Potable Groundwater Supplies and Low-cost Sanitary Engineering — How Compatible?

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Many developing countries are counting on groundwater to supply an increasing proportion of their demand for drinking. There is currently a drive, in many of these same countries, for major improvements in excreta disposal, involving the intensive use of unsewered sanitation or sewage treatment in stabilization ponds with effluent reuse. Under certain hydrogeological conditions these low-cost technologies may be in conflict with the use of groundwater for potable supplies. The principal potential hazards are identified, discussed and illustrated. A more detailed understanding of these problems, coupled with improved design, careful siting and integrated planning of the installations involved, is required to reduce the groundwater pollution hazard.

1. INTRODUCTION

In developing countries, groundwater is increasingly used as a supply of potable water because it is normally the cheapest and safest source. Commonly, large numbers of production boreholes of simple design tap comparatively shallow water-table aquifers and provide individually small, untreated and unmonitored supplies. Unfortunately, such sources may be vulnerable to pollution from the land surface.

Improvements in sanitation are also widely and urgently needed in these countries. It is now accepted that low-cost sanitary engineering technologies are appropriate for developing countries where purely domestic effluents are involved. These low-cost technologies fall into two categories.

(1) On-site excreta disposal units, such as the ventilated pit and the pour–flush latrine. These are suitable for the rural population, in villages and smaller towns and, in some circumstances, in parts of larger towns and cities.

(2) Wastewater treatment in simple unlined stabilization ponds. These are suitable for water treatment where waterborne domestic sewerage systems already exist, before discharge into a surface watercourse or, in arid regions, before reuse for agricultural irrigation or for basin groundwater recharge.

It has long been recognized that the natural soil profile can be an effective medium for the purification of human wastes, including the elimination of fecal microbes and the absorption and/or breakdown of numerous chemical compounds. However, not all soil profiles and their related hydrogeological environments are equally effective in this respect. Current knowledge of the migration and attenuation of the relevant water pollutants is largely derived from investigations in North America and Europe on the effectiveness of septic tank soakaways and on the operation of sewage land-disposal schemes.

It is important to recognize that there are significant differences between septic tanks and

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latrines. These differences result from the following.

(1) Much of the upper 2 m of the soil profile, which is biologically most active and contains the greatest concentration of pathogenantagonistic microbes, is normally removed during the latrine construction, but is normally present below septic tanks soakaways.

(2) The hydraulic loading of septic tank soakaways should be designed not to exceed about 30 mm/d, whereas substantially higher loading (80–120 mm/d) is inevitable in pour–flush latrines.

(3) Fecal bacterial populations and pathogen counts are much higher in many latrines in developing countries (10^9 FD/100 ml) than they are in septic tanks.

(4) The use of septic tanks is usually restricted to low density urban or rural settlements, whereas latrines are being installed at very high density in some villages and towns of developing countries.

There are also significant differences between sewage land disposal schemes and the wastewater reuse schemes being introduced in some arid regions of the developing world. These differences come about for the following reasons.

(1) The normal method of wastewater treatment in developing countries involves retention in shallow unlined oxidation or facultative stabilization ponds, which may have high rates of seepage loss, especially after initial construction and subsequent cleaning.

(2) In developing countries both the raw wastewater and the stabilized effluent often have very much higher fecal bacteria counts.

(3) It is not currently realistic in developing countries to consider chlorination of the stabilized effluent as part of the wastewater treatment process.

(4) In developing countries the land normally chosen for agricultural irrigation is freely draining, to reduce initial farming investment and to avoid soil management problems. The result is that the irrigation return waters can readily percolate to groundwater.

2. PRINCIPAL GROUNDWATER POLLUTION HAZARDS

2.1 PATHOGENIC BACTERIA AND VIRUSES

Early research on the attenuation of fecal bacteria in the subsurface, including some directly relevant work on latrines, showed that:

(1) in soils, fecal coliforms were rarely found to penetrate more than 1 m in depth and

(2) when fecal coliforms were discharged directly at the water-table, lateral migration did not generally exceed 10 m in the direction of ground water flow.

