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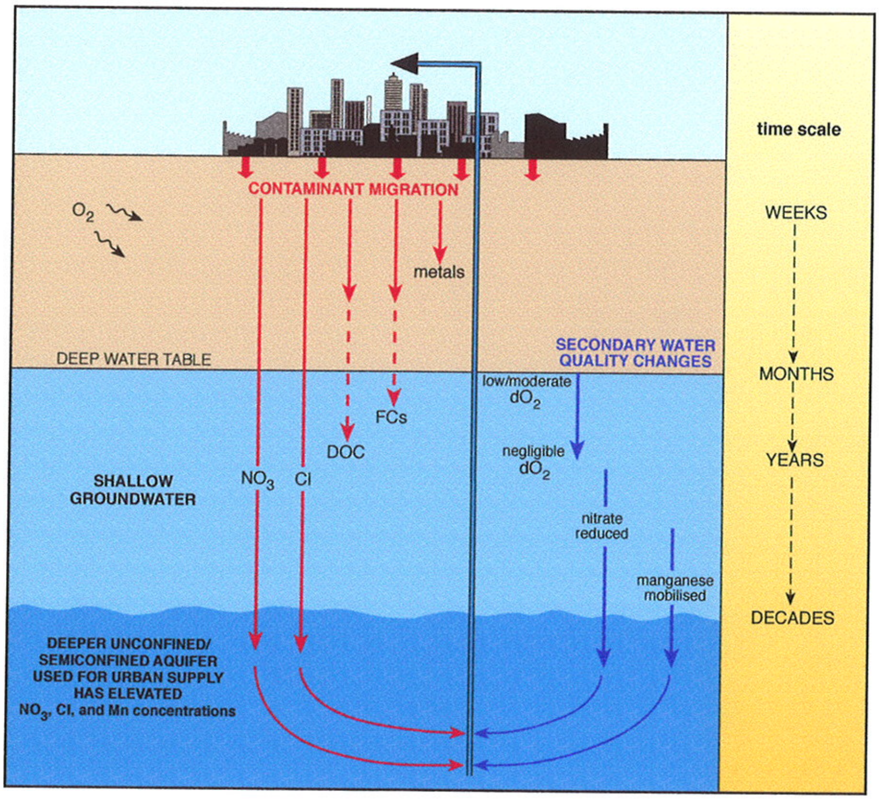
THE STUDY OF THE POLLUTION RISK TO DEEP GROUNDWATERS FROM URBAN WASTEWATERS: PROJECT SUMMARY REPORT

British Geological Survey, Hydrogeology Group



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Lawrence AR, Morris BL, Gooddy DC, Calow R and Bird MJ.

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The reports on the cities of Hat Yai in Thailand and Santa Cruz in Bolivia, on which the observations made in this report are based, are referenced;

British Geological Survey, Department of Mineral Resources Thailand, Prince of Songkhla University Thailand, 1997. Assessment of Pollution Risk to Deep Groundwaters from Urban Wastewaters : Hat Yai City Report. British Geological Survey Technical Report WC/97/16, BGS Keyworth, UK.

Gooddy DC, Wagstaff SJ and Lawrence AR, 1997b. Assessment of Pollution Risk to Deep Aquifers from Urban Wastewaters: Hat Yai Data Report. British Geological Survey Technical Report WC/97/44, BGS Keyworth, UK.

British Geological Survey and Cooperativa de Servicios Públicos "Santa Cruz" Ltda, 1997. Assessment of Pollution Risk to Deep Aquifers from Urban Wastewaters: Santa Cruz City Report. British Geological Survey Technical Report WC/97/11, BGS Keyworth, UK.

Gooddy DC, Wagstaff SJ, Morris BL 1997a. Assessment of Pollution Risk to Deep Aquifers from Urban Wastewaters: Santa Cruz Data Report. British Geological Survey Technical Report WC/97/45, BGS Keyworth, UK.

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EXECUTIVE SUMMARY

Groundwater is an important source of water supply for many cities being both relatively cheap to exploit and generally of good quality. However, in many developing countries the subsurface from which groundwater is obtained is also widely used for the disposal of wastewaters. As a consequence shallow groundwater is frequently polluted beneath many unsewered cities. Abstraction from deeper aquifers can reverse vertical water level gradients and induce substantial leakage from shallow layers. Currently the deeper groundwaters beneath cities are usually of high quality being largely derived from recharge which occurred many decades if not centuries previously. The security of this valuable resource and the risk posed by leakage of shallow urban groundwaters induced by pumping is of major concern.

Accordingly the Department for International Development (DFID) agreed to fund Project R5975 'The study of pollution risk to deep groundwaters from urban wastewaters' under the DFID Technology Development and Research (TDR) Programme. Results from this project suggest that leakage from beneath cities makes a substantial contribution to recharge to these deeper aquifers although travel times (from shallow layers) are typically 10-40 years. These timescales ensure that only the most mobile and persistent contaminants are likely to reach these deeper aquifers. For these reasons nitrogen, chloride and the chlorinated solvents represent the contaminants of most concern.

However, this research suggests that whilst nitrogen and chloride have penetrated to deeper aquifers, their concentrations do not represent a significant health concern even though, in the case of nitrogen, concentrations may exceed guidance values. Urban wastewaters in many developing countries appear to have lower concentrations (both actual and relative) of synthetic organics than corresponding wastewaters in N.America and Europe. This may change as the developing countries industrialise.

