtaining a copy of the program have been unsuccessful. From the above article, there appear to be a large number of input parameters, and it is not clear if the model can accommodate unsaturated flow. Nevertheless, it appears to be promising, and it is worth continuing to try and obtain a copy of the code.

BIOPLUME II

This code was developed for simulating transport of hydrocarbons in ground water under the influence of oxygen-limited biodegradation. The model analyses two-dimensional flow, and computes the changes in concentration with time due to convection, dispersion, mixing and biodegradation. The concept behind the development of the code was to be able to simulate natural biodegradation processes, as well as bioremediation processes. The code was developed specifically for modelling migration of hydrocarbon pollution, and the biodegradation formulations are therefore all formulated in these terms. The code cannot easily be used for simulating the movement, and possible retardation of a nitrate plume, although it does appear possible if some of the input data could be re-formulated. The documentation is adequate, although there are no verification examples. A further drawback is that only saturated flow may be modelled, although if the source code could be obtained, this could be easily rectified. The code is relatively easy to use, and the developers have wisely decided to make the code compatible with a commercially available program (the contour plotting program SURFER), which greatly simplifies post-processing of data.

FLAC

This code was designed for the analysis of continuum problems, such as problems of stress analysis in rock and soil mechanics. It is a two-dimensional explicit finite-difference code, and problems of groundwater flow can be analysed together with the stress analysis components. It was thus not specifically designed for flow problems, but has undergone extensive refinements over the past few years, and now provides a useful technique for analysing problems of this type. Saturated, as well as variably saturated flow can be analysed, although a very simplistic approach is used to account for the variation of hydraulic conductivity with moisture content. There is no facility for simulating the retardation or adsorption of contaminants, and the user is thus limited to deriving an upper bound estimate of the pollution migration. Despite this limitation, the code was the easiest to use, and the documentation by far the most comprehensive. The finite difference grid is easily set up, and plotting and viewing of the output is straightforward.

CTRAN/W

This code is marketed by Geoslope Inc. of Canada, and may be used to model the movement of a contaminant through soil. Unfortunately it has to be used in conjunction with a second code (SEEP/W), which is used to carry out the advective component of flow. Despite this annoying restriction, the code has many advantages. These include the ability to model variably saturated flow, transient boundary conditions, heterogeneous ground conditions, and adsorption and decay of a contaminant. The documentation is excellent, and verification examples are included. Although not a major concern, the code only runs under the WINDOWS environment, and it is therefore necessary to acquire this additional software in order to run CTRAN/W. Output from the program may be obtained in graphical format, and is easy to handle. A particular advantage of this code is the facility for 'particle tracking'. The position of a designated particle at any time after the initiation of movement may be followed with a view to, for example, determining whether or not it will reach a drinking water well. This provides a useful technique for evaluating the importance of retardation or decay processes in contaminant migration processes.

In summary, there is no code that has been derived specifically for the analysis of potential pollution from on-site sanitation. Codes that have been developed for the simulation of contaminant migration from waste disposal sites would seem to be most appropriate, as long as the deficiencies are accounted for. These codes must be capable of simulating flow in the unsaturated zone, and should be able to account for processes such as decay of a particular contaminant. Such codes are of necessity more complex than the more common saturated flow codes, and require a good deal more input data. These additional data may be difficult to obtain, and recourse will then have to be made to rules of thumb, e.g. the simplified curves of hydraulic conductivity versus moisture content produced by the I.G.W.M.C. (Van der Heijde, 1986). This is of course not satisfactory, and it is preferable that when computer simulation is planned for a particular project, care be taken to ensure that the necessary input parameters can in fact be determined.

Another point that must be noted is the need for training of personnel that may make use of these computer codes for the evaluation of the suitability of a particular site for on-site sanitation. As mentioned before, the EPA consider that of the order of months are necessary to become adequately familiar with a single code. It is suggested that a few (perhaps 3-4) codes are chosen for a thorough evaluation of their applicability to the problem of on-site sanitation. Depending on the outcome of the study, a hierarchy of 'preferred' codes could be formulated.

Finally, it should be stressed that although there are obviously a number of factors that make the widespread use of computer simulation difficult, it would be contrary to international trends if we were to neglect this technique as an aid in evaluating the pollution potential of on-site sanitation.

