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Assessing Risk to Groundwater from On-site Sanitation: Scientific Review and Case Studies

Groundwater Systems and Water Quality Programme

Commissioned Report CR/02/079N

ARGOSS

BRITISH GEOLOGICAL SURVEY

COMMISSIONED REPORT CR/02/079N

Assessing Risk to Groundwater from On-site Sanitation: Scientific Review and Case Studies

British Geological Survey

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Executive Summary

Many people in developing countries rely upon untreated groundwater supplies for their drinking water. During the United Nations Water Decade of the 1980s considerable progress was made with the provision of safe drinking water and sanitation in both rural and urban areas. Even so, in many of these countries a substantial portion of the population still do not have access to an adequate water supply whilst access to sanitation facilities continues to lag behind that of water supply (WHO and UNICEF, 2000).

It is recognised that further improvements in safe water and sanitation coverage will require both increased use of groundwater and greater utilisation of on-site sanitation systems. However there are concerns, that these two solutions may conflict. Under some hydrogeological conditions on-site sanitation systems may result in severe contamination of groundwater which could negate the anticipated health benefits.

Given these concerns, a project 'Assessing the Risk to Groundwater from On-site Sanitation (ARGOSS) was undertaken during 1998-2001 by the British Geological Survey, the Robens Centre for Public and Environmental Health, the Geology Department of the University of Dhaka, the Civil Engineering Department of Makerere University, Kampala and International Center for Diarrhoeal Disease Research Bangladesh and funded by the Department of International Development (DFID) under the Knowledge and Research Programme as part of the UK provision of technical assistance to developing countries.

The purpose of this project was to:-

(i) review and assess the risk posed to drinking water supplies by on-site sanitation systems

(ii) develop a manual that provides guidance on the safe design and siting of drinking water supplies with respect to on-site sanitation facilities.

This report provides a review and assessment of the risk posed by on-site sanitation systems to groundwater based on a survey of the scientific literature and detailed case studies undertaken by the ARGOSS project. The report is in two parts: Part 1 is a review of the scientific literature and provides the scientific basis for the guidelines presented in the companion ARGOSS manual (ARGOSS 2001). Part 2 describes the two case studies undertaken by the ARGOSS project team, in Uganda and in Bangladesh. A third case study is also presented describing the results of research in Argentina by the University of Rio Cuarto. This case study was not funded by the ARGOSS project.

The companion ARGOSS manual (ARGOSS, 2001) provides guidance on how to assess and reduce the risk of contamination of groundwater supplies from on-site sanitation systems and is aimed at those responsible for planning low cost water supply and sanitation schemes.

1 Introduction

1.1 PURPOSE AND SCOPE OF SCIENTIFIC REVIEW

The purpose of this report is to:

- review the scientific literature related to pollution of groundwater by on-site sanitation, principally relating to micro-organisms and nitrate
- identify the factors which control the movement and fate of these contaminants
- present the scientific basis for pollution risk assessment described in the companion ARGOSS manual
- identify areas of uncertainty which require further research

Comprehensive literature reviews on the impact of on-site sanitation systems on groundwater have been published (Lewis al 1982) and more recently by Ryneveld and Fourier (1997). These reviews observed that most of the scientific literature on pathogen behaviour was published in the United States and related to the disposal of septic tank effluent. A number of recent case studies from developing countries have been published but these are in general insufficiently detailed to provide definitive evidence for contaminant migration and behaviour. Nevertheless they do provide valuable data. Routine monitoring of rural drinking water supplies appears to be the exception in most developing countries.

1.2 SANITATION OPTIONS

The choice of sanitation technology depends on many economic, technical and social issues and each type of technology has advantages and disadvantages. Sanitation facilities may be water-borne or dry. There are many different forms of sanitation ranging from conventional and modified sewerage, to water-borne on-site systems such as septic tanks, aqua privies and pour-flush latrines to dry systems which are generally different forms of pit latrines, some of which may include urine separation (Figure 1.1). The choice of sanitation system is based partly on availability of water, but also on cultural reasons and anal cleansing methods. Sanitation systems can be divided into two principal categories:

(i) Off-site methods: these are forms of sewerage, where faecal and household wastes are carried away from the household. No treatment occurs at the household and the waste must be taken to a treatment plant before discharge into the environment.

(ii) On-site methods: these include septic tanks and all forms of pit latrines. In these systems the wastes are stored at the point of disposal and usually undergo some degree of decomposition on site. On-site systems either require periodic emptying or construction of new facilities once they fill up.

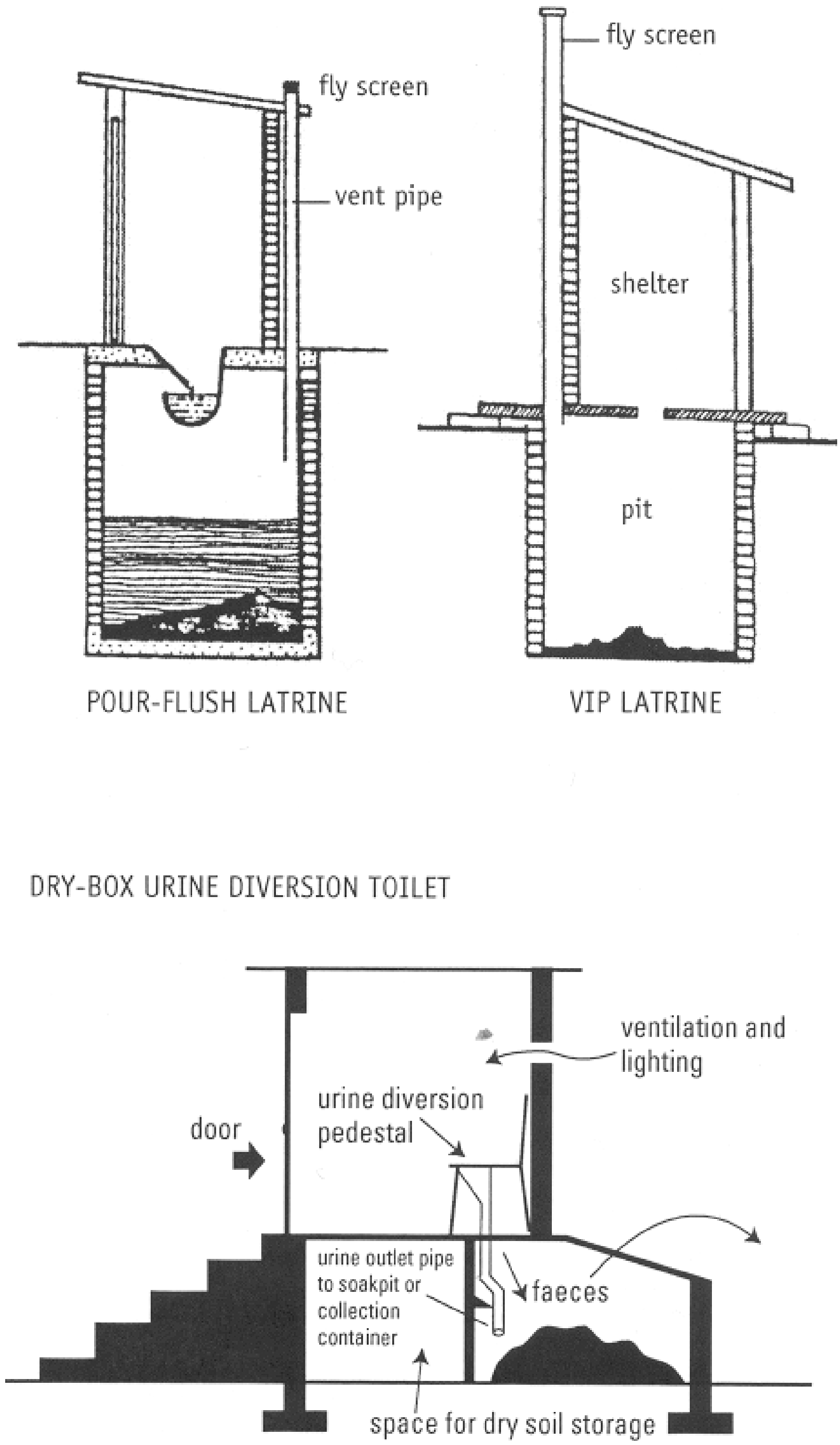


Figure 1.1 Examples of on-site sanitation design

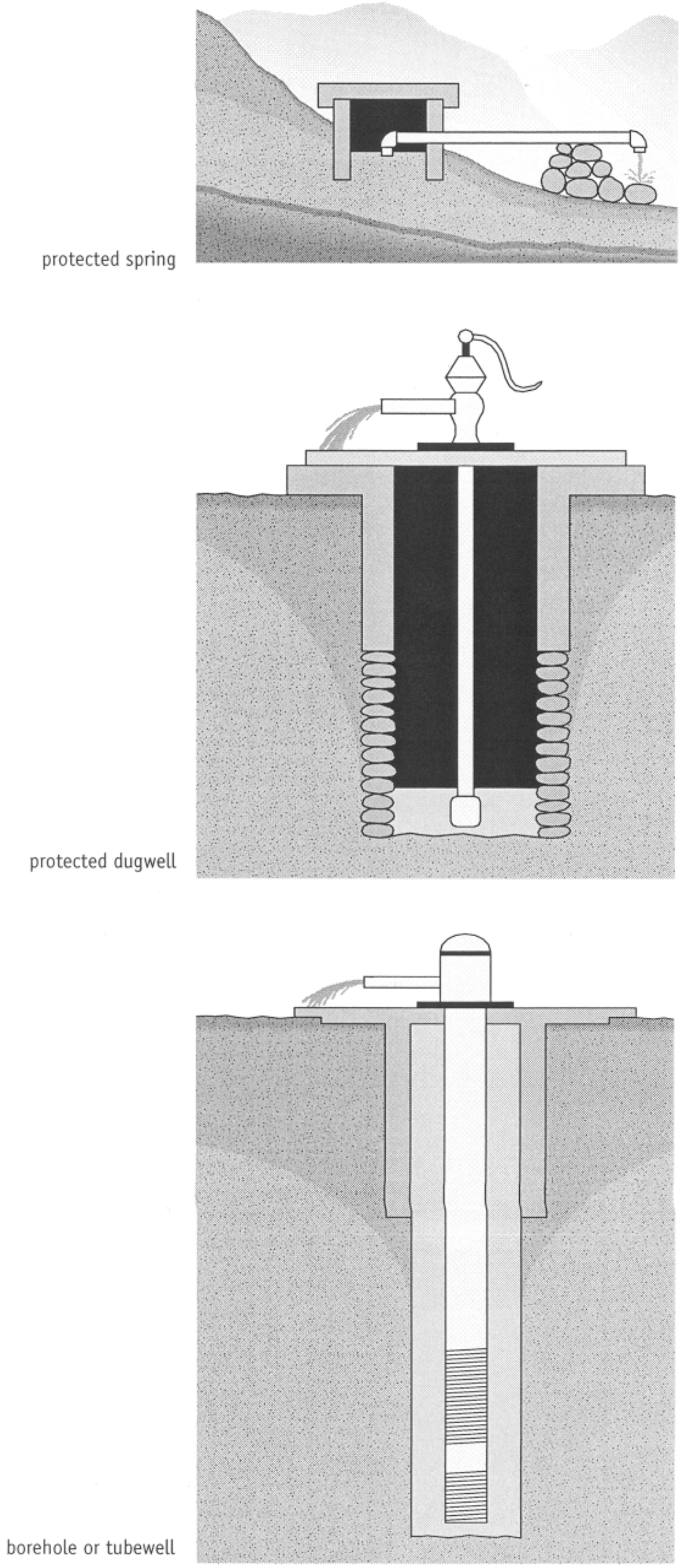


Figure 1.2 Typical groundwater supply designs

1.2.1 Off-site methods

Off-site methods are often found in urban areas where space constraints limit the potential for on-site facilities. They can provide a greater degree of convenience than on-site methods and ultimate responsibility for the treatment and disposal of waste usually lies with a utility or local authority. Conventional sewerage is very expensive and requires an in-house water supply to function properly. However, cost analyses have shown that modified sewerage becomes cheaper than on-site methods as population densities increase (UNHCS, 1986). Whilst sewerage is often viewed as the most desirable form of sanitation, it has several drawbacks. There is evidence from Europe that sewers may significantly contribute to microbiological and nitrate contamination of groundwater and therefore may represent a significant risk where groundwater is exploited for domestic supply (e.g. Barrett et al, 1999, Powell et al, 2001). Furthermore, sewerage requires treatment and can be poorly operated and managed and can lead to the discharge of inadequately treated wastes into the environment. Thus when considering the use of sewerage attention must be paid to the potential for groundwater contamination and ensure that systems are operated and designed with groundwater protection needs in mind.

1.2.2 On-site methods

On-site methods include both relatively expensive systems such as septic tanks that provide the same degree of convenience as a sewer (these are used in higher income parts of Kampala, Uganda, for example), and cheaper pit latrines (as used in middle and lower-income parts of Kampala and Iganga, Uganda and Dhaka, Bangladesh – see case studies in Part 2 of this report). On-site systems often represent a significant hazard to groundwater because faecal matter accumulates in one place and leaching of contaminants into the subsurface environment may occur. Septic tanks typically hold the solid component of wastes in a sealed tank where the matter decomposes anaerobically. Liquid effluent is usually discharged into a soakaway pit. In well-designed septic tanks, the solid matter does not represent a significant hazard, but the soakaway pits may cause both microbiological and chemical contamination.

Pit latrines, either dry or pour-flush, are more common in developing countries, especially in South Asia and Africa. Pit latrines are usually not sealed, although sealed pits may be used in urban areas or in areas of high water table. In general pit latrines are only appropriate where the water supply is off-plot and are not appropriate when large volumes of wastewater are generated. In most pit latrine designs, the liquid part of the waste is allowed to infiltrate into the soil, although some pour-flush latrine designs provide a soakaway. This infiltration of wastes (often containing micro-organisms and nitrogen, the latter of which may be oxidised to nitrate) represents an additional hazard to groundwater, particularly as this frequently occurs at some depth in the subsurface and thus by-passes the soil. The soil is the most biologically active layer and is where contaminant attenuation is greatest. However, biological communities also typically occur within the clogged zone that frequently develops where effluent infiltrates the subsurface (Kreissl, 1978) and are likely to include predatory micro-organisms capable of removing pathogens (Butler et al, 1954; Krone et al, 1958). This may help limit the risk of contaminant movement to deeper layers to some degree.

An important difference between 'dry' pit latrines and pour-flush latrines is the fluid loading to the pit latrine. In the former type the loading is likely to be less than 50 mm/d assuming a pit area of 0.8 m^2 and a maximum amount of liquid discharged to the pit of 33 l/d (Lewis et al 1982). However for pour-flush latrines the typical loading can significantly exceed 50 mm/d which increases the likelihood of groundwater pollution (Lewis et al, 1982).

1.3 WATER SUPPLY OPTIONS

In its natural state, groundwater is usually of good microbiological quality and as a result is often the preferred source of drinking water supply as treatment is limited to disinfection. In the rural and peri-urban areas, groundwater supplies are frequently untreated (see case studies in Part 2). However, the construction of groundwater supplies may allow a direct route for contamination of groundwater.

The principal forms of groundwater supply used for drinking water are shown in Figure 1.2 and are briefly discussed in the following sub-sections.

1.3.1 Dugwells

This is the traditional source of water in many areas of the world. These wells are usually relatively shallow, typically 5-20 m in depth and are brick or stone-lined, at least in their upper part. The depth of the well is usually dependent upon both the depth to water table (and the water table fluctuation) and the hardness of the strata. In areas of hard rock, the depth to bedrock may be the limiting factor deciding the depth of the well. The diameter of the well is very variable but is usually within the range 1-10 m; large diameter wells being particularly common in the Indian subcontinent.

These wells are frequently uncovered and, when used for domestic supply, water is normally withdrawn by bucket. They are highly susceptible to contamination, particularly by pathogenic micro-organisms and the various pathways by which contaminants can enter the wells are illustrated in Figure 1.3.

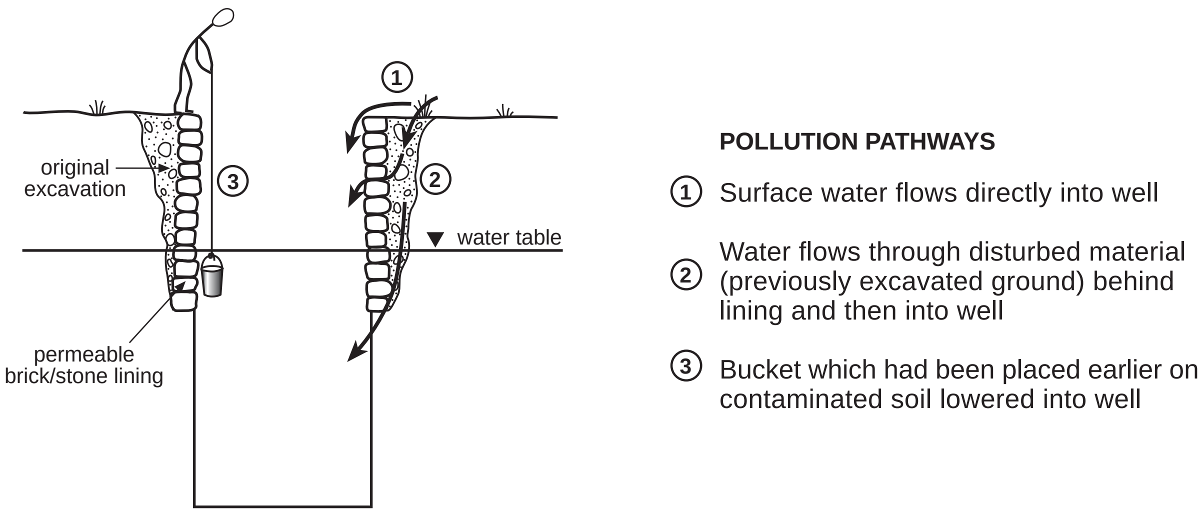


Figure 1.3 Routes for contaminant migration into open dug wells

Some wells are better constructed and have features designed to protect them from contamination:

(a) upper sections sealed with impermeable cement lining

(b) concrete surround to the well at least 1.5 m in diameter which slope away from well to prevent ingress of contaminated surface water

(c) drainage channel take water away from wells and prevent ponding

(d) covered well top, with a dedicated handpump or bucket.

1.3.2 Boreholes

In attempts to improve the bacteriological quality of well water and provide 'safe' drinking water supplies, many agencies have implemented programmes of shallow borehole drilling, especially in rural areas, to provide alternative sources to relativley unsafe open wells and surface waters.

These shallow boreholes are typically 100-150 mm in diameter, 20-50 m deep and fitted with a handpump. Whilst these drilled wells normally provide much better quality water than open wells (Lewis and Chilton, 1984; Howard, 2002) microbial contamination caused by inadequate sanitary sealing is not unusual and is especially a concern where hand drilling methods are employed in soft alluvial sediments. This is discussed in more detail in the Bangladesh case study (Part 2).

1.3.3 Springs

These may occur where groundwater discharges at the surface. They are generally protected by constructing a spring box around the eye of the spring, which is then backfilled with a fine gravel or sand to provide additional filtration. The areas backfilled is usually covered by layers of clay or finer material to limit percolation of surface water and the whole area is typically fenced and has a ditch to divert surface water away from the source. Water may be collected from spring directly by users or may feed piped systems by gravity. Springs can be susceptible to contamination and great care needs to be taken to protect the supply. The Uganda case study (Part 2) highlights the use of protected springs for water supply in both urban and peri-urban settings.

1.4 CONTAMINANTS ASSOCIATED WITH ON-SITE SANITATION AND POTENTIAL HEALTH CONCERNS

1.4.1 Microbiological contaminants

Different types of pathogens transmit infectious diseases. These have differing impact on health and transmission routes may vary. These should be understood in order to understand the health consequences of different types and levels of contamination. The pathogens that cause infectious diarrhoeal diseases that can be transmitted through contaminated water are grouped into three principal types of organisms: bacteria, viruses and protozoa. All these pathogens may be transmitted by other routes, including via contaminated hands, flies and animals. Helminths (or worms) are not included here as their size makes them unlikely to be present in groundwater supplies unless there is a direct entry for surface water, in which case pathogens of other types will also be present and are likely to represent a greater risk to health. Table 1.1 lists some of the major viral, bacterial and protozoan pathogens, the source of these pathogens and the associated diseases.

Bacterial pathogens cause some of the best known and feared infectious diseases, such as cholera, typhoid and dysentery which still cause massive outbreaks (or epidemics) of diarrhoeal disease and contribute to ongoing infections. As with all micro-organisms, theoretically a single bacterium is sufficient to cause infection within humans (WHO UNNICEF, 2000), however, in general, higher doses of bacteria are required to initiate symptoms than for other types of pathogens such as viruses. The control of such pathogens was the original target of the pioneers in sanitary improvements and this led to significant improvements in both water production process control and monitoring techniques. The control of bacterial pathogens in drinking water remains critical in all countries worldwide (Ford, 1999). Bacteria tend to be very susceptible to the natural processes which reduce their numbers (attenuation), which are described below. Therefore, reducing bacterial pathogens loads through simple protection measures is relatively easy and should be a major target of the planners of water and sanitation programmes.

Table 1.1 Illnesses acquired by ingestion of faecally contaminated water.

Pathogen	Source	Disease
VIRUSES		
Hepatitis A virus	Human faeces	Infectious hepatitis
Hepatitis E virus	Human faeces	Infectious hepatitis
Astrovirus, Calcivirus, Rotaviruses, Norwalk-type viruses	Human faeces	Diarrhoeal diseases
Coxsackieviruses and Echoviruses	Human faeces	varied symptoms and diseases
BACTERIA		
Campylobacter jejuni	Human and animal faeces	Diarrhoeal diseases
Enterohaemorrhagic *E. coli* O157:H7	Human and animal faeces	Hemorrhagic colitis
Enteroinvasive *E. coli*	Human faeces	Diarrhoeal diseases
Enteropathogenic *E. coli*	Human faeces	Diarrhoeal diseases
Enterotoxigenic *E. coli*	Human faeces	Diarrhoeal diseases
Salmonella typhi	Human faeces and urine	Typhoid fever
Shigellae spp.	Human faeces	Dysentery
Vibrio cholerae O1	Human faeces	Cholera
PROTOZOAN PARASITES		
Cryptosporidium spp.	Human and animal faeces	Diarrhoea
Giardia lamblia	Human and animal faeces	Diarrhoea

Viruses are much smaller organisms and cause a range of diarrhoeal diseases. Some viral diseases, for instance polio, are most effectively controlled through vaccination rather than control of water quality control. Epidemics of viral diseases have been recorded, for instance Hepatitis E Virus which is of particular concern for pregnant women (Grabow, 1997), although in general viral infections tend to lead to milder and self-limiting diseases. In general viral infections tend to lead to milder and self-limiting infections. Viruses are commonly transmitted through poor hygiene, drinking water is not the principal route. The severity of some viral infections also depends on when first exposure to the pathogen occurs. For instance, when first exposure to Hepatitis A Virus occurs in childhood, the symptoms are often relatively mild and a degree of lifelong immunity is acquired. When first exposure is in adulthood, the effects tend to be more severe. For other viruses such as rotavirus, this protection does not occur. The dose of viruses required to initiate symptoms tends to be very low and viruses are often less likely to be attenuated. Therefore, reducing the risks from viruses in drinking water is difficult without disinfection of the water supply. Protection measures may greatly reduce the numbers of pathogens in the water and therefore reduce the risks of infection, but controlling sources of viruses (for instance on-site sanitation) alone is unlikely to reduce the risk to an acceptable level.

Protozoa are relatively large organisms and include *Cryptosporidium* and *Giardia*. Protozoan pathogens cause diarrhoea, although in most cases this is relatively mild and self-limiting. In most developing countries, exposure to protozoan pathogens occurs through direct contact with animals, poor hygiene and contaminated food. Drinking water is unlikely to be the major route of transmission. Although infectious doses of many protozoa are very low, attenuation is often easy given the large size of the organisms. Therefore, control of protozoan pathogens in groundwater is relatively easy because of the size of the cysts and should be an easily achievable target, even though the actual health risk is relatively limited.

1.4.2 Chemical contaminants

The chemical contaminants of principal importance that are derived from on-site sanitation are nitrate and chloride. Each person excretes in the region of 4kg of nitrogen per year and under aerobic conditions it can be expected that a significant percentage of organic nitrogen will be oxidised to form nitrate.

Chloride is also abundant in human wastes (the ratio of chloride to nitrogen in human faeces is approximately 1:2). This ratio is useful in identifying the presence of sewage in groundwater (see case studies in Part 2). Each person on averages loses approximately 4g of chloride per day through urine (90–95%), faeces (4–8%) and sweat (2%). However, the chloride content can be very variable and depends on its concentration in drinking water. Nitrate is a health concern and WHO have set a Guideline Value of 50 mg/l as the safe level (11.3 mg/1 as N). Chloride is of less concern for health, but affects the acceptability of the water and thus may result in use of alternative, possibly more microbiologically contaminated, water sources. In both cases, environmental protection concerns also need to be addressed, as remediation of contamination is difficult. Nitrate and chloride are generally stable, especially in aerobic environments and therefore contamination is likely to build-up and persist in the longer term. Remediation of the aquifer or treatment of the water supply are expensive and difficult to achieve.

Nitrate is a natural and inevitable part of the normal mixed diet with vegetables being the principle dietary source when levels in drinking water are less than 10 mg/l. However, in situations where the concentration in drinking water is high, the total dietary intake may be substantially increased. Consequently, adults who drink water containing greater than 50 mg/l nitrate may consume as much as 50-90 mg/day from this source alone. For bottle-fed infants with milk formulas made up from such sources, this would represent an average daily intake of 8.3-8.5 mg nitrate per kg of body weight per day.

The principle health concerns associated with the ingestion of nitrate are:

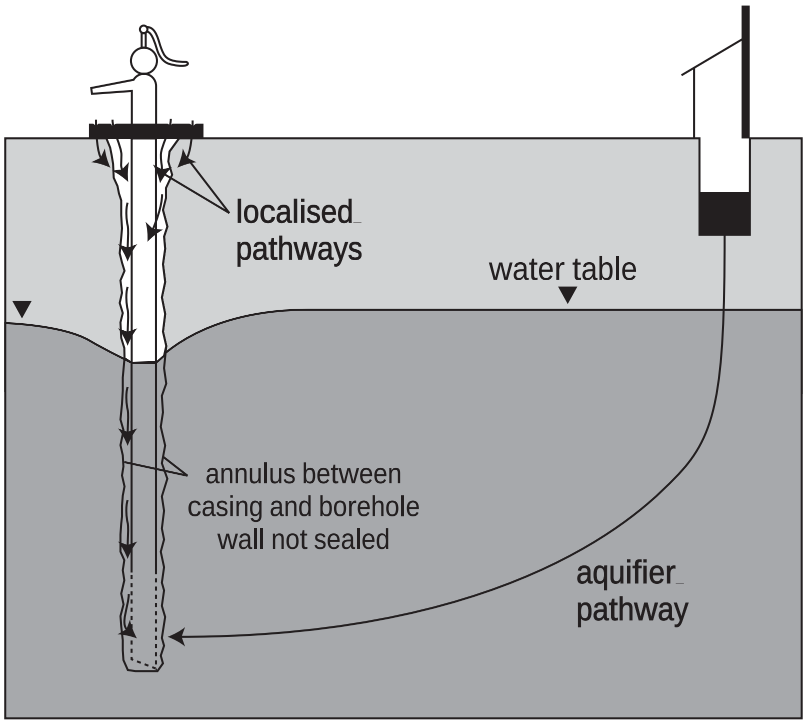
- Methaemoglobinaemia (or blue-baby syndrome)
- possible link with stomach cancer

Nitrate is in itself relatively non-toxic, however upon ingestion it is partially converted by bacteria in the mouth to nitrite. The formation of nitrite is especially important as it reacts with haemoglobin, the oxygen carrying constituent of red blood cells, to produce methaemoglobin which cannot transport oxygen.

In anoxic groundwaters nitrogen may be present as ammonium. Man's estimated average intake of ammonium through food and drinking water is in the range of 18 mg per day and exposure from environmental sources is thought to be insignificant and of no immediate consequence to health. In addition to this, there is no evidence to suggest that ammonium is carcinogenic. Although ammonium is not toxic in its own right, the by-products and consequences of its reactions with other compounds and their potential health effects should be considered. The principle by-products of concern are nitrite and N-Chloramines. The recommended WHO guideline value for ammonium is 0.2 mg/l.

1.5 RISK ASSESSMENT: CONTAMINANT PATHWAYS

This review is concerned with the fate and transport of contaminants associated with on-site sanitation and the potential risk posed to groundwater supplies. Contamination of groundwater supplies by micro-organisms derived from on-site sanitation systems can occur by two types of pathway (Figure 1.4).



(i) Aquifer pathway, where pathogens migrate through the subsoil from the base of the latrine to the water table and from there to the intake of the well or screen

(ii) Localised pathway that develops because of the poor design and/or construction of the water supply. This pathway provides a rapid bypass mechanism to the intake of the water supply limiting the residence time of the microbes in the subsurface.

The latter pathway is a common route for well contamination by micro-organisms although contamination can be relatively easily prevented by ensuring correct design of water supply and careful construction. Assessing the risk posed to groundwater by contamination via the aquifer pathway is more difficult and is the subject of this review.

2 Principles of Groundwater Movement and Aquifer Vulnerability

2.1 VARIABILITY IN THE AQUIFER TYPE AND VULNERABILITY TO POLLUTION

It has long been recognised that the subsurface can be very effective at purifying water. As water moves through the ground, natural processes reduce (or attenuate) the concentration of many contaminants including harmful micro-organisms.

The degree to which attenuation of contaminants, associated with on-site sanitation, occurs depends on the nature of the subsurface. It is important to understand how hydrogeological factors control contaminant movement and behaviour as this can lead to the classification of environments which can be used to help assess the risk posed to groundwater. The principal hydrogeological environments and their characteristics are summarised in Table 2.1.

Aquifer pollution vulnerability is a helpful concept widely used to indicate the extent to which groundwater at the water table can be adversely affected by an imposed contaminant load. This is a function of the intrinsic characteristics of the unsaturated zone. Some hydrogeological environments are inherently more vulnerable than others (Table 2.2). Areas of the same aquifer system may have different relative vulnerability due to spatial variations in the thickness of the unsaturated zone. Aquifer vulnerability can be subdivided into four broad classes:

- Extreme: vulnerable to most water pollutants with relatively rapid impact in many pollution scenarios;
- High: vulnerable to many pollutants except those highly absorbed and/or readily transformed;
- Low: only vulnerable to most persistent pollutants in the very long term; and
- Negligible: confining beds present with no significant groundwater flow

The vulnerability of the principal hydrogeological environments to pollution is generalised in Table 2.2.

2.2 PATHOGEN SURVIVAL IN THE UNSATURATED ZONE

The soil is the main zone in which surface pollutants are attenuated. However, pit latrines place the pollutant below this zone and so the unsaturated zone represents the first line of natural defence. It is essential that the unsaturated zone be fully considered in the evaluation of risks to groundwater supplies. Should it be ignored, evaluations will be excessively conservative. However, the role of the unsaturated zone can be complex and its ability to attenuate contaminants difficult to predict.

Maximisation of effluent residence times in the unsaturated zone is the key factor affecting the removal and elimination of bacteria and viruses (Lewis et al 1982). Useful reviews of factors affecting pathogen movement in the subsurface have been made by Romero (1970), Lewis et al (1982), Pekdeger et al (1985) and West et al (1998).

Table 2.1 Characteristics of principal urban hydrogeological environments.

Hydrogeological environment	Lithology	Description/genesis	Extent/dimension
Major alluvial and coastal plain sediments	Gravels, sands, silts, and clays	Unconsolidated detritus deposited by major rivers, deltas and shallow seas; primary porosity and permeability usually high	Usually both areally extensive and of significant thickness
Intermontane colluvial and volcanic systems	Pebbles, gravels, sands and clays sometimes interbedded with lavas and pyroclastics	Formed by rapid in-filling of faulted troughs and basins in mountain regions; deposits are unconsolidated, primary porosity and/or permeability of colluvium, modern basaltic/andesitic lavas and andesitic/rhyolitic pyroclasts usually high, but older volcanics are	Much less extensive than alluvial and coastal plain sediments but can be very thick
Minor sediments associated with rivers	Gravels, sands, silts and clays	Unconsolidated detritus deposited by rivers; primary porosity and permeability usually high	Not areally extensive and of limited thickness
Recent coastal calcareous formations	Limestones and calcareous sands	Usually composed of coral limestones and fringing skeletal detritus often only loosely cemented; porosity and permeability can be exceptionally high	Limited area, often forming strip-like aquifers that fringe coastline or form small oceanic islands
Loessic plateau deposits	Silts, fine sands, and sandy clays	Usually well-sorted windblown deposits of silt and fine sand, with some sandy clay deposits of secondary fluvial origin; low permeability generally makes sub-surface more suitable as receptor than aquifer	Very extensive although deposits may form isolated systems cut by deep gullies
Consolidated sedimentary aquifers	Sandstones	Marine or continental deposits compacted to form consolidated rocks; degree of consolidation generally increases with depth/age of deposition and increasing compaction reduces primary porosity and permeability; secondary porosity introduced by fractures of tectonic origin can form a very significant component	Difficult to generalise but can form extensive aquifers and be of substantial thickness
Weathered basement	Crystalline rocks	Weathering of older igneous or metamorphic rocks usually produces a deeply weathered mantle of moderate porosity and generally low permeability, underlain by fresher rock which may be fractured; the combination results in a low potential, but important aquifer system	Very extensive, but aquifers are normally restricted to the upper 20 m

Table 2.2 Principal hydrogeological environments and their associated pollution vulnerability

Hydrogeological environment		Natural travel time to saturated zone	Attenuation potential	Pollution vulnerability
Major alluvial and coastal plain sediments	shallow layers:	weeks-months	low-high	high
	deep layers:	Years-decades	high	low
Intermontane colluvial and volcanic systems	shallow layers:	months-years	low-high	low-high
	deep layers:	Years-decades	low-high	low-high
Minor sediments associated with rivers		days-weeks	low-high	extreme
Recent coastal calcareous formations		days-weeks	low	extreme
Loessic plateau deposits	shallow layers:	weeks-months	low-high	high
	deep layers:	Years-decades	high	low
Consolidated sedimentary aquifers	sandstones:	months-years	low-high	low-high
	karstic limestones:	days-weeks	low	extreme
Weathered basement	thick weathered layer (>20 m):	weeks-months	high	low
	thin weathered layer (< 20 m):	days-weeks	low-high	high

2.2.1 Flow in the unsaturated zone

The unsaturated zone is a complex arrangement of solid particles and pore spaces filled with air and water. The proportions of water and air change with time as infiltrating water moves from the ground surface to the water table. Water is held in the pores due to molecular forces of cohesion and adhesion, this pore water is thus under tension. The smaller the pores the greater the water tension. Therefore when drainage occurs, it is the larger pores that drain first as these do not have sufficient tension to retain the water. The rate of decrease of moisture content with increasing tension is a function of the pore size distributions and is illustrated in Figure 2.1. Coarser sediments with large pores (e.g. sands, sandstones) drain rapidly at low tensions whilst finer-grained sediments, clays and silts, drain relatively few pores as their water is strongly retained in the smaller pores.