An important, if surprising, result of this research is that secondary water quality changes, caused by a reduction in the redox potential of the groundwaters, can have a major and possibly more serious impact producing increases in iron and manganese groundwater concentrations. This water quality deterioration may require costly treatment. In addition arsenic (which appears to be associated with iron mobilisation) has been identified as a significant water quality concern; concentrations 20 x WHO guidelines have been observed in shallow groundwater as a result of redox changes caused by seepage to the ground of urban effluent containing a high organic load. Mobilisation of arsenic in deeper aquifers has also been confirmed. The health implications of excessive arsenic concentrations in drinking water are significant and therefore an important recommendation from this research is that urban groundwaters need to be monitored for arsenic especially where reducing conditions producing iron mobilisation prevail.

A simple methodology for assessing risk, based on relative importance of urban derived leakage, probability of significant water quality changes and the likely impact on the user and or use has been developed. A key requirement to any assessment of likely impact is information on groundwater use and user.

Where problems already exist or are likely to develop it is important to reduce contaminant loading at surface and/or reduce vertical head gradients. Where the problem is entrenched these measures are at best likely to provide a solution in the longterm only. Control and regulation of groundwater use is difficult to achieve, especially as in may Asian cities, groundwater is abstracted by a large number of private users. The cost of treatment or of providing alternative sources of water supply is prohibitive.

Whilst the continued use of the ground both for waste disposal and for groundwater abstraction for urban water supply can be sustainable under some circumstances, it is, as a general rule, good practice to restrict the deep groundwater for potable supply only. Shallow groundwater abstraction for non-sensitive uses should be encouraged.

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1. INTRODUCTION

This report summarises the findings and conclusions of project R5975 'The study of pollution risk to deep groundwaters from urban wastewaters' carried out by the British Geological Survey (BGS) between 1994 and 1997 with funding from the DFID under the DFID Technology Development and Research (TDR) Programme. It is directed towards the non-specialist reader.

The report reviews the risk posed to deep groundwaters by the disposal of urban wastewaters to the subsurface (via on-site sanitation systems or to surface-water courses). The principal objectives and scope of work are summarised in Section 3, the principles of aquifer vulnerability and water quality modifications are described in sections 4 to 6 and the results and their implications are discussed in sections 7 to 10. Summaries of the supporting field investigations carried out in two cities (Hat Yai, Thailand and Santa Cruz, Bolivia) are presented here; full details are presented in separate case study reports (BGS et al 1997a, b).

2. BACKGROUND

Many cities in developing countries are dependent upon groundwater. Perhaps the most important aquifers occur in unconsolidated alluvial and alluviovolcanic deposits, as they usually possess sufficient porosity and permeability to meet the high urban water demand. Aquifers within unconsolidated sediments supply some of the worlds largest cities including Mexico City, Beijing and Jakarta in addition to numerous medium-sized cities.

Alluvial deposits typically comprise permeable sands and gravels (aquifers) together with less permeable fine sands, silts and clays (aquitards). It is common in thick alluvial deposits for several aquifers to be present, separated by aquitards; water levels in these aquifers can be significantly different, producing vertical flow. Furthermore, in those deeper aquifers little influenced by pumping, groundwater can be old and may be derived from recharge which occurred hundreds, if not thousands, of years previously. These slow travel times in natural groundwater systems are a consequence of low hydraulic gradients and significant aquifer porosity. Thus, thick alluvial deposits may comprise complex multi-aquifer systems and exhibit significant water quality and age stratification (Box 2.1).

The introduction of deep pumping to obtain further or better quality water can transform the flow pattern in such systems by reversing vertical hydraulic gradients if they were previously upward, or by steepening naturally occurring downward gradients. Recent, possibly polluted, groundwaters circulating at the near surface, may migrate to deeper aquifers as a result of pumping-induced leakage (Figure 2.1). Eventually as a consequence, the water pumped from the deeper aquifer will be a mix of water derived from shallow layers (which may be modern - several years or decades old - and likely to contain various anthropogenically derived chemicals) and pristine groundwater derived from old recharge perhaps hundreds of years previously. This picture is further complicated by chemical transformation of the shallow groundwater during its migration to the deeper aquifer. These transformations result from precipitation, sorption, ion exchange, degradation, and dissolution processes and some of these are discussed in detail in Sections 5 and 6.

An attempt is made here to present a simplified classification of aquifer types (their hydraulic setting) as follows:

(i) *Unconfined aquifer* - the water table forms the upper surface of this aquifer, active groundwater circulation is typically - years - decades, whilst the travel time from ground surface to the water table is generally weeks - months - years.

(ii) *Semi unconfined* - this aquifer is not overlain by an impermeable layer, but its upper surface underlies the shallow unconfined aquifer which is typically less permeable than the deeper system. Active circulation is typically decades to centuries. Travel time from ground surface to the aquifer is normally years - decades.

(iii) *Semi confined* - a semi permeable confining layer overlies this aquifer; circulation is typically decades - centuries or more. Travel time from ground surface to the aquifer is generally decades.

These three aquifer types and their hydraulic settings are illustrated in Figure 2.2. This is very much a simplification and in natural systems there is often a gradation from unconfined to semi-unconfined to semi-confined conditions.