A serious limitation of these early investigations, however, was that they were essentially limited to unconsolidated sediments with a mean grain size of less than 300 μm, at sites where the saturated groundwater flow-velocity probably did not exceed 1.0 m/d (Lewis *et al.*, 1982).

Numerous incidents of fecal groundwater pollution from excreta and wastewater disposal systems have been recorded, and no doubt countless more have occurred at sites with:

(1) only a thin cover (less than 3 m) of permeable unconsolidated sediments overlying fissured non-porous bedrock aquifers;

(2) a groundwater table which is seasonally or perenially shallow (at less than about 3 m depth) or

(3) a high hydraulic loading of wastewater (greater than about 30 mm/d).

For example, gross pollution with fecal bacteria of some water-supply boreholes in a Botswana village was traced to pit latrines generally of the order of 20–40 m distant (Lewis *et al.*, 1980). The hydrogeological conditions involved were a low-transmissivity weathered basement aquifer with low rates of seasonal rainfall recharge and a groundwater table at 4–6 m depth. Pathogenic species were detected, including some strains exhibiting resistance to common antibiotics. Much longer lateral distances of baterial travel have been documented, even in unconsolidated strata (Lewis *et al.*, 1982). Islands and peninsulas of caustic limestone with only thin soil cover (e.g. Jaffra, Sri Lanka and Yucatan, Mexico) are also known to be especially vulnerable to groundwater pollution from unsewered sanitation.

Less is known about the migration and attenuation of excreted pathogenic viruses; they are too small (0.01–0.1 μm) to be retained by porous-media filtration and their infective dose may be orders-of-magnitude less. Retardation of virus penetration into the subsurface is believed to be largely dependent upon surface absorption and subsequent degradation.

Infiltration into and through the unsaturated zone (above the water-table) affords the most important line of defence against fecal pollution of the underlying aquifers. Unsaturated vertical hydraulic conductivity decreases, often dramatically, with decreasing moisture content and increasing moisture tension (Fig. 1). In consequence, for many soils and some rocks actual groundwater flow rates in the unsaturated zone do not average more than about 0.2 m/d for hydraulic loadings of up to about 30 mm/d, and unsaturated zone residence times are such as to allow gross attenuation of fecal pathogens though a combination of mechanical straining, surface adsorption and degradation by antagonistic bacteria. However, under conditions of heavy artificial hydraulic loading or the occurrence of very high intensity infiltrating rainfall, moisture tensions will fall and groundwater flow velocities may be much higher, especially in consolidated