9 EXISTING GUIDELINES FOR MINIMISING ENVIRONMENTAL IMPACT OF ON-SITE SANITATION

One of the earliest guidelines to ensure that on-site sanitation did not cause pollution of groundwater was provided by Dyer and Bhaskaran (1945), who recommended minimum distances between an on-site latrine and water withdrawal points. Based on field studies, they concluded that in sandy soils bored latrines could be placed as close as 6m to a water supply well. This was soon found to be unrealistic, and a minimum distance of 15m became accepted practice, until more rigorous guidelines were developed (Lewis et al, 1980b).

Recognition that guidelines of the above type were overly simplistic, because they ignored factors such as site hydrogeology, later workers suggested guidelines that attempted to address this deficiency. Romero (1972) consolidated data from a number of international case studies to produce a graph such as Figure 9.1 shown below.

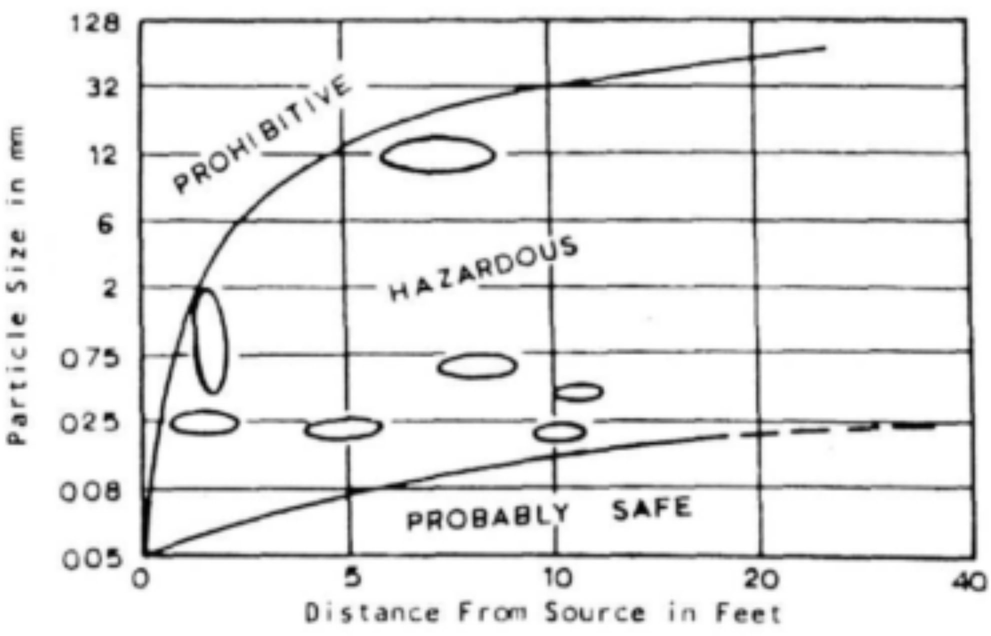
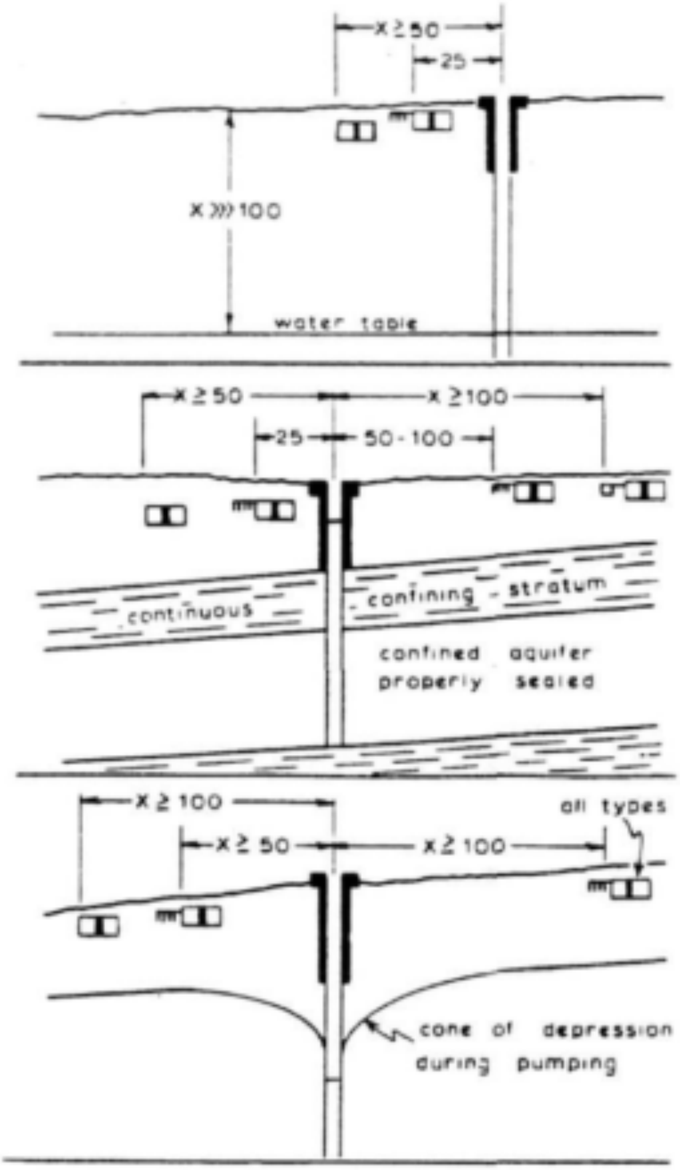