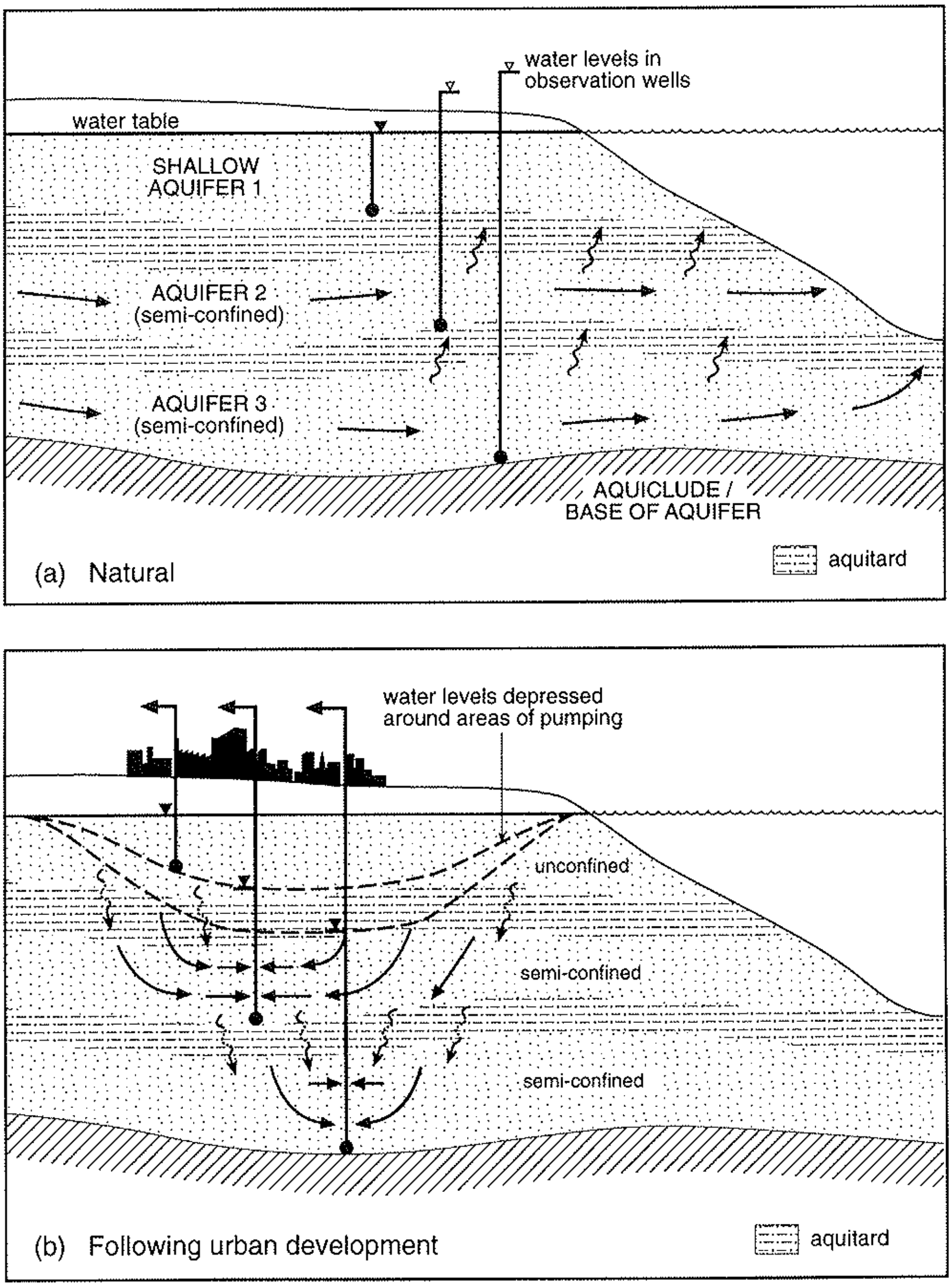
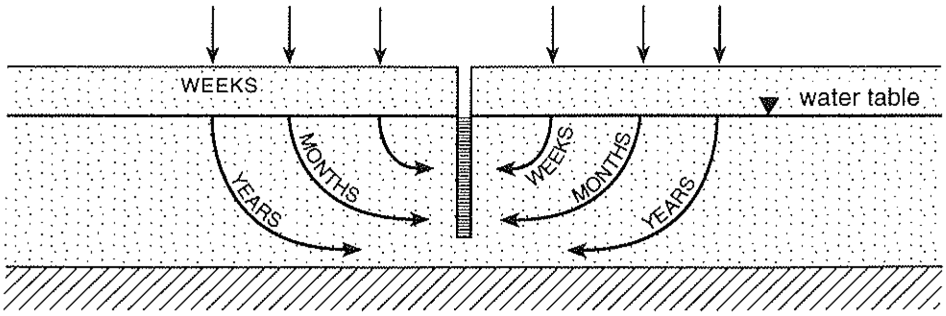
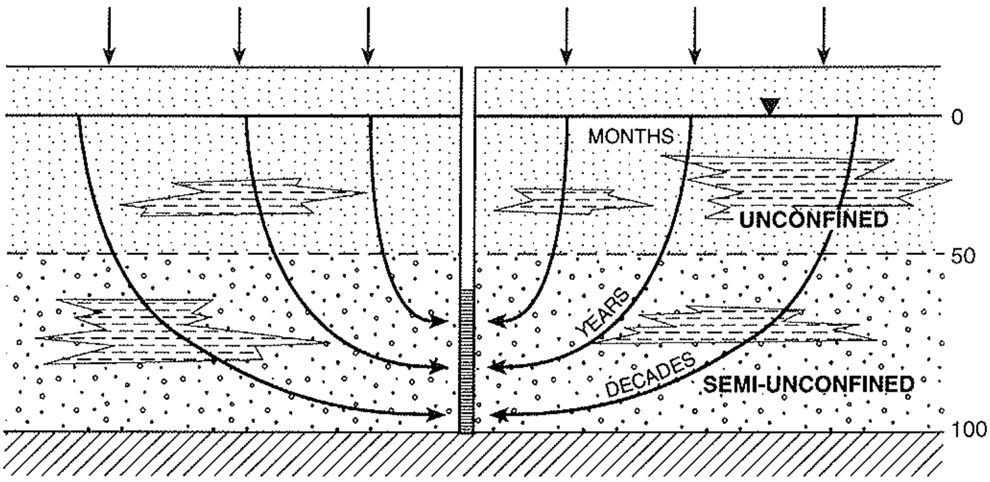