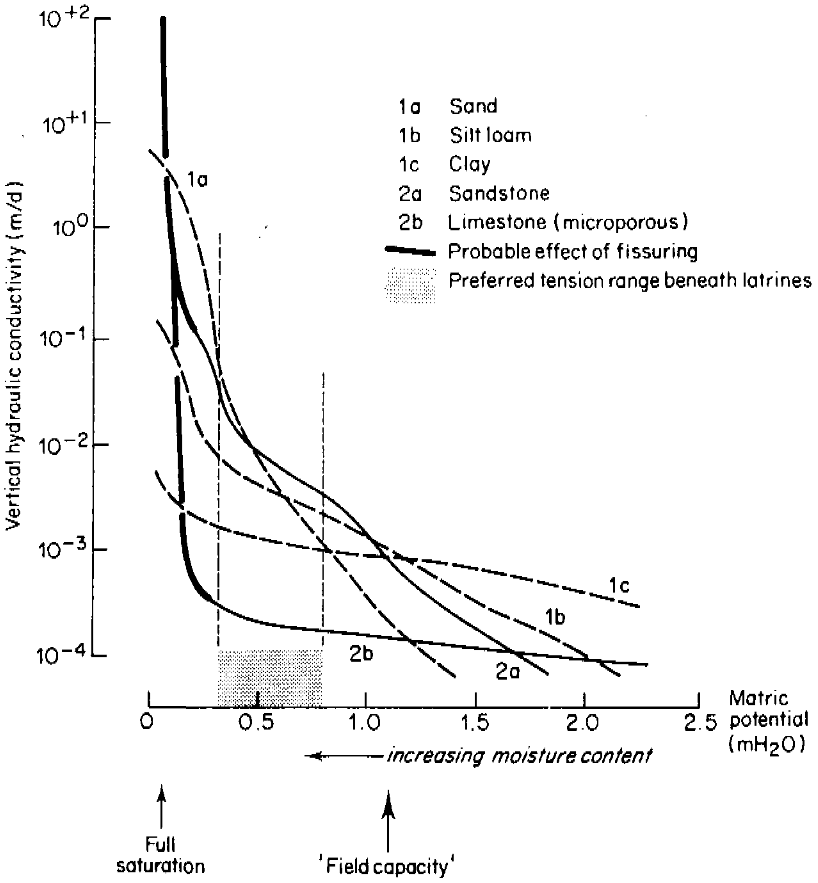

Fig. 1. *In situ* hydraulic conductivity–moisture potential (tension) relationship for representative unconsolidated and consolidated sediments.

fissured formations (Fig. 1), reducing greatly the opportunity for pathogen elimination. In soils, fecal pathogens normally survive less than 30 days, with 90% population reduction in 10–15 days, but under cool, moist and alkaline conditions a hardy residual fraction may survive for many months.

The processes responsible for the elimination of excreted pathogens, both bacteria and viruses, will generally be much less active below the water-table, where longer survival periods can be expected. Since many aquifers are heterogeneous and contain some highly permeable flow paths of relatively small cross-sectional area, groundwater flow velocities may often exceed 10 m/d, and reach 100 m/d or more near production boreholds in fissured formations. Thus, in many hydrogeological environments increasing lateral separation between the potential source of fecal pollution and a potable groundwater supply source is not an effective way of increasing protection against pollution, as large and often impractical distances are required.

Unfortunately, there is pressure to reduce the normally accepted minimum spacing of 15 m between excreta disposal unit and groundwater source (under favorable site conditions) to as little as 5 m in some developing countries, such as in Bangladesh and in parts of India and Sri Lanka. This often results from lack of space in very densely populated settlements, but can also occur in more-prosperous and well-organized settlements served by latrine sanitation, with the tendency for individuals to construct private digwells or tubewells to replace or augment properly sited communal groundwater sources.

2.2 NITROGEN SPECIES AND SALINITY

The build-up of nitrogen compounds in groundwater, while not representing as immediate a hazard to health as contamination by fecal microbes, is likely to be far more widespread and persistent.

An indication of the magnitude of groundwater nitrate pollution which is possible from on-site excreta disposal units can be derived from the following considerations: a population density of 100 persons/ha represents a discharge of some 500 kg N ha^{-1} year^{-1} to the ground which, if oxidized and leached by 100 mm/year of infiltrating rainfall, would result in the local groundwater recharge containing 500 mg NO$_3$-N/l (2200 mg NO$_3$/l). The concentration will be lower if:

(1) nitrogen remains immobile in organic forms until removed from latrines for land application or

(2) ammonification of denitrification of latrine contents has led to gaseous discharge of nitrogen to the atmosphere.

However, even if only 10% of the nitrogen in excreta is nitrified and leached, local groundwater recharge may reach very high nitrate concentrations in relation to WHO limits for potable water (Fig. 2); 50 ml NO$_3$-N/l (220 mg NO$_3$/l) in the case cited above.

A further decrease could only result if any of the following processs are significant:

(1) there is dilution from regional groundwater flow;

(2) there is denitrification within the groundwater system itself or

(3) there is significant dilution from the effluent liquid itself, as may be the case for pour–flush latrines.