FIGURE 9.1: Biological pollution travel in unsaturated soil (after Romero, 1972)

This graph shows the travel of bacterial pollution in unsaturated soil profiles as a function of the particle size of the soil. Three zones are indicated on the graph. These define distances from the contaminant source that could be considered probably safe, hazardous or prohibitive. Although an improvement on simplistic, minimum distance guidelines, Figure 9.1 is still deficient, particularly since it is impossible to characterise a soil by a single particle size, as has been implied in this figure.

In the same publication Romero (loc.cit) provides an improved version of the 'minimum distance' guidelines. An example is shown in Figure 9.2, which suggests minimum horizontal distances from septic tanks, that are dependant on underlying hydrogeological conditions.



Legend:

Septic tank with absorption trench or seepage bed

Septic tank with seepage pit

Septic tank with no absorption system

Note:

- All dimensions in feet
- All diagrams cut sections of unconsolidated sand and gravel unless otherwise indicated
- Diagrams apply to areas in which the zone of saturation is greater than 10 feet from the source of contamination

FIGURE 9.2: 'Safe' distances between septic tanks and drinking water wells (after Romero, 1972)

A more recent attempt to account for varying hydrogeology, and how it impacts on pollution from on-site sanitation. Lewis et al (1980b) proposed a 'pollution risk array', which is illustrated in Figure 9.3. This figure indicates the type of soil profile within which the pollution risk may be minimal or high, or where insufficient knowledge is available to make a judgement. Although useful, the figure only illustrates relative pollution potential, and does not quantify the pollution risk associated with different soil types.