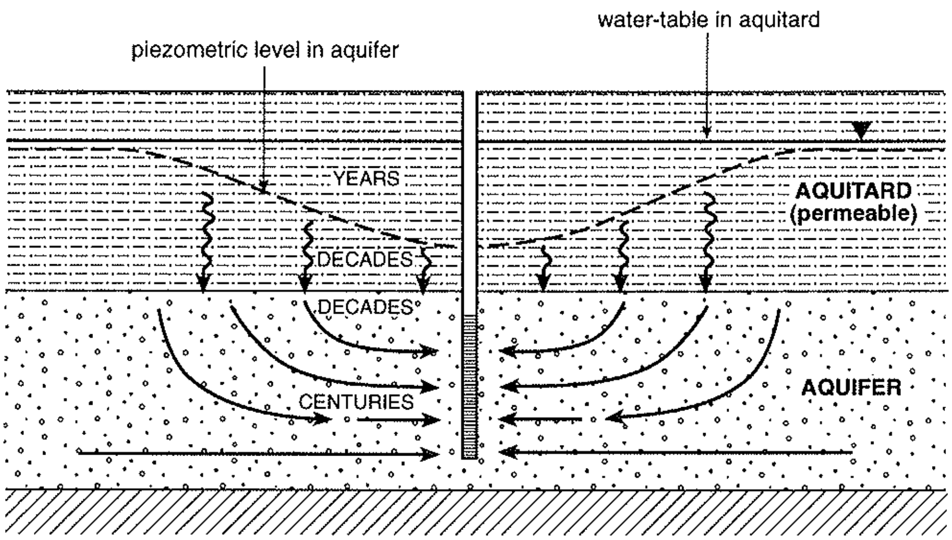
Figure 2.1 Modifications to groundwater flow induced by pumping from deep aquifers



(a) Shallow unconfined aquifer (<50m deep)



(b) Semi-unconfined aquifer (50-100m deep)

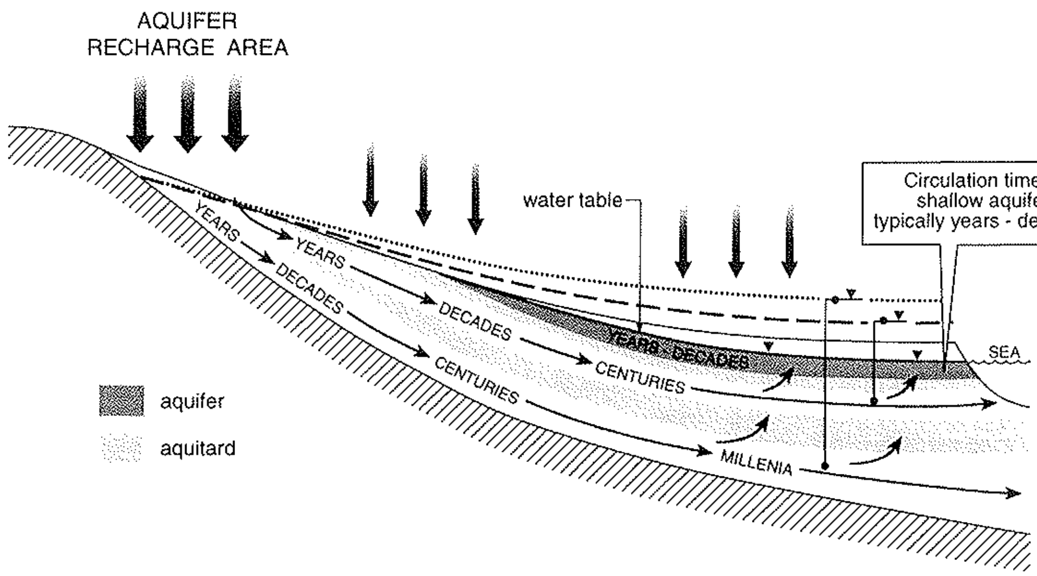


(c) Semi-confined aquifer

Figure 2.2 Aquifer types and their hydraulic setting

BOX 2.1 GROUNDWATER FLOW

Groundwater systems are dynamic and normally water is continuously in slow motion from zones of recharge to areas of discharge. In unconsolidated alluvial aquifers, the shallow aquifer is often unconfined with a water table, which is the level to which the ground is fully saturated. It receives recharge both from rainfall and from rivers. Thus the recharge is recent in origin and travel times are typically measured in months, years or decades. For deeper confined aquifers, the discharge point from the aquifer may be a long way from where the recharge occurred, and this drives groundwater flow. Hundreds or even thousands of years may elapse in the passage of water through this subterranean part of the hydrological cycle, since flow rates do not normally exceed a few metres per day and can be as low as 1 metre per year or less.



Groundwater flowpaths and travel times in deep groundwater system

Down-gradient and away from recharge areas, the aquifer may become deeper, as it is progressively covered with other alluvial beds. Some of these strata may be permeable, forming their own *aquifers*, others may be poor transmitters of water, restricting but not completely preventing the vertical passage of water, so the aquifer is then said to be semi-confined, below an *aquitard*. Yet other beds are practically impermeable and even more effective as confining layers, called *aquicludes*. The head or surface to which the water from a given aquifer layer will rise is called the *piezometric* head or surface.