Nevertheless, it must be expected that unsewered sanitation schemes will have a major impact on groundwater nitrate concentrations, especially in arid regions without significant regional groundwater flow. Not much systematic field data on nitrate pollution in groundwater of developing countries has yet been collected, but such pollution has been demonstrated in Botswana (Lewis *et al.*, 1980), where domestic water-supply boreholes and pit latrines have co-existed in certain major villages for some time. In 1976 a survey of two villages drawing their groundwater supplies from a shallow weathered basement aquifer with a seasonal rainfall recharge averaging about 50 mm/year found the nitrate concentrations of groundwater from within the urban limits were generally in excess of 20 mg NO$_3$-N/l, and in some cases 50 mg NO$_3$-N/l. By contrast, in the surrounding largely uncultivated areas concentrations were generally less than 5 mg NO$_3$-N/l. For one village this result has been confirmed by more-detailed sampling in 1982, and seasonal fluctuations in concentration have been demonstrated. As a rule, chloride concentrations will increase with those of nitrate and, in the case of these two villages, values always exceed 200 mg/l

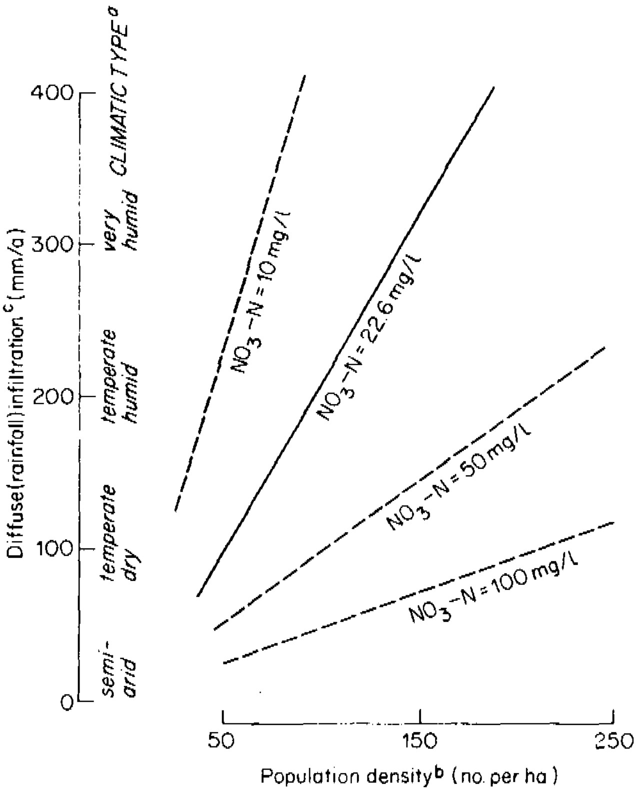

Fig. 2. Grossly simplified estimation of impact of unsewered sanitation schemes on nitrate concentration in local groundwater. [a]General indication only, depends on soil–vegetation system and proportion of surface runoff. [b]Assumes initial generation of 5 kg N/year per capita but 90% return, directly or indirectly, to atmosphere. [c]In some cases dilution from regular aquifer throughflow, streambed infiltration or effluent liquid itself will also be significant.

(and in most cases 300 mg/l). To overcome the pollution problem in these two villages potable water-supplies have had to be reticulated to village standpipes from boreholes outside the urban limits, and efforts are being made to protect these boreholes from new settlements and from cattle enclosures.

Stabilized wastewater effluent will generally contain nitrogen in excess of 20 mg N/l, mainly in organic and ammoniacal forms, but chloride and sulfate contents will vary widely, primarily with the corresponding levels in the associated water-supply. For the purposes of the present discussion, values from Lima, Peru, will be used since good data are available (Fig. 3). In this arid coastal region, as in various parts of Mexico and elsewhere, wastewater is increasingly being used for agricultural irrigation, and many of the non-plant nutrients will be leached from the soil by irrigation return water and recharge local groundwater. Concentrations will depend essentially on the rate of irrigation and on any infiltrating rainfall; very large concentrations of chloride and sulfate may be reached in the absence of the latter, especially at irrigation rates only slightly in excess of crop water requirements (Fig. 3). The situation with respect to nitrate is more complex because of such factors as the variable rate of nitrogen mineralization in soils, crop uptake and gaseous loss associated with soil denitrification (Lance *et al.*, 1976; Foster and Young, 1980). However, the gaseous loss associated with soil denitrification is least likely for cultivated