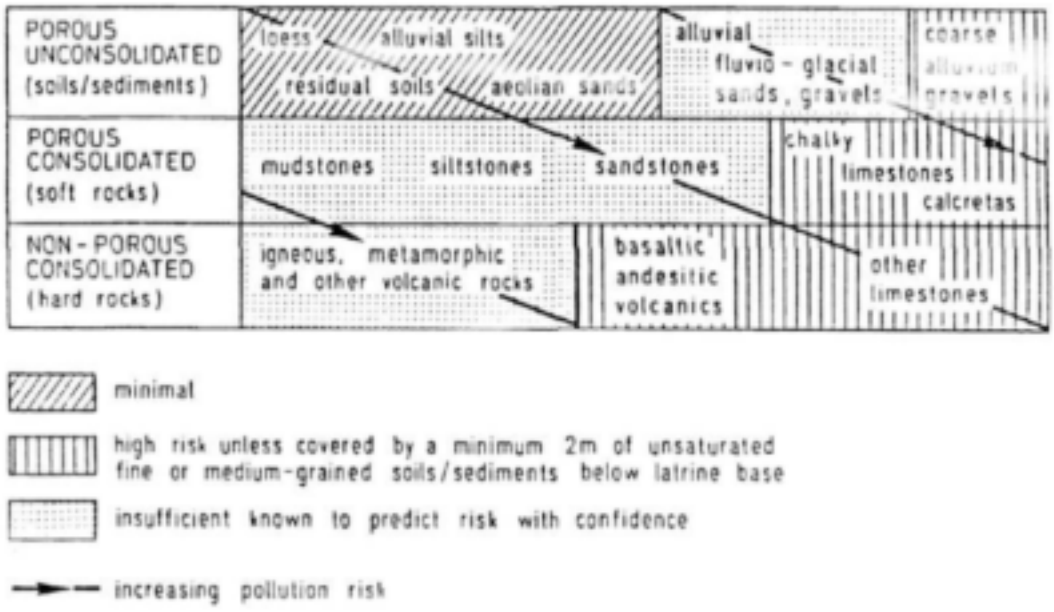
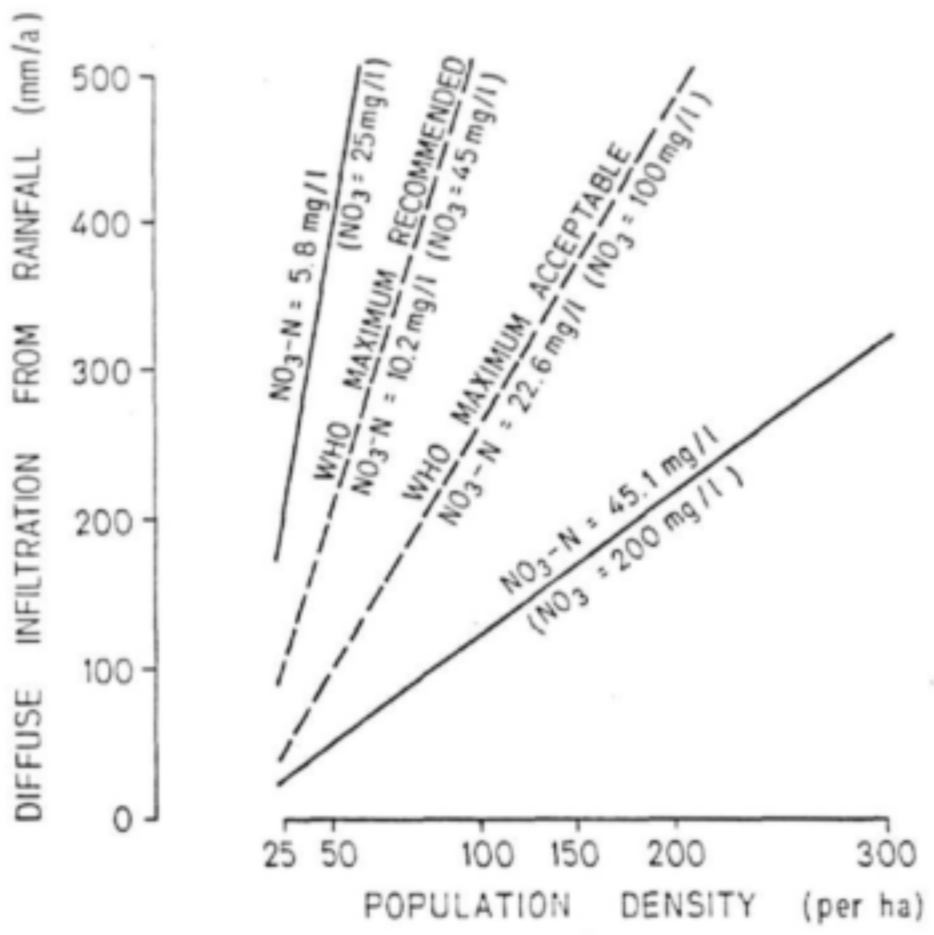


FIGURE 9.3: Classification of soils and rocks in an array of relative pollution risk (after Lewis et al, 1980b)

As noted in the report, the most pervasive form of pollution resulting from on-site sanitation is likely to be nitrate pollution. Lewis et al (1980b) present a theoretical relationship between the density of a settlement using on-site sanitation, the rainfall infiltration into the soil, and the resulting nitrate concentration in water infiltrating local groundwaters (see Figure 9.4). This approach, which is based on a simple mass balance, has also been utilised by Palmer (1981), and Muller (1989).