Deep alluvial systems may be made up of many such layers, and form complex multiple layer systems. In such systems, either upward or downward vertical leakage may occur, driven by the difference in piezometric head between adjacent layers. For instance, low-lying coastal plains often form groundwater discharge areas for the deeper aquifers and although the permeability of the aquitard hinders upward flow, substantial leakage can occur because of the large areas involved and the significant upward gradients possible. In contrast, downward gradients are often created where the deeper aquifers are pumped [and quite steep gradients can be created around wellfields or other zones of intensive pumping].

3. THE PROBLEM

3.1 Key issues

Many of the cities which are located on alluvial deposits and which obtain a significant component of their water supply from underlying aquifers also dispose to the ground much of their wastes. Key issues, therefore, are whether these practices are compatible with the long term use of the aquifer for potable supplies and what measures are required to mitigate the negative impact of waste disposal on groundwater quality.

In the early stages of city development it is often the shallow unconfined aquifer which is utilised for water supply. However, as development proceeds and the loading of urban wastewater increases, this upper aquifer becomes severely polluted. Widespread contamination of shallow urban aquifers has been extensively reported (Gunasekaram 1983, Sahgal et al 1989, Morris et al 1994). As a consequence the shallow groundwater is largely abandoned, at least for public supply and deeper aquifers are usually exploited instead. Sometimes deeper aquifers are preferred anyway being considered either immune to pollution from shallow subsurface disposal or buffered by drawing from deep throughflow.

Initially water quality in these deeper aquifers is excellent, but with time a deterioration in quality is commonly observed (Boonyakarnkul et al 1992). This deterioration is usually attributed to the downward leakage of shallow contaminated groundwaters (BGS and MoPH 1994) and as referred to in Section 2, results from deep pumping which causes profound changes to the water level distribution within the sediments. For example, steep downward vertical gradients have been observed in Jakarta as a consequence of pumping from deeper aquifers (Figure 3.1). An important issue is therefore how severe is water quality deterioration likely to become and what are the social and economic implications.

3.2 Project description

3.2.1 *Objectives*

Key questions that are considered in this report are:-

(i) How important can leakage from beneath the city be to overall recharge to a deeper urban aquifer?

(ii) How long does it take for water to move from the shallow to the deep aquifers?

(iii) What pollutants present in the shallow aquifer are likely to migrate to deeper aquifers and, what attenuation or retardation might be anticipated?

(iv) What are the likely social and economic implications of the above?

(v) What water management principles should be followed to mitigate the impact of leakage of contaminated groundwaters, or where the problem is already entrenched, how can it best be rectified?

Whilst answers to these problems for a given city area are likely to be location specific, the questions are fundamental to many hundreds of individual urban water supply settings, and this report seeks to provide generic answers which can be adapted to the detail required for application to the particular.

3.2.2 *Scope of project*

Two cities were studied; where deeper aquifers are being exploited for urban water supply in contrasting hydrogeological environments; Hat Yai, Thailand and Santa Cruz, Bolivia. The details of these two case studies are presented elsewhere (BGS et al 1997a, b).

This report reviews these case studies, together with other published data in order to provide:

- an assessment of the generic problem
- an understanding of the processes
- a methodology for assessing the problem
- an indication of management options to mitigate problems