The hydraulic conductivity therefore decreases with reduced moisture content and this decrease can be very considerable (Figure 2.2). For fine-grained sediments, the hydraulic conductivity is generally low and is less affected by changes in moisture content however, for coarse sediments and especially where fissures are present, the hydraulic conductivity can increase several orders of magnitude as the moisture content of the unsaturated zone approaches saturation.

While natural flow rates in the unsaturated zone of almost all formations do not generally exceed 0.2 m/d in the short term, and less when averaged over longer periods, water flow and pollutant penetration rates in fractured formations may be more than an order-of-magnitude higher, given high rates of fluid loading (for example from septic tanks). Thus rock type, and especially the grade of consolidation and whether there are fractures, will be key factors in the assessment of aquifer pollution vulnerability, especially in relation to microbial pathogens.

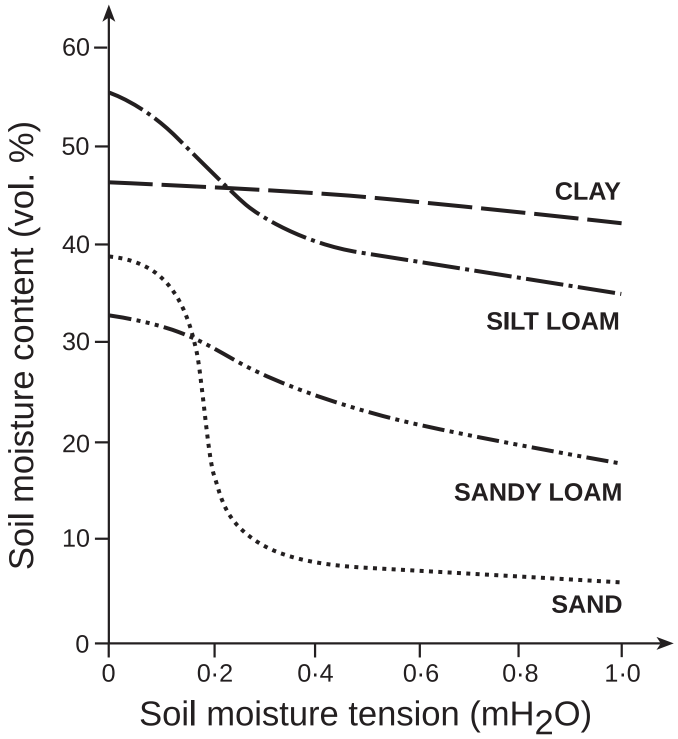


Figure 2.1 Soil moisture retention curves for four different soil materials (Bouma et al, 1972)

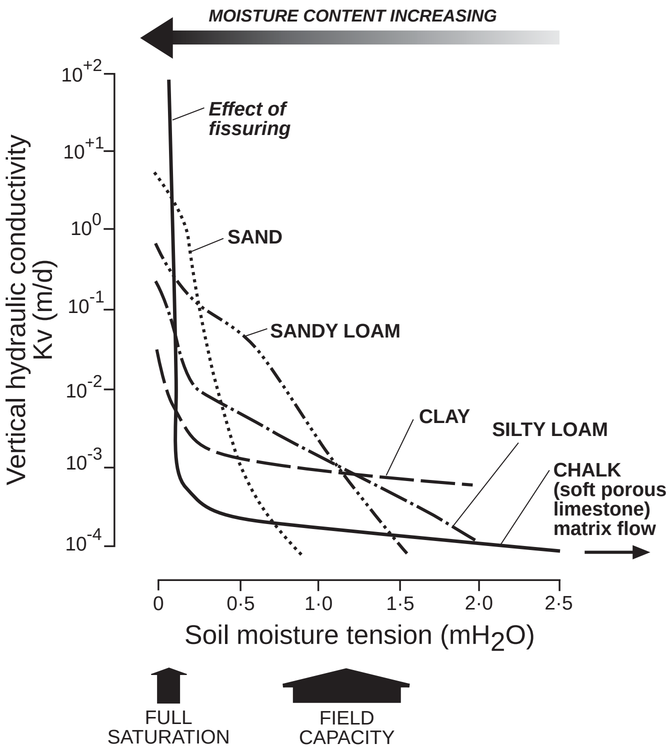


Figure 2.2 In-situ unsaturated hydraulic conductivity as a function of moisture potential (after Lewis et al, 1982)

2.2.2 Filtration

Filtration is very effective in the clogged zone that develops around pit latrines. Several processes contribute to pore clogging and include:

(i) blockage of pore by solids filtered directly from the effluent

(ii) accumulation of biomass from the growth of micro-organisms

(iii) production of slimes by some bacteria

Ziebell et al (1975) found that the bacterial populations were very high in the clogged zone but reduced rapidly with distance; bacteria populations at 30 cm distance from the clogged zone were similar to those in a ‘control’ soil sample. Caldwell and Parr (1937) observed that faecal coliforms were detected some 10 m downstream of a newly constructed latrine that penetrated the water table. However, after three months the plume dimensions were reduced and the formation of a clogged zone around the latrine was postulated as the cause. Poynter and Slade (1977) suggested that the removal of bacteria and viruses is essentially a biological process.

Filtration in the saturated zone, away from clogged zones, is unlikely to be an important mechanism except in the fine-grained sediments. Filtering effects are only significant when the average size of the particle or microbe exceeds approximately 5% of the average pore size (Harvey and Garabedian, 1991). Bacteria and protozoa are most likely to be affected by filtering, but viruses are typically much smaller (Figure 2.3) and so are less likely to be removed by filtering.

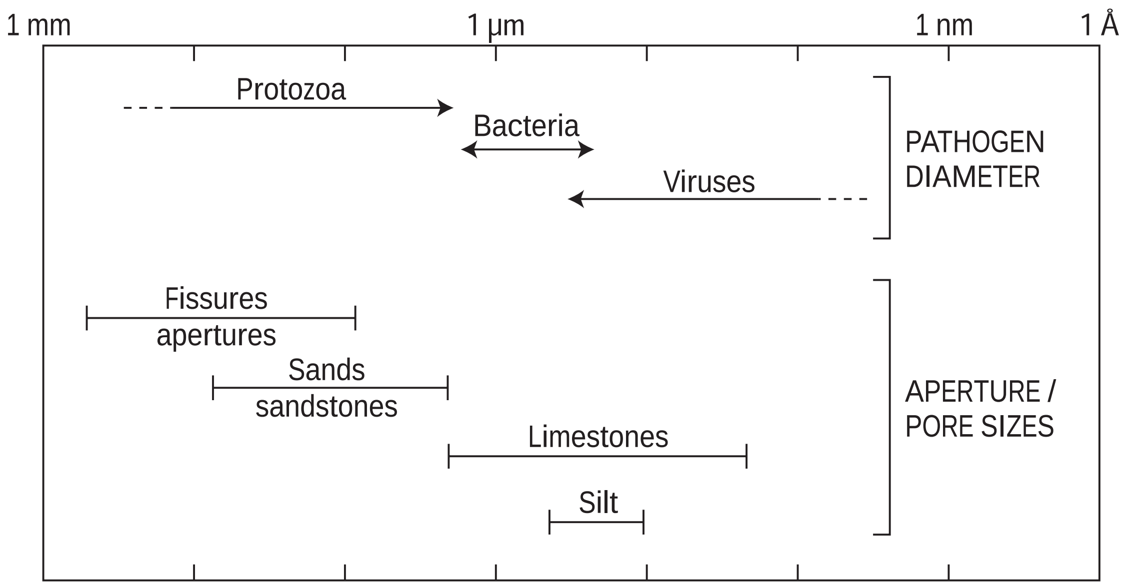


Figure 2.3 Pathogen diameter compared with aquifer matrix apertures.

2.2.3 Adsorption

Bacteria and viruses can be adsorbed in the subsurface. Complex physiochemical interactions occur between the microbe, the surrounding water and rock particles. Three types of adsorption mechanism exist: physical, chemical and ion exchange (Yavuz Corapcioglu and Haridas, 1984). Viruses are most likely to be affected by adsorption because of their size (West et al, 1998). Soils/rocks which contain clays are more likely to have a higher sorption capacity due to the shape of the clay platelets and their large surface area (Savage and Fletcher, 1985). Under most natural pH conditions, microbes suspended in water have a net negative charge as do most mineral surfaces in the subsurface. Therefore there is a tendency for the microbes to be repelled by the solid particles and remain mobile (West et al, 1998). However, under certain groundwater pH conditions or mineral characteristics this may change and adsorption may occur. Reversible adsorption of bacteria and viruses in a sandy soil under highly acidic groundwater conditions was demonstrated (Dowd and Pillai, 1997). Similar effects were described by Goldschmid et al (1973).

Adsorption is a reversible process and changes in groundwater chemistry (especially pH) can act to release microbes. Heavy rainfall and the infiltration of water of different chemical composition (lower ionic strength or pH) can flush out microbes previously sorbed (Sobsey et al, 1980; McCaulou et al; 1994 and Fourie and van Ryneveld, 1995).

During the process of migration through the unsaturated zone, viral retardation may occur. For clay-rich soils Sobsey et al (1980) showed 99.995% of viable virons were removed, however removal rates in sandy and organic soils were less (but still above 95%). Elution of viruses from sandy and organic soils was considerable but less so for clay-rich soils, although up to 20% elution could be seen. There is evidence that rainfall effects are effective in eluting viruses (Bales et al, 1993).

The factors influencing the movement of bacteria and viruses through soils is summarised in Table 2.3. This is also relevant to the unsaturated zone, given the soil zone is bypassed by pit latrines.

Table 2.3 Factors influencing movement of bacteria and viruses through soil (after Gerba et al 1975).

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

2.2.4 Factors affecting pathogen survival in the unsaturated zone

Micro-organisms, like all life forms, have a limited life span. Die-off rates vary enormously from a few hours to up to several months depending on the type of organism and environmental conditions. Useful reviews of pathogen survival are given by Lewis et al (1982), Bitton et al (1983), Feachem et al (1983), Yates et al (1985) and West et al (1998) and are summarised below and in section 2.3.2 in relation to the saturated zone.

BACTERIA

The dominant controls on bacteria survival rates appear to be moisture and temperature.

Kligler (1921) investigated the survival of salmonella typhi and shigella dysenteriae in different soil types at room temperature. Some bacteria survived for 70 days in moist soils although 90% died within 30 days. In dry soils, no bacteria survived longer than 20 days and with acid soils, irrespective of moisture content, this time was reduced to 10 days.

Mirzoev (1968) showed that bacterial die-off was slowed down or suspended at low temperatures. At temperatures as low as –45°C, shigella dysenteriae were detected 135 days after they had been added to the soil.

The survival of faecal streptococcus was investigated for five Oregon soils (Kibbey, et al, 1978). They found the die-off rates varied between the different soils but were generally longest in soils maintained under cool, moist conditions and this was attributed to the lack of antagonistic activity by soil micro-flora (Table 2.4). A review of the literature (Feachem, et al, 1983) confirmed that survival times for various micro-organisms varied widely. In general it appeared that thermotolerant coliforms only survived for 10 weeks, with a 90% reduction taking place within 2-3 weeks. However, under cool moist conditions thermotolerant faecal coliforms can survive many months. Conversely under hot and arid conditions complete elimination of the faecal indicator bacteria occurs within two weeks.

Table 2.4 Average 95% population reduction times (T_{95}) for faecal streptococcus in five Oregon soils (Kibbey et al 1978)

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

VIRUSES

As with bacteria, survival rates for viruses vary widely but the major controlling factors appear to be moisture and temperature (Lefler and Knott, 1974; Yeagar and O'Brien, 1979). Keswick and Gerba (1980) considered that virus survival is expected to increase with depth of penetration of the subsurface.

2.3 PATHOGEN SURVIVAL IN THE SATURATED ZONE

2.3.1 Groundwater flow in the saturated zone

In the saturated zone, water movement is greater because of the higher hydraulic conductivities (compared to unsaturated hydraulic conductivities, as discussed above). Groundwater movement is often, but not always, lateral and the flow rate is a function of the saturated hydraulic conductivity, the hydraulic gradient and the aquifer porosity.

In most hydrogeological environments, the hydraulic gradient is low, less than 0.01 and frequently less than 0.001, thus flow velocities although more than in the unsaturated zone are often relatively low (less than 2 m/d). Important exceptions do occur, for example coarse sand and gravel aquifers and especially fractured consolidated aquifers can have groundwater velocities in excess of 10 m/d and even 100 m/d. By increasing the separation between a water supply and an on-site sanitation unit, the travel time (and the opportunity for attenuation of contaminants) can be increased. This has been the basis for recommending 'safe' separations between water supplies and latrines (Subrahmanyan and Bhaskaran, 1950), ignoring the important role of the unsaturated zone.

Even in finer-grained and lower permeability aquifers, lateral separations are not always a reliable method of protection because very few aquifers are uniform and relatively thin but permeable layers may occur which provide a rapid pathway for lateral groundwater movement. The presence of such layers is difficult to predict.

In most hydrogeological environments some layering or stratification is present and this normally results in significant differences between the horizontal and vertical hydraulic conductivity (ratio of K_V:K_H varies from 1 to 0.01 or less). This in turn can lead to long travel times for groundwater to flow downwards from the surface of the water table to depth. For example, in the porous sedimentary aquifer beneath Santa Cruz (Bolivia) it has been estimated that modern fronts of water migrate at the rate of 2-5 m per year (Morris et al, 1994), whilst in finer-grained sediments beneath Hat Yai (Thailand) rates of 1-2 m/yr are more typical (Barker and Lawrence, 1994). Thus the vertical separation between the water table and the screen of a borehole can provide an important time delay allowing significant reduction in the risk of less persistent contaminants, including many micro-organisms, arriving at the screen. The formula below for predicting travel time from the water table to the borehole screen was developed based on a well and its image (minimum value for vertical flow) and was used to produce the travel time graphs included in the ARGOSS manual (ARGOSS 2001).

$$t = \frac{4\pi n}{Q}\left(\sqrt{\frac{K_h}{K_v}}\right)^3 \int_0^d \left[\frac{1}{(d-z)^2} + \frac{1}{(d-z)^2}\right]^{-1} dz = \pi(2\pi - 16/3)\frac{n}{Q}\left(d\sqrt{\frac{K_h}{K_v}}\right)^3 \approx 2.984\frac{n}{Q}\left(d\sqrt{\frac{K_h}{K_v}}\right)^3$$

where t is travel time from water table to well intake;
n is kinematic porosity;
Kh/Kv is ratio of horizontal to vertical hydraulic conductivity;
d is depth of well intake;
z is depth below water table; and
Q is pumping rate.

An important process for reducing contaminant concentrations in the saturated zone is dispersion (the spreading of the contaminant plume) and dilution. These processes are normally only significant where initial contaminant concentrations are low and/or permitted concentrations in drinking water relatively high.

2.3.2 Factors affecting pathogen survival in the saturated zone

Information on bacterial survival in groundwater is limited although it is generally accepted as being longer than in surface water due to the absence of sunlight. Kudryavtseva (1972) reported that coliforms introduced into saturated fine-grained alluvial sand survived for up to 3.5 months whilst a pathogenic serotype of E-coli, similarly inoculated into the groundwater survived for 3 months.

McFeters et al (1974) measured the survival of various faecal indicator bacteria and enteric pathogens in well water using membrane chambers. T_{50} values (time required for a 50% reduction) of the various cultures are given in Table 2.5.

Survival rates for various micro-organisms reported in the literature are presented in Table 2.6. Sobsey et al (1980) suggest that T_{99} for poliovirus was 42 days and for reovirus was 35 days in non-sterile conditions.

Table 2.5 Half time (time for 50% reduction) of various bacterial cultures in well water, at 9-12^oC (McFeters et al 1974)

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

Table 2.6 Example of half-lives of pathogenic microbes derived from experimentation. (Environmental conditions not specified) [from West et al 1998]

Microbe	Decay constant (hr^{-1})	Half-life (hr)	Reference
Viruses			
Poliovirus (in groundwater)	0.0013	533.2	Bitton et al 1983
	0.0088	78.8	Keswick et al 1982
Viruses (in well water)	0.0004 - 0.0037	1732.9 - 187.34	Bitton et al 1983
Enteroviruses	0.004	173.3	Gerba 1985
PSD-2 and MS-2 (*E. coli*) bacteriophage in groundwater	0.0033	21.0	Dowd & Pillai 1997
Bacteria			
Salmonella spp.	0.0078	88.9	Dowd & Pillai 1997
(in groundwater)	0.055	12.6	Gerba 1985
Klebsiella spp. (in groundwater)	0.0013	533.2	Dowd & Pillai 1997
Escherichia coli	0.038	18.2	Gerba 1985
	0.013	53.3	Keswick et al 1982
Streptococcus spp.	0.015	46.2	Gerba 1985
	0.0096	72.2	Keswick et al 1982
Shigella spp.	0.028	24.8	Gerba 1985
Faecal coliforms	0.064	10.8	Gerba 1985

In on-site sanitation systems, nitrate is leached as a consequence of the microbial degradation of organic nitrogen to ammonium ions which, are in turn, biologically oxidised to nitrite and nitrate as follows:

$$NH_4^+ + 2OH^- + 30_2 \Rightarrow 2NO_2 + 4H_2O$$

$$2NO_2^- + O_2 \Rightarrow 2NO_3^-$$

These reactions occur under aerobic conditions. Where the unsaturated zone is thin or soils are fine-grained and waterlogged, oxidation of the organic nitrogen may be incomplete and nitrogen may, instead, be 'lost' to the atmosphere by volatilisation of ammonium or by the biochemical reduction of nitrite to nitrogen gas.

Nitrogen compounds in excreta do not represent as immediate a hazard to groundwater but can cause more widespread and persistent problems. It is possible to make a semi-quantitative estimate of the persistent and mobile contaminants like nitrate and chloride in groundwater recharge. The estimate is based on the following equation (Foster and Hirata 1988).

$$C = \frac{1000aAF}{0.365A.U. + 10I}$$

where C is the concentration (mg/l) of the contaminant in recharge
a is unit weight of nitrogen or chloride in excreta (4 and 2 kg/cap/a)
A is population density
F is proportion of excreted nitrogen leached to groundwater (0-1.0)
U is non-consumptive portion of total water use (l/d/cap)
I is natural rate of infiltration (mm/a)

Greatest uncertainty surrounds the proportion of the nitrogen load that will be oxidised and leached in groundwater recharge. A range of 20-60% has been reported in the literature (Walker et al 1973, Kimmel 1984, Thomson and Foster 1986) although in the karst limestone aquifer beneath Merida the fraction of nitrogen leached to groundwater approach 100% (Morris et al 1994). Nitrate concentrations beneath unsewered urban areas can be especially high because of the greater population density and hence nitrogen load, concentrations in excess of 20 mgN/l are reported to be widespread in Santa Cruz (BGS and SAGUAPAC 1994), Lucknow (Sahgal et al 1989), Merida (Morris et al 1994) and Jaffna (Gunasekaram 1983) whilst concentrations in excess of 50 mgN/l were observed in both Jaffna and Lucknow.

It is evident from the above equation that troublesome nitrate concentrations are often likely to develop, except where water use is very high and/or population density is very low. This equation was used in the ARGOSS manual (ARGOSS 2001) to estimate approximately the likely nitrate concentration in groundwater. Especially high concentrations are likely to occur in those arid regions with low per capita water usage.

However, in anaerobic groundwater systems (typical of areas underlain by fine-grained sediments with shallow depths to water table) nitrate concentrations are often less. This is, as mentioned above, partly because there is less available oxygen to oxidise the organic nitrogen leached from the pit latrine and partly because any nitrate that is leached is often reduced to nitrogen gas by denitrifying bacteria present in the aquifer.

When assessing the potential risk of widespread contamination of groundwater by nitrate or chloride from on-site sanitation, the other possible sources (e.g. agriculture) should also be considered. Whilst quantifying the relative contribution from each source is likely to prove difficult, where potentially high nitrogen loadings are indicated, it would probably be worthwhile monitoring for nitrate in groundwater. Both nitrate and chloride may show significant seasonal fluctuations in shallow groundwater (exemplified in the Uganda case study, Part 2), although levels are expected to be more stable in deeper groundwater. Therefore, when assessing the risk of widespread nitrate or chloride contamination, it is important to recognise the possibility of seasonal peaks. Where such information is not available, it may be necessary to set-up a monitoring network (as was undertaken in Uganda, see Part 2). In general the likely level of nitrate contamination of groundwater will depend on:

- the quantity of recharge, which controls the degree of dilution (the higher the rainfall, the lower the nitrate content for a given population density)
- population density, which relates to the contaminant load (the higher the population density the higher the nitrate content)
- type of on-site sanitation system, which determines the proportion of nitrogen leaching
- other sources of nitrate in the environment, for example large concentrations of livestock animals may contribute a significant nitrate load
- the nature of the sub-surface and the hydrogeological environment including the potential for denitrification.

High groundwater nitrate concentrations in rural areas have also been attributed to leaching from on-site sanitation systems. For example, Hutton et al (1976) and Lewis et al (1980) suggested that pit latrines were responsible for the high nitrate concentrations observed in shallow village wells in eastern Botswana.

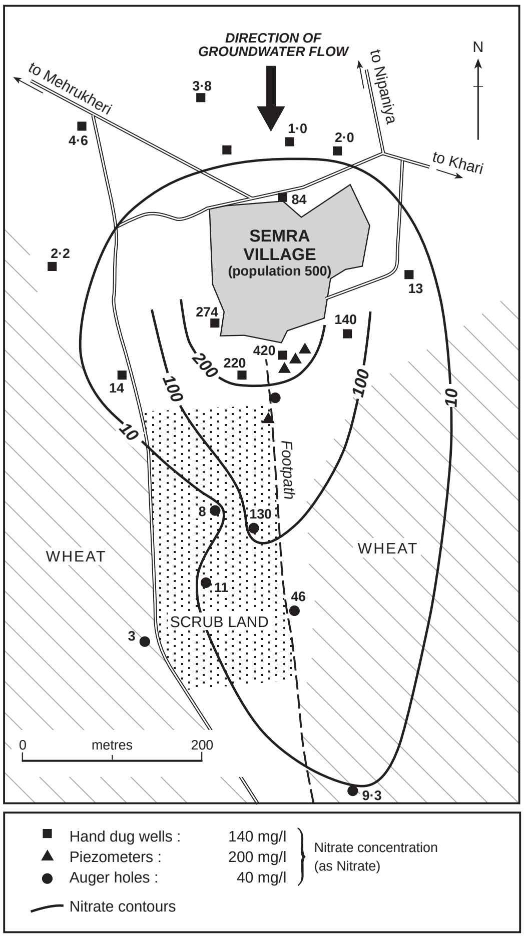


Figure 2.4 Groundwater nitrate pollution plume in an Indian village (Cook and Das, 1980).

A case study of groundwater pollution in central India showed a plume of high nitrate groundwater emanating from the village (Figure 2.4) (Cook and Das, 1980). Nitrate concentrations up to 100 mgN/l were observed in groundwater in some village wells and was attributed to on-site sanitation and leaching of animal wastes within the village.

3 Field investigations of pathogen migration

In the preceding sections water movement in the unsaturated and saturated zones and the factors controlling pathogen fate and transport were reviewed. However a number of field investigations have been undertaken dating back to the 1920s which provided empirical evidence for the spread of microbiological contaminants from pit latrines. It was this empirical evidence that was used as the basis for ‘set back’ distances or separations between pit latrines and drinking water supplies (e.g. Subrahmanyan and Bhaskaran, 1950). The summary below is drawn largely from the review by Lewis et al (1982).

3.1 BACTERIA IN THE UNSATURATED ZONE

3.1.1 Conditions favouring attenuation of bacteria

Kligler (1921) was one of the earliest researchers to investigate the relationship between pit latrines and the spread of waterborne infectious diseases. Field and laboratory studies were conducted to determine the viability of pathogenic bacteria, and their extent of penetration in the soil underneath a pit latrine. The field studies were conducted in a variety of soils with pit latrines that had been in use for 1-3 years. He concluded that pit latrines and septic tanks, if properly constructed, are unlikely to cause the spread of bacterial intestinal infections. The pathogenic organisms were found to die off rapidly in faeces, and bacteria were only transported 0.9-10.5 m in the soil type studies. The pit latrine appeared to involve minimal pollution risk in sandy or clay type soils provided the groundwater level did not rise higher than 3-4 m below ground level, i.e. 1.5-2.5 m beneath the base of the pit latrine.

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

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

The conclusion that can be drawn from these early studies is that at least 2 m of sandy soil is required beneath a pit latrine to prevent pollution of any underlying groundwater (see Table 3.1). In recent years, because of fears of pollution, the fate of septic tank effluent has been the subject of increasing attention, especially in North America. Almost 20 million housing units, representing 29% of the United States’ population, dispose of domestic waste through individual on-site sanitation systems, discharging approximately 3 x 10^{12} m^3/yr of water to the soil. A survey by officials, consultants, water well drillers and other water resource officials in 35 states, disclosed that septic tanks and cesspools rank highest in the total volume of wastewater discharged directly to the ground, and were the most frequently reported source of groundwater contamination (Miller and Scalf, 1974).

Table 3.1 Penetration and removal of bacteria in the unsaturated zone – On-site excreta disposal (from Lewis et al 1982)

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

[1]Field data
[2]Laboratory column experiments
[3]Mound disposal system
[4]Lysimeter studies

3.1.2 Conditions less favourable for attenuation of bacteria

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

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

3.1.2.1 THIN SOIL OVER FRACTURED BEDROCK

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

3.1.2.2 HIGH OR SEASONALLY HIGH GROUNDWATER CONDITIONS

Viraraghavan (1978) conducted a study in Ontario, Canada, to monitor the horizontal movement of faecal bacteria from the end of a septic tank disposal field located in a sandy clay soil. The groundwater level at the site fluctuated between near ground level during the spring snow-melt to a depth of 2.5-3.0 m during late summer. At the time of the investigation the water table was only 0.15 m below the tile drain (0.6 m below ground level). The septic tank effluent contained 1.6×10^5 faecal coliforms/100 ml. The depth of the unsaturated zone available for purification was limited so relatively high levels of organisms (100/100 ml) were found in the groundwater even in the most distant observation well (15.25 m). Another study (Brandes, 1974) reported a reduction in total coliform bacteria from 8×10^6/100 ml in the septic tank effluent to 4×10^3/100 ml in the groundwater 7.5 m from the tile drain. It was thought that the fill material (stones and boulders) used for the absorption field allowed the effluent to penetrate to the water-table which fluctuated between 0.5 and 2.1 m.

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

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

(b) the initial period (first l00 days) was critical until the bacteria film built up on the sand surfaces sufficiently to provide the retentive power necessary;

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

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

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

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

3.2 BACTERIA IN THE SATURATED ZONE

3.2.1 Movement in porous-unconsolidated sediments

Caldwell conducted a series of detailed studies measuring pollution travel from a variety of pit latrines. The first study (Caldwell and Parr, 1937) measured pollution travel from a 5.1 m pit latrine in a shallow (3.6 m) perched water table located in a coarse sandy stratum. Faecal material from a family of six was added daily to the latrine. The natural groundwater velocity was found to be in the range 0.6-2.5 m/d. Initially faecal organisms travelled some 4.5 m in 3 days. The chemical contaminants reached 10.6 m after 9 days. Later the chemical plume was traced to 26 m but could not be detected at 31 m. After 2 months, faecal coliforms were present in 90% of the samples at 4.5 m, and were occasionally detected at 10.6 m. A conclusion of this study was that the clogging process was an important defence mechanism limiting the extent of bacterial penetration. After the onset of clogging in the latrine (3 months), the migration of organisms was inhibited and within 7 months bacterial pollution was limited practically to the latrine. However, chemical contamination of the groundwater still occurred, although this was somewhat diminished.

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

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

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

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

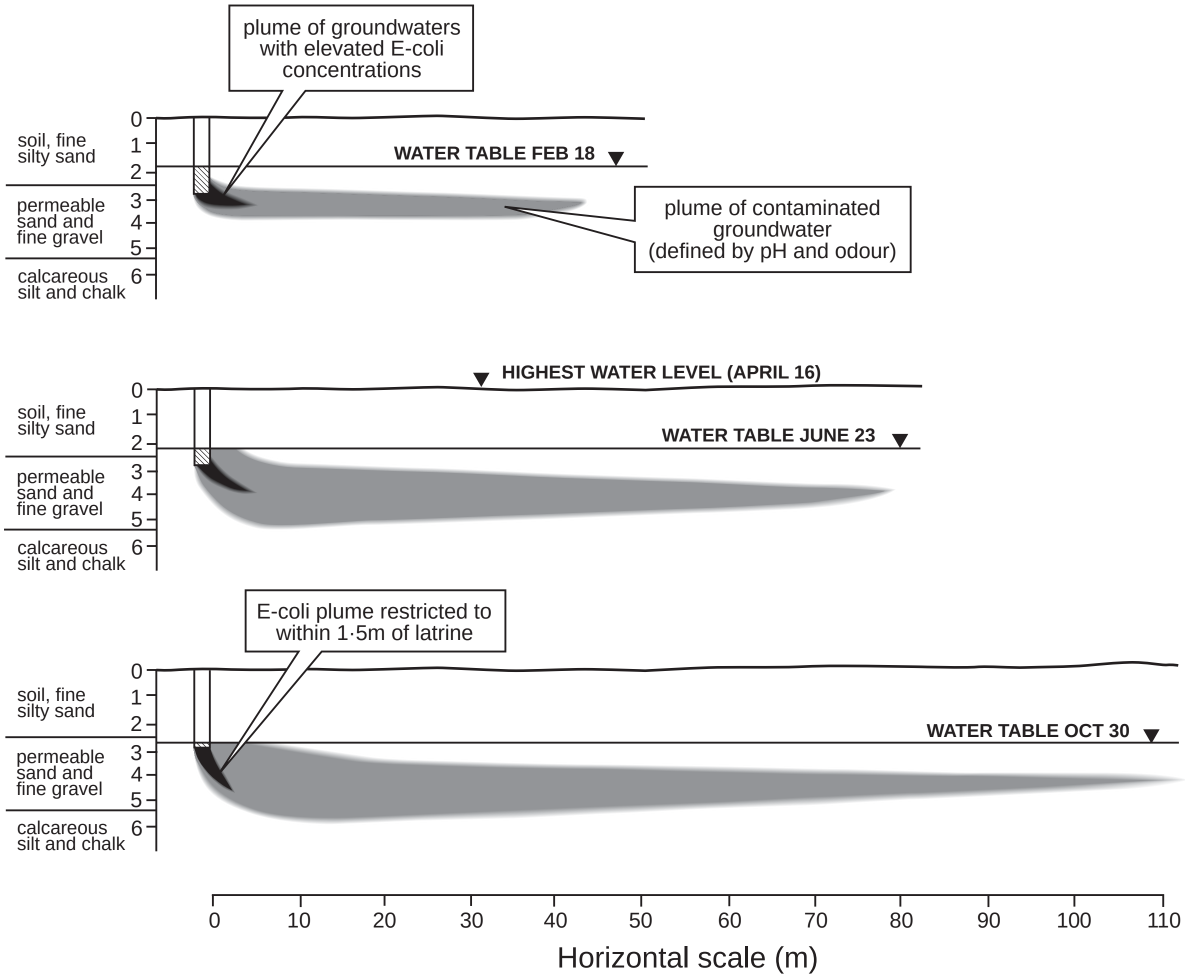


Figure 3.1 Pollution plume migration from a pit latrine penetrating the water table (Caldwell, 1938a).

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

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

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

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

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

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

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

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

More recent studies have shown that lateral movement led to reductions of thermotolerant coliform densities within 5m from both pit latrines and refuse pits and suggested that a 10 day travel time was adequate for control of microbiological quality (Chidavaenzi et al, 2000).

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

The results of a number of studies investigating faecal bacteria transport in the saturated zone are summarized in Table 3.2. This shows that the linear travel of pollution is governed primarily by the groundwater flow velocity, and the viability of the organisms. Maximum linear diffusion appears to be the distance the groundwater flows in a period of around 10 days. This implied survival time contrasts sharply with laboratory studies and controlled field studies which suggest a possibility of survival of 100 days or more. The distance over which faecal bacteria can be traced will depend not only on the groundwater velocity and their death rate, but also on their initial concentration, dispersion within the groundwater body, the sample volume tested and the sensitivity of the method used to detect them.

Further, other processes (e.g. filtration and adsorption) may retard pathogen transport and thus the distances migrated by pathogens are likely to be significantly smaller than that predicted for an equivalent water molecule for the same time period. In many countries, groundwater protection zones around water supply boreholes are defined on the basis of travel time and designed to provide protection against contamination. Typically the inner zone is defined on the basis of protection against pathogens. The travel times used to prescribe the inner protection zone vary from 400 days, by Thames Water, UK to 10 days in Switzerland, with most water companies in the UK and other European countries using 50-60 days. In a review of protection zones (Adams and Foster, 1979), it was suggested that, based on an earlier review (Lewis et al, 1982), no proven source of groundwater pollution by pathogens had exhibited more than 20 days travel and thus 50 days was a zone that provided reasonable safety. However it is important to realise that zero risk is unobtainable and that in setting a travel time based distance it is important to consider the margins of safety that should be incorporated, the likely health impacts from poor water quality and inadequate sanitation and resource constraints.

A 50 day travel time distance is the preferred level of protection, with travel time being based on movement from pollutant source to drinking water supply. However a 25 day travel time may be more realistic in some circumstances, where there are space (or distance) constraints, accepting that the risk of contamination although low is higher than for the 50 day travel time distance.

3.2.2 Movement in fissured rocks

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

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

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

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

3.3 VIRUSES

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

Table 3.2 Summary of bacterial travel in the saturated zone (includes travel through unsaturated zone) (from Lewis et al 1982)

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

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

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

A recent study in Israel (Marzouk et al, 1979) indicated that 20% of 99 shallow groundwater samples (3 m) analysed contained enteric viruses; the source of these viruses was thought to be septic tank effluents. Viruses were isolated from 12 samples (20-400 1) which contained no detectable faecal bacteria in the 100 ml sample volumes tested. Hence, enteric viruses may be present in water that shows little or no sign of bacterial pollution.