Lewis et al (loc.cit) and Muller (1989) present data which may be used to verify the reasonableness of this graph, but simultaneously include other data which do not fit the findings. Figure 9.4 should therefore be regarded merely as a reasonable first estimate. Many other factors (such as the magnitude of denitrification processes in the unsaturated zone) need to be quantified before a result such as that summarised in Figure 9.4 can be used with a high degree of confidence.



Estimation of the effects of unsewered sanitation schemes on the NO_3-N concentrations in local groundwaters.

Assumptions:

- Generation of 5 kg N/capita/annum and effluent volume of 1 to 10 l/capita/d
- 90 % of N lost directly to atmosphere from excreta disposal unit
- Dilution from diffuse infiltration and effluent liquid only

FIGURE 9.4: Estimated effect of on-site sanitation on local groundwater, as a function of population density (after Lewis et al, 1980b)

In terms of regulations, Table 9.1 shows the recommended safe distances between a range of on-site sanitation systems and domestic wells that are in use in the United States, Romero (1972).

TABLE 9.1: Recommended safe distances between domestic wells and various on-site sanitation systems, (after Romero, 1972)

Source of pollution	Recommended distance (m)			
	California	Colorado	FHA*	USPHS•
Septic tank	16	30m from point of juncture between well casings and aquifer	16	16
Sewer lines with watertight joints	3	As above	3	3
Percolation field	30	---	30	30
Absorption bed	30	8m minimum horizontal distance	30	-
Seepage pit	30	As above	30	30

* Federal Housing Authority
• United States Public Health Service

In South Africa, the approach to limiting pollution from on-site sanitation has generally been to adopt the 'minimum distance' approach. De Villiers (1987) summarised the guidelines for areas where the groundwater is used for household purposes, and the water source is located downgradient from a latrine as follows:

- 15m from the water source if the water table is quite shallow (1m to 5m below the bottom surface of the pit or soakaway)
- 30m from the water source if the water table is very shallow
- 7.5m from the water source if the highest seasonal water table is more than 5m below the bottom surface.

He noted that these criteria are not applicable to areas where fissured rock, limestone or very coarse subsoils occur.

The same approach as the above was followed in guidelines for the provision of services for developing communities (the so-called 'Green Book', Department of Development Aid, 1988).

A particularly simple approach to the problem is provided by the National Building Research Institute (NBRI, 1984), who state that VIP latrines are suitable for use in residential areas with a maximum of 250 persons per hectare.

The suitability of existing guidelines is discussed in Section 10.

10 CONCLUSIONS AND RECOMMENDATIONS

10.1 Current state of knowledge

In reviewing the current state of knowledge on the environmental effects of on-site sanitation, particular attention was given to its applicability to the South African situation.

It was found that most of the available literature addressed one of two approaches to sanitation provision. Both of these approaches limit the use of on-site sanitation to certain densities of settlement:

- In developing countries, where waterborne sewerage is largely unaffordable, substantial improvements in health are obtained by the use of on-site sanitation. Possible pollution of the groundwater somehow appears less significant in the face of the alternatives (high standards for a few, and minimal coverage for the rest of the population). Consequently, contamination of the groundwater is not considered unless the water is to be used for drinking purposes.

- In developed countries, where waterborne sewerage is largely affordable, it is used primarily for reasons of convenience, but also on the assumption that there is no associated pollution risk. Where on-site sanitation has been used on a large scale (as in the case of septic tanks in the USA), it has been at relatively low densities.

The situation in South Africa straddles the above two scenarios. It does not have the financial resources of a highly developed country; at the same time it is a water-scarce country and is concerned not to pollute the water resources it does have. Use of on-site sanitation in high density applications in a region of scarce water resources, as is being considered for the large metropolitan areas in South Africa, is a significant departure from existing approaches.

Although there has been considerable research carried out on septic tanks and related groundwater contamination problems, there has been relatively little work carried out on contamination from LOFLOS (aquaprivies) or VIP's, which are the main forms of on-site sanitation currently being considered in South Africa.