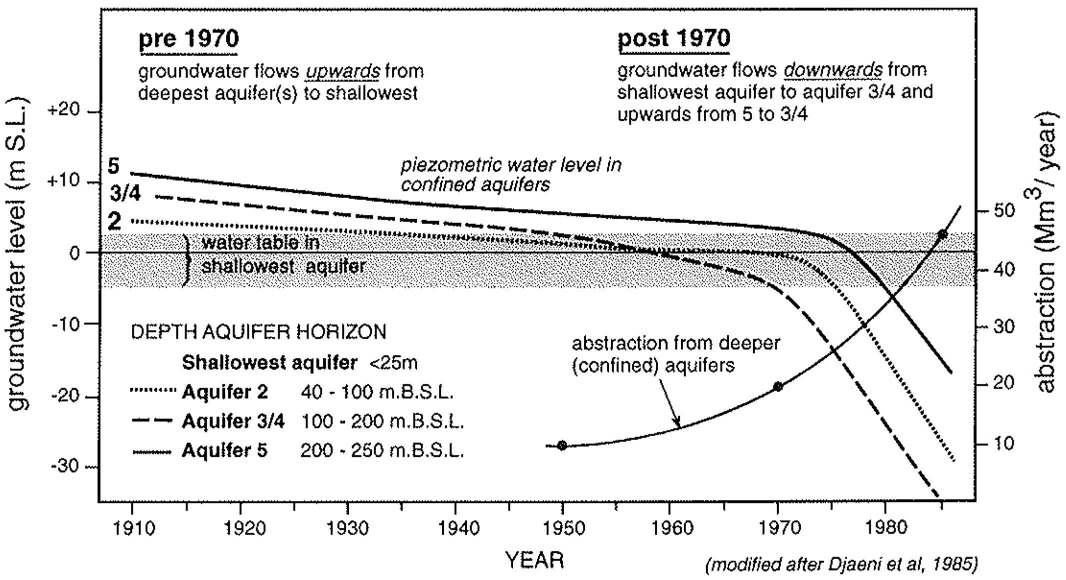


Figure 3.1 Reversal of vertical groundwater gradient beneath Jakarta, Indonesia

4. AQUIFER TYPES AND VULNERABILITY TO POLLUTION

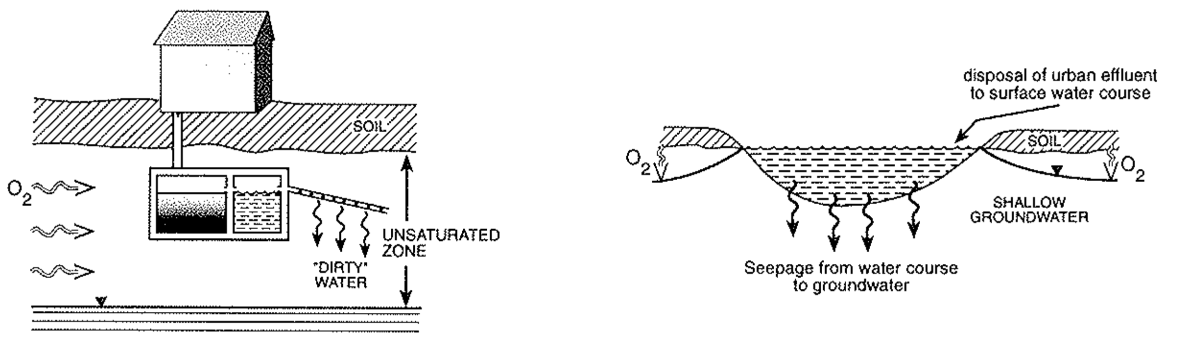
There are many ways of classifying aquifer pollution vulnerability. The scheme adopted here is based on the depth of/to the aquifer and the thickness of its unsaturated zone (Table 4.1). This classification is broadly based on the travel time (from ground surface to the top of the aquifer). The greater the travel time the greater the opportunity for the aquifer to attenuate contaminants by filtration, sorption, dilution, volatilization and degradation processes.

Travel times from the surface to the water table are typically in the range days-years depending upon the thickness of the unsaturated zone, although where the recharge rate is so high as to impose a significant hydraulic loading, more rapid movement is possible.

For deeper unconfined (or semi-confined) aquifers, travel times to the upper boundary of the aquifer are likely to be in the range years-decades (Figure 2.2). In Santa Cruz, the rate of downward migration of modern urban recharge due to pumping was estimated to be about 3 m/a (BGS et al, 1997b) so that travel times from the surface to intermediate depth aquifer horizons (50-100 m) are approximately 20-30 years. For deeper aquifer horizons in Santa Cruz (>100 m) 'modern' water is only now arriving at the upper surface in some areas indicating travel times of more than 30 years (Box 4.1). The age of the pristine deep groundwater beneath Santa Cruz is uncertain but maybe many centuries old.

In Hat Yai, it has been estimated that it takes 20-40 years for water to migrate from the surface and to 'break through' the aquitard into the underlying semi-confined aquifer (BGS et al 1997a). This estimate was based on results of modelling and on analysis of the water quality deterioration in the aquitard (Box 4.2). It is apparent in both cases that currently pumping-induced vertical flow makes a major contribution to recharge to the deeper aquifers.

In unsewered cities, urban wastewaters are either disposed to the ground via septic tank soakaway systems/ latrine cesspits or to surface water courses (Figure 4.1). The latter option is preferred where the water table is shallow as septic tank systems do not operate effectively under these conditions. Both methods of disposal by-pass the soil zone which is the most effective layer at removing contaminants (Box 4.3). Below the soil in the unsaturated zone, most attenuation processes continue and, to greater effect than in the underlying saturated zone, with the exception of dilution. Where the depth to the watertable is sufficiently deep, complete removal of faecal pathogens and degradable compounds within the unsaturated zone is possible.



a) Disposal of domestic wastewater to soakaway/septic tank (common in unsewered cities with deeper water table).

b) Disposal of wastewater to shallow water course (common in cities where water table is near ground surface).

Figure 4.1 Routes for the disposal of urban wastewater to the subsurface

For most compounds, aerobic degradation is considerably more effective than anaerobic degradation and thus the unsaturated zone, with its readily available (and effectively inexhaustible) supply of oxygen, plays a crucial role in removing many compounds although some compounds (e.g. some chlorinated solvents) are more effectively degraded under anaerobic conditions. The depth to the watertable is therefore of major importance when assessing the pollution risk and, as will be seen in Section 6, has important implications for secondary water quality effects caused by redox changes.

It is possible for contaminants to migrate to depth more rapidly than the rates suggested by Table 4.1. This can occur where the contaminants 'short-circuit' the unsaturated zone by migrating downwards directly to the saturated zone via problem boreholes (abandoned, long-screened, or poorly constructed boreholes). In practice, the volumes of water that short-circuit the unsaturated zone and penetrate to the deeper aquifer are usually very small when compared with the volumes that infiltrate through the whole profile. Nevertheless contaminants that 'short-circuit' the unsaturated zone can produce troublesome concentrations in individual boreholes. This localised pollution is usually, but not exclusively, a microbiological rather than a chemical problem. This is because of the much lower levels of microbiological, compared to chemical, contamination that is permitted in drinking water.

Table 4.1 Classification of unconsolidated alluvial aquifers and their vulnerability to pollution