4 Summary

The following section summarises the main findings from the preceding literature review. This provides the basis for the guidelines presented in the companion ARGOSS manual (ARGOSS 2001). Contamination of groundwater supplies by on-site sanitation systems can occur via two main pathways:

(a) the aquifer pathway, where pathogens migrate from the base of the pit latrine to the water table and from there to the intake of the well or screen

(b) localised pathways that develop through the poor design, construction and/or operation and maintenance of the water supply. Such pathways provide a rapid bypass mechanism to the intake of the water supply, limiting residence time in the subsurface

The latter pathway is probably the most common cause of well contamination by micro-organisms (for example, see Bangladesh and Uganda case studies) although contamination can be relatively easily prevented by ensuring correct design of water supply and careful construction. Assessing the risk posed to groundwater by contamination via the aquifer pathway is more difficult and was the subject of the literature review and is summarised below:

- Of the microbial contaminants in human excreta, bacteria and viruses provide the most common concern due to their small size and their mobility in the subsurface. The major inorganic pollutants associated with on-site sanitation are nitrate and to a lesser extent chloride.
- Where the soil zone has been passed, as is the case with pit latrines, the unsaturated zone provides the first and most important line of defence. For fine-grained unconsolidated sediments (e.g. clay, silt and fine sand) even a relatively thin unsaturated zone (~2 m) can provide sufficient attenuation of pathogenic bacteria. In the ARGOSS manual, a minimum thickness of 5 m was proposed to provide an extra safeguard for these fine-grained sediments. Rates of water movement in the unsaturated zone do not normally exceed 0.2 m/d in the short term, even in coarse-grained sediments, and less when averaged over longer timescales. Where the unsaturated zone extends beyond 10 m, travel times to the water table are probably in excess of 25 days. Exceptions do occur and rates of water movement in gravels and in fissured rocks can be much higher. The unsaturated zone cannot be relied upon to attenuate pathogens in such hydrogeological settings.
- Rates of water movement in the unsaturated zone are also dependent on moisture content, especially in coarser sediments (the higher the moisture content the faster the water movement). Thus where fluid loading from on-site sanitation systems is high (>50 mm/d) then the delay, and hence attenuation of pathogens, in the unsaturated zone cannot be relied upon.
- A clogged layer develops around the base of the latrine which enhances bacteria and virus removal by filtration and by predation by antagonistic microbes. This clogged layer may take several months to develop around new latrines.
- The most important processes for attenuating pathogens in the unsaturated zone are predation by other micro-organisms, sorption (especially viruses) and, for the larger micro-organisms in fine-grained sediment, filtration.
- Survival of micro-organisms in unsaturated zone is very variable and is dependent upon moisture content and temperature. Pathogens can survive for many 10's of days although half-lives are typically 5-30 hours.

- At the water table micro-organisms can be transported considerable distances horizontally, depending on the permeability of the aquifer and the hydraulic gradient. In finer-grained sediments (clays, silts and fine sands) rates of groundwater flow are frequently less than 2 m/d. However, in gravels and especially in fractured rocks rates of water movement can exceed 10 m/d or even 100 m/d.
- The processes responsible for retarding and attenuating pathogens are, in most cases, much less effective than in the unsaturated zone. Survival rates for pathogens are likely to be at least as high in the unsaturated zone.
- Many field studies indicate that lateral migration of microbial pollutants in the saturated zone is limited to the distance groundwater flows in period of not more than 15 days, despite survival of individual micro-organisms for much longer times. This can be explained by the fact that transport of micro-organisms is likely to be retarded by the processes of filtration, adsorption and dispersion.
- The concept of a 'safe' distance based on the travel time of a water in a specified period is useful and was the basis for the lateral set back distances or separations between water supplies and on-site sanitation systems. Separations (between water supply and on-site sanitation system) equivalent to the distance that a water molecule would travel in 25 (or preferably 50) days appears to represent a low risk (but not a guarantee) that contamination of the water supply would occur. However, aquifers are rarely uniform and predicting rates of lateral water flow is difficult especially if permeable sand or gravel layers, or fractures, are present.
- Many geological formations are layered or stratified, often producing high rates of horizontal to vertical permeability; this, combined with groundwater flow being normally horizontal, makes rates of downward movement, below the water table, much slower (than lateral flow) and less variable. Attenuation of pathogens over quite short vertical depths are possible.
- The oxidation of organic nitrogenous matter present in human excreta, principally in the form of urea, produces nitrate. The proportion of organic nitrogen that is oxidised and leached to the water table varies considerably and depends principally upon the availability of oxygen. Nitrate concentrations arriving at the water table will frequently exceed drinking water guideline values; mixing and dilution within the aquifer may help to delay the arrival of unacceptable nitrate concentrations at the water supply.

5 Conclusions

This review has shown that on-site sanitation systems can pollute groundwater, especially where the subsurface is permeable and the depth to water table is shallow.

The main contaminants of concern are pathogenic micro-organisms (viruses and bacteria) and nitrate. Whilst microbiological contamination is perhaps of more immediate concern, nitrate may represent a more persistent problem in the longer term. Nevertheless, the subsurface can, and often does, provide a very effective means of attenuating contaminants and in many hydrogeological environments, especially those characterised by slow intergranular water movement (as opposed to fissure flow), removal of pathogenic micro-organisms can be very effective.

Carefully designed and sited low-cost groundwater supplies can be compatible with on-site sanitation in many circumstances. The ARGOSS manual (ARGOSS 2001) provides guidelines for the proper design and location of water supplies. However, there are several areas of uncertainty which still require further attention.

- The extent (and cause) of contamination of shallow low-cost drinking water wells is poorly recognised in many developing countries because of lack of monitoring.
- There is little data on virus transport in groundwater; especially in tropical environments.
- The possibility of developing novel designs of on-site sanitation systems which may help reduce contaminant leaching especially in those environments most susceptible to contamination (e.g. fissured aquifers, and high water table conditions).

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Appendix A

An investigation of the impact of on-site sanitation on the quality of groundwater supplies in two peri-urban areas of Dhaka, Bangladesh

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A.1 INTRODUCTION

In Bangladesh, as in many developing countries, the development of groundwater resources for potable use has increased substantially over the last three decades. Bangladesh depends on groundwater sources to supply a high percentage of its domestic demand for potable water. In rural and periurban areas several million shallow hand tubewells (both private and public) have been constructed to provide drinking water supply to the communities. Although groundwater is generally of better quality than surface water, it can become contaminated and there is special concern that on-site sanitation systems may in certain circumstances contribute to contamination of drinking water supplies. In Bangladesh safe excreta facilities are not very common and pit latrines are generally constructed very close to hand tube wells, which may result in contamination of supplies by microbiological and chemical contaminants contained in faecal matter via a number of pathways. Polluted water may cause considerable health and related social problems. This case study was undertaken to examine the relationship between groundwater quality and on-site sanitation. The specific objectives were to:

1. assess the risk posed by pit latrines to the chemical and microbiological quality of the groundwater;
2. assess the risk posed by environmental factors (e.g. condition of tube wells, distance to pit latrine etc.) on water quality of tube wells;
3. compare evaluated groundwater quality with drinking water quality standards;
4. recommend measures for protecting existing groundwater resources from contamination due to on-site sanitation.

A.2 CASE STUDY AREAS

A.2.1 Location and population

The two study areas selected represent contrasting geology, hydrogeology, land use and sanitation settings. Dattapara, to the north of Dhaka city (Figure 2.1), is situated on the elevated Madhupur Tract where the oxidised sands and gravel beds of the Pliocene Dupi Tila aquifer are overlain by a thick sequence (15 m) of Pleistocene Madhupur Clay (Figure 2.2). The area is not affected by annual monsoonal flooding. Keraniganj, to the south-west of Dhaka city (Figure 2.3), is situated on the Recent alluvial flood plains of the Ganges and its tributaries. The aquifer is unconfined to leaky confined with a top layer of silt and clay of varying thickness (Figure 2.4). The low-lying and agricultural areas here are inundated every year by normal monsoonal flooding. The river Buriganga marks the boundary between the elevated Madhupur Tract and the Recent floodplains.

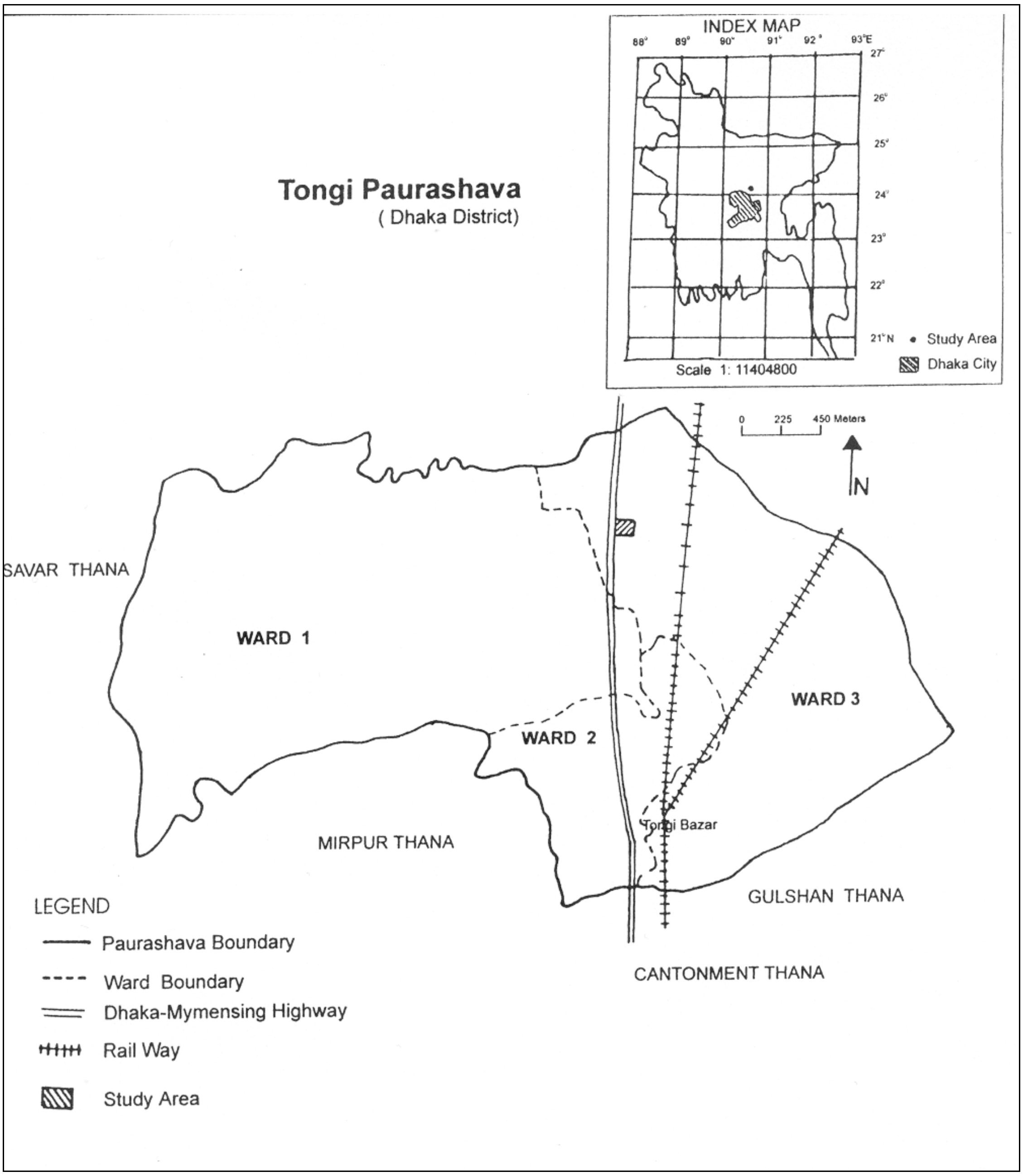


Figure A2.1 Location map of Dattapara study area

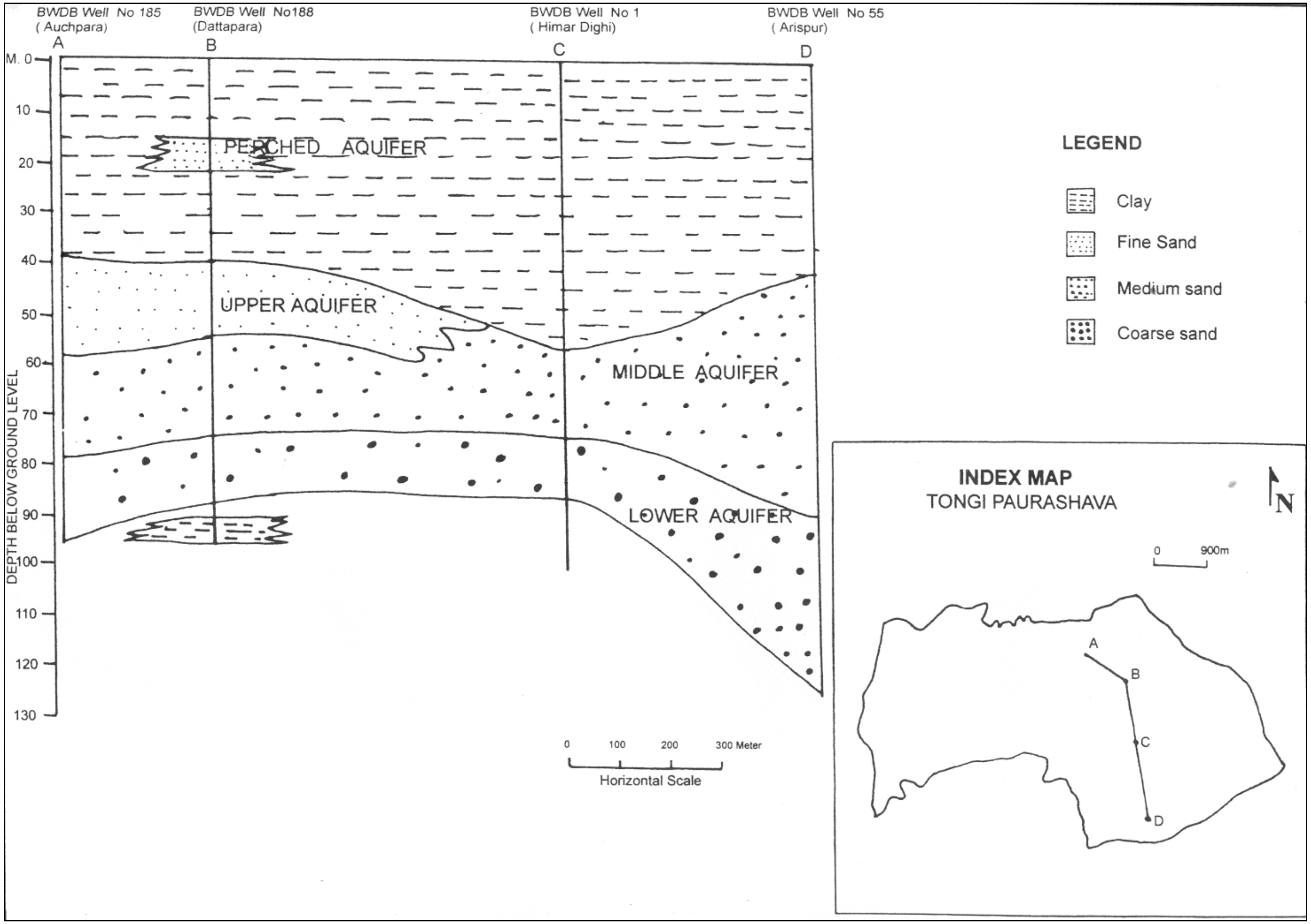


Figure A2.2 Geological cross-section through Dattapara study area

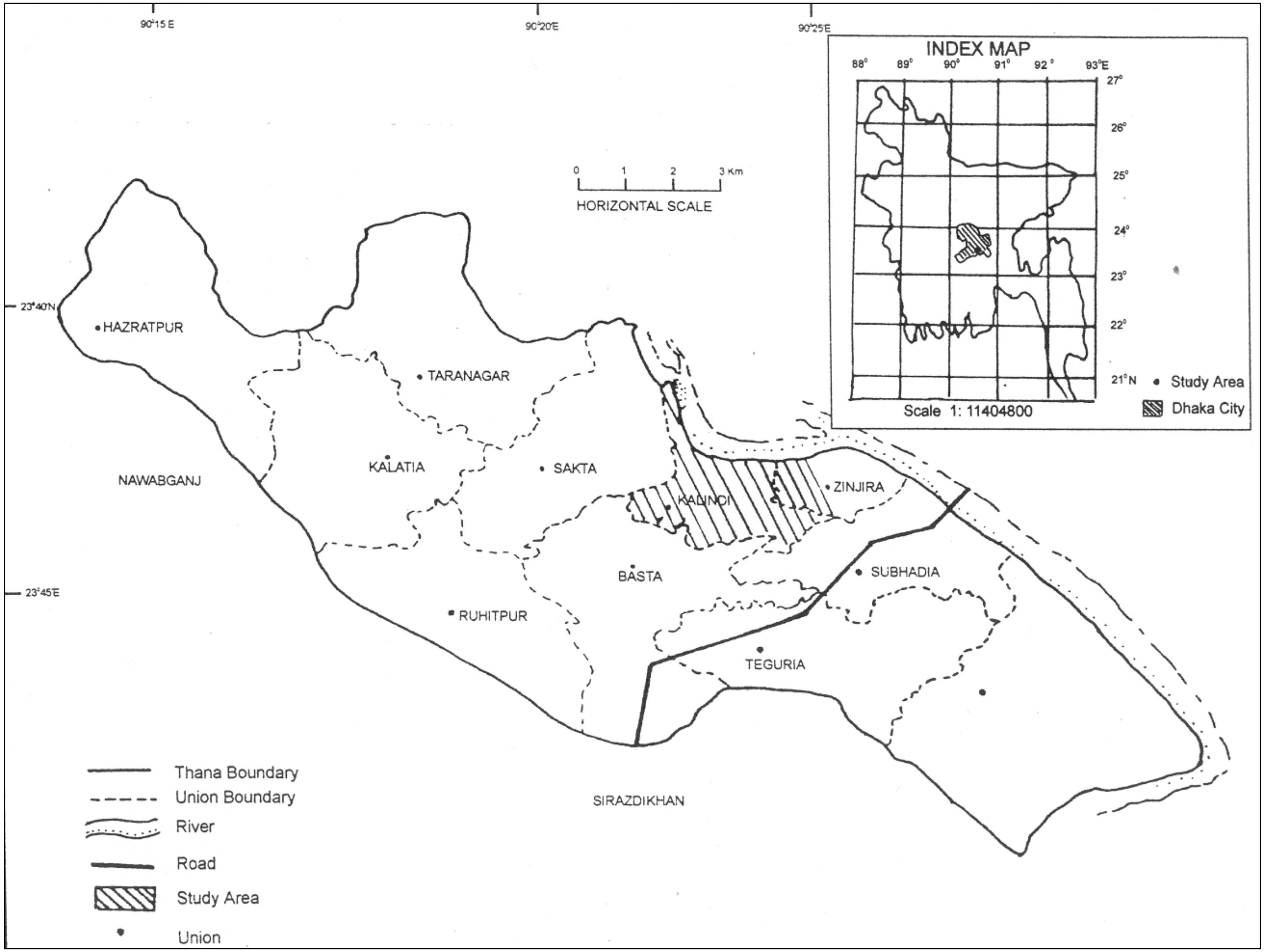


Figure A2.3 Location map of Keraniganj study area

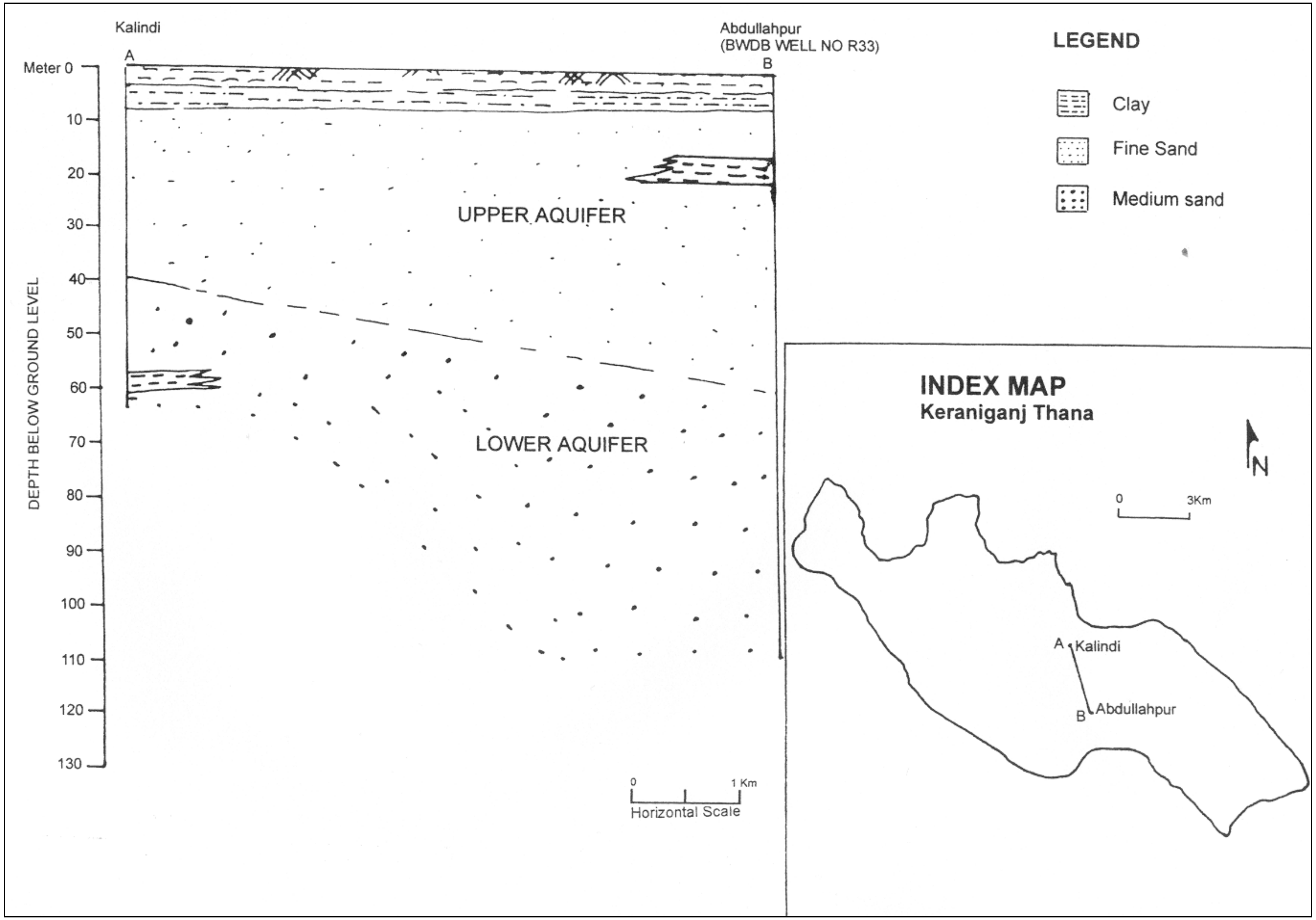


Figure A2.4 Geological cross-section through Keraniganj study area

Dattapara is a very densely populated area where almost every house has an on-site sanitation system, in the form of a pit latrine, and a hand-pumped tubewell within close proximity of each other. The area was developed as a small shanty town for the slum dwellers of Dhaka relocated in early 1974. Dattapara (Ershad Nagar) is a settlement within Tongi Paurashava (Ward No. 3), Gazipur Sadar Thana, Gazipur district. The Thana occupies an area of 446.38 km^2 including 0.31 km^2. river and 54.52 km^2. forest area. Dattapara study area covers an area of 0.45 km^2 to the east of the Dhaka-Mymensingh highway.

Gazipur Sadar Thana in which the Dattapara study area is located, has a population of 685,325. There are 116,163 households, and the population density is 1748 persons per km^2. The decadal population growth rate is 55.96% and annual compounded growth rate is 4.54% (Bangladesh Population Census, 1991). The Dattapara study area, however, is a densely populated slum area, with a total population of 27899 (source: Terre des Hommes (TDH), a Dutch NGO, January 1999) representing 5300 households. The population density is 61,998 persons km^2 (619.98 person/ha).

Keraniganj was a typical village by the Buriganga until forty years ago. With the rapid urbanisation in Dhaka, the area started developing as a sub-urban growth centre and accordingly the population density and land use patterns changed. However, the population density is still well below that of Dattapara. The area has two different types of latrines, viz. pit latrines and hanging latrines.

Keraniganj is a Thana of Dhaka district and is located on the south bank of the River Buriganga. The study was carried out at three mauzas (Gokpar, Kalindi and Mandail). Keraniganj Thana occupies an area of 166.87 km^2 including 9.79 km^2 river area. The study area covers an area of 2.7 km^2. The Dhaleswari River borders Keraniganj on the west and the south, and the Buriganga River borders the area to the north and to the east.

Keraniganj has a population of 617,411, the second largest of Dhaka district Thanas. The population density is 3932 persons per km^2. The decadal population growth rate is 46.61% and annual compounded growth rate is 3.90%. There are about 94765 households in the Thana (Bangladesh Population Census, 1991).

A.2.2 Drinking water supply and sanitation

In both Keraniganj and Dattapara, hand-pumped tubewells are the source of drinking water for over 95% of the households (Bangladesh Population Census, 1991). Tubewells in Dattapara are generally constructed privately or by TDH and the Government. The tubewells, which are constructed to a shallow depth (15-20 m), are sufficient in number to meet the local demand of water in the wet season but not in the dry season when water levels drop. Few people in either area use piped water, dug wells, ponds or surface water bodies as a source of water.

According to the Bangladesh Population Census (1991) 21.19% of households in Keraniganj have off-site sanitation, 74.63% have on-site sanitation, while 4.18% households have no sanitation facilities. Two different types of latrines are used, pit latrines and hanging latrines. A hanging latrine is a covered platform constructed over a pond. Almost every house in Dattapara has a pit latrine; the total number is 3681 (TDH, 1998). Most pit latrines in the study areas are of the "pour-flush" type. These latrines are generally placed in approximately 1.5 to 3m deep excavations with overflow being discharged via a canal system.

Other potential sources of contamination in the study areas (especially in Dattapara) include drainange ditches which convey sewerage effluent; ponds and stagnant pools; piles of rubbish; excreta of children, cattle, chickens etc around tubewells.

A.2.3 Climate

and its surroundings are characterised by a hot and humid summer and a cool and pleasant winter (from November to February). Temperature ranges between 30°C and 38°C in summer while the minimum average temperature during winter is 9°C. The climate of Dhaka City is characterised by a wet season, March to October, with moderately high temperatures (average monthly maximum 32-35°C) and a dry season (November to February) with moderately low temperatures (26-30 °C).

Table A.1 shows the monthly rainfall figures for Keraniganj and Dattapara for the period of the study. On average, more than 80 percent of rainfall occurs between May and September, the monsoon period.This was the case during the period of the study. Table A.1 shows the variability that can exist in rainfall over relatively small distances. In 1998 the monthly maximum rainfall in Keraniganj was 434 mm compared with 887.5 mm at Dattapara.

Table A.1 Monthly rainfall in millimeters in Keraniganj and Dattapara (1998).

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Keraniganj (BWDB Station No. R412)	45	10	114	128	210	72	434	345	204	30	91	0
Dattapara (BWDB Station No.R17)	42	3.1	58	82	236	59	888	394	359	48	97	0

A.2.4 Physiography

The Madhupur Tract occupies the central part of the Greater Dhaka District. The rest of the region is covered by the flood plains of the Jamuna-Padma-Meghna rivers. The Madhupur Tract stands higher than the surrounding flood plains. On the west it is separated from the Jamuna flood plain by a near vertical scarp, while on the south it dips almost imperceptibly beneath the recent flood plain. On the east, the Madhupur Tract and flood plain meet at a low angle, but they can be distinguished easily.

In the physiographic divisions of Bangladesh, Keraniganj falls in the Young Brahmaputra (Jamuna) flood plain. The Ganges River flood plain in the south and the Madhupur Tract in the east border this physiographic unit. The area slopes regionally towards the south with gentle and low relief. Levees are found in the elevated parts of the area wherein flood basin and ox-bow lakes are located in the lowest depressions. The average elevation of the area is 3 to 4 metres above mean sea level (masl). The highest elevation is 7 masl near Jainpur in the north-western part of the area

Dattapara is situated on the elevated Madhupur Tract. Red soils of the Pleistocene Madhupur Clay Residuum usually form the surface of the Madhupur Tract and are roughly delimited by the Bangshi, Banar-sitalakhya and Buriganga rivers. Average elevation of the area is 6 to 8 masl.

A.2.5 Drainage Systems

Two principal rivers, the Dhaleswari and the Buriganga, with their numerous interconnecting small channels form the drainage system of Keraniganj Thana. A large number of interconnecting small channels and khals, some of which are ephemeral in nature, are present in the investigated area. Standing water bodies are also common in the area.

The Dattapara slum area is devoid of a natural drainage system. Artificially constructed narrow concrete drains criss-cross the whole slum area to carry out wastewater. However, these drains do not work efficiently due to poor construction and lack of proper maintenance. As a result stagnant water bodies form in places. In addition to these, a small number of ponds are also present in the investigated area.

Flooding is a natural recurring phenomenon in Bangladesh. Extensive and devastating river floods occurred in Bangladesh in 1954, 1955, 1962, 1966, 1970, 1974, 1984, 1987, 1988 and 1998. However, the extent and maximum depth of flooding are very variable on the local scale and controlled by the distribution of the different flood phases. Flooding disrupted the monitoring programme for two months in Keraniganj and for a month in Dattapara, as all the wells were flooded.

In Keraniganj river levels start to rise in early July and reach a peak at the end of August, beginning to recede in September. About 90% of the area is submerged during peak floods. The duration of flooding varies from 15 to 60 days and the depth of water from 0.5 to 2.5 m.

At Dattapara flooding starts later and duration is less compared to Keraniganj. Here river levels start to rise in early August and peak at the end of August. The duration of flooding varies from 20 to 25 days with a depth of 2 feet water.

A.2.6 Geology

Dattapara is situated on the elevated Madhupur Tract where the Dupi Tila formation is unconformably overlain by a thick sequence (15 m) of Pleistocene Madhupur clay. Keraniganj is situated on the recent flood plains of the Ganges and its tributaries. Here the Recent Alluvium overlies a top layer of silt and clay. The river Buriganga demarcates the boundary between the elevated Madhupur Tract and the recent floodplains. However, the correlation of strata between Keraniganj and other parts of Dhaka City is very uncertain owing to the paucity of good quality borehole information in this area. A simplified cross-section of the geology is included in Figure 2.2.

DUPI TILA FORMATION

The Dupi Tila consists of mainly yellowish brown to pale grey medium to coarse sand or sandstone, with minor kaolinitic clay bands. The sands are weakly cemented by secondary clay and iron oxides. The Dupi Tila consists of alternating fluvial channel and floodplain deposits (Johnson and Alam, 1990). Beneath the Madhupur Tract the sands contain on average 52% quartz, 29% feldspar (mainly plagioclase) and 12% mica as well as about 4% kaolinite and 3% chlorite which is lodged in pits on, and adhering to, the surface of sand grains.

The Dupi Tila was followed by a period of erosion and subsequent deposition of the lower alluvial sequence of the Dihing series. This probably occurred at the onset of one of the glacial phases of the Pleistocene. Davies (1989) suggests that the coarsest layers may have been derived from fault initiated gravity slides in the vicinity of the hinge zone. The grain size of these deposits indicates a declining energy environment (WASA, 1991). Krishnan (1953) considered the formation to be of Pliocene age.

MADHUPUR CLAY AND SAND FORMATION

The Madhupur clay and sand Formation unconformably overlies the Dupi Tila Formation. It outcrops over more than half of Dhaka city, and thickness varies from 6 to 25 m, with an average of 10 m. This formation also extends a few meters below the flood plain areas of Dhaka city. The lower subunit (Bhaluka Sand) consists of pale yellowish brown silty sands and sands. It is highly micaceous and cross-bedded (Monsur, 1994). The middle subunit (Mirpur Silty Clay) consists of light brown sandy clay to clayey sand. The upper subunit (Dhaka Clay) consists of red clays with reddish yellow spots. The middle and upper subunits also contain iron concretions, pipe stems, plant roots and manganese spots (Monsur, 1994). The two upper beds (Kalsi Beds) of this formation are sandy clay to silty clay with iron concretions (Monsur, 1994).

A.2.7 Hydrogeology

AQUIFER SYSTEM OF THE STUDY AREAS

The two study areas represent two different aquifer systems. At Dattapara, the Pliocene Dupi Tila aquifer is confined or semi-confined as it is overlain by a thick sequence (15 m) of Pleistocene Madhupur clay. In Keraniganj Recent alluvium forms the unconfined to leaky confined aquifer with a top layer of silt and clay of varying thickness. The aquifer characteristics of these two localities are discussed below.

Dattapara

A geological cross section based on data from four boreholes in the Dattapara region (Figure 2.2) reveals a confined to semiconfined aquifer system with three distinct subunits – an upper, middle and lower aquifer. A thick clay sequence (aquiclude) overlies the whole aquifer system. The upper aquifer is discontinuous, extending from the base of the aquiclude down to a depth of 50 to 60 m from the ground surface. Its thickness varies from 0 to 20 m, and is composed of very fine to fine sand. The water level within this confined aquifer is typically at a depth of 8 m below the ground surface. The long term hydrograph of Dattapara area indicates seasonal fluctuations of water levels and a possible small long term decline. The middle aquifer extends from the base of the aquiclude (and from the base of the upper aquifer where the latter is present), down to a depth of 80 to 90 meters from the ground surface. It is between 15 and 50 m thick, and is composed of medium sand. The lower aquifer extends from the base of the middle aquifer down to a depth of 85 to 125 m from the ground surface, with thickness ranging from 15 to 45 m. It is composed of medium to coarse sand and some gravel. In addition, there is a local perched aquifer located 15-20m below the ground surface. Most of the hand tubewells of the Dattapara slum area tap this aquifer.

WASA (1991) reported that vertical permeability of the overlying aquiclude in Dattapara varies from $6.5x10^{-4}$ to 1.5×10^{-2} m/d with an average value of about 4.6×10^{-3} m/d. It can neither yield freely to wells nor transmit appreciable water to the adjacent aquifer below it. As estimated by Barker et al. (1989), its specific yield ranges between 0.02 and 0.27 with an average value of 0.084

Aquifer test data indicate that the main (lower) aquifer is good to excellent, with permeability values between 11 and 18 m/d (WASA, 1991), and storativity of 0.02 to 0.05 (BWDB, 1998). The transmissivity of the aquifer varies from 131 to 3352 m^2/d with an average value of 1600 m^2/d (WASA, 1991), which indicates the high degree of water transmitting capacity of the aquifer. While no separate aquifer tests have been conducted on the upper and middle aquifers, they are thought to have hydraulic characteristics suitable mainly for hand tubewells and/or shallow wells of small yield.