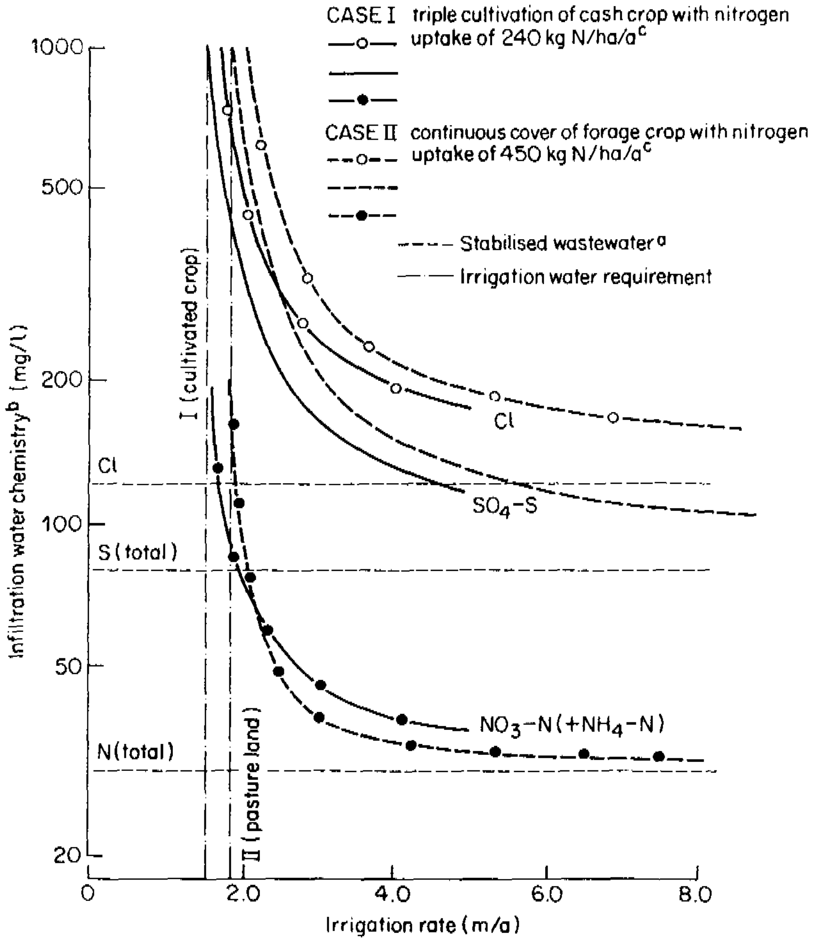

Fig. 3. Grossly simplified estimation of potential nitrate and salinity problems in groundwater recharge from agricultural land irrigated with wastewater in arid climates. [a]Composition may vary significantly with water-supply quality, unit water consumption, proportion of industrial effluent, groundwater infiltration to sewers and stabilization lagoon characteristics. [b]Assumes wastewater is the only source of irrigation water and that local rainfall is insignificant. [c]Additional to that generated by soil mineralization (no losses by soil denitrification or other processes).

soils, and it must be anticipated that very high NO_3-N concentrations will be found in irrigation return waters (Fig. 3).

2.3 TRACE ORGANIC COMPOUNDS

Not enough is known about the migration, transformation and attenuation of organic compounds in groundwater systems. However, the corresponding pollution risk should not be dismissed, even where sewage is entirely of domestic origin, because it is likely to contain increasing numbers and concentrations of synthetic organic compounds derived from common community chemicals such as detergents, disinfectants and solvents.

The migration of most soluble organic compounds will be retarded by absorption, and the more biodegradable types will be eliminated.

However, increases in the concentrations of anionic ABS detergents and phenols have been reported in groundwater from some areas where land disposal of sewage effluents is practised and in such areas an increase in total (dissolved) organic carbon is also commonly observed (e.g. Hughes and Robson, 1973; Baxter, 1982). In the absence of measurable biochemical oxygen demand (BOD), this must represent an increase in more-persistent (essentially refractory) organic compounds. Little is known about these compounds, but the potential for generating widespread trace organic contamination of groundwater must be recognized. The risk of pollution will be greater where artificial hydraulic loading is highest and where waterborne sewage systems are involved.