Environmental impact of on-site sanitation is a multi-disciplinary topic, covering a wide number of fields, including

- Water quality
- Health
- Microbiology
- Geohydrology
- Risk assessment

There appears to have been relatively little work done which integrates the different fields.

Sanitary wastes contain both chemical and microbial contaminants, which, in sufficient quantities, are potentially harmful to human health and to the aquatic environment. The critical question is whether these contaminants will be transported to the groundwater or surface water in sufficient concentrations to pose a pollution hazard.

From the literature, it appears that once these contaminants reach the groundwater, transport is relatively fast. For this reason, although there will be a certain amount of contaminant decay within the groundwater, the point of compliance for acceptable contamination levels is taken as the water table level, if the water is to be used for drinking purposes.

There are several processes which may retard or prevent altogether the transportation of contaminants through soil. These are particularly important in the unsaturated zone where the rate of water movement is low. These processes include amongst others filtration and adsorption, as well as the various chemical processes such as nitrification and denitrification. As a result of these processes, the risk of contamination from bacteria and viruses (as long as they are not washed out onto the surface) is very low. There is, however, likely to be contamination from nitrates. (Consideration of mass balances alone indicates that this is the case). The primary concern therefore is nitrate pollution.

It is not clear to what extent nitrate can be denitrified in the soil to produce nitrogen gas which will escape into the atmosphere (and hence not pollute the groundwater). The optimum conditions for the proliferation of nitrate reducing bacteria have not been investigated. It is also not clear to what extent their growth can be induced by artificial means.

Once the nutrients (such as nitrate) get into an impoundment, their effect on the ability of the water body to support life is also dependent on a number of factors (including whether the impoundment is nitrogen or phosphorus-limited).

In summary we are talking about long term effects that are too significant to ignore, but difficult to determine.

To ensure correct perspective is maintained when evaluating the environmental effect of on-site sanitation, it is necessary to determine what the contamination from other sanitation options, such as waterborne sewerage, are likely to be. The often mentioned assumption that waterborne sewerage causes no contamination is very probably erroneous, and the potential contamination from this source is very relevant in the context of the relative costs of various sanitation options. Comparison of relative amounts of contamination should not stop at different sanitation systems. The relative impact of contamination arising from mine waste disposal, solid waste landfills, and agricultural activities also need to be quantified. Only in this way can informed decisions as to the true cost and benefit to South Africa of implementing various sanitation options be made.

10.2 Suitability of existing guidelines

There are numerous factors which influence the extent of the above processes, which vary from site to site. It is therefore not appropriate to produce guidelines in the form of an unqualified set of rules for the suitability of a site for on-site sanitation.

Virtually all existing guidelines are formulated in terms of safe distances between on-site sanitation and source of drinking water or in terms of acceptable densities of population utilising on-site sanitation. A particular difficulty is that decision-makers may be faced with the evaluation of a site that does not conform to any of the idealised hydrogeological profiles implicit in the above guidelines.

The goal of the above guidelines is the protection of the groundwater. They do not specifically address the problem of potential pollution of surface water resources, i.e. they do not address the situation where contamination of the groundwater may be acceptable (because it will not be used for drinking purposes), but where surface water resources need to be protected.

10.3 Objectives that guidelines need to address

It needs to be recognised that compliance requirements may vary, depending on the objective of these requirements. If groundwater is to be used for drinking purposes, then there will be a particular compliance requirement, whereas if surface water resources are the primary concern, and protection of the groundwater is not a consideration, some other compliance requirement will prevail.

To ensure this required flexibility, guidelines should provide a methodology for evaluating the effect of on-site sanitation on the environment, specifically water resources, and for evaluating whether compliance requirements will be met.

This methodology should address the nature and extent of contamination from on-site sanitation.

More specifically, the following must be defined:

- Nature of contamination: is it microbiological or chemical contamination that is of concern? As outlined above, nitrate is highly likely to be the more mobile of the contaminants. On the other hand, the infectious doses of viruses (or even some types of bacteria) are much smaller than is that of nitrate.