Keraniganj

It is quite difficult to delineate precisely the aquifer system of this area due to a lack of good quality borehole information. However, an attempt has been made to delineate a generalised aquifer system; Figure 2.4 shows two aquifers, upper and lower, which are of fine and medium grained sand respectively. The upper (shallow) aquifer is at a depth of between 5 to 7 metres from the ground surface, and its thickness varies from 30 to 60 metres. The water-level in the aquifer is typically at a depth of about 3 m below the ground surface. The long term hydrograph of Keraniganj shows no evidence of long-term water-level fluctuations. The lower (deeper) aquifer is located at a depth of between 40 to 70 metres from the ground surface with an undefined base. Most of the hand tube wells of this area are constructed down into the lower aquifer.

RECHARGE OF GROUNDWATER

Recharge to the aquifers below Dhaka City occurs through direct precipitation, flood waters and by horizontal inflow from surrounding areas (WASA, 1998). The rate of percolation from these sources depends on the available rechargeable surface area and the vertical permeability of the unsaturated zone. At Dattapara, recharge from rainfall and floodwater is thought to take place very slowly as the aquifer is overlain by a relatively thick (about 40m) clay layer. Recharge by horizontal inflow from floodwaters of the low-lying areas is thought to occur only during the period from July to September. Thus rainfall and floodwater can hardly replenish the aquifer system. In Keraniganj recharge from rainfall and floodwater is high even where the presence of an overlying silty-clay layer produces leaky or semi confined conditions. Recharge by horizontal inflow from low lying areas and from the Buriganga river is thought to occur throughout the year.

A.2.8 Design and construction of hand pumped tubewells

Hand pumped tubewells are constructed by the "sludge" method which involves manually driving 38 mm GI pipe with a 65-75 mm cutter at the end. A drilling fluid, a mixture of mud, cow dung and water, is used to stabilise the hole. Soil samples are forced up through the hollow GI pipe and retrieved at the surface. When visual inspection of these samples indicates that fine-medium sand is reached, drilling is stopped. The rods are then pulled out and the well screen and casing lowered into the hole. The formation is allowed to collapse around the casing although this does not necessarily ensure that there are no voids between the formation and the casing.

The casing is normally made of PVC (38 or 50 mm ID) except for the upper length which is of GI pipe. This latter pipe is normally embedded in a concrete block to keep the casing pipe rigid and a concrete platform is placed at the surface (Figure 2.5).

Good design and maintenance of the weathered completion is necessary to reduce the risk of contamination of the well by 'localised' pathways resulting from poor design or completion of the well head. Features that need to be included in the design (ARGOSS, 2001) are:-

- cement apron sloping away from the tubewell and extending more than 1.5 m (cement should remain sound with no cracks)
- drainage channel (should remain in good working order, not cracked, broken or blocked)
- firm attachment between handpump and apron
- protective fence

The standard of design and maintenance of handpump tubewells in Bangladesh is very variable. A major concern in the current design and construction of handpump tubewells is that there is no provision for a sanitary cement seal between the borehole wall and the tubewell casing. The absence of a sanitary cement seal can result in:-

- contaminated groundwater, at the water table, being in direct contact with the tubewell casing; any defect or crack in the casing could permit ingress of contaminated water into the tubewell
- contaminated groundwater migrating down the outside of the casing, bypassing the fine-grained formation, to the screen of the tubewell.

Thus, handpump tubewells in Bangladesh must be considered to have a significant pollution risk as a result of their design providing localised pathways for contamination.

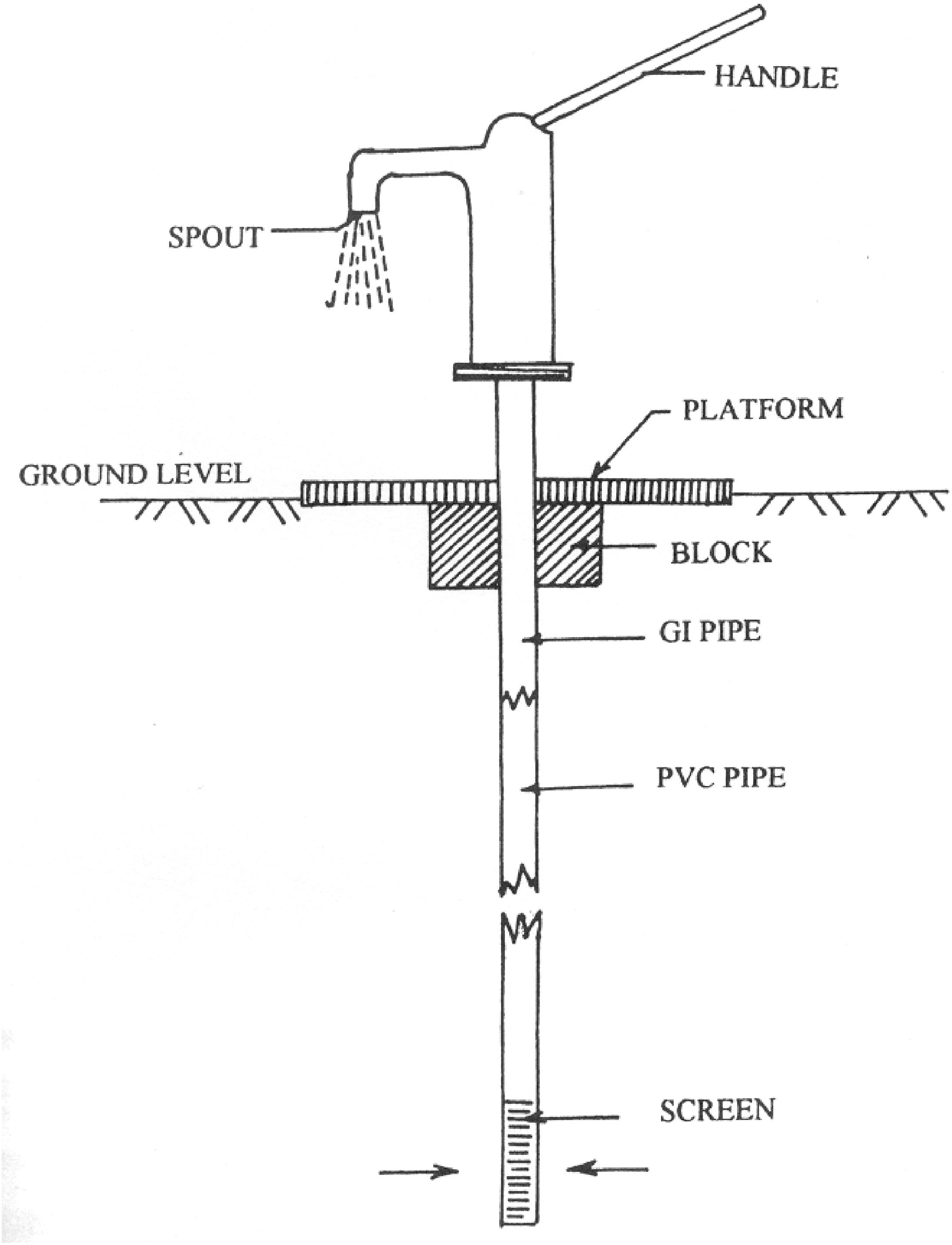


Figure A2.5 Design of hand-pumped tubewells used in Bangladesh

A.3 METHODOLOGY

A.3.1 Site Selection and monitoring

Within Dattapara and Keraniganj 50 hand-pumped tubewells were selected for monitoring (Dattapara - D1 to D50; Keraniganj - K1 to K50). Figures 3.1 and 3.2 show the positions of monitoring tubewells in Dattapara and Keraniganj, respectively. The ranges of depths of the tubewells monitored in each area are shown in Table A.2. Monitoring was carried out between February 1998 and October 1999.

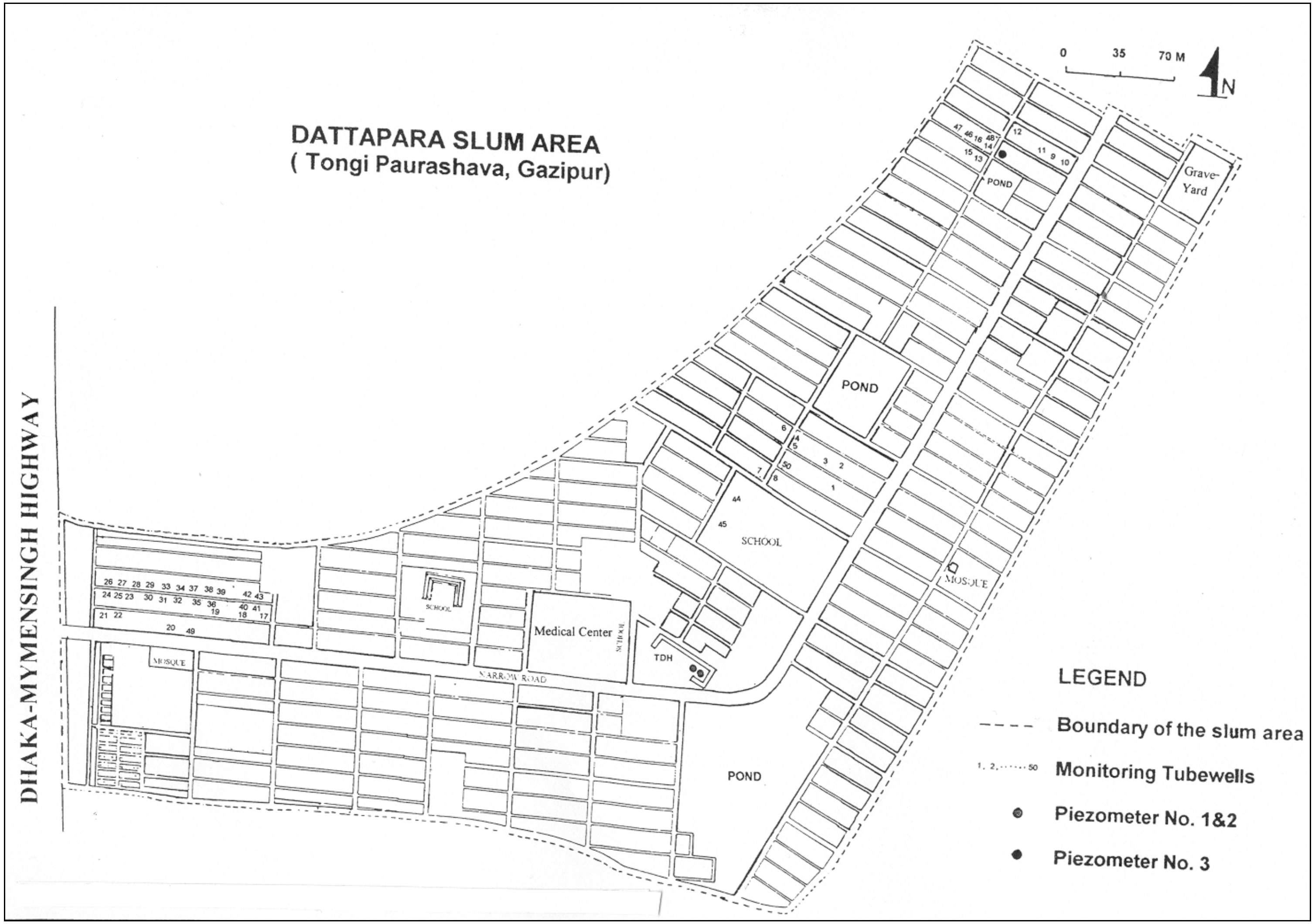


Figure A3.1 Locations of monitoring tubewells and piezometers in Dattapara study area

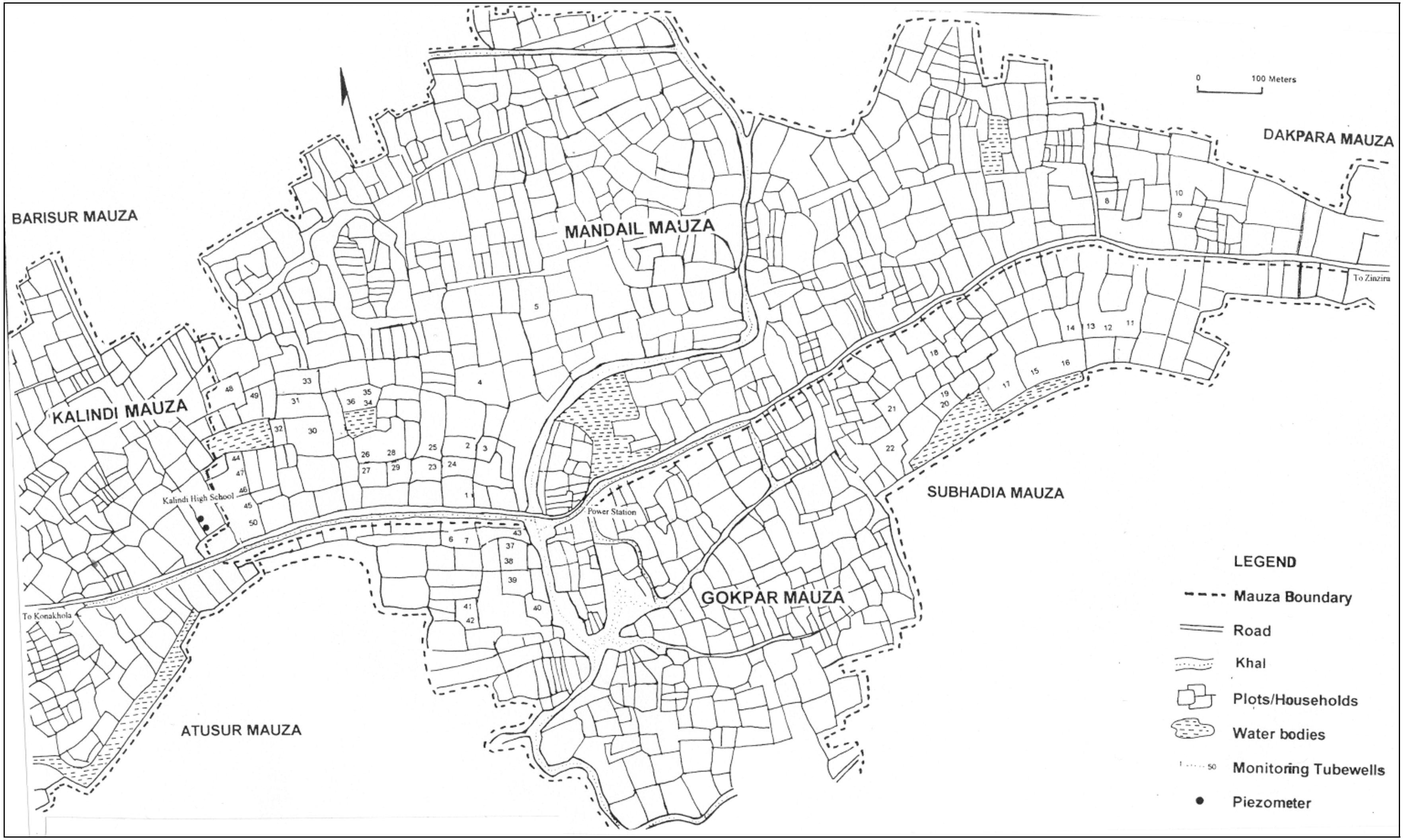


Figure A3.2 Locations of monitoring tubewells and piezometers in Keraniganj study area

TableA.2 Depths of tubewells monitored in Dattapara and Keraniganj.

Depth range (m)	Number of tubewells	
	Dattapara	*Keraniganj*
≤ 10	0	0
11-20	46	0
21-30	1	0
31-40	0	2
41-50	0	11
51-60	1	6
61-70	2	20
71-80	0	5
81-90	0	5
91-100	0	1

As part of the first and last monitoring rounds, sanitary survey inspections were carried out to obtain information on the condition of the tubewells, distance from pit latrine and presence of other potential sources of contamination. A hazard score for each tubewell was obtained based on sanitary survey forms. A subset of the 50 tubewells at each site were chosen for more frequent monitoring. Both shallow and deep wells with high and low hazard scores from the sanitary surveys were selected. Initially 20 tubewells from each area were included for this routine monitoring. This was subsequently reduced to 12 tubewells from Keraniganj and 15 from Dattapara.

The frequency of monitoring and the determinands measured are presented in Table A.3. The measurements undertaken in the full suite are provided in Table A.4.

Table A.3 Routine Monitoring: Sampling frequency and parameters measured.

Approx. date of sampling	Interval since previous sampling	Chemical/ microbiological parameters	Wells sampled	Sanitary survey?
End Feb 98		Full suite[1]	50 each site	yes
End Apr 98	8 weeks	SEC[2] + FC[3], FS[4]	50 each site	no
Early Jun 98	5 weeks	Full suite	20 each site	no
Mid Jun 98	2 weeks	SEC + FC, FS	20 each site	no
End Jun 98	2 weeks	Full suite	20 each site	no
Mid Jul 98	2 weeks	SEC + FC, FS	Keraniganj - 12, Dattapara - 15	no
End Jul 98	2 weeks	SEC + FC, FS	Keraniganj - 12, Dattapara - 15	no
Mid Aug 98	2 weeks	SEC + FC, FS	Keraniganj - 12, Dattapara - 15	no
End Sep 98	2 weeks	Full suite	Keraniganj - 12, Dattapara - 15	no
Mid Oct 98	6 weeks	SEC + FC, FS	Keraniganj - 12, Dattapara - 15	no
End Nov 98	6 weeks	SEC + FC, FS	Keraniganj - 12, Dattapara - 15	no
Mid Jan 99	6 weeks	Full suite	50 each site	no
End Feb 99	6 weeks	SEC + FC, FS	Keraniganj - 12, Dattapara - 15	no
Mid Apr 99	6 weeks	Full suite	Keraniganj - 12, Dattapara - 15	no
Early Jun 99	6 weeks	SEC + FC, FS	Keraniganj - 12, Dattapara - 15	no
Mid Jun 99	6 weeks	Full suite	Keraniganj - 12, Dattapara - 15	no
End Jun 99	2 weeks	SEC + FC, FS	50 each site	no
Mid Jul 99	2 weeks	SEC + FC, FS	Keraniganj - 12, Dattapara - 15	no
Early Aug 99	2 weeks	SEC + FC, FS	Keraniganj - 12, Dattapara - 15	no
End Aug 99[5]	2 weeks	Full suite	50 each site	Yes

[1] Full suite – Parameters listed in Table 3.3
[2] SEC – Specific Electrical Conductance
[3] FC – Faecal Coliforms
[4] FS – Faecal Streptococci
[5] Not all sites were accessible due to flooding; monitoring was undertaken between August and October

Table A.4 Parameters determined in monitoring programme.

	Parameters: Chemical	Parameters: Microbiological
Field determination	bicarbonate temperature SEC dissolved oxygen pH alkalinity redox potential	none
Laboratory determination	Major ions including: chloride, nitrate, ammonium	Faecal coliforms Faecal streptococci

The sampling procedure was as follows: the tubewells were flushed-out for 5 minutes prior to sampling. The pump spout was flamed prior to the sampling of water for bacteriological analysis to create sterile conditions. Water samples for bacteriological analysis were collected in sterile nalgene plastic bottles and and kept in in a cool box with ice packs during transportation to the laboratory. The samples were processed within 6 hours of collection.

A full description of methods for sampling and analysis can be found in Khandkar (1999).

A.3.2 Drilling and Installation of Piezometers

As part of this study piezometers were drilled at both sites, principally to collect water level information but also to obtain samples for water chemistry analysis. At Keraniganj two piezometers were drilled at depths of 15.2 m and 58.5 m (24 July 1998). At Dattapara three piezometers were drilled at depths of 15.2 m, 15.2 m and 54.3 m (1 August 1998). One of the shallow piezometers at Dattapara was drilled within 1 m of a pit latrine to ascertain the depth below the pit latrine that micro-organisms could be detected in the sediment. Sediment samples were collected every metre to a total depth of 10 m. These samples are analysed for faecal coliforms and faecal streptococci. The sediment sampling method involved pushing a split-spoon sampler holding a sterilised 1.5 inch tube into the sediment. The tube was recovered at the surface, sealed at both ends with wax and carried to the laboratory.

The piezometers were included in the tubewell monitoring round, following the same frequency and measuring the same determinand set.

A.4 RESULTS

An overview of the key results from the case study is provided here. Data collected during the study is described in detail in Khandkar (1999).

A.4.1 Sanitary inspections

Sanitary inspections were carried out during the full sampling point surveys in February 1998 and August-October 1999. The results of these surveys are shown in Tables A1 and A2 in the Appendix to this chapter. These results are summarised in Table A.5, which gives the percentage of well locations for each period and case study site for which the answer to the inspection question was positive. It also provides average numbers of pit latrines occurring within 5, 10 and 15 metres of the well.

Table A.5 Summary of sanitary inspections undertaken in Dattapara and Keraniganj during the period of the project.

Sanitary inspection question	Percentage of well locations at which answer to question is yes (Number of wells inspected in brackets)			
	Dattapara		Keraniganj	
	Feb 98 (50)	Aug 99 (46)	Feb 98 (50)	Aug 99 (46)
1. Pit latrine within 3 m	14	35	48	63
2. other source of pollution within 3 m	42	43	64	54
3. drainage poor, causing stagnant water within 2 m	40	37	56	52
4. hand pump drainage channel faulty, permitting ponding	38	28	60	54
5. cement floor less than 1 m	88	93	58	76
6. ponding on cememnt floor	36	30	36	30
7. is there a cement floor	88	89	76	78
8. cement surround cracked	30	24	32	26
9. handpump loose at point of attachment to base	12	13	28	26
10. slot on top of hand pump	24	11	34	37
11. slot used to prime pump	24	9	34	22
12. pit latrines used	100	98	94	96
Average numbers of wells where pit latrines occur within specified distance				
< 5 m	0.3	0.5	0.6	0.7
< 10 m	0.9	1.1	1.3	1.3
< 15 m	1.7	2.2	1.6	1.7

The sanitary inspection survey in February 1998 showed the tubewells in both Dattapara and Keraniganj had a wide range of risk scores (risk score is the sum of the number of positive answers to questions 1 to 12 on the sanitary survey form). No tubewells in either Dattapara or Keraniganj had a score of zero. In Dattapara 64% of the tubewells had a low score (1-5) while 42% had a moderate score (6-8) and 12% a high score (9-12). In Keraniganj conditions were similar with 54%, 26% and 20% of the monitored tubewells having low, moderate and high risk scores respectively.

The survey also demonstrated the potential for contamination from pit latrines: 95 out of 100 wells had a pit latrine within 15 m, with 2 latrines within 15 m as the average. 38 of the wells had a pit latrine within 5 m. Physical conditions of the 100 monitoring wells showed that 1 in 10 of the tubewells were loose at the point of attachment to the concreter base; that ponding occurs on or around the concrete apron at 6 out of 10 well sites and that in all cases the cement surround is inadequate ie it extends less than 1 m beyond the well or is cracked.

The results of the sanitary inspection undertaken 15 months later showed not unexpectedly similar results although the number of wells which had a pit latrine within 3 m had increased significantly in Dattapara and Keraniganj. The other main difference is that fewer tubewells were fitted with handpumps which had a slot on the top, which was used to allow water into the tubewell to help prime the pump.

A.4.2 Microbiological water quality

Sampling rounds including all study wells at Dattapara and Keraniganj, were undertaken in May 1998 and in January, June and August to October 1999. As explained in Chapter 3, flooding in Keraniganj in 1999, meant that sampling of wells could not be undertaken in one round and had to be done over a period of three months as flood waters abated. A summary of the microbiological results is given in Table A.6. Those samples which had faecal coliform counts of greater than 1000 per 100 ml are thought to have been contaminated by poor quality water used for priming as counts of this number are very uncommon in Bangladesh groundwaters (pers. comm. Dr Bilqis Amin Hoque).

Table A.6 Summary of microbiological analyses from sampling rounds of all wells in Dattapara and Keraniganj. Numbers of wells per range of FC/100 ml.

Ranges of measured FC/100 ml	Dattapara				Keraniganj			
	May 98	Jan 99	Jun 99	Aug 99	May 98	Jan 99	Jun 99	Aug-Oct 99
0	37	42	22	21	20	27	23	16
1-9	6	5	14	13	21	19	14	22
10-49	2	1	3	9	2	3	5	5
50-99	0	0	1	3	2	0	1	1
100-999	0	0	4	0	0	0	4	2
≥1000	5	0	2	0	5	0	0	0

A subset of the study wells at each site were chosen for more frequent sampling, 10 wells at each site. These time series data are presented in Tables A.7 and A.8.

The data provide some evidence that the microbiological water quality in Dattapara is better than in Keraniganj, even though the average depth of the tubewells in Keraniganj is significantly greater than in Dattapara. For example, based on the full sampling round data, the percentage of wells with a zero faecal coliform count is on average 61% for Dattapara compared with 43% for Keraniganj. However, it is noteable that the percentage of wells with a faecal coliform count of less than 10 is comparable at the two sites, 80% Dattapara and 81% Keraniganj. There appears to be some correlation between the microbiological water quality and both the preceding and current rainfall conditions at the time of sampling (Figure 4.1) with water quality being best in the dry season and poorest during the wet season. In June 1999 and in August-October 1999, periods of very high rainfall have the poorest microbiological water quality, with faecal coliform counts of greater than 10 occurring in, on average, 20% of the wells compared with 4% in January 1999.

Table A.7 Microbiological analysis from higher-frequency monitored wells at Dattapara.

Sample date		Rainfall (mm)	Faecal coliform per 100 ml D01	D07	D17	D20	D23	D24	D26	D31	D33	D49
1998	Apr	82										
	May	236	19	1000	0	1	1000 0	0	0	17	1000	8
	Jun	59	200 14 24	7 4 1	0 0 0		2 2 1	0 2 0	0 0 65	0 0 0	24 0 0	5 0 0
	Jul	888	39 3	14 68	8 0		24 0	5 5	74 1	27 5	52 0	41 7
	Aug	394			9		0	2	0	33	10	0
	Sep	359										
	Oct	48	31	1	0		0	0	0	0	0	0
	Nov	97	2	0	11		0	0	0	0	0	0
	Dec	0										
1999	Jan	0	0	0	0	0	1	0	0	1	0	3
	Feb	0										
	Mar	0										
	Apr	7	NA	1	2	0	29	0	0	0	16	0
	May	431										
	Jun	423	NA 3	28 1000	1 9 5	0 0 0	15 390 32	0 0 145	1 0 0	0 0 0	0 0 0	NA NA NA
	Jul	595	3	17	7	0	13	0	1	0	1	NA
	Aug	463	3	12	4 70	0 4	2 23	0 24	0 0	0 1	0 13	NA 0

Table A.8 Microbiological analysis from higher-frequency monitored wells at Keraniganj.

Sample date		Rainfall (mm)	Faecal coliform per 100 ml									
			K04	K10	K14	K17	K21	K22	K23	K26	K41	K42
1998	Apr	128										
	May	210	74	4	0	2	6000	9	1000	1	10	1000
	Jun	72	16 8	0 3	0 4	9 0	37 1	190 53	8 0	6 0	62 1	41 12
	Jul	434	160	30	12	35	6	59	4	100	4	36
	Aug	345										
	Sep	204	4		0	3			0	30		
	Oct	30										
	Nov	91										
	Dec	0										
1999	Jan	0	0	2	0	1	8	18	0	11	3	7
	Feb	0										
	Mar	0										
	Apr	7	0	310	0	0	36	1	1	7	5	14
	May	431										
	Jun	423	55 0 5	9 33 18	0 2 3	1 1 1	17 14 0	17 510 4	1 0 0	1 0 0	48 2 2	1 630 2300 0
	Jul	595	97	5	1	0	0	2	3	0	2	23
	Aug	463	21 20	8 12	0	3	7	3	0 3	0 3	0	15
	Sep	553			0	145						
	Oct	372					0	12			3	8

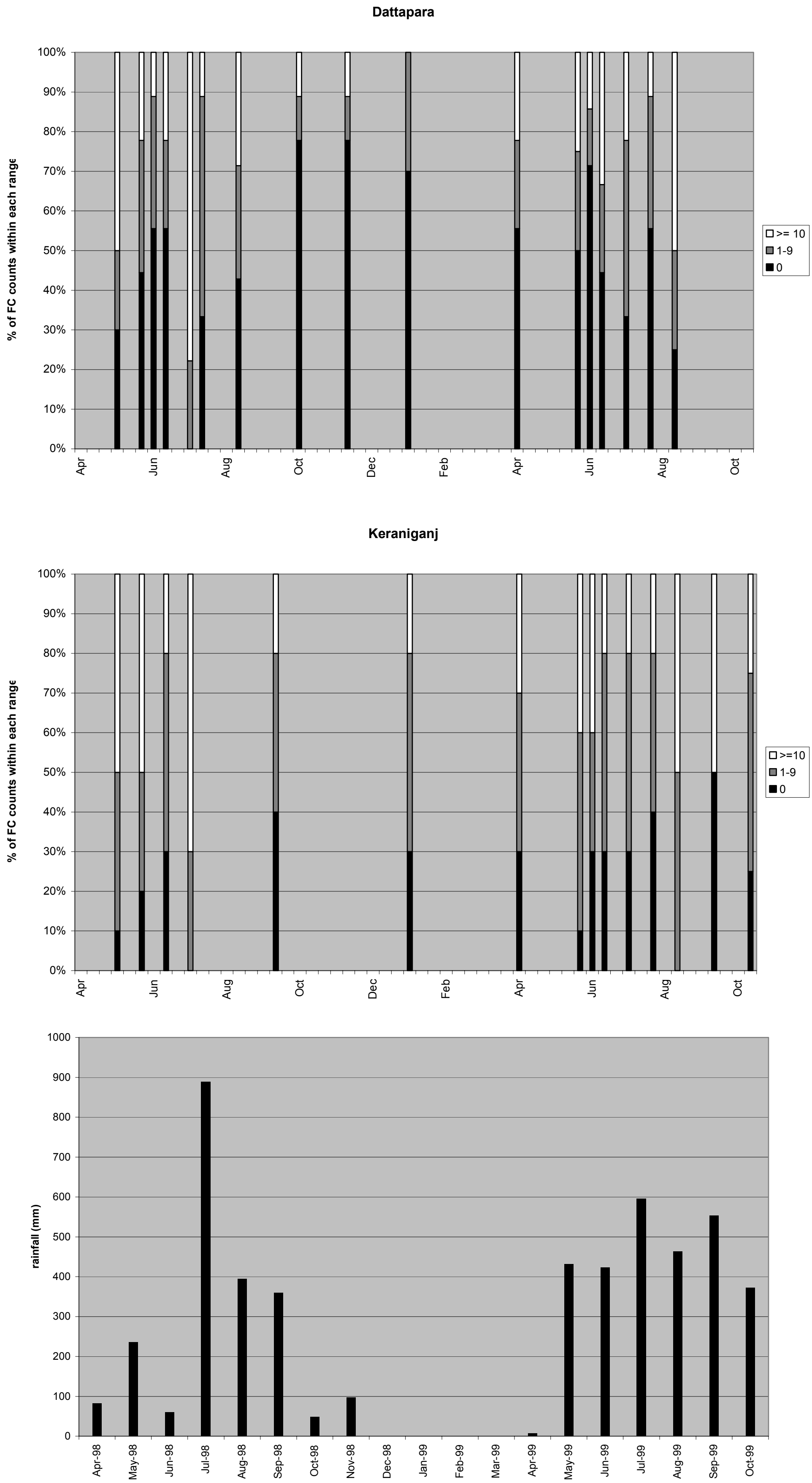


Figure A4.1 Percentage of tubewell samples within ranges 0, 1-9 and ≥10 faecal conforms/100 ml for Dattapara and Keraniganj, and monthly rainfall, for the period of study.

A.4.3 Comparison of microbiological water quality and sanitary inspections

There appears to be no correlation between the overall risk score from the sanitary inspections and the microbiological water quality. To test whether any individual risk factor might have a significant influence on the microbiological water quality the approach for simple frequency analysis of sanitary inspection risk data set-out in ARGOSS (2001) was followed. The key water quality objective has been defined as <1FC/100ml. The procedure assesses the importance of each risk, by categorising well samples for each sample round into those that meet the water quality objective and those that fail. The number of times each individual risk factor was reported from sample sites where the water quality mets the objective and the number of times each risk factor was reported at sample sites where the water quality exceeded the objective target has been counted. Using the total number of samples in the category, the percent of samples in each category when the risk factor was reported to be present has been calculated.

As suggested in ARGOSS (2001), if the frequency of reporting of a particular risk is higher for samples that exceed the water quality objective than for those that meet the objective, this is evidence of a positive association between the risk factor and water quality. This indicates that water quality is likely to be worse when the risk factor is present. The size of the difference between the two groups of samples is likely to reflect the strength of the association.

If the frequency of reporting for the risk is the same between the two groups, or lower for the group which exceeds the water quality objective, it is unlikely that there is a strong positive association between the risk factor and water quality (however, the factor may contribute to overall poor management or be associated with the formation of other risk factors).

It should be noted that unfortunately, due to logistical difficulties, the sanitary inspections were not undertaken at the same time as the microbiological samples were taken. However this does not invalidate the results because many of the factors will not vary significantly. The major difference between the two sanitary inspections, some 15 months apart, was an increase in the numbers of wells that were located within 3 m of a latrine. The analysis of the sanitary inspections and microbiological water quality is presented in Tables A9, A10, A11 and A12.

Table A.9 Dattapara 1998 – sanitary inspection February, microbiological analysis May

Hazard	Number meeting objective where hazard occurs	Number failing objective where hazard occurs	Numbers as percentages		Percentage difference
1	6	1	16	13	-4
2	14	6	38	75	37
3	14	4	38	50	12
4	13	3	35	38	2
5	33	7	89	88	-2
6	13	4	35	50	15
7	32	7	86	88	1
8	11	2	30	25	-5
9	4	2	11	25	14
10	7	3	19	38	19
11	7	3	19	38	19
12	37	8	100	100	0
Numbers of wells where pit latrines occur within specified distance					
< 5 m	9	1	24	13	-12
< 10 m	24	5	65	63	-2
< 15 m	37	8	100	100	0

Table A.10 Dattapara 1999 – sanitary inspection August, microbiological analysis Aug to Oct

Hazard	Number meeting objective where hazard occurs	Number failing objective where hazard occurs	Numbers as percentages		Percentage difference
1	5	11	24	44	20
2	6	14	29	56	27
3	4	13	19	52	33
4	4	9	19	36	17
5	20	23	95	92	-3
6	5	9	24	36	12
7	21	20	100	80	-20
8	3	8	14	32	18
9	2	4	10	16	6
10	1	4	5	16	11
11	1	3	5	12	7
12	20	25	95	100	5
Numbers of wells where pit latrines occur within specified distance					
< 5 m	5	13	24	52	28
< 10 m	15	19	71	76	5
< 15 m	21	25	100	100	0

Table A.11 Keraniganj 1998 – sanitary inspection February, microbiological analysis May.