3. CONCLUSIONS AND RECOMMENDATIONS

In general, it can be concluded that for a given condition of hydraulic and pollutant loading:

(1) the depth of water-table and the character of strata in unsaturated zone will be the principal factors controlling the degree of penetration of pathogens and biodegradable organics into groundwater and

(2) the amount of dilution by regional flow and local recharge will be the dominant factors controlling the final concentration of nitrates and of the more-persistent trace organic compounds in groundwater.

However, it remains difficult to quantify the groundwater pollution risk for a sufficiently wide variety of hydrogeological conditions. A more-detailed understanding of the character and scale of these problems is required to allow effective planning of low-cost groundwater supplies and sanitary engineering measures on an integrated basis.

Except perhaps in the most unfavourable hydrogeological conditions, the groundwater pollution risk associated with the low-cost sanitary engineering measures being introduced is not such as to warrant the abandonment of such activities. Rather, it should be more widely recognized that a hazard exists, and that significant degradation of groundwater quality may often result unless substantial design modifications to the sanitary engineering measures are made. Some of the groundwater quality problems are likely to be surprisingly slow to appear in water-supply sources, but to prove exceptionally persistent once having appeared.

Numerous measures can be considered to reduce the risk of pollution of groundwater supplies, and may prove to be technically feasible, scientifically justified and economically viable under some conditions. These include the following.

(1) For sewered sanitation to:
  (a) incorporate artificial filter media in the base of latrines to improve retention and elimination of pathogens;
  (b) discourage excessive deepening of latrines and reduce fluid hydraulic loading by increasing latrine area–depth ratio, thus increasing pollutant retention time in the unsaturated zone;
  (c) impose minimum separations between groundwater supply boreholes and pit latrines considerably greater than the normally accepted minimum of 15 m;
  (d) ensure effective emptying of latrine pits when contents are stabilized and application of the nitrogen over a broad area, rather than allowing it to accumulate locally in the ground as very often occurs at present and
  (e) artificially suppress mineralization of organic nitrogen in latrines, thus reducing the seepage of ammonium and/or nitrate in the groundwater.
(2) For waste water stabilization ponds and irrigation reuse areas to:
  (a) control location with respect to hydrogeological conditions and to groundwater supply installations and prohibit the use of groundwater for potable supplies with a defined zone;
  (b) consider lining, or otherwise sealing, the base of ponds to reduce seepage;
  (c) increase pond area–depth ratio in order to reduce artificial hydraulic, pathogen and nutrient loading, and increase pollutant residence time in the unsaturated zone;

(d) select the types of crop grown, to reduce mineralization and/or increase uptake of nitrogen and thus reduce nitrate leaching losses and

(e) control irrigation schedules to maximize soil denitrification and dilute the concentrations of other mobile non-nutrient salts which have been leached.

REFERENCES

Baxter, K. M. 1982. The effects of the disposal of sewage effluents on groundwater quality in the United Kingdom. *Wat. Services*, **86**, 219–220.

Foster, S. S. D. and Young, C. P. 1980. Groundwater contamination due to agricultural land-use practices in the United Kingdom. *UNESCO Studies and Reports Hydrology*, **30**, 268–282.

Hughes, J. L. and Robson, S. G. 1973. Effects of waste percolation on groundwater in alluvium near Barstow, California. *Proc. 2nd AAPG–USGS–IAHS Symp. Underground Waste Management & Artificial Recharge*, Vol. 1, pp. 91–129.

Lance, J. C., Whisler, F. and Rice, R. C. 1976. Maximizing denitrification during soil filtration of sewage water. *J. Environ. Qual.*, **5**, 102–107.

Lewis, W. J., Farr, J. L. and Foster, S. S. D. 1980. The pollution hazard to village water supplies in eastern Botswana. *Proc. Inst. Civ. Engrs II*, **69**, 281–293.

Lewis, W. J., Foster, S. S. D. and Drasar, B. S. 1982. The risk of groundwater pollution by on-site sanitation in developing countries: a literature review. *IRCWD (Dübendorf) Report 01-82*, pp. 1–79.