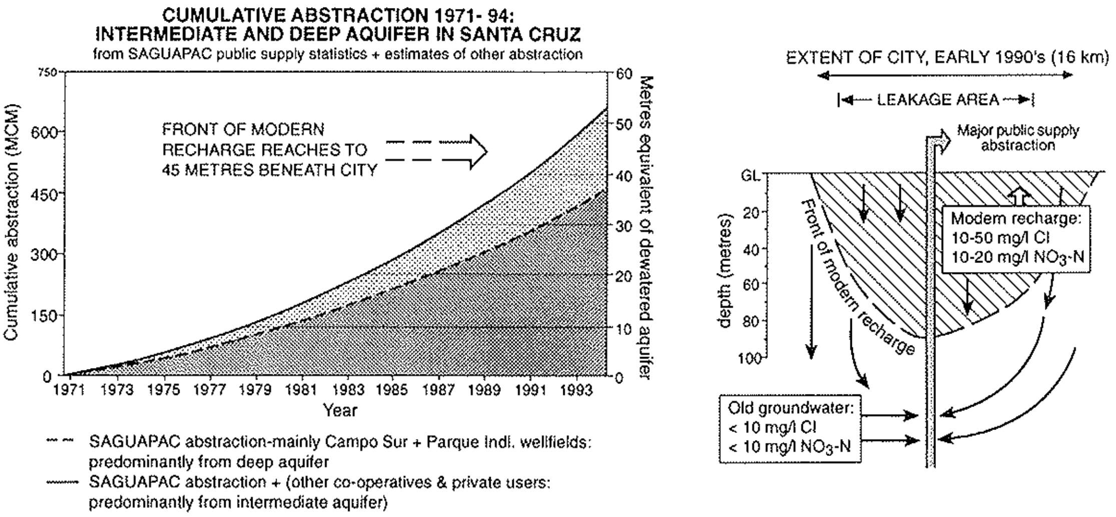
	AQUIFER TYPE		TRAVEL TIME[1]	AQUIFER VULNERABILITY TO POLLUTION	EXAMPLE		
					CITY	TRAVEL TIMES	EVIDENCE
1	**UNCONFINED:** **Depth to water table (m)**	(a) 0-5	days-months	high-very high	Hat Yai (upper aquitard)	days-weeks	Rapid response of water table to rainfall (BGS et al 1997a)
		(b) 5-10	weeks-months	high	Santa Cruz (shallow groundwater)	days-weeks	Rapid response of water table to rainfall (BGS et al 1997b)
		(c) >10	months-years	moderate-high	Lima, Peru	years	Urban recharge indicators (Geake et al 1986)
2	**SEMI-UNCONFINED:** **Depth to top of deep aquifer (m)**	50-100	years-decades	moderate-low	Santa Cruz (intermediate depth aquifers)	20 years	Penetration of chloride beneath city (BGS et al 1997b)
		>100	decades	low	Santa Cruz (deep aquifers)	>30 years	Penetration of chloride beneath city (BGS et al 1997b)
3	**SEMI-CONFINED:** **Depth to top of deep aquifer (m)**	<50	decades	low	Hat Yai (semi-confined aquifer)	20-40 yrs	Simple modelling combined with water quality data (BGS et al 1997b)
		<100	decades-centuries	low	León, Mexico (deep aquifer)	30-40 yrs	Penetration of chloride from peri-urban wastewater irrigation induced by deep pumping (BGS et al 1996a)

[1] approximate and generalised time for water to migrate under intergranular flow conditions from ground surface to upper surface of aquifer

BOX 4.1 DOWNWARD LEAKAGE OF CONTAMINATION INDUCED BY PUMPING: SANTA CRUZ, BOLIVIA

Santa Cruz de la Sierra, Bolivia is a low-rise, relatively low-density fast-growing city, whose municipal water supply is derived entirely from wellfields within the city limits, extracting from deep semi-unconfined outwash plain alluvial aquifers. Public supply is provided by cooperatives of which the largest is SAGUAPAC, providing for almost two-thirds of the population. Water is obtained from about 50 (90-350 m deep) boreholes which provide 98Ml/d (1994). There are also many private wells, (some 550 in 1991), used for industrial, commercial and some residential supplies. These wells are generally less than 90 m deep and draw water principally from the shallow aquifer.

Groundwater in the deeper aquifer, below 100 m, is of excellent quality, similar to the shallow aquifer upgradient of the city, and represents the natural condition. However the uppermost aquifer above 45 m shows substantial deterioration with elevated nitrate and chloride. Local peaks are typically in the range 10-35+ mgN/1 and up to 135+ mg/l respectively beneath the more densely populated districts, and are derived from the disposal of domestic wastewaters to the ground via on-site sanitation systems. These represent a major source of urban recharge, which is then drawn down in response to abstraction from the deeper semi-confined aquifers. The extent and travel time of the leakage-induced recharge front can be tracked by the chloride, which acts as a tracer. The front has penetrated to over 90 m in the most intensely pumped area (1994) although the average depth of penetration beneath the city is probably closer to 45 m.



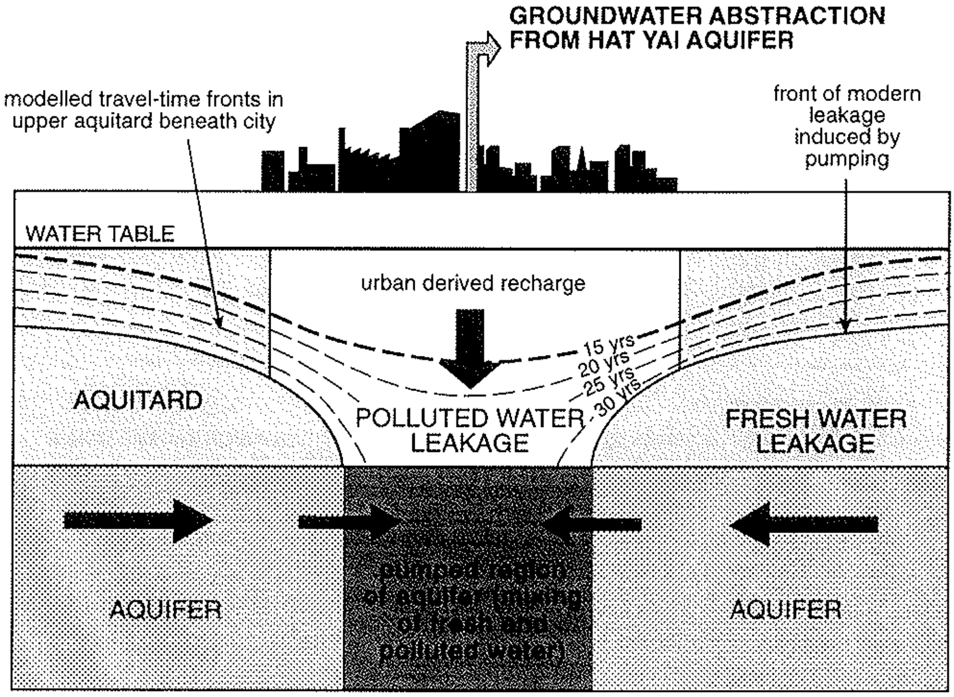
Development of abstraction and vertical leakage in Santa Cruz aquifer system, 1971-94