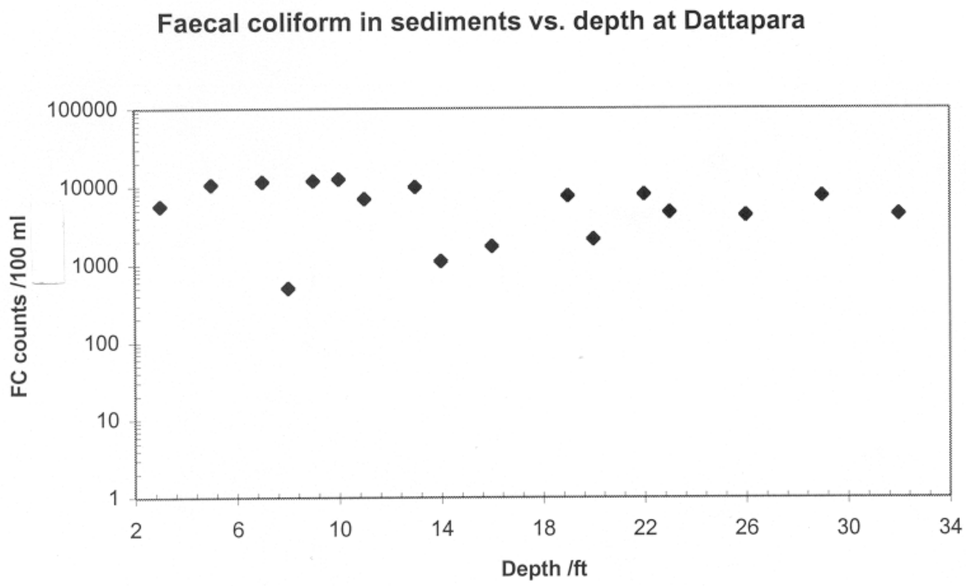
Hazard	Number meeting objective where hazard occurs	Number failing objective where hazard occurs	Numbers as percentages		Percentage difference
1	6	14	30	56	26
2	16	13	80	52	-28
3	12	14	60	56	-4
4	16	12	80	48	-32
5	10	15	50	60	10
6	9	8	45	32	-13
7	14	20	70	80	10
8	7	8	35	32	-3
9	7	6	35	24	-11
10	8	8	40	32	-8
11	8	8	40	32	-8
12	20	23	100	92	-8
Numbers of wells where pit latrines occur within specified distance					
< 5 m	8	14	40	56	16
< 10 m	13	19	65	76	11
< 15 m	17	23	85	92	7

Table A.12 Keraniganj 1999 – sanitary inspection August, microbiological analysis Aug to Oct.

Hazard	Number meeting objective where hazard occurs	Number failing objective where hazard occurs	Numbers as percentages		Percentage difference
1	12	17	75	57	-18
2	8	17	50	57	7
3	6	18	38	60	23
4	8	17	50	57	7
5	11	24	69	80	11
6	2	12	13	40	28
7	15	21	94	70	-24
8	1	11	6	37	30
9	1	11	6	37	30
10	3	14	19	47	28
11	1	9	6	30	24
12	15	29	94	97	3
Numbers of wells where pit latrines occur within specified distance					
< 5 m	13	17	81	57	-25
< 10 m	16	23	100	77	-23
< 15 m	16	29	100	97	-3

A.4.4 Sediment microbiological data

The results of the sampling of sediment during the drilling of the piezometers showed that high numbers of faecal coliforms and streptococci were present down to 10 m (Figure 4.2). This is a surprising result in a fine-grained material as it would be expected that most of the bacteria should be attenuated by this depth. The consistency of the numbers of faecal bacteria with depth suggests that there was no cross-contamination during drilling, sampling or analysis although this remains a possibility. It may be that the bacteria are present at depth but not mobile. Desorbing agents were used to extract the faecal bacteria from the sediment samples and so the analysis accounts for both 'free' faecal coliforms and streptococci and those that have been attenuated by being adsorbed to the sediment grains or filtered. However, it is still surprising to find as many as 2000 FC/100 ml at 10 m depth, whether 'free' or adsorbed/filtered.



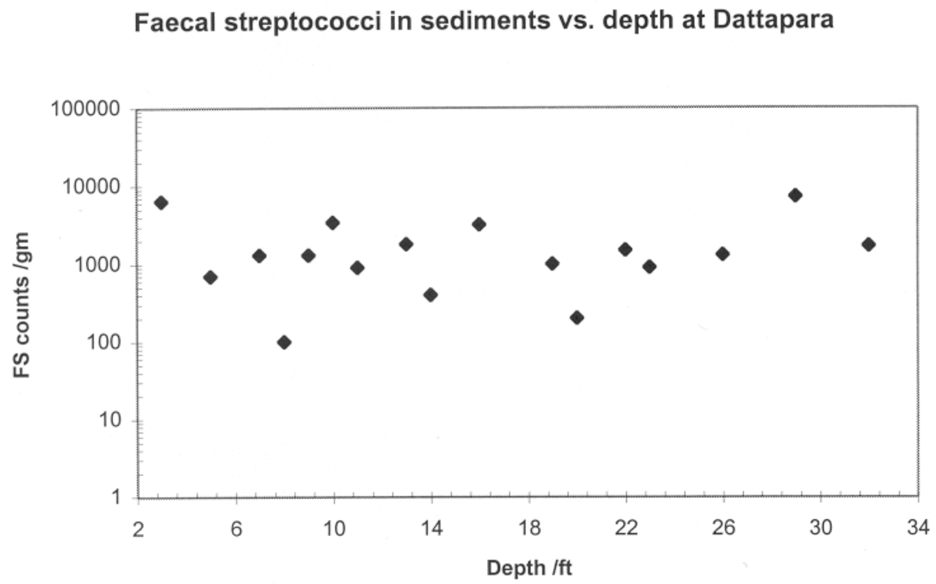


Figure A4.2 Profile through unsaturated zone of faecal coliform/100 ml at Dattapara

A.4.5 Results of hydrochemical water sampling

Presentation and detailed analysis of all the hydrochemical data collected within the Dattapara and Keraniganj study areas can be found in Khandkar (1999). Here the key data are presented providing the evidence of the occurrence of pollution from on-site sanitation.

CHANGE IN HYDROCHEMISTRY WITH DEPTH

Hydrochemical data collected during the project show that the groundwaters in Dattapara and Keraniganj, as across much of Bangladesh (Davies and Exley, 1992), are, in their natural form, of Ca – HCO_3 type. However, water samples show a marked difference in shallow groundwater chemistry in Dattapara. It is thought that the relatively high chloride concentrations here are primarily as a consequence of leachate from pit latrines. Chloride concentrations measured in

February 1998 (which are typical of the chloride concentrations from the whole period of monitoring) are plotted against tubewell depth for Dattapara and Keraniganj in Figure 4.3. The chloride concentrations in tubewells in Keraniganj, which are all greater than 30 m deep, are generally less than 50 mg/l. In Dattapara, there are only three tubewells greater than 50 m depth. The chloride concentrations in two of these tubewells were lower than the majority of the shallow tubewells.

It is not possible to make a comparison of the shallow groundwater quality in Dattapara with that in Keraniganj using the tubewells. However, Figure 4.3 also includes groundwater chloride concentrations measured in the piezometers drilled in Dattapara and Keraniganj during the study (again these are typical of the chloride concentrations from the whole period of monitoring). The chemical data from the single shallow piezometer in Keraniganj indicates the relatively good chemical water quality of the shallow groundwaters and that they differ little from the deep groundwaters. The water quality in the Dattapara piezometers corroborates the findings from the tubewell data, that the shallow groundwaters here are of relatively poor quality compared with the deep groundwaters. The higher population density in Dattapara may provide a explanation for the difference in groundwater chemistry between the two sites.

TEMPORAL CHANGES IN HYDROCHEMISTRY

There is no consistent trend in the concentrations of the contaminants associated with on-site sanitation within the study sites over the period of the project although an upward trend in SEC in Dattapara of approximately 240 μS/cm per year is found in over half of the shallow tubewells. However, there was no increasing trend found in groundwater chloride concentrations during the study.

NITRATE CONCENTRATIONS

Nitrate concentrations in groundwaters sampled in Dattapara and Keraniganj during the project are all well below the WHO guideline of 50 mg/l. All concentrations are below 10 mg/l with only two greater than 5 mg/l and the majority below 1 mg/l. It is thought that oxidation of organic nitrogen in the leachate from pit latrines is incomplete because of the lack of oxygen in the system, a result of a shallow water-table and general waterlogged conditions. Incomplete oxidation of organic nitrogen will produce ammonium which may be volatilised to the atmosphere or sorbed to sediments. Further any nitrate that is leached to the water-table is likely to undergo denitrification due to the reducing conditions. In aerobic environments, nitrate is stable but in anaerobic environments nitrate can be reduced naturally by micro-organisms present in the sub-surface to harmless nitrogen gas.

A.5 DISCUSSION

A.5.1 Microbiological water quality

The WHO recommended guidelines for bacteriological quality of drinking water is <1 FC/100 ml, however in many developing countries water quality is considered to be good, or at least acceptable, if bacteriological quality is <10 FC/100 ml. In this study, the quality of tubewell water in both Dattapara and Keraniganj is generally good (<10 FC/100 ml) although sporadic highs were observed in most boreholes. Some of the very high faecal coliform numbers detected (>500 FC/100 ml), especially in the first round of sampling, were attributed to the use of poor quality water for priming the handpump. The sporadic nature of the contamination both spatially and temporally suggests that contamination is not widespread and that it may be a localised problem and a consequence of inadequate well-head protection measures. Further evidence for this is provided, especially in Keraniganj, by the long travel times for water to migrate through the aquifer from the water table to the tubewell screen which should ensure that faecal coliform are reduced to low numbers (<1 FC/100 ml).

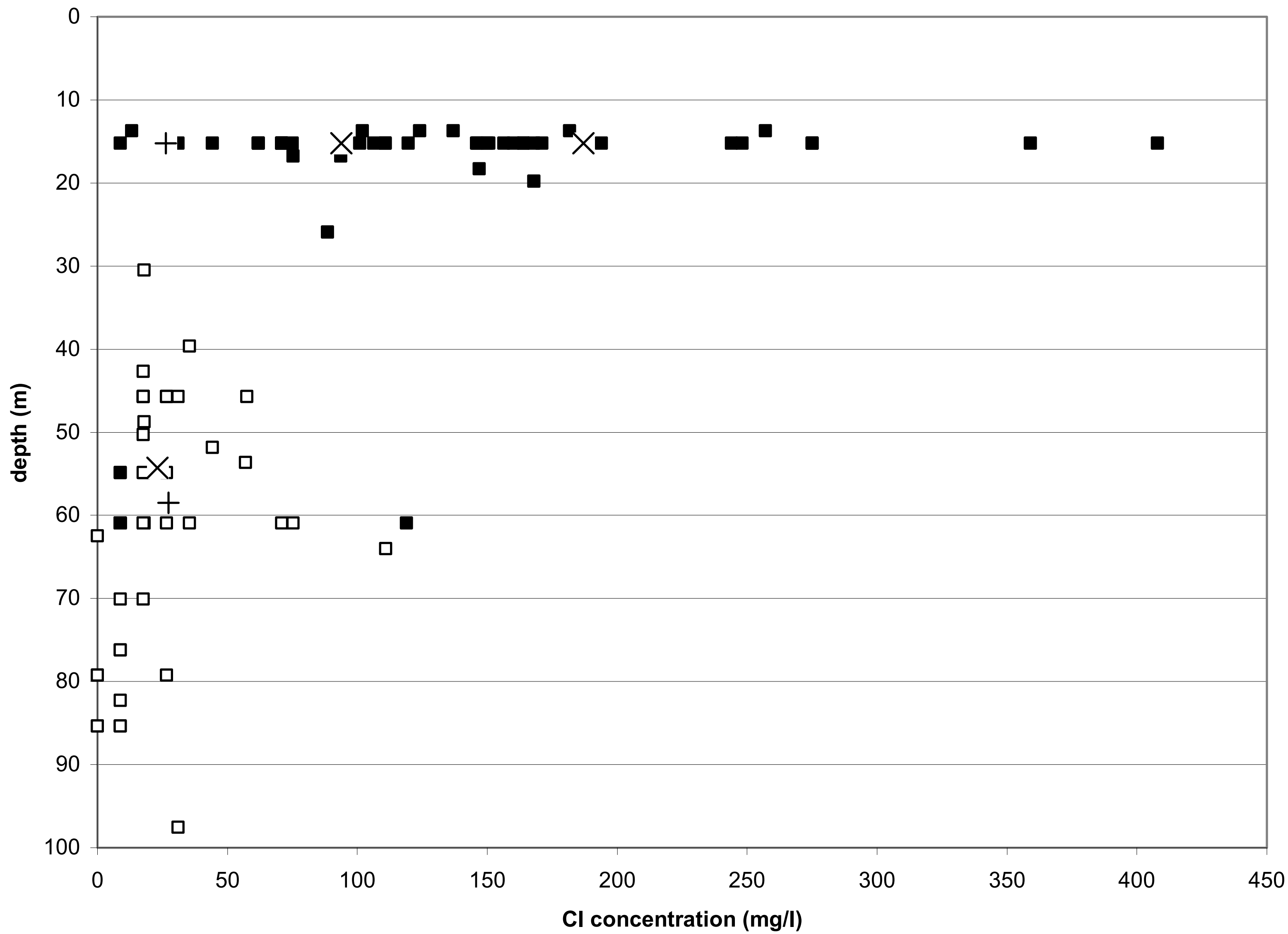


Figure A4.3 Groundwater chloride concentration for February 1998 against depth of case study tubewells and piezometers for Dattapara and Keraniganj

The results of sanitary surveys show a significant percentage of tubewells can be classed as moderate-high sanitary hazard (Section 4.1). However, no consistent relationship between individual hazards and water quality over different seasons was apparent for Dattapara or Keraniganj. For example, the presence of Hazard 1 (a pit latrine within 3 metres) appeared to correlate with poor quality in Keraniganj for May 1998 (difference +26) but not in August 1999 (difference -18). Likewise in Dattapara there was a negative correlation in May 1998 (-4) but a positive one in August 1999 (+20). Thus whilst improvements could be made in both the design and maintenance of the well-head completion and in keeping potential sources of contamination away from the well, no single hazard was identified as the major contributing factor to the sporadic incidents of poor quality water. This leads to find out probable pathways for contaminants (particularly micro-organisms) migration to hand tubewells from pit latrines. There are five possible pathways through which contaminants can move to tubewells.

Pathway 1: Downward percolation through aquifer/aquitard

Contamination of tubewell by downward percolation of micro-organisms from pit latrines to tubewells through fine-grained silty clay layer is very unlikely. This is because predicted travel times from the water table to the screen exceed 50 days and thus the risk of micro-organisms reaching the tubewell screen is very low. However, the results of soil sampling (Section 4.4) to 10 m depth at Dattapara appears to contradict this view, as a depth of 10 m in the silty clay layer beneath Dattapara probably represents about 1-2 years travel time (assumes a recharge of 250-500 mm/a, an effective porosity of 5% and simple piston displacement). One explanation why Faecal Coliforms and Faecal Streptococci are observed as deep as 10 m (with apparently no attenuation) is that the Faecal Coliforms are sorbed to the sediment (and therefore are not mobile). This would increase their survival rate.

Pathway 2: Lateral spreading at water table and seepage through defective casing

A more likely contaminant pathway to the tubewell (for micro-organisms) is the ingress of water (containing high numbers of bacteria) close to the water table through a defective casing joint and pit latrines could be a source of leachate at shallow depths, particularly where the unsaturated zone beneath the latrine is thin (Macdonald et al, 1999). The joints of the PVC casing are glued, and openings may occur either where the solvent based cement deteriorates with age (or where incompletely applied) or where the casing cracks due to excessive pressure when the casing is lowered into the ground. The quality of the casing and the solvent cement is variable and people may try to reduce costs by using cheaper (and probably inferior) materials.

Pathway 3: Percolation behind casing

Another pathway is percolation of shallow contaminated water down the back of the casing where either the formation has not collapsed fully around the casing, following completion, or where more permeable sand has been used to backfill the annulus between the formation and the casing. This pathway is less likely where a significant thickness of fine silty sand occurs as this is liable to collapse around the casing especially below the water table. Further, it might be anticipated that if this was the most important pathway then the deeper tubewells should be generally less contaminated. However, there was no inverse correlation between numbers of faecal coliforms and tubewell depth.

Pathway 4: Seepage from surface through cracked (or missing) platform then as (2) or (3) above

Polluted water at the surface can seep through or beneath the cracked platform surrounding the tubewell and enter the tubewell, either through eroded or defective casing pipe or by percolation down the outside of the casing to the screen. This is a possible pathway but its importance as a pathway is uncertain because no correlation between sanitary hazards and Faecal Coliforms is observed. Pathways and their probability for tubewell contamination are summarised in Table A.13. It is likely, however, that this is the cause of sporadic poor quality water observed in both Dattapara and Keraniganj, densely populated areas with a shallow water table that is contaminated.

Table A.13 Pathways and their probability for tubewell contamination.

Pathways	Probability	Reasons for probability
(1) Downward percolation through sediments to screen. Evidence: soil sampling indicates no reduction inb FCs to 10 m soil depth.	Unlikely pathway but evidence is conflicting and needs to be resolved.	• sediments have significant capacity to attenuate micro-organisms and travel time likely to exceed time for FCs to be attenuated to 'safe' levels (10-50 days) • no inverse correlation of FCs in water with depth of tubewells.
(2) Seepage through defective casing joints at shallow depth - PVC joints are glued and may crack when installing - GI pipe (near surface) may corrode especially as shallow groundwater may be quite corrosive.	Probable that pathway is responsible for some contamination.	• shallow water table would increase this risk • increase FCs observed in tubewells where water filled ditch was close by (may indicate shallow water table) • slightly higher in FCs observed in deeper tubewells in Keraniganj compared to Dattapara may be due to shallow water table.
(3) Percolation behind casing to screen. Evidence: - annulus behind casing not sealed with cement - annulus may be backfilled with more permeable sand on completion of tubewell.	Pathway is possible and may be responsible for some contamination.	• no inverse correlation of FCs with depth of tubewells. (This might be expected as the deeper tubewells the annulus less likely to be "open" • below water table sand is likely to collapse around casing.
(4) Surface water seeps through cracked (or missing) platform then as (2) or (3) above. Evidence: contaminated water at ground surface may seep in and around tubewell where platform is absent or cracked.	Pathway is possible but its importance as a pathway is uncertain.	• no correlation between sanitary hazards and FCs.
(5) Prime pump with dirty water Evidence: know that many (up to 20%) of tubewells need priming - non-potable water is sometimes used	Pathway is certain. It is probably that this pathway is responsible for some contamination.	• any water used to prime a tubewell is likely to have some FCs (because of handling etc|).

Pathway 5: Priming pump with dirty water

Priming of the pump is not uncommon in Dattapara and Keraniganj and provides an obvious pathway for contamination. A survey of 2000 tubewells in Bangladesh concluded that up to 20% of all tubewells are primed (Hoque, 1998). Even when good quality tubewell water is used to prime the pump the opportunity for introducing contamination must be high.

Although, as mentioned earlier, poor quality water (>10 FC/100 ml) occurs sporadically in most tubewells some trends can be discerned. Firstly, water quality is generally better in the dry season than the wet (Section 4.2) but correlation is low. The reason for the seasonal variation may be (a) runoff picks up contamination at the surface (from animal excreta, waste dumps etc) and seeps into the subsurface around the well (because of poor sanitary well head completions or (b) the water table is at shallower depth and is thus more prone to contamination.

Secondly, water quality is generally better in Dattapara than Keraniganj although the correlation is weak; the main differences between Dattapara and Keraniganj are that the geological succession beneath Keraniganj is more permeable (fine silt/sand compared to silty-clay) and the water table is shallower. Thus, the water table beneath Keraniganj may be more susceptible to contamination. These correlations (between bacteriological water quality and both season and location) albeit that they are rather weak, provide some additional evidence that the mechanism for periodic contamination of the tubewell may be the entry of poor quality water through defective casing (or casing joint) at or near the water table.

A.5.2 Chemical water quality

Chemical water quality for both Dattapara and Keraniganj is generally good, at least for the parameters measured; nitrate concentrations are low despite the high loading of nitrogen from the pit latrines estimated at 3000 kgN/ha/annum. Reasons for the low nitrate concentrations are probably twofold:

(i) low rates of nitrification of the organic nitrogen in pit latrine probably due to lack of oxygen in the poorly permeable clay soils

(ii) denitrification in the shallow groundwater.

In aerobic groundwater nitrate is stable, however where oxygen is depleted, nitrate can be reduced to nitrogen or nitrous oxide gas, a process known as denitrification. The dissolved oxygen content of groundwater can be readily depleted where oxidation reactions occur. A shallow water table, such that aerobic degradation of the organics present in the urban infiltration is incomplete within the unsaturated zone is probably the most important environmental factor that contributes to denitrification (Lawrence et al, 1997). The dissolved oxygen and Eh of the groundwater beneath Dattapara were 1% (of saturation) and -250 mV respectively; conditions amenable for active denitrification.

Ammonium concentrations in groundwater can be relatively high which is indicative of incomplete oxidation of organic nitrogen.

Chloride concentrations can be relatively high (Section 4.5) especially in shallow groundwater and a reducing trend with depth is observed. The origin of the chloride is almost certainly a consequence of pit latrine leachate. In the case of Dattapara with a population density of 624 per ha, the chloride loading is estimated at 1250 kg/ha/annum. The chloride concentration in the leachate from the pit latrines could be more than 400 mg/l, assuming an annual infiltration of 300 mm. It is thought that a front of modern (high chloride) water will migrate downwards in response to pumping from deeper tubewells resulting in increasing chloride concentrations in time at depth.

A.6 CONCLUSIONS

1. Bacteriological water quality is generally acceptable (<10 FC/100 ml) although sporadic highs do occur throughout the year but more especially in the wet season.
2. Given the significant depth of the tubewells, particularly in Keraniganj, the low permeability of the shallow sediments and the sporadic nature of the contamination, the pathway of migration for the faecal coliform bacteria to the tubewell is thought to be localised – that is due to the design and construction of the water supply. Such pathways provide a route for small quantities of water to rapidly bypass the aquifer.

3. Well completions are poor and the sanitary surveys showed that many tubewells have moderate-high sanitary hazard score. However, no single hazard factor could be identified from the sanitary survey as being the principal cause of well contamination.
4. It is likely though that the lack of cement seal around the outside of the tubewell casing is a major contributing factor. All tubewells lack this cement seal and therefore the outside of the tubewell casing could be in direct contact with water of poor bacteriological quality, especially at or near the water table. Thus should the casing have a crack or the casing joint be non-watertight then this poor quality water could be drawn into the tubewell as a result of the difference in pressure between inside and outside of the casing during pumping. This is thought to be the main cause of contamination.
5. Despite the very high nitrogen loadings in the pit latrines, nitrate concentrations in the tubewells are surprisingly low and reflect the lack of sufficient oxygen in the subsurface to oxidise much of the available organic nitrogen. Given the low concentrations of both nitrate and ammonium in groundwater, it is probable that the bulk of the nitrogen loading in the pit latrines is 'lost' by volatilisation of ammonium within the unsaturated zone. Further, any nitrate that is leached to the water-table is likely to be reduced to nitrogen gas (or to an oxide of nitrogen) by denitrifying bacteria.
6. Chloride concentrations are relatively high, especially at shallow depths, and are a consequence of the high chloride loading in the pit latrines. Chloride concentrations at depth are likely to increase with time as the front of modern (high chloride) migrates downward in response to deep pumping.

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Appendix B
Impact of on-site sanitation on groundwater supplies in Kampala and Iganga, Uganda

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B1. INTRODUCTION

This section provides a detailed description of the methodology and findings of the case study work performed in Uganda as part of the ARGOSS project (1997-2001). This component of the project was performed by the Robens Centre for Public and Environmental Health, University of Surrey and the Public Health and Environmental Engineering Laboratory, Department of Civil Engineering, Makerere University.

The aim of the study was to evaluate the risk to groundwater quality in two major shallow groundwater source types in Uganda. The objectives were as follows:

1. To contribute to the ARGOSS manual on the risk to groundwater from on-site sanitation
2. To evaluate the trends in water quality in shallow groundwater in Uganda
3. To assess the importance of on-site sanitation and other risk factors in causing groundwater contamination
4. To develop a statistical model of contamination as a means to understand the causes of contamination

B2. METHODOLOGY

The Uganda study was performed in two urban areas of Uganda: the capital city Kampala and a smaller town, Iganga. These urban areas were located in different parts of the country, have very different demographic characteristics and have a different hydrogeological regime.

B2.1 Site descriptions

Kampala is the Capital city of Uganda and is experiencing rapid growth, through in-migration and natural increase. Population increases during the day are significant and may contribute a further 300,000 to 400,000 people. The City is spread over 160 km^2 and is split into 5 Divisions and 98 Parishes. There is a wide range of socio-economic conditions within the City, but a significant proportion of the population is poor (Howard and Luyima, 1999).

A recent pilot surveillance programme established that were at least 225 protected springs in the city, with a further 75 unprotected springs (Howard and Luyima, 1998). Although there is a well-developed primary and secondary piped water infrastructure throughout the City, the numbers of people with at least one household tap is limited (Howard and Luyima, 1999). The remaining population obtain water from neighbours with a tap, from public taps and from protected springs (Howard et al, 1999). It has been estimated that over 60% of the low-income population with access to protected springs within a Parish use these for part or all their domestic water needs, including drinking and cooking (Howard et al, *in press*).

Some of the protected springs in Kampala have been in existence for many years (some since at least the colonial period) and a number were rehabilitated following the Amin and Obote years that led to an almost total collapse in water supply infrastructure. The springs are built using a simple design transferred from rural areas. The quality of these springs is of concern and there is evidence of a link between contamination during the cholera epidemic in 1997/8 and the use of protected springs (Howard and Bartram, in preparation; Nasinyama et al, 2000). Sanitation was also poor in the city particularly in low-income areas at the time of the outbreak (Government of Uganda, 1997).

Iganga is located in eastern Uganda and lies on the main road linking Kampala and Jinja to the Kenyan border. The town is relatively small with a population of under 50,000 but is increasing in size given its advantageous position on the major trade route between Uganda and Kenya. Iganga was selected as it provided a very different setting to Kampala in terms of its size and density of population, topography and the main water sources available.

The town had been provided with a piped water supply fed by mechanised deep boreholes, but the system was not reliable or entirely comprehensive. There were a large number of shallow boreholes fitted with handpumps in the town, with a smaller number of dug wells and protected springs, which the population relied upon.

B2.2 Study areas and sampling methods

Within Kampala, two study areas were selected based on population density. A total of 15 protected springs were selected in high-density settlements and 10 protected springs in low-density areas. Sampling was undertaken on a monthly basis between March 1998 and April 1999, with a smaller set of springs from each of the two study areas visited on a weekly basis at the start of the wet season in September 1998 and March 1999. In Iganga samples were taken from 13 boreholes fitted with handpumps, two dug wells both fitted with handpumps and one protected spring. Only monthly samples were taken given the lack of any indication of microbiological water quality change and the distance from Kampala.

B2.3 Hydrogeology of the study sites

As in much of sub-Saharan Africa, and many other tropical regions, the main aquifer type in Uganda is weathered crystalline rock. In these environments, groundwater is transmitted by fractures in the bedrock and unconsolidated weathered materials that form the overlying mantle, or regolith (Figure B1). Permeability within the bedrock and overlying weathered mantle derives from the prolonged, *in situ* decomposition of bedrock (McFarlane, 1991; Nahon and Tardy, 1992; Thomas, 1994; Taylor and Howard, 1998a). As a result, weathered aquifer systems occur in areas where weathering has been undisturbed by Pleistocene glaciation or by significant aeolian erosion. In sub-Saharan Africa, the bedrock comprises primarily Precambrian ‘basement’ rocks that include granite and gneiss (Key, 1992; Houston, 1995). Younger sedimentary terrains featuring limestone and sandstone occur in central and southern Africa as well as coastal regions of West Africa.

Due to a primarily *in situ* development, mantles of weathered rock constitute the progressive degradation of bedrock materials and therefore produce highly anisotropic aquifers. The thickness of the mantle is regularly in the order of tens of metres and depends upon a wide variety of factors which include tectonic setting (which influences the duration of weathering), bedrock lithology, and climate (Ollier, 1984; Wright, 1992; Thomas, 1994). The weathering dynamics in the unsaturated zone of weathered mantles remain poorly understood. However, work by Taylor and Howard (1999b) reveals two key characteristics about weathered mantles:

- The texture of the weathered mantle features a bimodal particle-size distribution as sand-sized, primary minerals are progressively weathered to clay-sized, secondary minerals at shallower depths. The poorly sorted texture of the weathered mantle has important implications: localised variations in the parent-rock matrix give rise to high degree of spatial heterogeneity in weathered mantle lithology; and, as recognised by McFarlane (1992), secondary structures of the bedrock such as quartz stringers are translated into the composition of the weathered mantle producing preferential pathways for subsurface flow and contaminant transport. In addition, in common with many shallow aquifers of an unconsolidated nature, the development of root channels also result in preferential flow paths.
- Less aggressive weathering is associated with saturated conditions and the persistence of coarse-grained materials at the base of the weathered mantle. Lithological studies in Malawi (McFarlane, 1992) and Malaysia (Eswaran and Bin, 1978) support this observation and indicate similarly that the aquifer in the weathered mantle comprises a poorly sorted, muddy-sand.

Fractures in the underlying crystalline bedrock (Figure B1) tend to be both subhorizontal and discontinuous. Groundwater exploration strategies in crystalline terrains commonly attempt to locate regional and hence, more continuous, fracture systems in the bedrock. Packer testing (Howard et al., 1992; Taylor and Howard, 1999a) and detailed geophysical studies (Houston and Lewis, 1988) indicate that fracture density generally increases towards the surface. This is consistent with the suggestion of other authors (Davis and Turk, 1964; Acworth, 1987; Wright, 1992) that fracturing results from decompression (i.e. sheeting) resulting from the removal of overlying rock in solution and erosion of pre-weathered (unconsolidated) material at the surface. Double (outflow) packer testing of fracture zones in the bedrock shows that transmissivities are typically less than $1m^{2/}d$ (Figure B2) and that one or two highly productive zones account for the bulk transmissivity of the bedrock

Figure B2 demonstrates the difference between typical aquifer properties in areas of low and high relief (deep weathering and land surface stripping). Essentially, increased run-off (and hence erosion) in areas of high relief prevents the formation and accumulation of extensive and thick weathered mantles.

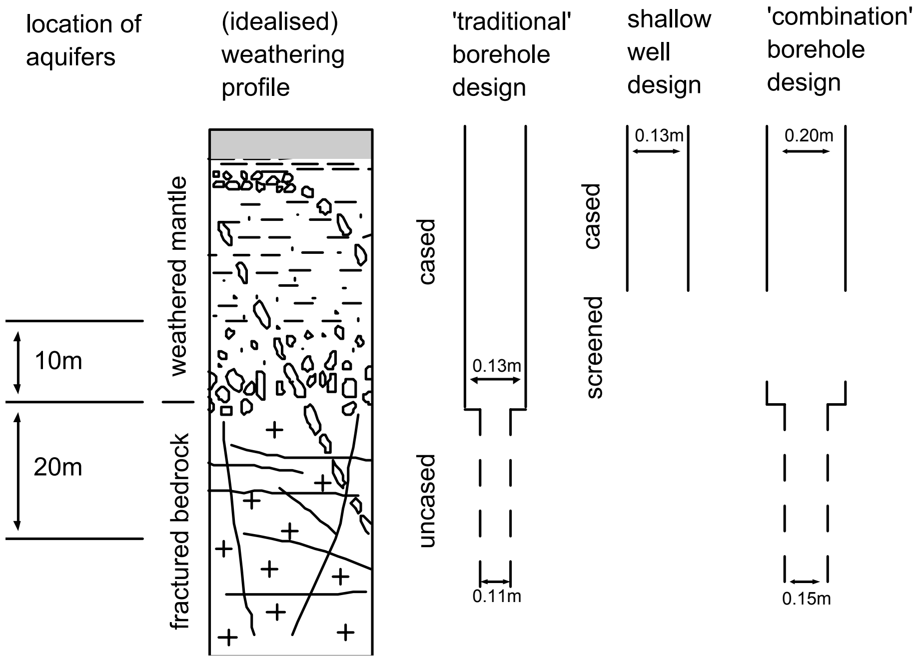

Figure B1. Idealised cross section through the Iganga aquifer

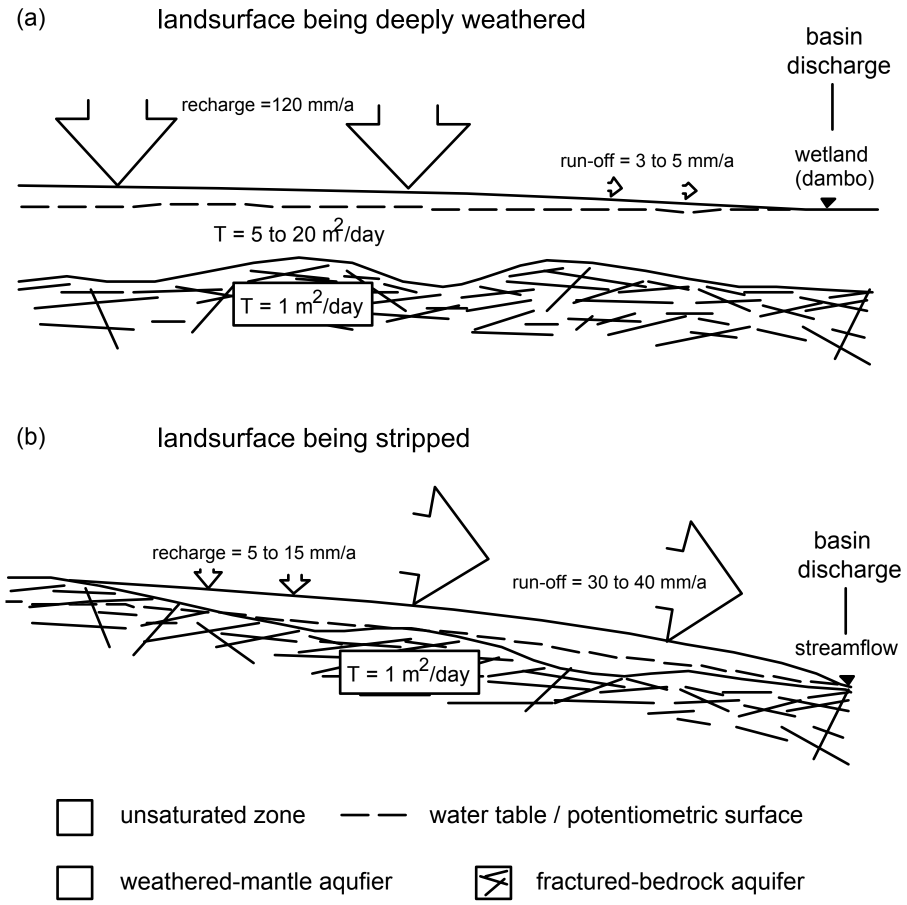

Figure B2. A comparison of the groundwater flow systems in areas of low and high relief

Kampala and Iganga are in areas of contrasting weathered zone development. Kampala is built on a number of hills, the pronounced topography being controlled by differential weathering of various grades of meta-sedimentary bedrock types. The topography results in thin weathered mantles of limited extent (a stripping environment), which produces shallow, localised groundwater flow systems with relatively short residence times. These systems discharge to springs that form at geological boundaries on the lower slopes of hills and at the edges of dambos (lowland swampy areas).