- Extent of contamination: all evidence from the literature indicates that movement of viruses and bacteria in the subsurface is likely to be very limited, unless particularly unfavourable conditions (e.g. limestone formations) exist. Nitrates, however, appear to be far more mobile, and may travel extensive distances, even in relatively impermeable soil. In formulating guidelines, it is the extent of this travel that needs to be defined. Does it continue inexorably, or is it ultimately arrested by natural denitrification processes?

10.4 Recommended strategy for the evaluation of the environmental impact of on-site sanitation

In seeking to provide access to adequate sanitation for all its inhabitants, one of the options the country has is the widespread use of on-site sanitation at relatively high densities. It needs to be emphasised that the opportunity now exists for the actual planning of relatively high density use of on-site sanitation, particularly in the fast-growing urban areas.

Simplistic guidelines that consist of a few, easy to follow rules are unable to take account of the multitude of variables that influence the potential environmental effect of on-site sanitation. The following strategy is therefore suggested:

- Define compliance requirements that must be met. If not implicit in these requirements, define the point of compliance, and allowable mass loading or concentration of contaminants.

- To estimate the risk of pollution of water resources by viruses or bacteria use the 'residence time' approach. This entails a calculation of how long it would take a 'particle' of water to travel from a latrine to the point of compliance. Techniques for doing this could vary from simple, hand calculation techniques, to sophisticated finite element computer analyses, depending on the complexity of the hydrogeological conditions underlying the latrine. If the travel time exceeds about 150-200 days, then according to survival times recorded in the literature, microbiological contamination should be eradicated.

- To estimate the risk of pollution of water resources by nitrates use a mass balance approach. This approach requires knowledge of a number of factors, including the proportion of nitrogen leached from the on-site sanitation system, the amount of rainfall that infiltrates the sub-surface, and the rate of denitrification in the subsurface. Although very rough estimates of these factors have been made by various authors, the sensitivity of the computed nitrate levels to these factors require that they are more accurately defined. Although potentially useful, the mass balance approach is currently inadequate to assess whether compliance requirements for a particular site can be met.

- Until such time as adequate data relating to the input parameters that are required for the above approach become available, it will be necessary to carry out field monitoring of on-site sanitation schemes if the water resources are to be protected. This approach is necessary to provide an early warning system that contaminant levels may build up to hazardous levels at some time in the future, and to allow alternative sanitation strategies to be implemented, or remedial measures to be taken.

10.5 Recommendations for further research

To address the uniqueness of the South African situation (high density use of on-site sanitation, coupled with the need to protect scarce water resources), the following research topics need to be addressed:

- Quantification of the 'source' (ie the contaminant loading applied to the soakaway). In particular, the volume and characteristics of the contents of VIP pits, and the effluent from LOFLOS need to be determined.

- Quantification of the movement of these contaminants in the subsurface. Of particular concern is the determination of the likely extent of the nitrate contaminant plume resulting from high-density use of on-site sanitation systems. Questions that need to be addressed in this work include whether or not the nitrate plume will ultimately be arrested by denitrifying bacteria, and whether it is possible to initiate or accelerate such a process (e.g. by lining a pit latrine wall with an optimum soil mixture) or by the addition of a substance such as glucose to accelerate bacteria growth.

- Quantification of the effect of these contaminants on the groundwater and surface water resources (by both modelling and field monitoring). A critical question is the contribution of nutrients originating from on-site sanitation systems to eutrophication of surface water resources.

- Development of measures to contain and/or render harmless these contaminants. These may include the use of appropriate materials for pit lining, the addition of glucose to the soil adjacent to the pit or soakaway, and the implementation of pit-emptying programmes.

- An assessment of the cost of the pollution, including treatment of affected water to acceptable standards, remediation of contaminated soil, and the implementation of preventative measures.

Further topics which impact on the viability of on-site sanitation, and which need to be researched include:

- Treatment, disposal and re-use of sludge from on-site sanitation systems
- Installation, operation and maintenance of on-site sanitation systems, including the management of these operations
- Communication of appropriate information to communities, particularly with regard to on-site sanitation, to enable them to make an informed choice of sanitation system

Much of the existing research has concentrated on the negative impacts of on-site sanitation. There are considerable benefits to public health and to the environment to be gained from the use of on-site sanitation, and this needs to be taken cognisance of in future work.

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