Kampala has a shallow aquifer from the weathered regolith and site investigations showed the presence of preferential flow paths, although it is not known how far these extend. Recharge in Kampala is not well-understood in part as rainfall occurs throughout the year, but is likely to primarily occur during the two periods of heavier rainfall (March to May and September to November).

By contrast the bedrock beneath Iganga comprises predominantly very high-grade gneisses and granites. Prolonged deep weathering has resulted in a relatively thick weathered mantle and a landscape of low relief. The aquifer is considered to be of regional extent, with a high residence time. Figure B1 shows an idealised profile through the Iganga aquifer. Recharge in Iganga is likely to primarily follow the seasonal rains.

In Kampala, low to middle-income communities often obtain groundwater untreated from protected springs. The design of these springs is very simple. An excavation is made to intersect with the eye of the spring and an area between the eye and proposed retaining wall is filled with permeable material such as hard core. The backfill is capped with protective material (plastic sheeting and clay) and grass planted on top. A diversion drainage ditch is constructed to take uphill surface runoff away from the spring. One or more galvanised steel pipes with plastic liners act as discharge points from the retaining wall.

In Iganga, water is obtained from shallow tubewells. These penetrate the full thickness of weathered overburden, and are screened in the high permeability zone above the intact bedrock. The depth to the intake screen is largely unknown given the lack of archived information, but is generally in the region of 10-20 metres.

B2.4 Water quality parameters used in the study

The study tested for a range of pollution indicators in the shallow groundwater. These included microbiological, chemical and physical parameters. Samples were taken from the commonly used outlets of the groundwater sources. Samples were collected in sterilised bottles, which were rinsed three times with water from the source prior to collecting the sample. For boreholes and dug wells, the spout of the handpump was disinfected by flaming prior to sampling. All samples were stored in a cold box kept below $4^{o}C$ and analysis was performed within 6 hours of sampling.

The microbiological quality was assessed using a variety of indicator bacteria. There are inherent weaknesses in this approach as the value of indicator bacteria in predicting the presence of protozoa and viruses is questionable (Ashbolt et al, 2001). However, they are reasonable indicators of bacterial pathogens and more generally they are a reasonable index of recent faecal contamination in water.

Although no testing was undertaken for bacteriophages, which are increasingly accepted as indicators of virus presence (Grabow, 2001), the discussion section highlights data from a tracer study in Iganga and other work performed in Uganda. Protozoan analysis was not performed, given the difficulties of analysis and lack of available capacity in Uganda.

B2.5 Indicator bacteria

The principal indicator bacteria used in the study were thermotolerant coliforms and faecal streptococci. Thermotolerant coliforms are an accepted surrogate for *E. coli*, the principal indicator of microbiological quality currently used in drinking water supplies (WHO, 1985, 1993). In temperate climates, it is estimated that 95% or more of thermotolerant coliforms are *E. coli*. However, there is research that suggests that in tropical waters a lower proportion of thermotolerant coliforms are *E. coli* and that there are other thermotolerant coliform species with an environmental source (Hazen and Torranos, 1990). Some research has suggested that environmental sources of *E. coli* exist in tropical areas, as they have been isolated in pristine water environments (Hazen and Torranos, 1990). However, this research appears to have ignored rodent and other animal sources and in Uganda such an assertion is difficult to support in the study areas given the wide array of sources of faeces in the environment and obvious pathways into the water sources.

Faecal streptococci are accepted as a possible alternative indicator to coliforms. Previous research in drinking water has shown that faecal streptococci have a stronger relationship to diarrhoeal disease than *E. coli* (Moe et al, 1991) and are the indicator of preference for recreational water (Bartram and Rees, 2000). Faecal streptococci have greater persistence in water and do not multiply in polluted environments (Ashbolt et al, 2001). As a result their use has been recommended for testing whether or not groundwater has received contaminated recharge (WHO, 1993; WHO, 1996).

Analysis of the thermotolerant coliforms was carried out using the membrane filtration technique in the Public Health and Environmental Engineering Laboratory. The water was filtered through 0.45μm Gelman nitro-cellulose filter papers and then placed on a pad soaked in Difco membrane lauryl sulphate broth (MLSB). The plates were incubated for one hour at ambient temperature (approximately 25°C), to aid bacterial resuscitation, before transferring to 44°C +/- 0.5°C for a further 23 hours incubation. Yellow colonies were counted as being thermotolerant coliforms.

Faecal streptococci were isolated and enumerated by membrane filtration and growth on Membrane Enterococcus Agar (Slanetz and Barltey Agar, Oxoid Ltd). Measured volumes of water were filtered through 0.45μm Gelman nitro-cellulose filter papers. The filters were placed on membrane enterococcus agar and preincubated at ambient temperature for one hour to aid bacterial resuscitation. The plates were then incubated at 44°C +/- 0.5°C for a further 47 hours. After incubation all red, maroon and pink colonies that were smooth and convex were counted and recorded as faecal streptococci.

B2.6 Sanitary inspections and hazard assessments

At each site a sanitary inspection was performed when the sample was taken. The sanitary inspections followed a standardised format and were adapted from those used elsewhere in urban areas of Uganda and reported in the literature (Howard, 2001; WHO, 1997). The inspection form excluded questions on latrines, which were assessed through a separate exercise. The data was stored in a dedicated sanitary inspection and water quality database called ‘Sanman’ (Howard, 2001).

B2.7 Chemical and physico-chemical parameters

Samples were taken from each site and tested for nitrate and chloride. A limited number of analyses for ammonia and phosphate were also carried out. Samples were taken in sterile plastic one-litre bottles, which were rinsed three times in sample water before filling. Bottles were filled to the brim to reduce de-gassing. Analysis was performed using a HACH DREL 2000 spectrophotometer and a Palintest Photometer 3000.

Physico-chemical parameters (electric conductivity, pH and temperature) were measured on-site using Hanna Instruments waterproof meters and probes inserted into a flow cell. Water was allowed to flow through the cell until the readings for electric conductivity stabilised before readings were taken for all other parameters.

B3. RESULTS AND DISCUSSION

B3.1 Microbial contamination in Kampala

The data were analysed to assess trends in the microbiological quality of water related to various factors in order to be able to develop a fuller understanding about the causes of contamination. This looked at both potential sources of faeces within the environment (hazard factors) as well as a range of potential routes for microbiological entry into the water supply. The relative importance of different factors in causing contamination was tested. Statistical analysis was performed using SPSS.

In addition to investigating the factors that appear to be most associated with contamination being present, their influence upon the presence of different bacteria was also evaluated. The microbiological data exhibited a log-normal distribution, although the model fits the thermotolerant coliform data more reliably than the faecal streptococci. Statistical analysis was performed using appropriate non-parametric tests as the sanitary inspection data was categorical and the raw microbiological data non-normal. However, this is accepted practice when analysing water resources data and the tests performed show no loss of power (Helsel and Hirsch, 1992).

The analysis of the data looked at several relationships that were suspected as being important and then drawing on the results of these analyses, attempted to produce statistical models for contamination of the protected springs.

B3.1.1 *Relationship of contamination to population density*

The areas around each spring were divided in high and low density based on a qualitative assessment of density in the immediate vicinity of the springs. The relationship to population density was suspected as being important in determining contamination. In more densely populated areas, the loading of faecal material is expected to be higher. This loading will result irrespective of whether excreta are disposed of through sewers, on-site sanitation or is discarded on the surface.

There was a significant differences in microbiological quality between high and low-density areas. Mann-Whitney U tests were performed on this data to assess whether there was a difference in average contamination between the two population density groups. This showed a highly significant difference between the two groups for both median thermotolerant coliforms ($U = 13$, $p = 0.001$) and faecal streptococci ($U = 13$, $p<0.001$).

There is potentially confounding of this data in that the high-density areas were, in general, of lower socio-economic status than the low-density areas as determined by a socio-economic index reported elsewhere (Howard, 2001). It was believed possible that as a result, these areas had lower rates of sanitation coverage. As no detailed estimates exist for sanitation coverage in Kampala this is difficult to assess. However, the latrine assessment did not indicate significantly larger numbers of sanitation facilities in low-density areas and it is therefore considered unlikely that socio-economic status caused major confounding.

B3.1.2 *Relationship to rainfall*

Rainfall data from the Makerere University meteorological station were collected from the Department of Meteorology. Previous studies have shown that the quality of water is often dependent on rainfall (Wright, 1986; Gelinas et al, 1997) and therefore the data were analysed to assess whether rainfall exerted a significant control over microbiological contamination. The relationship to rainfall initially appears largely straight forward, with the monthly median number of thermotolerant coliforms and faecal streptococci from all sites in Kampala showing a broadly similar pattern as the monthly rainfall as shown in Figure B3. However, with further investigation, the exact nature of the relationship is more complex.

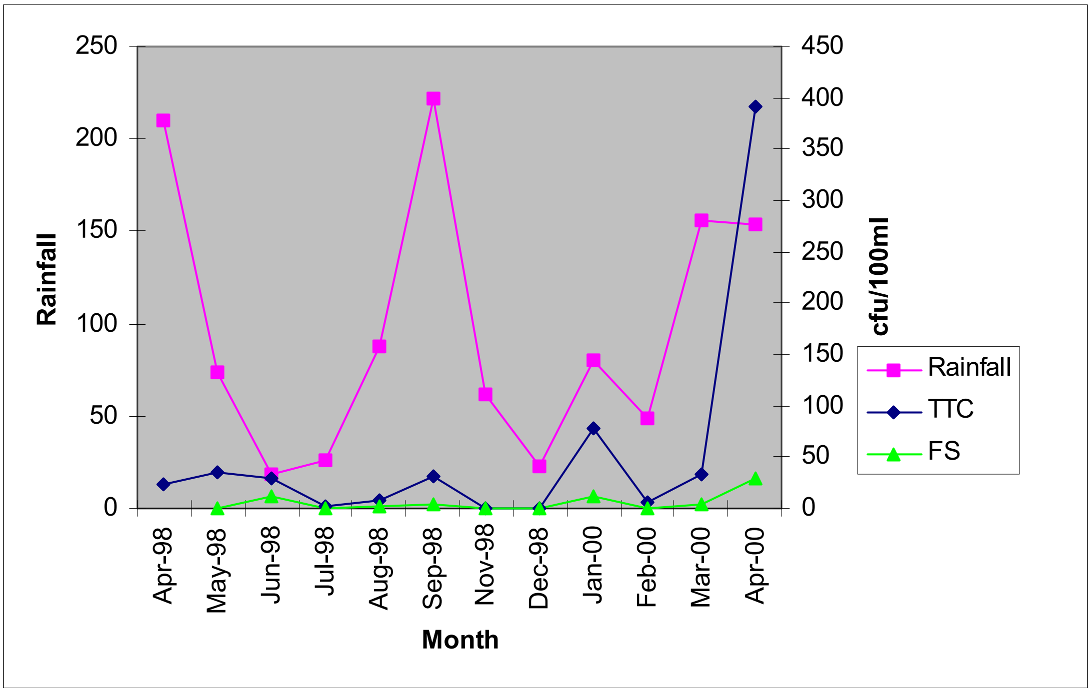

Figure B3 Rainfall and microbiological quality, Kampala

The initial statistical analysis looked at relationships between the median contamination from each day's sampling against monthly rainfall and rainfall amount in the 24, 48 and 120 hours prior to sampling. For the thermotolerant coliforms, a strong correlation is seen between median contamination and rainfall over previous 24 hours ($r = 0.920$, $p = 0.01$) and over the previous 48 hours ($r = 0.785$, $p = 0.01$). The correlation with rainfall in the previous 120 hours ($r = 0.514$, $p = 0.05$) and monthly total rainfall ($r = 0.616$, $p = 0.05$) are less significant. The stronger relationships between average contamination and rainfall over relatively short time periods suggest that there is likely to be rapid recharge into the springs and that this exerts a significant influence of over contamination. The still significant relationship with 5 day and monthly total rainfall suggests that there may also be some a residual contaminated recharge derived from sub-surface leaching, although this may be from a pit latrine, solid waste dump or contaminated surface water.

Median faecal streptococci density also showed a highly significant correlation with rainfall in the previous 24 hours ($r = 0.989$, $p = 0.01$) and rainfall over the previous 48 hours ($r = 0.960$, $p = 0.01$) again were strong. There was no significant correlation with monthly rainfall ($r = 0.333$, $p = 0.317$).

These results suggest that short-term rainfall is a good predictor of the degree of contamination and give further weight to the importance of rapid recharge by contaminated surface water as the major cause of microbiological deterioration rather than sub-surface leaching from contaminant sources such as pit latrines. If the latter were a major cause of contamination, the numbers of faecal streptococci would be expected to be sustained for longer periods of time given their environmental resistance.

B3.1.3 *Sanitary risks*

Assessment was made of the sanitary integrity of the protected springs and possible sources of faecal pollution in the environment in order to evaluate which factors were most likely to be influencing microbiological quality. The sanitary inspection used eight factors dealing with potential sources of pollution, potential pathways for pollutant entry into the water sources and general factors believed to contribute to the development of pathways. In addition to sanitary inspection, an assessment was done of latrine and solid waste proximity to the springs.

The data from the sanitary inspections provide a good overall estimate of operation and maintenance of the protected springs (Howard and Luyima, 1998). The average level of risk noted in high-density areas was significant higher than in low-density areas (Mann-Whitney U Test: $U = 27.5$, with $p = 0.001$). This suggests that although population density may be associated with significantly higher faecal loading in the environment, the risks arising from poor operation and maintenance are also significantly higher in these areas compared to low-density areas. This suggests that the higher microbiological loads in shallow groundwater in areas of greater population density may arise from surface sources due to poorly disposed of faecal matter able to recharge the springs rather than sub-surface leaching from pit latrines.

The median contamination and median sanitary risk scores were analysed as there are broad relationships previously reported in the literature that suggest sanitary risk score is a useful predictor of level of contamination (Lloyd and Bartram, 1991). In this assessment, the median value was used as the microbiological data included results that had levels of contamination that could not be counted. These were given default values of 30,000 cfu/100ml, which was chosen because this was above the highest level of contamination directly recorded from colony counts at the highest dilution of the sample tested. However, as this figure is not a directly observed reading, its use would have distorted the mean value derived. Furthermore, there were a small number of very high actual counts recorded that exert significant influence on the mean. Exclusion of these outliers was considered, but rejected as it was felt that they were important to include within the analysis, an approach generally recommended within water resources research (Helsel and Hirsch, 1992). The use of the median in effect allows the outliers to remain in the analysis but reduces their influence (Helsel and Hirsch, 1992).

When using the data from all the springs there is a strong correlation between median risk and median thermotolerant coliforms ($r = 0.529$, $p < 0.01$). The same is true for faecal streptococci ($r = 0.582$, $p < 0.01$). However, when the data from high density and low density areas were analysed separately, no significant relationship was noted with median risk. This implies that population density and average risk interact and that other factors are also important in determining levels of risk. A more detailed analysis of the influence of multiple factors is described in section 6.5 below.

A basic hazard analysis was performed in relation to the water sources to investigate whether these influenced the microbiological quality of the water. The hazards (essentially sources of faeces) included were on-site sanitation and solid waste disposal, the latter being included given anecdotal evidence of faecal disposal in waste dumps from the ‘wrapper’ method of faecal disposal. Fresh faeces were observable on several occasions on the ground in high-density areas, suggesting that surface disposal of faecal matter takes place in Kampala.

On-site sanitation facilities within 10, 20, 30 and 50m showed no significant difference between low and high-density areas in Kampala when analysed using the chi-square statistic. Waste presence within 10, 20, 20 and 50m was significantly higher in high-density than low-density sites, suggesting that this may be a significant hazard in high-density areas. This would support the hypothesis that the higher microbiological loads in areas of greater population density arises from washing in of faecal material disposed of on the ground surface and not sub-surface leaching from pit latrines.

B3.1.4 *Assessment of specific risk factors on contamination*

Although overall sanitary risk score appears to be a key factor in contamination some individual risks factors (depending on whether they are hazard, pathway or indirect factors) may be more important than others in causing contamination. Relationships between individual risk factors and differing levels of contamination based on the raw data were explored in order to develop a model of spring contamination. This was done in the first instance by using the chi-squared statistic (χ^2).

The first round of analyses investigated relationships between individual risk factors and faecal streptococci. The faecal streptococci data were transformed into two different sets of binary categorical data: above/below 0 cfu/100ml and above/below 10 cfu/100ml, which were selected on the basis of the range of results obtained. The data were converted into a binomial format simply through classifying whether they met the criteria (i.e. above 0 or 10 cfu/100ml) or not. The results for the individual risk factors were already in binomial categorical form. The associations between different factors and presence of faecal streptococci are shown in Table B1. This table indicates that only three risk factors provide results for faecal streptococci presence that are statistically significant to the 99% confidence level: the presence of an eroded backfill area, the presence of surface water uphill and faulty masonry. The absence of a fence is significant to the 95% level. The strong relationship between faecal streptococci and population density and faecal streptococci and rainfall within 24, 48 and 120 hour periods should be noted (all significant at the 99% level). Within increasing levels of faecal streptococci, only erosion of the backfill area and surface water uphill show relationships significant at the 99% level, whilst faulty masonry and waste within 20m show a relationship at the 95% level. Again, the relationships to population density and rainfall are noted as high. It is interesting to note that latrine presence has little or no relationship to presence of faecal streptococci, which suggests that these are not a significant source of these bacteria in the environment. This initial analysis suggests that relatively few factors directly influence faecal streptococci presence and that these are indicative of very rapid recharge from an environment with faecal contamination. The relationships between different risk factors and both presence and numbers of thermotolerant coliforms are more complex. The greater categories used – thermotolerant coliforms below/above 0 cfu/100ml, below/above 10 cfu/100ml and below/above 50 cfu/100ml reflect the more common isolation of these bacteria. The categories were selected because the absence of thermotolerant coliforms forms one component in the WHO Guidelines for Drinking Water Quality (WHO, 1993). The use of 10 cfu/100ml is justified as this is a level suggested by WHO as an appropriate relaxation for small water supplies (WHO, 1997), whilst 50 cfu/100ml is the Ugandan national guideline for untreated supplies.

For presence of thermolerant coliforms, erosion of the backfilled area, absence of a fence and waste within 20m are the sanitary risk factors giving chi-squared results significant at the 99% level (Table B2). Surface water collecting uphill, other pollution sources (for instance animals) uphill and waste within 10m are all significant at the 95% level. Population density again shows a very strong relationship to the presence of thermotolerant coliforms, as does rain have fallen within 48 and 120 hours. Interestingly, the relationship with 24-hour rainfall is not significant.

At 10 cfu/100ml, χ^2 for erosion of the backfill area, surface water uphill, waste within 10m and waste within 20m are all significant at a 99% level. Similar χ^2 values are seen with high population density and rainfall within the previous 24, 48 and 120 hour periods. Flooding of the collection area, other pollution uphill and latrines within 30m all have χ^2 values significant at the 95% level. At the highest level of contamination, erosion of the backfill area, faulty masonry, surface water collecting uphill and waste within 10, 20 and 30m of the spring are significant at the 99% level. Population density, rainfall within 24, 48 and 120 hour periods are also significant at this level. Other pollution source uphill is significant at the 95% level.

These initial results suggests that far more factors influence thermotolerant coliform contamination of shallow groundwater and that a variety of sources of these bacteria are found in the environment.

B3.1.5 *Statistical models of contamination*

Taking into account the results of the analysis above, logistic regression models (Table B3) were developed for the levels of contamination for both coliforms and streptococci. The models were backward stepwise models analysed through SPSS. Initially all co-variates where odds ratios showed relationships significant at least to the 95% level were included. In a few cases, factors where the relationship was not significant at the 95% level, but very close to this level of significance were also included.

Several overlapping factors were included in the initial models (waste within 10, 20 or 30m; latrines within 30m and 50m; rainfall within the previous 24, 48 and 120 hours). This was done for two reasons:

1. If these overlapping factors turned out to be most important (e.g. only rainfall included in final models) this was important to note.
2. Whilst these factors overlapped, it did not mean that their interactions with other factors would be the same and this was also important to investigate.

Where overlapping factors were included in the final iterations, the one with the most significant relationship was retained in the developing the final model.

Table B1 Faecal streptococci associations with risk factors

Risk factor	Median FS above 1			Median FS above 10		
	χ^2	Df	p	χ^2	df	p
Masonry protecting the spring faulty	7.136	1	0.008**	4.902	1	0.027*
Backfill behind wall eroded	10.640	1	0.001**	9.585	1	0.002**
Spilt water floods collection area	0.019	1	0.890	0.163	011	0.686
Fence absent or faulty	4.642	1	0.031*	0.282	1	0.595
Animals have access within 10m of spring	0.499	1	0.480	0.300	1	0.584
Surface water collects uphill of spring	20.808	1	0.000**	49.325	1	0.000**
Diversion ditch uphill faulty	0.171	1	0.679	2.187	1	0.139
Other pollution sources uphill of spring	1.273	1	0.259	0.014	1	0.906
Latrine within 30m uphill of spring	0.558	1	0.455	0.036	1	0.850
Latrine within 50m uphill of spring	0.364	1	0.546	0.385	1	0.535
High population density	19.010	1	0.000**	20.594	1	0.000**
Waste collects within 10m uphill of spring	2.080	1	0.149	3.576	1	0.059
Waste collects within 20m uphill of spring	1.439	1	0.230	5.553	1	0.018*
Waste collects within 30m uphill of spring	0.291	1	0.590	1.065	1	0.302
Rainfall within previous 5 days	6.982	1	0.008**	9.094	1	0.003**
Rainfall within previous 2 day	23.595	1	0.000**	32.777	1	0.000**
Rainfall within previous 1 day	11.383	1	0.001**	22.950	1	0.000**

NB: ** - significant at 99%
* - significant at 95%

Table B2 Faecal coliforms associations with risk factors

Risk factor	Median FC above 1			Median FC above 10			Median FC above 50		
	χ^2	df	P	χ^2	df	p	χ^2	df	p
Masonry protecting the spring faulty	0.511	1	0.511	3.181	1	0.074	8.430	1	0.004**
Backfill behind wall eroded	20.030	1	0.000**	17.958	1	0.000**	23.107	1	0.000**
Spilt water floods collection area	2.997	1	0.083	4.489	1	0.034*	1.313	1	0.252
Fence absent or faulty	10.213	1	0.001**	2.478	1	0.115	0.398	1	0.528
Animals have access within 10m of spring	1.856	1	0.173	0.097	1	0.756	0.386	1	0.534
Surface water collects uphill of spring	6.277	1	0.012*	27.334	1	0.000**	58.892	1	0.000**
Diversion ditch uphill faulty	0.810	1	0.368	1.258	1	0.262	1.313	1	0.252
Other pollution sources uphill of spring	4.732	1	0.030*	5.995	1	0.014*	3.902	1	0.048*
Latrine within 30m uphill of spring	3.718	1	0.054	4.462	1	0.035*	2.210	1	0.137
Latrine within 50m uphill of spring	0.939	1	0.531	1.665	1	0.197	0.086	1	0.770
High population density	28.834	1	0.000**	41.449	1	0.000**	54.957	1	0.000**
Waste collects within 10m uphill of spring	4.945	1	0.026*	14.248	1	0.000**	29.536	1	0.000**
Waste collects within 20m uphill of spring	10.641	1	0.001**	22.479	1	0.000**	25.120	1	0.000**
Waste collects within 30m uphill of spring	1.730	1	0.188	4.718	1	0.030*	8.1015	1	0.004**
Rainfall within previous 5 days	10.247	1	0.001**	18.819	1	0.000**	8.779	1	0.003**
Rainfall within previous 2 day	30.784	1	0.000**	32.211	1	0.000**	24.064	1	0.000**
Rainfall within previous 1 day	1.213	1	0.271	8.604	1	0.003*	18.322	1	0.000**

NB: ** - significant at 99%
* - significant at 95%

Table B3 Logistic regression models

Dependent variable	Co-variates included in model	Number of steps	Final co-variates in model	Selected model co-variates
FS above 1	R1, R2, R4, R6, R8, R9, R11, R12, R15, R16, R17 Step 1 (full model) *–2LL* = 332.309; χ^2 = 67.882, df = 11, p = 0.000	7	R2, R4, R6, R16, R17 Step 7 (final stepwise model) *–2LL* = 338.413, χ^2 = 61.779, p = 0.000	R2, R4, R6, R16 *-2LL* = 343.265; χ^2 = 56.927; p = 0.000
FS above 10	R1, R2, R6, R7, R10, R11, R12, R13, R15, R16, R17 Step 1 (full model) *–2LL* = 316.798; χ^2 = 100.894, df = 11, p = 0.000	8	R6, R11, R16, R17 Step 8 (final stepwise model) *–2LL* = 323.822, χ^2 = 93.870, p = 0.000	R6, R11, R16 *-2LL* = 336.068; χ^2 = 81.624; p = 0.000
FC above 1	R2,R3, R4, R6, R8, R9, R11, R12, R13, R15, R16, R17 Step 1 (full model) *–2LL* = 245.338; χ^2 = 85.357, df = 11, p = 0.000	7	R2, R4, R8, R11, R16 Step 7 (final stepwise model) *–2LL* = 249.073, χ^2 = 81.623, p = 0.000	R2, R4, R8, R11, R16 Step 7 (final stepwise model) *–2LL* = 249.073, χ^2 = 81.623, p = 0.000
FC above 10	R1, R2, R3, R6, R8, R9, R11, R12, R13, R14, R15, R16, R17 Step 1 (full model) *–2LL* = 326.514; χ^2 = 101.416, df = 13, p = 0.000	8	R2, R3, R6, R11, R15, R16 Step 8 (final stepwise model) *–2LL* = 334.045, χ^2 = 93.885, p = 0.000	R2, R3, R6, R11, R16 *-2LL* = 342.069; χ^2 = 85.861; p = 0.000
FC above 50	R1, R2, R6, R8, R9, R11, R12, R13, R14, R15, R16, R17 Step 1 (full model) *–2LL* = 293.183; χ^2 = 129.791, df = 12, p = 0.000	5	R1, R2, R6, R9, R11, R12, R16, R17 Step 5 (final stepwise model) *–2LL* = 293.954, χ^2 = 129.020, p = 0.000	R1, R2, R6, R9, R11, R12, R16 *-2LL* = 304.691; χ^2 = 118.283; p = 0.000

R1. Masonry protecting the spring faulty
R2. Backfill behind wall eroded
R3. Spilt water floods collection area
R4. Fence absent or faulty
R5. Animals have access within 10m of spring
R6. Surface water collects uphill of spring
R7. Diversion ditch uphill faulty
R8. Other pollution sources uphill of spring
R9. Latrine within 30m uphill of spring
R10. Latrine within 50m uphill of spring
R11. High population density
R12. Waste within 10m uphill of spring
R13. Waste within 20m uphill of spring
R14. Waste within 30m uphill of spring
R15. Rainfall within previous 5 days
R16. Rainfall within previous 2 days
R17. Rainfall within previous 1 day

Table B3 summarises the co-variates included in each model, the number of iterations and the covariates included at the final step. The table also provides –2LL, χ^2, degrees of freedom and significance for the first and final steps. In each case, the –2LL value (the log likelihood score of the goodness of fit) increases as the model develops, suggesting an increasingly strong model. The final models developed removed overlapping factors described above by selecting the specific component (e.g. waste rainfall within 48 hours) that had the strongest relationship as defined by the model parameters. Table B3 also provides the final models developed. The models illustrate interesting differences and similarities between the different bacteria and allow conclusions to be drawn as discussed further below.

FAECAL STREPTOCOCCI

The logistic regression models for faecal streptococci over 0cfu/100ml and over 10 cfu/100ml are four and three factor models respectively. Isolation of faecal streptococci colonies appears to be primarily related to the erosion of the backfill area, lack of fence, surface water uphill and rainfall occurring within the previous 48 hours. For presence of at least 10 cfu/100ml, the factors are even more restricted and only surface water uphill, population density and rainfall within the previous 48 hours are included. These are consistent with the results from the odds ratio analysis.

In both cases, the only factor that could act as a contaminant source or hazard factor is surface water that collects uphill of the spring. This may at first glance seem unexpected, but does fit well with the limited coverage with latrines in Kampala leading to build up of faecal matter in drainage channels, solid waste dumps and surface water. It is also likely that a significant proportion of this faecal matter may be animal rather than human in origin as both wild animals (rodents etc) and domestic animals (in particular goats and sheep) are found in low-income areas. It is interesting to note that both eroded backfill and lack of a fence are included, suggesting that direct contamination routes predominate. It is likely that the erosion of the backfill and surface water uphill interact as water inundates the backfilled area. In this case rainfall will act as a primary climatological control as it will aid both the washing in of contaminants into the backfill area and replenishment of the surface water. The importance of these routes is supported by the fact that no other 'hazard' factor is included in the model after half of the iterations are completed.

Stagnant surface water may act as a contaminant source independently of erosion of the backfill area through direct infiltration of localised recharge paths. However, population density is not included in the final model for isolation of at least 1 cfu/100ml, which implies that the control on the presence of faecal streptococci is primarily related to pathway factors allowing direct ingress of poorly disposed of faecal matter and not on sub-surface microbial loading.

The risk factors controlling whether faecal streptococci are recorded at above 10cfu/100ml are more limited and restricted to surface water uphill, population density and rainfall within 48 hours. This suggests that whilst the presence of faecal streptococci contamination is independent of overall microbiological loads, increasing contamination is more dependent on the load of the faecal streptococci in the environment. The absence of eroded backfill as an important factor in the final model (this was removed at a relatively early iteration) implies that for increasing faecal streptococci loads a source of faecal matter is more important. This is supported by the fact that waste within 10m (a further potential source of faeces) was only omitted at the second to last step.

THERMOTOLERANT COLIFORMS

Analysis of thermotolerant coliforms, over 0 cfu/100ml and over 10 cfu/100ml incorporate 5-factor models, whereas in the final condition (over 50 cfu/100ml) a 7-factor model is developed. Rainfall within 48 hours, population density and erosion of the backfill area are common to all these models.

In the model for isolation of at least 1cfu/100ml; the lack of fence and presence of other pollution (for instance animals etc) uphill are included in the model. The incorporation of the fence factor is the same as for the presence of faecal streptococci and this, with the presence of 'other pollution' uphill suggests that presence of thermotolerant coliforms is determined by a direct recharge route. This model illustrates that there is significant general faecal contamination in the environment that is easily and rapidly washed into the springs when rainfall occurs. Surface water collecting uphill was not an important factor and was excluded on the second step and appears less important as a hazard than latrines within 30m which was excluded on the second to last step.

In the model representing over 10 cfu/100ml being found, the fence and other pollution risks factors are omitted and surface water uphill and flooding of the collection area are included. For flooding of the collection area, it is likely that this can be taken as more general indicator of poor maintenance of the spring. The surface water uphill will again function as a source of coliforms in the environment that may either directly inundate the spring or may recharge local preferential flow paths. The importance of hazard factors is supported by the exclusion of latrines within 30m of the spring on the last step on the model development.

When investigating heavy contamination (over 50 cfu/100ml), faulty masonry, surface water uphill, latrine within 30m and waste within 10m are all included. This suggests that for heavy contamination, multiple sources of microbes are more important than pathways and suggests that these levels show a much wider contamination of the shallow aquifer. This is supported by the fact that many of the pathway and contributory or indirect risk factors did not show any significant relationships with this condition.

B3.1.6 *Discussion of the implications of the data analysis*

For both bacteria, the strong relationships with short period rainfall and measures of poor sanitary completion supports a general theory of rapid recharge and deterioration in microbiological quality in response to rainfall. This suggests that entry of recharge water occurs close to the spring with limited opportunities for attenuation and is likely to result both from localised interflow through preferential flow paths and direct ingress through poorly maintained infrastructure. Stagnant surface water and solid waste material both appear to be more important than latrines as sources of these bacteria, which would fit well with the low sanitation coverage. However, it should be noted that latrine presence is a more significant source of thermotolerant coliforms than faecal streptococci.

The importance of sanitary protection points to poor operation and maintenance as a major cause of contamination and this would in part explain why springs in high-density areas tend to have much higher levels of contamination, as poorly disposed of faecal matter is able to directly flow into the spring. The poor operation and maintenance in high-density areas may be due to a number of factors. These include: deterioration in rapidly growing urban centres as communities become more heterogeneous and experience in-migration. This may have a negative impact on operation and maintenance as the users of water supplies become less willing to be involved in community-based programmes that they had little or no involvement in establishing.

It could also be supposed that in higher-density areas, the population will be more likely to use piped water sources and therefore have less interest than low-density communities in sustaining point sources. However, the latter appears not to be true in Kampala and water usage studies have indicated that protected spring use is no lower in many of the high-density areas than low density areas and indeed in some cases the opposite is true (Howard et al, *in press)*. The use of particular source types is more dependent on socio-economic status than population density.

In terms of difference in frequency of isolation of the different types of bacteria, two explanations may be put forward for these results. Firstly, the numbers of faecal streptococci in fresh human faeces may be anywhere between 1 and 4 logs less than thermotolerant coliforms.

The numbers of thermotolerant coliforms can then be expected to remain at higher levels over longer periods of time than the faecal streptococci as initial densities of streptococci are far lower making isolation increasingly difficult with die-off. In this argument, no external influences are exerted on survival or presence (a reasonable assumption if transport is rapid). The generally much higher numbers of thermotolerant coliforms than faecal streptococci after recent rainfall may offer some support to this explanation. This may fit well with the greater importance of latrines for thermotolerant coliforms than faecal streptococci as the travel times are more extended and densities of faecal streptococci become too low for easy isolation. However, as noted above, the greater environmental resistance of faecal streptococci would suggest that this scenario is less likely as it would be expected that sub-surface leaching of the faecal streptococci would continue for a sustained period of time.

An alternative explanation is that the faecal streptococci isolated are derived from purely faecal sources, whereas the thermotolerant coliforms have at least a partial environmental source. This warrants further explanation. There is evidence of thermotolerant coliform presence in tropical waters in the absence obvious sources of faeces in the environment (Hazan and Torranos, 1990). Certainly some thermotolerant bacteria (e.g. *Klebsiella spp, Enterobacter spp* and *Citrobacter spp*) have environmental as well as faecal sources. However, as noted above, the vast majority of the thermotolerant coliforms identified in this study are *E. coli*. This therefore suggests that environmental bacteria are not causing confounding of the results.

It is possible that the *E. coli* can multiply within an environment that is contaminated with faeces. This would offer an explanation as to why the thermotolerant coliforms persist at far greater numbers for much longer periods of time, which would not have been expected simply from initial greater numbers released. This may, in part, explain why levels of coliforms are much higher in high-density areas than low-density areas given overall higher nutrient levels in the shallow groundwater. There is some evidence of the ability of thermotolerant coliforms to grow in topical soils providing there are sufficient nutrients, primarily glucose and salts (Byanppanahalli and Fujioka, 1998). It is not known whether this occurs in Uganda, but the impact is likely to be limited and it is suspected that extended survival rather than multiplication is occurring.

In another study undertaken in the same region, indicators unique to human faeces (sorbitol-fermenting bifidobacteria) and animal faeces (*Rhodococcus coprophilus)* were measured together with thermotolerant coliforms and faecal streptococci. For full details of this study please see the project report (Howard et al, 2001).

This study concluded that the definite spikes in numbers of sorbitol-fermenting bifido bacteria indicated recharge within 7 days and in all likelihood shorter time periods. The results from this study suggested that direct recharge is likely to be a more important mechanism for faecal contamination rather than a single source of faeces (for instance a pit latrine). The study also concluded that faecal streptococci have a stronger relationship with indicators of confirmed human faeces than the thermotolerant coliforms.

This conclusion is important in relation to health risks as the stronger relationship between faecal streptococci and indicators unique to human faeces suggests that they are more useful as an indicator of relevance to health. Such a conclusion would be line with current thinking on the use of indicator and index bacteria (Ashbolt et al, 2001).

The study by Howard et al. (2001) also analysed for coliphage as indicators of virus presence. In the wet season, coliphage were identified at 5 springs also included in the ARGOSS study. In the dry season only 2 springs in the ARGOSS study were shown to have coliphage present (and in one of these, no presence was recorded in the wet season). The presence of coliphage suggests that there is a risk of virus presence in shallow groundwaters, although the data set was too limited to draw firm conclusions regarding their source. The source with the most persistent presence of coliphage in the wet season was in a slaughterhouse, suggesting that an animal source is likely. At one site there was an indication that a pit latrine could be the source, although cattle were kept above this spring. However, some degree of caution should be exercised when interpreting bacteriophage data as phage have a range of isoelectric points, as do other enteric viruses. This means that no particular phage is truly representative of the behaviour of all other enteric viruses (Grabow, 2001).

B3.1.7 *Conclusions on microbial contamination in Kampala*

In conclusion, it appears that local recharge by contaminated surface water is the principal cause of microbial contamination in the Kampala protected springs. The evidence of the results of both faecal streptococci and thermotolerant coliforms supports a theory of rapid recharge where localised pathways are more important in causing deterioration in water quality. The importance of direct pathway factors and of surface water collecting close to the springs support this idea of rapid response. Other potential sources of bacteria appear more important for coliforms than streptococci, which may relate to either a residual contamination supplied through baseflow related to greater numbers released and/or extended survival, or multiplication in nutrient rich environment.

Whilst latrines do not appear to be significant influences on microbiological quality of water, their association with thermotolerant coliform presence at increasing levels suggests that latrines may be leading to low-level extended contamination and may cause higher contamination at specific sites. This implies that they may be residual sources of more persistent pathogens in groundwater, although it is debatable how important this would be in comparison to other transmission routes. In general, very limited lateral separations appear sufficient to reduce risks from contamination from on-site sanitation in Kampala. The data from this study suggest that far from causing major concern in relation to contamination of groundwater sources, on-site sanitation may in fact reduce risks and lead to improvements in water quality.

B3.2 Microbial contamination in Iganga

Of the 152 samples analysed from boreholes in Iganga, only seven showed any presence of thermotolerant coliforms. Of these seven samples, three were recorded at Bikadho, two from Nkono II and one each from Bugumba B and Bulubandi. The highest level recorded was only 12 cfu/100ml from Bikadho, with Bulubandi showing 3 cfu/100ml. The remaining results were all 1 cfu/100ml. Only five samples showed the presence of faecal streptococci, three samples from Bulubandi (range 1-3 cfu/100ml), one from Nkono II (3 cfu/100ml) and one from the Market Mosque (290 cfu/100ml). However, the last result is believed to have been derived from contamination during either sampling or analysis, as the level is very high and this source did not show any other faecal streptococci presence during the study.

The sanitary inspection of the boreholes incorporated 9 factors dealing with potential sources of pollution, potential pathways for pollutant entry into the water sources and general factors believed to contribute to the development of pathways. In addition to these sanitary inspection, an assessment was done of latrine and solid waste presence and proximity in relation to the boreholes.

The average sanitary risk for the boreholes was low, with a mean of 38.8% and a median of 33%. This is lower than the average sanitary risk recorded for the protected springs in Kampala and suggests that operation and maintenance of the boreholes in Iganga is better than the springs in Kampala.

Analysis of risk factors showed no significant relationships with presence or levels of microbiological contamination. At only one dug well was a latrine found within 10m, for 3 boreholes latrines were found within 20m with the rest having latrines over 20m from the source. Solid waste was not commonly found close to the sources.

Overall, the data for the boreholes in Iganga indicate that this technology is very robust with respect to bacterial pathogens. This almost certainly derives in part because of the deeper intake of water and the greater potential for attenuation. However, as Iganga is a lower density settlement, faecal loading in the environment is lower and this may also contribute to the general absence of faecal indicator bacteria. However, the presence of obvious sources of microbiological contaminants close to the boreholes suggests that the better quality is primarily determined by the nature of the technology rather than the lower faecal loading.

Given the nature of the technology, the absence of obvious preferential flow paths and lack of direct pathways into the boreholes, the risk of protozoa presence is believed to be minimal, although this was not verified through analysis. In common with other boreholes, a residual risk from viruses may be considered a possibility. This was particularly the case in Ignaga where the nitrate:chloride ratio indicates a faecal source of the nitrogen (see next below for full details). However, tracer tests performed in Iganga showed that there was virtually no breakthrough of coliphage tracers from the injection to the monitoring wells, despite breakthrough within 19 hours of a conservative chloride tracer. Although this test had some methodological problems, it appears unlikely that a significant risk from viruses is likely to be present in the Iganga boreholes.

The good microbiological quality of water from boreholes in Iganga as determined by indicator bacteria is mirrored in tests performed in other towns in the east of Uganda under project R6874 and the result described by Howard and Luyima (2000). Boreholes were tested over a 12-month period in Tororo, Mbale and Soroti using the Oxfam-DelAgua water testing kit. Of the 43 samples taken from boreholes in Tororo, only one showed the presence of thermotolerant coliforms at a very low level. In Soroti, a total 14 out 197 samples showed presence of thermotolerant coliforms in the range of 1-12 cfu/100ml. However, in Mbale thermotolerant coliforms were found more frequently (40 out of 45) with a range of 1cfu/100ml to too numerous to count.

Analysis of the data suggests that in both Soroti and Mbale, sources of contamination were more important than pathways into the source. In the case of Soroti, this was latrines being uphill of the borehole, latrines within 10m of the borehole and surface water within 10m of the borehole. For Mbale, the principal factors were latrines uphill, latrine proximity and other pollution within 10m. Increasing contamination was associated with the proximity of the latrines.

This analysis suggests that in both Soroti and Mbale, latrine proximity is a concern for microbiological quality. Although it was not possible to undertake a hydrogeological assessment in Soroti given the lack of data, the monitoring data suggests that 10m would be a minimum for latrine-borehole separation, but that 20m would be preferred. As the hydrogeology of Soroti is believed to be similar to Iganga, such findings would support the use of 20m as a horizontal separation in Iganga.

The situation in Mbale, however, is likely to be somewhat atypical, as the geology is different from that of the other towns. Whilst still underlain by the Pre-Cambrian granitised gneisses that make up much of the basement geology of Uganda, it is located adjacent to the Mount Elgon-Wanale massif which is a tertiary volcanic-sedimentary complex. It is likely that preferential flow paths have developed and that fracture aquifers may contribute significantly to the groundwater regime. Data from the early 1990s from springs in this area indicate significant water quality problems and notably raised turbidity that was linked to a fracture aquifer.

Furthermore, three of the boreholes tested had only recently been sunk and siting of these appears to have been poorly planned. At least two boreholes showing consistent high levels of contamination were sunk in low-lying, swampy areas with large low-income urban settlements on high ground. In one case, the borehole was actually sunk through an old termite mound in order to raise the wellhead above a level of flooding. In these areas it appears likely that pit latrines within the vicinity will penetrate the saturated zone, at least during wet periods and that there is insufficient depth of unsaturated zone between the base of the pit and groundwater. Thus in Mbale, horizontal separation from latrines may be more important than elsewhere, particularly where boreholes have been poorly sited.

The dug wells in Iganga showed more frequent contamination than the boreholes and pronounced seasonality. However, in both cases at least three samples showed an absence of thermotolerant coliforms, usually in samples taken in the dry season. Faecal streptococci were also regularly found in the dug wells, although the numbers varied significantly from those of the thermotolerant coliforms, but showed a similar seasonal trend.

The sanitary inspection of the dug wells incorporated 9 factors dealing with potential sources of pollution, potential pathways for pollutant entry into the water sources and general factors believed to contribute to the development of pathways. The average sanitary risk score is low for the dug wells, with a mean of 24.5% and a median of 23%. This suggests that operation and maintenance of relatively good and better than the protected springs in Kampala. Analysis of the particular factors involved in deterioration in water quality was not feasible given the small data set. However, in the case of one well, the almost complete absence of any risk factors other than a latrine within 10m suggests that contamination is derived from the pit latrine. At the second dug well, there were a significant number of risk factors identified and it is therefore difficult to say with certainty which of these were most important, particularly given the very shallow water table. It is likely that several factors led to contamination.

One protected spring was also tested in Iganga and in general the microbiological quality was reasonable, with many samples showing an absence of both thermotolerant coliforms and faecal streptococci. The average sanitary risk score was high, with a mean of 59.1% and a median of 62%. However, as this spring was located well away from any human settlement and appeared to come from a fracture aquifer it is likely that the low levels of contamination noted is likely to be due to low faecal loading in both the immediate surroundings and the recharge area.

The data from Iganga indicates that the risk of microbiological contamination of boreholes is limited. The technology is robust with regard to microbiological quality in general. The limited impact of sanitary risks found at the source itself indicates that hazards would be more important than pathways in causing contamination. This is supported by evidence from other towns in Uganda where contamination was primarily linked to the presence of pit latrines. In general, a horizontal separation between the boreholes and pit latrines of 20m seems adequate for most boreholes where these are sunk into alluvial aquifers. Where the boreholes are sunk into fracture aquifers a greater distance is required, which may be difficult to determine accurately given the current lack of data.

The dug wells and protected spring were both more contaminated, but the limited data set precluded detail analysis of risk factors and contamination found. However, at least one dug well showed evidence of direct contamination from a pit latrine and this suggests that greater horizontal distances are required to protect the wells. A figure in the order of 20m is likely to be adequate. However, it is far from certain that on-site sanitation would be more important than other risk factors. In general, the vulnerability of dug wells to contamination is high and this technology is unlikely to be robust with respect to water quality without significant improvement in the design.

B3.3 Chemical contamination in Kampala

Sampling and analysis for nitrate, chloride, ammonia and phosphate was carried out. Significant levels and trends in ammonia and phosphate concentrations were not detected. The nitrate and chloride data however provides some interesting information. Beneath high-population density areas of Kampala, nitrate levels are very high (as are thermotolerant coliform counts, as described above). In low-density population areas of Kampala nitrate levels are less elevated than elsewhere. Median data are presented in Table B4.

Table B4. Median dry season (previous month's rainfall <100mm) data from Kampala and Iganga (after Barrett et al., 1999)

Study area	NO_3 (mg/l)	Cl (mg/l)
Monsoonal rain[1]	0.6	0.9
Weathered mantle[2]	0.1	21
Fractured bedrock[2]	2.2	24
Iganga, Uganda	44	32
Kampala[3], Uganda	74	54
Kampala[4], Uganda	21	17

[1]Taylor, unpublished data, [2]Taylor and Howard (1994), [3]Kampala high-density,[4]Kampala low-density

Site by site analysis indicates that some sites respond rapidly to rainfall (rises in chloride levels, decreases in nitrate), but that others do not, implying a greater or lesser susceptibility to surface runoff/interflow and baseflow. In total, ~75% of the high-density population sites exceed the WHO Guideline value of 50mg/l NO_3 at least once during the study. This value is the same in medium density sites, but falls to 20% in the low-density population sites monitored. Clearly nitrate loading to groundwater is linked to population density. A plot of dry season nitrate and chloride data (Figure B4) demonstrates a clear correlation between the two parameters. Such a close relationship on a regional scale indicates a single dominant source for both species, and the relatively high nitrate to chloride ratio (>1) is indicative of a faecal origin (Morris et al., 1994). The ratio of nitrogen to chloride in human waste is approximately 2:1 (equating to ~8.8:1 nitrate to chloride). The ratio observed in the dry season (month preceding sampling <100mm rainfall) in Kampala's high and medium population density areas is 1.1:1, which implies that approximately 13% of nitrogen in human waste is being leached to groundwater.

However, observation of temporal variations in chloride and nitrate indicate that the relationship is more complex. Figure B5 clearly shows that chloride and nitrate do not follow the same seasonal pattern. Indeed, nitrate concentrations are seen to actually decrease during the onset of the rainy season. This may be explained in the context of the nitrogen cycle. As previously discussed, bacterial indicators demonstrate that the protected springs in high-density population areas of Kampala are susceptible to sewage contamination during the rainy season as a result of ingress of the contaminated water that results from the presence of surface waste, leading to the contamination of surface runoff and surface water bodies. This rapid contamination of spring water by sewage produces an initial dilution of groundwater nitrate concentrations, as the contaminated recharge waters will contain little or no nitrate (the nitrogen being present initially in organic form). The subsequent rise in nitrate levels, as the rainy season progresses reflects the nitrification of sewage derived nitrogen in the groundwater (which is generally oxidising at shallow levels).

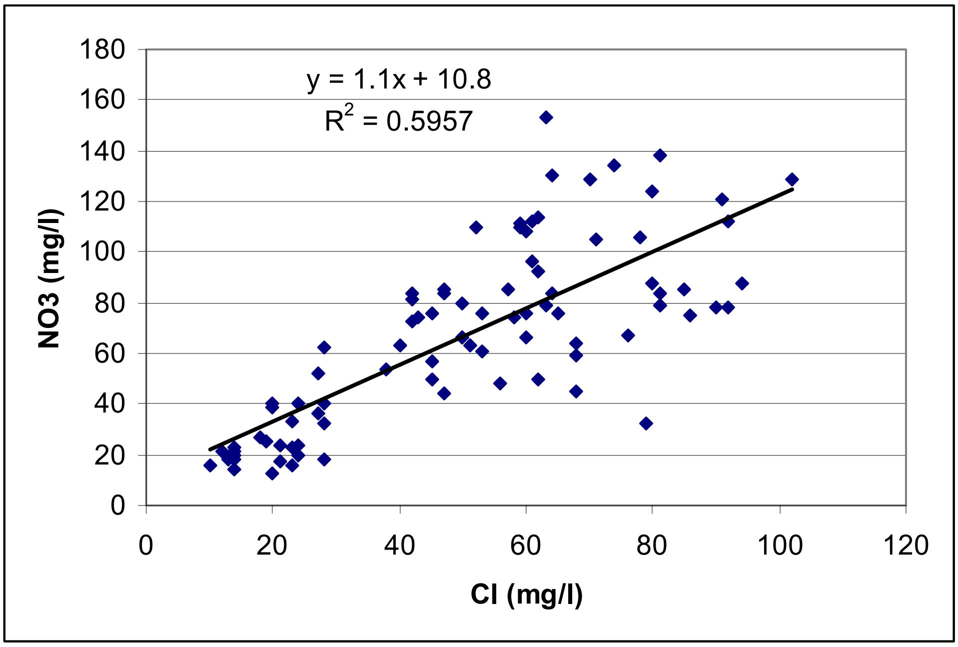

Figure B4. Correlation between chloride and nitrate in high and medium density population areas of Kampala during the dry season (where previous months rainfall <100mm)

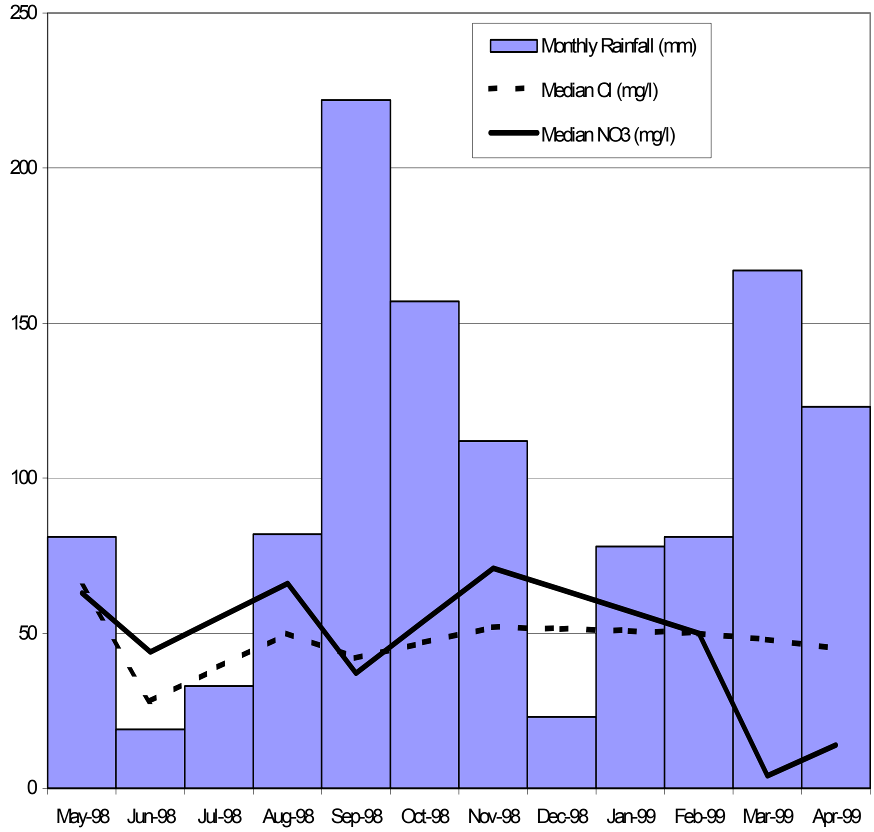

Figure B5. Temporal variations in median chloride and nitrate concentrations in high-density population areas of Kampala

B3.4 Chemical contamination in Iganga

In Iganga, 50% of the sampled sites exceeded the WHO Guideline value of 50mg/l NO_3 at least once during the study (71% in high-density population areas and 38% in low-density population areas). Table B4 indicates that median NO_3 and Cl values are elevated compared to pristine waters, but that the median NO_3 value does not exceed the WHO Guideline value. However, as faecal contamination is proportional to population density, and urban populations and densities are expected to increase in small towns such as Iganga, it is likely that the median NO_3 concentration will increase over time. Figure B6 is a plot of nitrate and chloride concentrations from Iganga during the dry season (previous month's rainfall <100mm). As seen in Kampala, there is a correlation between the concentrations of the two species indicating a single dominant source. The ratio of nitrate to chloride (1.5:1) indicates 17% leaching to groundwater of nitrogen in human waste.

As seen in Kampala, there is an 'inverse' in the nitrate to chloride ratio during the rainy season (Figure B7). This would imply relatively rapid penetration of sewage contamination during the rainy season, although the absence of microbial indicators suggests that transport times are sufficient to allow microbial attenuation.

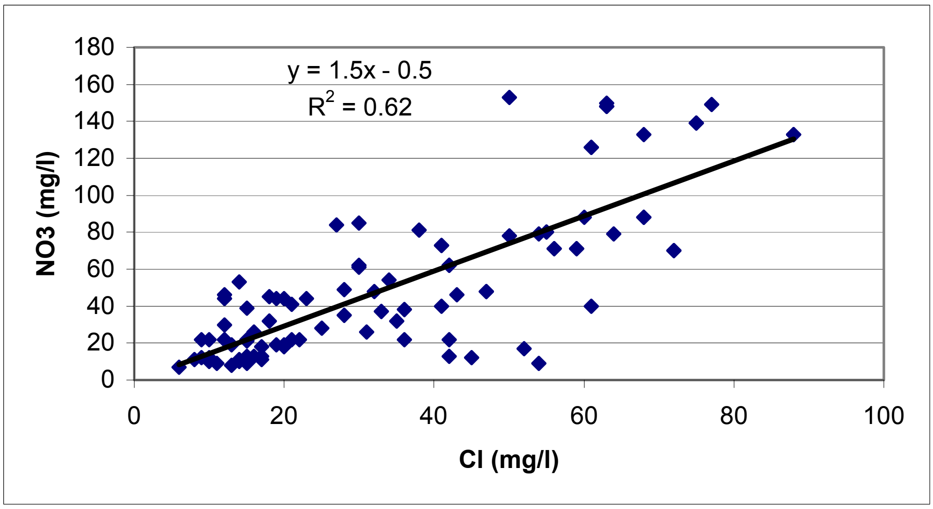

Figure B6. Correlation between chloride and nitrate concentrations in Iganga

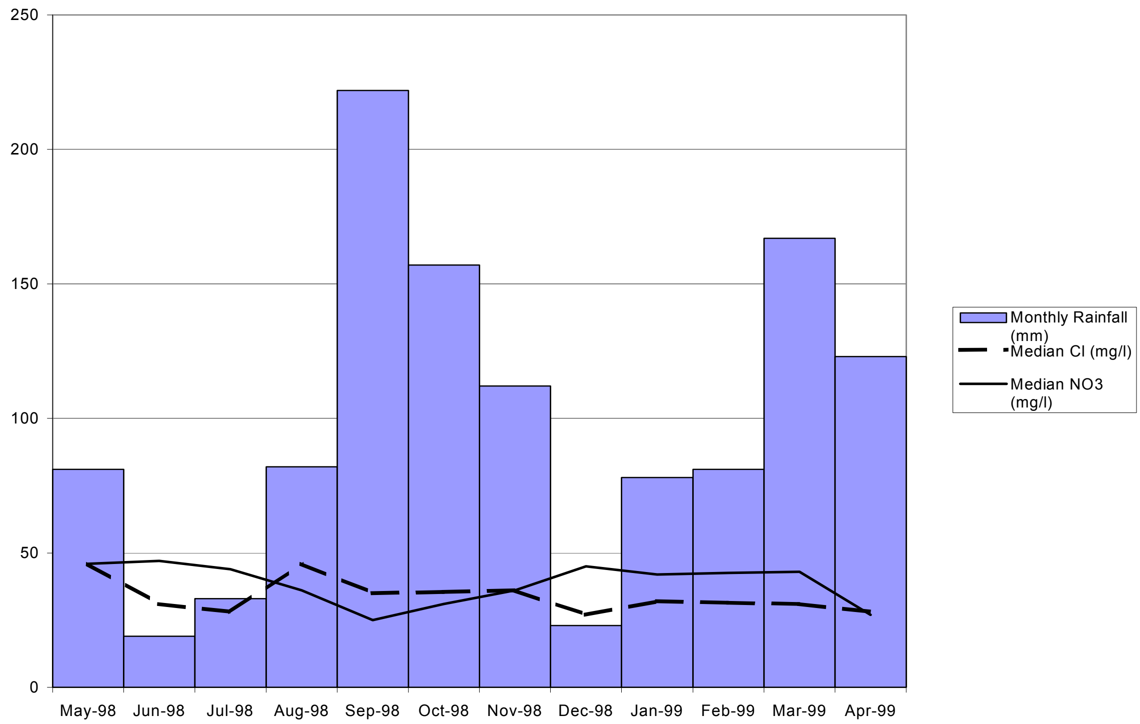

Figure B7. Temporal variations in median chloride and nitrate concentrations in Iganga

B4. CONCLUSIONS

The study from Uganda allows a number of conclusions to be drawn that are of broader relevance.

The study showed that the collection and analysis of water quality monitoring and sanitary inspection data can be an effective way of establishing the importance of on-site sanitation as a source of contamination of shallow groundwater. This therefore offers the potential for countries and organisations that lack detailed hydrogeological data to still be able to develop an understanding of the relative risk of on-site sanitation in causing groundwater contamination. Although the results may be somewhat qualitative - i.e. a definitive number cannot be derived representing a safe separation distance - it does provide a useful working tool.

The data from Uganda show that the quality of water from point water supplies is partly technology dependent. The evidence suggests that boreholes are robust with regard to microbiological contamination and are far less vulnerable to contamination under most conditions than protected springs. The data from dug wells was too limited to draw conclusions but suggest a significantly worse performance than boreholes and probably poorer quality than springs. The protected springs, although clearly vulnerable to contamination, can be re-designed to reduce contamination to acceptable levels.

Microbiological contamination of protected springs results from very rapid recharge of contaminated surface water through preferential flow paths and direct ingress into poorly maintained infrastructure. Sub-surface leaching is far less significant. The control of microbiological contamination in such supplies therefore requires greater attention to operation and maintenance of infrastructure than proximity of latrines. It is likely that horizontal separation distances need only be nominal to be acceptable. This is likely to be mirrored in other areas where such technologies are used. In the rare occasions that boreholes showed contamination, sub-surface leaching appeared to be more important. However, in most cases a separation of 10m is likely to be sufficient. This appears likely to be applicable in most areas, except where fracture aquifers predominate.

Nitrate contamination is likely to occur from sub-surface leaching from on-site sanitation and in higher-density areas the levels are frequently in excess of the WHO Guideline Value. Where groundwater does not provide the long-term source of water (e.g. Kampala) this may not be a significant problem unless pollution of alternative sources are suspected. However, in towns such as Iganga that are groundwater dependent, nitrate contamination represent a significant long-term resource availability issue that requires action to reduce contaminant loads.

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APPENDIX C
RISK OF GROUNDWATER POLLUTION BY ON-SITE SANITATION, GEOCHEMICAL MODELS AND GEOINDICATORS, RIO CUARTO CITY, ARGENTINA

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C1. INTRODUCTION

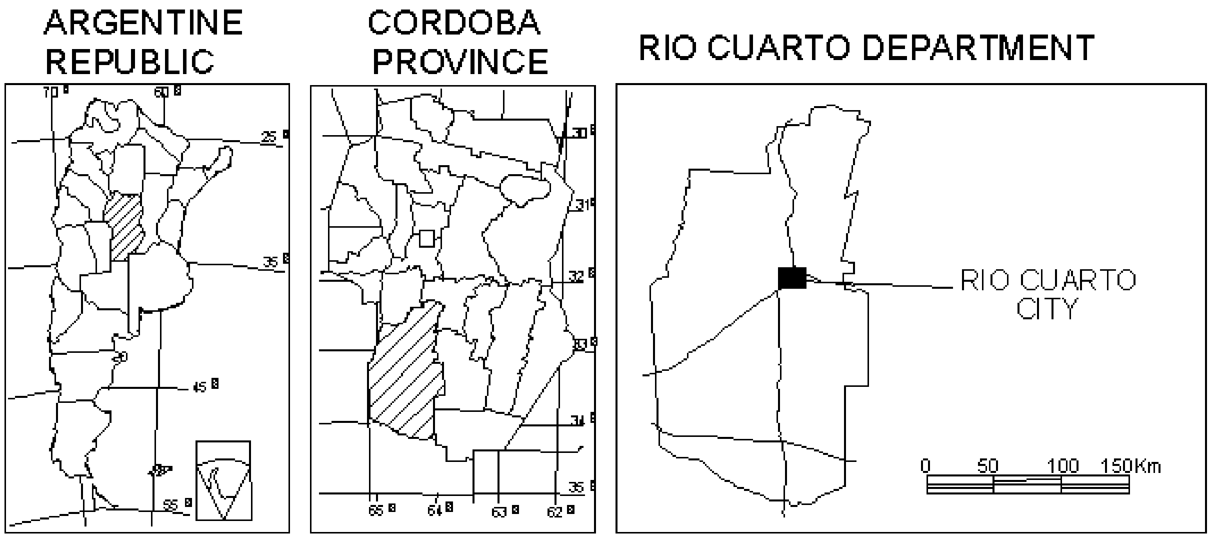
The supply of piped water and sanitation systems is considered a key element for the improvement of health in developing countries. In relation to the removal of excreta, low cost on-site sanitation systems are the most common option for low income sectors. However, there is serious evidence that these systems can cause the degradation of groundwater quality, as a result of microbiological and chemical contamination.
According to the World Bank Report (1995), pollution problems in Argentina can be divided into two groups: A) larger metropolitan areas (Buenos Aires, Còrdoba, etc) where there is a predominance of ground and surface water contaminated with untreated domestic sewage and industrial effluents, air and noise pollution from transport and industrial sources and inappropriate disposal of hazardous solid and industrial wastes. B) smaller cities or towns (e.g. Rìo Cuarto city) where surface and groundwater pollution from untreated domestic sewage and inadequate collection and disposal of urban solid wastes prevail. Thus, the main health problems in the country are diarrhea, due to poor sanitary systems, and respiratory problems, as a result of air pollution. The report highlights that groundwater pollution is the most important in the country.
Rio Cuarto city, with 150, 000 inhabitants, is an example of a medium sized city; coverage of piped water and sewerage are 75 and 60% respectively. The city uses groundwater for all water supplies and the distribution of piped running water is nowadays the responsibility of “Ente Municipal de Obras Sanitarias” (EMOS). The supply is derived from shallow groundwater adjacent to the Rio Cuarto river, by adits and vertical wells of different size and yield.
In periurban neighborhoods, where there is no running water, groundwater is also used for water supply and it is obtained from shallow domestic wells. On-site sanitation systems are widely used in these areas and for this reason, there is a significant risk that groundwater could become contaminated.
For this reason, an investigation of the potential problem was carried out in successive stages of work whose main objective was to generate scientific information which would allow priorities in the planning and use of groundwater to be established. In order to achieve this general objective, other specific ones were established, among which it is important to highlight:
- to establish the hazard, vulnerability and the risk of contamination of the phreatic aquifer, due to the presence of on-site sanitation systems.
- to carry out the geochemical characterization of groundwater, verifying the existence of pollution in areas with different risk degrees (different aquifer vulnerabilities and pollution loadings).
- to monitor groundwater geoindicators to evaluate and control the degree and the rates of environmental changes.

C2. Characterization of the study area

C2.2 Location and climate

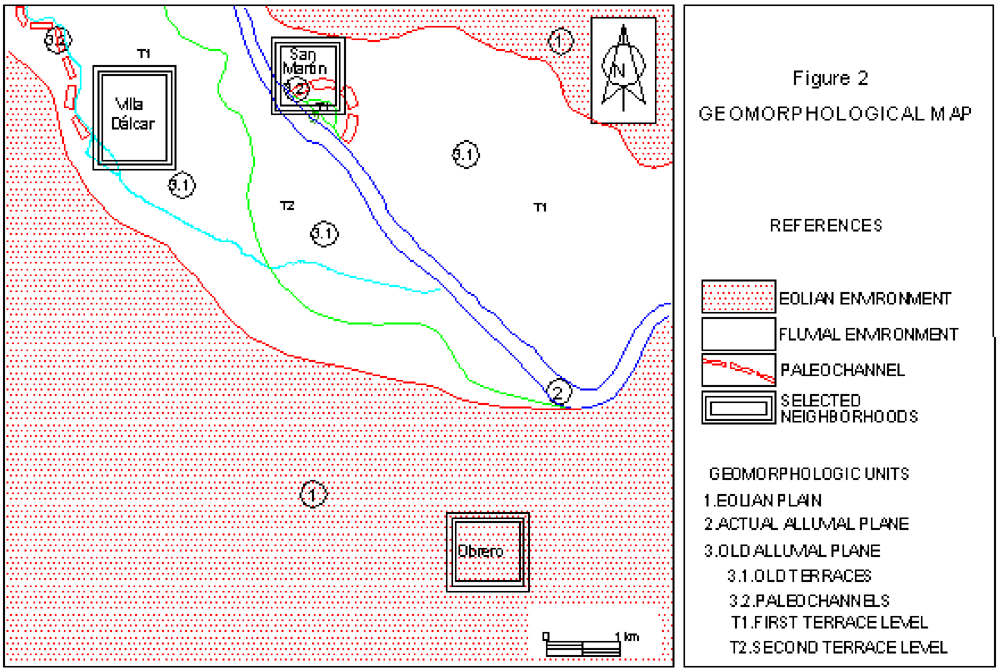
The study area is Rio Cuarto city, which has a surface of 5,816 ha and presents a mean altitude of 438 meters above sea level. It is situated between 33° 03’ and 33° 07’ South latitude and 64° 21’ and 64° 18’ West longitude. (Figure 1)
The studied area has a sub-humid mesothermal climate, with a monthly mean temperature of 16.5ºC. The annual mean rainfall is of 890 mm, most (85%) of which is concentrated in spring-summer. The annual mean evapotranspiration is of 804 mm and most excess rainfall occur between the months of November and November.

FIGURE 1: LOCATION MAP



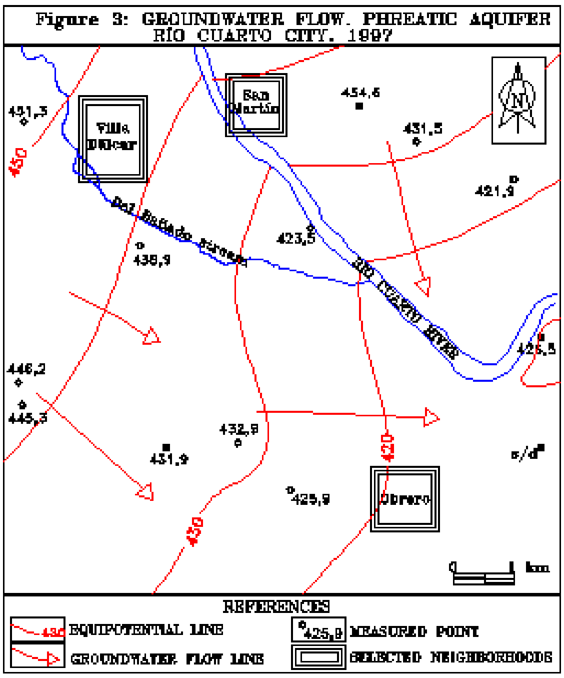
C2.2 Geology and Geomorphology

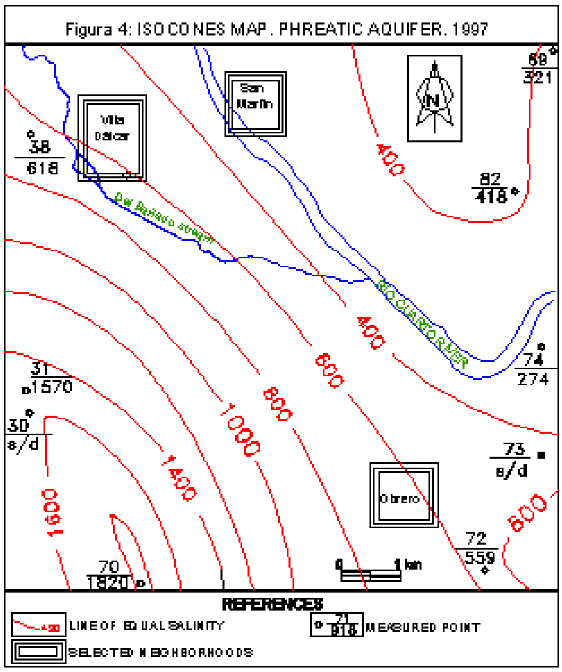
The area studied is situated in the Argentine Geological Province “Chaco Pampeana Plain”. The underlying geological structure is important; NW-SE faults exert strong influence on the course of the Rio Cuarto river whilst N-S and NE-SW lineations are prominent. The outcropping lithology is made up of Quaternary sediments of aeolian and fluvial origin. The relief is slightly undulating and two geomorphological environments can be distinguished. (Figure 2). In one of them, a typically fluvial morphology predominates (terraces, abandoned meanders, etc.). This relief feature is linked to the migrations of the Rio Cuarto river, and the sediments are principally coarse (fine, medium and coarse sands, and gravels). The other environment is of an aeolian origin and it includes loess with intercalated calcrete lenses.



C2.3 Hydrogeology

A phreatic aquifer is present in the shallow aeolian and fluvial sediments. The general groundwater flow is NW- SE and the morphology of the water table is slightly undulating. The Rio Cuarto river is a gaining stream with perennial regime. The mean hydraulic gradients of groundwater are of 5 x 10 $^{-3}$ (Figure 3). In relation to quality, the least content of total dissolved solids (TDS) content varies from 150 mg/l which is associated with the deposits of the alluvial plain of the Rio Cuarto river to 1700 mg/l within the aeolian deposits (Figure 4). The greatest content of total dissolved solids is associated to the environment of the aeolian plain (1700mg/l) (Figure 4).





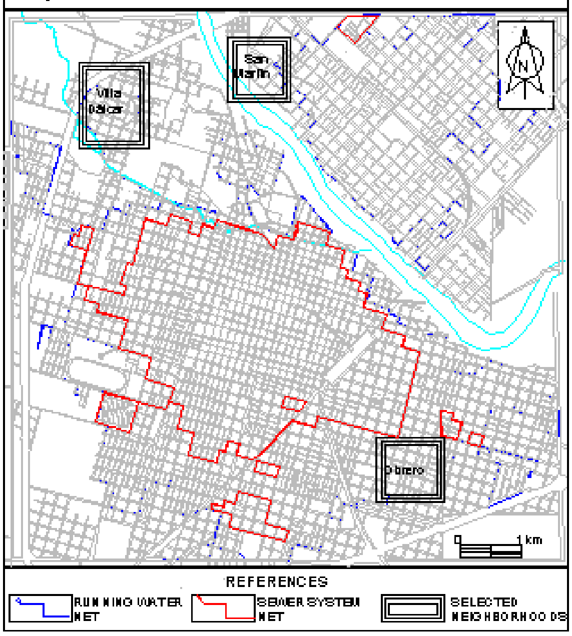
C3. Contamination Risk: a tool for planning groundwater

It is important, where a groundwater contamination hazard is identified, to assess the risk for the aquifer in order to identify priority areas for monitoring and, where necessary, managing or reducing the risk. However, such assessments do no substitute for inspection and monitoring in the field. Initially, an assessment of the risk was made based on the methodology developed by Foster and Hirata (1988). This approach considers the interaction between: a) the hazard, that is to say, the contaminating recharge that is, will be or might be applied to the subsoil as the result of human activity and b) the contamination vulnerability of the aquifer due to the natural characteristics of substratum.
In the present study the determination of contamination risk of the phreatic aquifer was carried out in relation to the problems with on-site sanitation systems in the area of Rio Cuarto city. In the analysis the downtown area was not included since it possesses running water and sewerage systems. (Figure 5).

C3.1 Groundwater Vulnerability

Using as a base the methodology proposed by Foster and Hirata (1988), vulnerability was determined through three successive steps: a) determination of groundwater occurrence, b) definition of lithology in the unsaturated zone, c) determination of depth of the static level of the aquifer. For the three steps, the corresponding desk and field work were done, which allowed to make thematic maps. According to the methodology, indexes between 0 and 1 were given, where 1 indicates the highest vulnerability. For the hydraulic character, value 1 was given to all the area, since the aquifer is of a phreatic type. For the lithology of the unsaturated zone, and following the modifications of Blarasín et al. (1993) method, the highest proposed indexes were 0,7 for the coarsest materials and 0,5 for the finest. In relation to water depth, 8 intervals were established, with value 1, for the greatest vulnerability, corresponding to the least water level depth. The indexes that correspond to these three variables were multiplied successively in order to obtain the final vulnerability indexes (Figure 6). Three different types were obtained: low, medium and high vulnerability.

Figure 5: RUNNING WATER AND SEWER SYSTEM NETS-1999



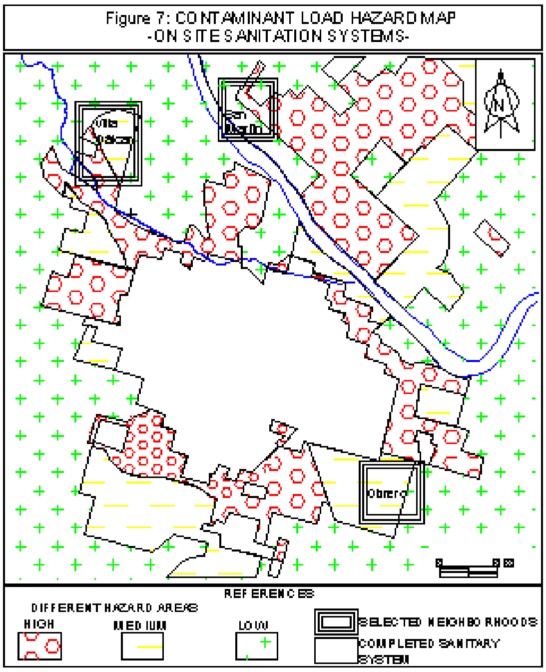


Figure 7: CONTAMINANT LOAD HAZARD MAP -ON SITE SANITATION SYSTEMS-

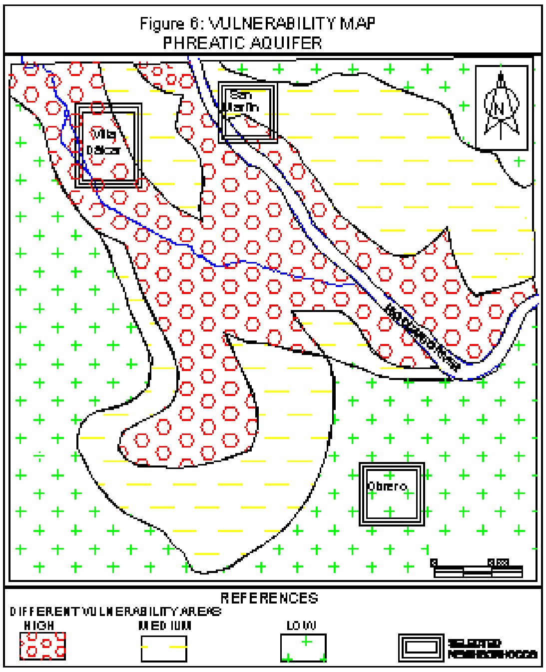


Figure 6: VULNERABILITY MAP PHREATIC AQUIFER

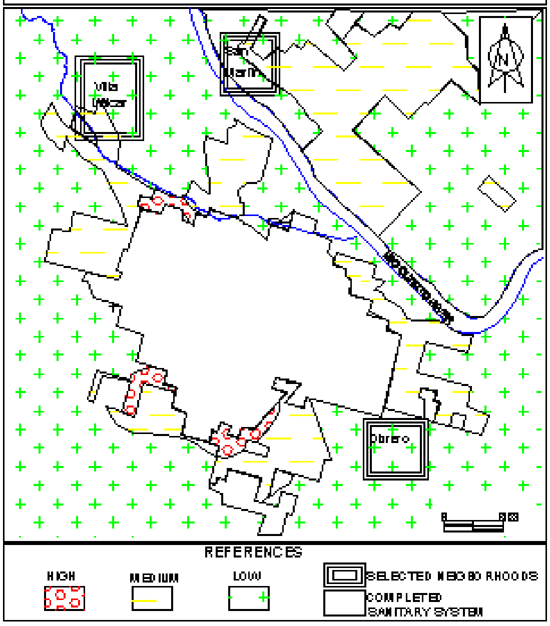
C3.2 Contaminant load hazard

According to the methodology of Foster and Hirata (1988), it is necessary to establish four characteristics of the contaminant load to the subsoil for each contaminating activity. They are: a) type of contaminant involved, b) time of load application, c) method of disposition in subsoil, d) contamination intensity (or relative concentration). In the present study, and following modification by Blarasin et al (1993), these four characteristics were assessed using indexes between 0 and 1, being 1 the highest hazard. The item “type of contamination” was given value 1, since this variable is constant in all the areas without sewered systems and it is considered that the existence of excreta discharge always implies a great hazard, specially if it is taken into consideration that some of these contaminants, like nitrates in an oxidised (aerobic) environment, are extremely mobile and persistent. The item “time of load application” was also given value 1, since this event occurs continuously. In relation to “method of disposition”, value 1 was given to the greatest hazard to sectors where sanitation systems have their base in the saturated zone, and value 0,5 when the base is in the unsaturated zone. The relative concentration of the contaminant came about from the comparison with the map of house density per hectare of Rio Cuarto (Rozovich, 1997), using as criteria “the number of on-site sanitation systems is equivalent to the number of houses”. In this way, value 1 was given to areas with high urban density, value 0,5 to those of medium density and value 0,1 to those of low density. In this way the final hazard index, and the corresponding map, result from the product of the given values to these four characteristics (Figure 7).

C3.3 Contamination Risk

According to the indexes given to the vulnerability and the contaminant load hazard, the mathematical product of these indexes to obtain the risk result in 55 different risk classes. The indexes were grouped in class intervals for the final map, obtaining 3 classes, from very low risk (between 0,02 and 0,09) to high risk (between 0,55 to 0,63). The areas of lower risk are more common in peripheral sectors, which are less populated and have a deeper water table, and where the sediments of the unsaturated zone are very fine. The presentation of this map with the traffic light code, in which red colour indicates the greatest risk and green colour the lowest risk, is of great use for those who take decisions in the planning of groundwater (Figure 8).

Figure 8: CONTAMINATION RISK MAP
-ON SITE SANITATION-



C4 GEOCHEMICAL CHARACTERIZATION AND CONTAMINATION MODELS

As it has already been stated, methodologies of a preventive type (risk maps) should not substitute inspection and field monitoring. In the verification stage of geochemical models, different neighborhoods of the city for various reasons were selected. These reasons included (i) differences in pollution risk, (ii) water quality as measured by Neighbors Associations, and (iii) the geological conditions. As a summary, three examples of the great diversity of situations found are explained. In all the cases, at least thirty domestic water wells were included in the survey in each of the studied neighborhoods, with a density of one well per hectare. The samples obtained were then subjected to chemical and microbiological analysis. Regarding the contamination danger that the presence of on-site sanitation implies, nitrate, TDS, chloride, and the microbiological features of the water were especially considered, since they are good indicators of that problem.

C4.1 General aspects of contamination by nitrates

The presence of nitrates in groundwater in urban environments, is principally caused by the presence of some source of human and/or animal wastes. According to the CAA (Argentinean Alimentary Code) the established limit of nitrate in water is 45 mg/l, and according to the WHO, the limit is 50 mg/l. Water with higher contents of nitrate may cause metahemoglobinaemia in children, and may have implications in diabetes cases, etc.

When the nitrate in groundwater derives from nitrogenous organic substances, it is the result of microbial degradation into ammonia ions which are biologically oxidized until they become nitrites and nitrates through reactions 1 and 2.

(1) $2\ NH_4 + 2\ OH + 3\ O_2 \Rightarrow 2\ NO_2 + 2\ H + 4\ H_2O$ (Nitrosomonas)

(2) $2\ NH_2 + {}^4O2 \Rightarrow 2\ H_2O + NO_3$ (Nitrobacter)

C4.2 General aspects of contamination by bacteria

In general, microorganisms may be in suspension in water, adhered to particles in suspension or to the solid surfaces of the materials of the aquifer. Their mobility in groundwater will vary according to their state. Microorganisms preceding from human excreta may include: helminth eggs, protozoa, bacteria and viruses. Bacteria, unlike viruses, can multiply themselves outside of their primary host environment if the nutrient supply is guaranteed, as in the case of an aquifer where the effluents come from on site sanitation systems. Bacteria form authentic micro ecosystems in groundwater, and faecal coliforms bacteria are adequate indicators of faecal contamination.

For some authors, viruses are the most important potential contaminants coming from the discharge of septic systems. These viruses may appear even in water that has been subjected to routine bacteriological analysis and that has been considered satisfactory.
Microbiologically contaminated water may cause diseases like cholera, hepatitis, diarrhea, and meningitis, among others.

C4.3 Model of contamination in reducing environment in fluvial sediments: Villa Dalcar neighborhood

The phreatic aquifer is made up of fine and coarse sands and gravels of fluvial origin. The water is fresh and calcium bicarbonate, with TDS values ranging between 294 and 644 mg/l and is of the calcium bicarbonate water type. This neighborhood is located in a topographic low and the groundwater level is situated close to the ground surface (2-4 m). The cesspools discharge effluents at the same water level in all cases. The water wells in houses have depths ranging from 7 to 12 m. The population density of this area ranges between 8 and 80 persons/ha. In this neighborhood 54% of the groundwater was inadequate for human consumption due to bacteriological contamination. There was a recount of microorganisms in 67% of the samples, 54% show total coliforms and 31% of these have faecal coliforms. These results allowed an interpretation which is summarized in Figure 9a. It is apparent that significant and widespread contamination of the aquifer by faecal bacteria occurs as a result of the direct discharge from cesspools to groundwater (Figure 10a).
In this neighborhood, no nitrate was detected in groundwater (Figure 11a). This is due to a lack of dissolved oxygen (anaerobiosis) in the upper part of the aquifer as a consequence of a greater availability of nutrients because of the direct discharge of nutrients to groundwater. However, nitrite is present in some cases, indicating denitrification. The presence of dissolved iron in the water results from these anaerobic conditions. During water abstractions, the dissolved iron gets in contact with an environment with available oxygen (aerobic) and it oxidizes (ferrous iron), and it precipitates as oxides and hydroxides in sanitary installations and pipes. The oxides usually have a reddish color with a black one on some occasions due to the presence of manganese or organic matter. The bad smell and taste of the water are characteristic of this area. The most contaminated samples show an increase of TDS and chlorides.

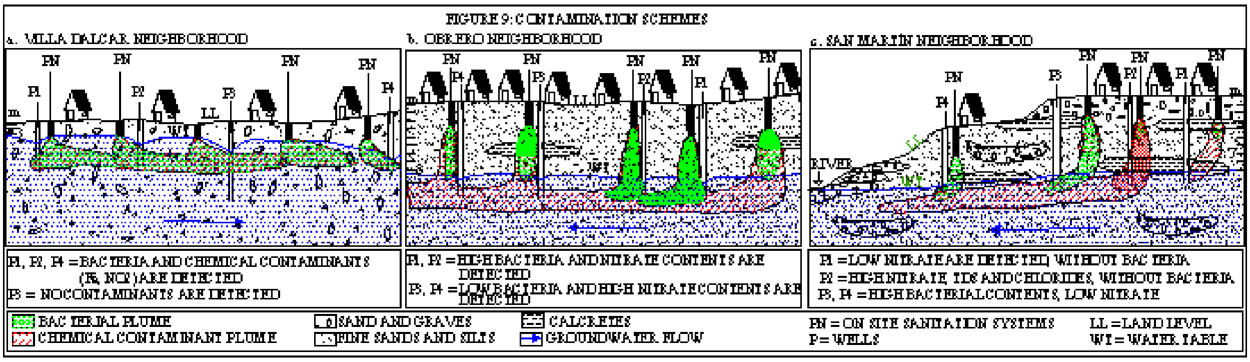


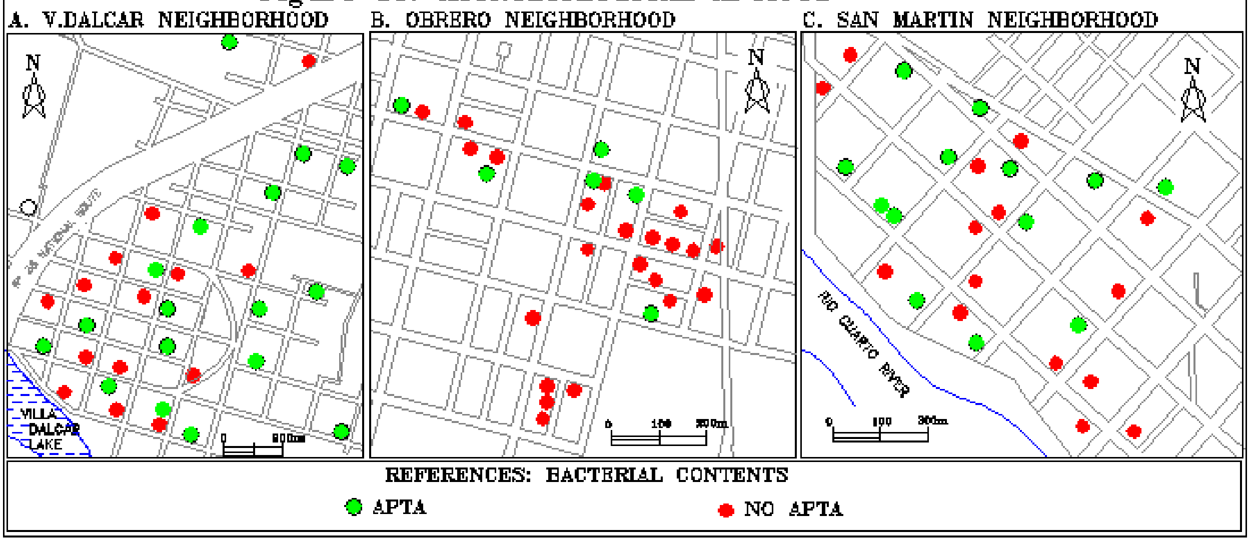
Figure 9 Conceptual model of the three neighborhoods

C4.4 Model of severe contamination in oxidizing environment in aeolian sediments: Obrero neighborhood.

The sediments that comprise the aquifer of this neighborhood are of aeolian origin, silts and fine sands, with frequent intercalations of calcretes.
The water is of the sodium bicarbonate type, with a saline content (TDS) of between 400 and 2700 mg/l. The water table is found at an average depth of 9 m and the domestic wells have a depth of between 12 and 14 meters.
On site sanitation systems have an average depth of 5 m, that is to say, they have their bases in the unsaturated zone. The mean urban density of the area is of 150 inhabitants per hectare.
Of the total number of samples analyzed, 87% are inadequate for human consumption, due to the deterioration of the chemical quality and the bacteriological contamination (Figure 10b).
In this neighborhood, the plume of microbiological contaminants (Figure 9b), reaches groundwater, in most of the cases, but just in a few occasions there are faecal coliforms, *Escherichia coli* or

Pseudomonas aeruginosa. When these do not reach the aquifer is because the decontaminating processes typical of the unsaturated zone are acting (action of mechanical filtration and biochemical transformations).
TDS, chloride and nitrate are other important indicators of contamination. The 82% of the samples exceeded the nitrate established limit for human consumption, the more contaminated samples presenting contents ranging between 250 and 400 mg/l (Figure 11b). The nitrates derived from the nitrogenous organic matter coming from cesspools, which oxidizes in the unsaturated zone, with the participation of the aerobic bacteria already mentioned. The highest salinity levels (TDS) are observed in the more contaminated samples. The average salinity of the samples is of 951 mg/l (Table 1), the salinity of the uncontaminated samples is of 545 mg/l, while the most contaminated sample has a TDS value of 2737 mg/l. In the five most contaminated nitrate samples the chloride average is of 211 mg/l, three times more than the average (69 mg/l) of the samples which are not contaminated.
As it can be observed, in spite of the availability of the unsaturated zone, whose thickness and grain size indicates a good attenuation capacity, the contaminating load is sufficient to produce significant of degradation of groundwater.

Figure 10: MICROBIOLOGICAL APTITUD



C4.5 Model of low contamination in an oxidizing environment in fluvial sediments: San Martin neighborhood

The phreatic aquifer is formed of fluvial fine to coarse sand deposits and gravel, with intercalations of silty materials and calcrete lenses. The water table is between 6 and 9 m whilst domestic abstraction wells are typically 6 to 10 m deep. The water is of the calcium bicarbonate type and is fresh with a mean salinity (TDS) of 354 mg/l, with values ranging between 180 mg/l and 932 mg/l.
The sanitation systems have an average depth of 4 m, i. e. they do not reach the groundwater level. This aspect and the low population density (average: 67 persons/ha) produces the contamination model shown in Figure 9c. In just a few cases, the microbiological plume reaches groundwater. There is no evidence of faecal coliform bacteria as species indicator. However, total aerobic and coliform microorganisms were found, for this reason, 54% of the samples are not adequate for human consumption. Nitrogenous organic matter coming from cesspools oxides with microbiological intermediation, in the unsaturated zone. In this way, the nitrates reach groundwater. Only five samples (18%) exceeded the established limit for nitrates, and the 50% presented contents between 1 and 20 mg/l. The mean salinity of the five contaminated samples is of 684 mg/l, which doubles the salinity average (281 mg/l) of the uncontaminated ones. In addition, in those samples with nitrate contents superior to 50 mg/l, the average of normal chloride would be four times more.
In spite of the fact that the unsaturated zone sediments are not very permeable, the state of low degradation of groundwater is caused, mainly, by the low density of inhabitants and the fact that the water table is deeper than in the fluvial sequences described for Villa Dalcar neighborhood.

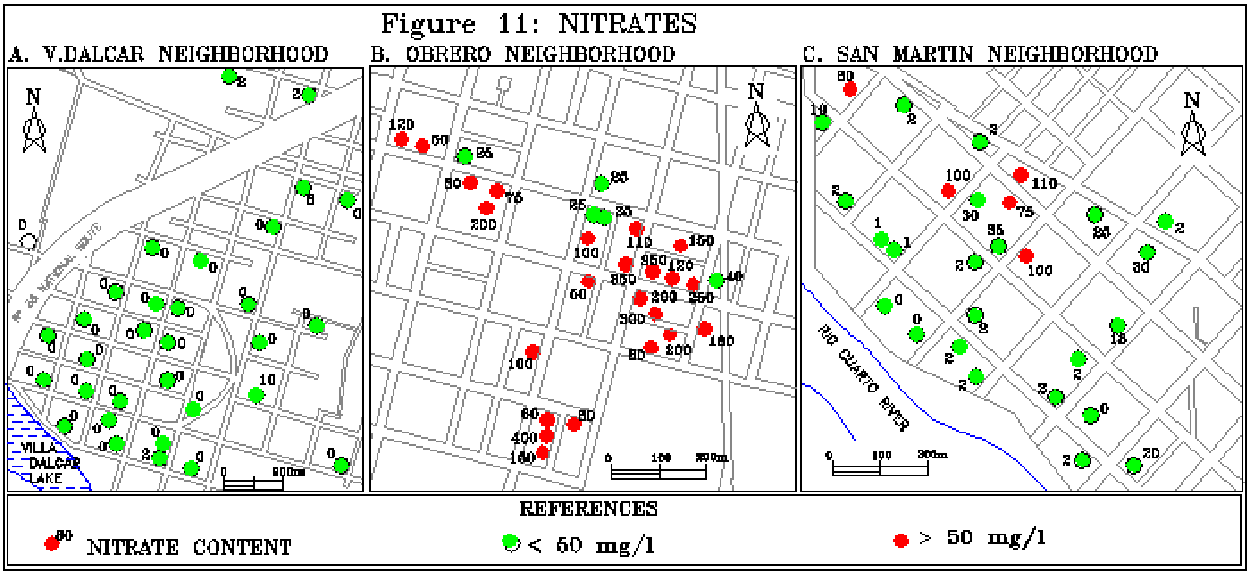


Figure 11: NITRATES

C5. GEOINDICATORS AS TOOLS TO MONITOR CONTAMINATION DUE TO ON-SITE SANITATION SYSTEMS

In addition to the production of risk maps, and to the definition of geochemical contamination models in specific sectors of the city, it is considered of fundamental importance to monitor the environmental changes that are produced as a consequence of the presence of on-site sanitation systems. The geoindicators are an interesting tool to show those changes and to interact with economists, politicians and all those who are in charge of taking decisions.

The geoindicators are measurements (magnitudes, frequencies, rates, and trends) of processes and geological phenomena that occur in or close to the Earth surface and are subjected to modifications that are significant in the understanding of environmental changes through periods of 100 years or less. The geoindicators describe processes and environmental phenomena that are capable of changing without human interference, even though human activities may accelerate, decelerate or deviate natural changes (Goudie, 1990).

The geoindicators help to answer the following basic questions:

1. What is happening in the environment? (conditions and trends)
2. Why is this happening? (causes, relations between human influences and natural processes)
3. Why is this significant? (economic, ecological and health effects).
4. What are we doing about it? (implications for planning and policies).

If the pressure- state- response is followed, the geoindicators can be used to assess:

1) The pressures on environment caused by human activity.
2) The resulting conditions or state in the assessed environment.
3) The political answers to correct undesirable situations.

C5.1 Groundwater as a means to measure environmental changes

Three types of changes in groundwater can be taken into consideration (Edmunds, 1996), which are sensitive to processes that occur in a time scale of 50 to 100 years, the most interesting period to measure geoindicators: 1) Variations in groundwater level (due to human and natural causes). 2) changes in natural chemistry. 3) Impact because of contamination due to human activity on groundwater quality.

The challenge is to select parameters that are representative of the changes in these three processes and that are easily worked at a practical level. It is considered that, in the time scale mentioned, the changes that take place in shallow aquifer systems (0-100 m), probably provide the most relevant indicators.

According to Edmunds (1996), the geochemical processes that can be considered as objectives for the monitoring are the attenuation of acids and mineral dissolution, the redox reactions, and the changes in salinity. In this way the primary geoindicators associated to these processes are alkalinity (like HCO3), the consumption of O_2 and the chloride, electric conductivity and total dissolved solids, respectively. For urban, industrial, and agricultural pollution in a large number of instances, pollution may be recognized indirectly by three proxy indicators, an increase in Cl, DOC and HCO_3 from fermentation of organic matter (Edmunds,1996). These three indicators are all mobile in relation to the complex mixture with which they are associated. They may serve, therefore, as early warning of pollutant migration and are recommended as the most universally applicable primary indicators.

C5.2 The problem of groundwater in the urban zone of Rio Cuarto city in the context of the geoindicators

In order to know what is happening in the environment, the conditions and responses of two main aspects, **water levels and quality**, are being studied in different neighborhoods.

a. **Water levels**: the measure of water level is an element of fundamental importance in order to actualize vulnerability maps, since if the level goes up or down, the degree of vulnerability will also change. Knowledge of water levels is necessary to (i) confirm water resources and (ii) evaluate flood risks and the potential impact on buildings of shallow watertable. An example of this effect on buildings is the significant increase of the water table level registered in a low lying sector to the South of the city, where nowadays groundwater is being pumped, in order to prevent the watertable flooding underground services and basements.

Water quality: it is necessary to continue controlling the contents of major and minor ions, and some important properties (HCO_3, CO_3, Cl, SO_4, Na, K, Ca, Mg, pH, F, As, NO_3, NO_2, DOC, STD, Alkalinity, Hardness). TDS, HCO_3, NO_3, and Cl have been selected as the most important indicators and they will be measured every three months.

Table 1: Indicators of interest to measure in Rio Cuarto city

PRESSURE INDICATORS	**Levels**	Number of inhabitants, endowment, quantity and type of industries, house density, open air spaces, km and frequency of irrigation of dust streets, quantity, distribution, and size of swimming pools.
	Quality	Quantity of sanitation system/ha, nº of sanitation systems in the saturated zone, nº of inhabitants/ha (hydraulic charge), location, type and size of wastes sites, etc.
STATE INDICATORS	**Levels**	Variation in function of time.
	Quality	About groundwater: STD, COD, O_2, HCO_3, Cl, NO_3, faecal coliforms. About health: fluorosis cases, arsenic poisoning, diarrhea, and parasitic illnesses.
RESPONSE INDICATORS	News on radio, TV and the press, decisions of Neighbors Associations, political decisions of the Town Council, works of EMOS (km of pipe/year, families supplied with water and/or water and sewerage/ year), planning decisions of the Secretary of Public Works, planning proposals of the municipal territory, among others.	

It is very important to carry out this monitoring using indicators that can demonstrate the consequences that contamination can have on economy, ecology and health. For instance, if the contamination problems reduce the effective resources available and when this is combined with the great consumption of water (450 l/person/day), the result is that water becomes more scarce and expensive. Certain local problems have forced EMOS to construct deeper wells which sometimes places them in less permeable aquifer sediments materials and as a consequence increases costs. In many cases, the degradation of the groundwater may be irreversible (or at least difficult and expensive to put right). In relation to health, the most frequent diseases are diarrhea, and parasitic illnesses, however in the last few years there have been several cases of hepatitis. However, for the city the statistics do not allow the cause of these diseases to be differentiated, for example whether it is a consequence of contaminated water or contaminated food. Such evidence would require epidemiological studies

Monitoring, using the appropriate indicators and the evaluation of this data should lead to useful planning decisions in these neighborhoods. For instance, improving the location of the wells of EMOS, preserving the water resources, improving the design of on site sanitation systems, etc. It is important to highlight that the investigations that have been done have provided interesting results, like the ones obtained in Villa Dalcar, Quintitas Golf, and San Martin neighborhoods, which did not have running water or sewerage systems. These studies had the support of Neighbors Associations and the media, and the numerous formalities related to this problem also included its treatment by the Town Council. These are considered important response indicators in order to correct undesirable situations. Finally the Town Hall of Rio Cuarto city, taking into consideration that a great number of families and even a school were involved in the problem of contaminated groundwater, decided to supply the neighborhoods with running water. In this way, more than 500 families have benefited.

C6. CONCLUSIONS

- For the first time, the contamination risk and the hydrogeochemical model of the groundwater, that is used for human consumption, were defined in Rio Cuarto city, for areas without sanitation systems. This is considered the fundamental base for some control measures of great urban-social impact that are being applied in relation to population health care. However, it should not be forgotten that groundwater is being polluted. The hydrochemical and hydrodynamic models defined constitute the scientific base to carry out works for the recovery of the aquifer, especially if it is taken into consideration that still there are families that do not have running water.

- The obtained risk for the city varies between very low and very high. The lowest risk areas are the peripheral, less populated ones, which have a greater water table depth and where the sediments of the unsaturated zone are predominantly fine. The categorization of the city in different zones according to different risk degrees is a fundamental base to decide the extension of sanitation systems and the settlement of new neighborhoods.

- The risk maps are dynamic, that is to say, they change with time, since the vulnerability of a certain sector (related to changes in water depth) and the hazard of the contaminant load are also variable. This indicates that the maps should be modified periodically, so the decisions taken are the correct ones.

- The association of important counts of total aerobic microorganisms, total coliforms, *Pseudomonas aeruginosa* and *Escherichia coli* and chemical indicators like nitrates and the increase of the concentration in total salts and chlorides is evidence of contamination derived from the discharge of excreta. The studied neighborhoods present deterioration of groundwater chemical quality and different scenarios of contamination could be determined. In those sectors where predominantly aerobic conditions were found, nitrates were present in all the samples. The samples with the highest nitrate values showed an increase in two other indicators typical of this problem: total dissolved solids and chloride. In those neighborhoods where anaerobic conditions prevail there was no evidence of nitrates (even though species reduced from nitrogen may exist). In the most contaminated samples there is dissolved iron and there is also an increase in TDS and chlorides.

- In order to reduce the concentration of nitrates, it is necessary to stop the discharge of effluents, since the only natural active mechanism to obtain this reduction is the dissolution of this element in the groundwater regional flow.

- Of the environmental factors, water table depth and the grain size of the sediments of the unsaturated zone are the most significant in controlling the arrival of contaminant plumes at the aquifer because of their effect on available oxygen, vulnerability, nutrient transport, etc.

- In relation to human activities, the most important factors in the contamination models described are the population density and the relative positions and depths of cesspools and water wells in the different houses of the neighborhoods. The age of the wells does not seem to be relevant.

- In all the studied neighborhoods, more than 50% of the analyzed samples are not adequate for human consumption because of their microbiological characteristics. The arrival of total coliforms, faecal coliforms, *Escherichia coli* and *Pseudomonas aeruginosa* to the aquifer is variable, depending on all the aspects mentioned in the two previous conclusions.

- With respect to the solution of this problem, it is suggested that the construction of on-site sanitation systems have some kind of control, since they are done without taking into consideration the adequate depths, the distances in relation to water wells, etc. In addition, considering that the vertical permeability of the aquifer is smaller than the horizontal, an option to avoid the arrival of the contaminants is the deepening of water wells and an appropriate design. In this way it would be possible to avoid the vertical groundwater flow that carry away contaminants from the upper part of aquifer.

- There are several specific challenges for the future work with geoindicators. One of them is to define the thresholds or critical loads involved. In this way, it will be possible to express more specifically the relative stability of a particular environment. For those chemical elements present in groundwater, which cause already known diseases, it is necessary to do geomedical studies to establish thresholds of the region, since the applicable values to certain countries may be inappropriate or lack of significance for different climate, geological and social conditions.

- For the case of Rio Cuarto city, of the primary state indicators, the piezometric levels and HCO_3, Cl, TDS, and NO3 are being measured. In addition, the experience in the city indicates that primary excreta contamination bioindicators, associated to these geochemical processes, are the faecal coliform microorganisms. Among the secondary indicators, all the major ions, arsenic, fluorine and their effect on health are being measured. Pressure indicators are perceived through the making of hazard contaminant load maps. Among the response indicators, as social and political awareness of the problem, there are diffusion measures of the media, management measures carried out by neighborhood organisms and those done by the Town Hall, like the decision of providing running water to the families of the studied neighborhoods.

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