Groundwater abstraction first started in about 1970; by 1994 the cumulative abstraction was equivalent to the volume of available water stored within the upper 50 m of the aquifer beneath the city. This indicates firstly that the front of modern, high chloride, recharge is moving down at a rate of about 2 m per year and secondly that downward leakage from beneath the city accounts for the bulk of the local recharge to the deeper aquifers.

BOX 4.2 INDUCED LEAKAGE OF CONTAMINATED GROUNDWATER TO THE SEMI-CONFINED AQUIFER; HAT YAI, THAILAND

Hat Yai is situated on a semi confined alluvial aquifer in the south of Thailand. The city is dependent on groundwater for domestic and industrial purposes. Pumping from the semi-confined Hat Yai Aquifer within the city centre has produced steep vertical gradients leading to leakage of shallow groundwater from the overlying aquitard. This shallow groundwater within the aquitard has been found to be heavily polluted and contains high concentrations of ammonia and chloride. Incipient contamination of the underlying Hat Yai Aquifer has been observed close to the centre of major abstraction and where vertical water level gradients are steepest.

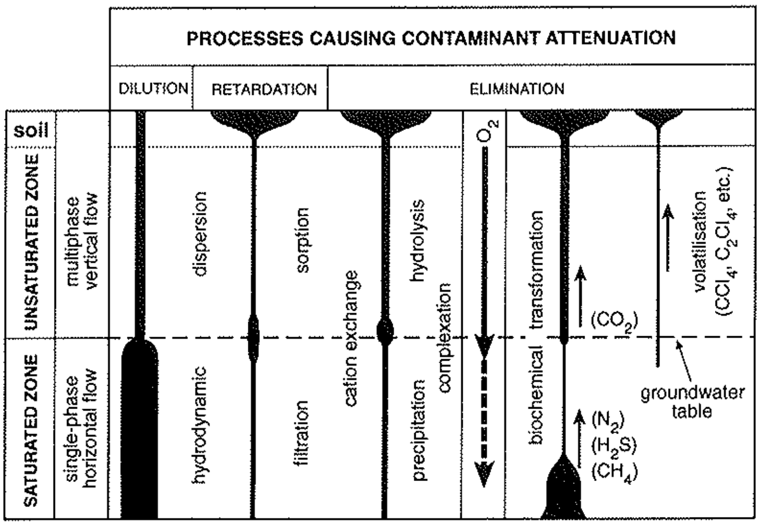
Comparison of the water quality monitoring data with simple modelling suggests that the urban derived leakage accounts for at least 25% of the recharge to the deeper aquifer. Modelling also predicts travel times in the range 30-50 years for the front of modern, polluted recharge to move through the aquitard into the underlying Hat Yai aquifer.



Travel times for downward migration of contaminated urban wastewater in response to deep pumping, Hat Yai

BOX 4.3 ATTENUATION OF CONTAMINANTS BELOW THE GROUND SURFACE

When waterborne pollutants pass below the ground surface many, but not all, are subject to naturally occurring attenuation processes. If the contaminant enters via a natural soil profile, this can be a potentially effective system in itself as soils are usually aerated by soil fauna and are biologically very active. This favours bacterially-mediated degradation, which for many compounds is more effective in an aerobic environment. Soils also may have high clay and/or humus content, providing opportunities for sorption and ion exchange. However in an urban setting, many pollutants enter the subsurface directly, via structures which by-pass the soil zone such as septic tanks, latrines or soakaway pits. As these structures are designed to dispose of waterborne wastes quickly, they will almost invariably impose a higher hydraulic loading, or surcharge, than pollutants entering via the soil zone. The increased percolation velocities which result bring about a corresponding reduction in residence times and in opportunity for contaminant degradation. Contaminant attenuation processes continue to a lesser extent at depth, in the unsaturated zone, and the diagram below illustrates qualitatively their relative importance in the soil, then above, at, and below the groundwater table.



(line width indicates relative importance of process in corresponding zone)

Processes promoting contaminant attenuation in groundwater systems *(Foster & Hirata, 1988 after Gowler 1983).*

The role of the unsaturated zone is especially important for shallow aquifers beneath urban areas with on-site domestic wastewater disposal, because the infiltrating effluent contains much organic load, generating a high biological oxygen demand (BOD). A thick unsaturated zone offers aerobic conditions, and as for many organic compounds aerobic degradation is more effective than anaerobic decomposition, this will favour pollutant removal as part of a natural bioremedial process.

Some areas of Santa Cruz have a moderately thick unsaturated zone (more than 10 m), where denitrification and other bacterially-mediated degradation reactions have a chance to occur. Conversely, in Hat Yai, the water table is shallow and as this does not permit septic tank systems to function properly, urban wastewater is disposed directly to slowly flowing rivers/drainage canals. Although effluent in such watercourses may be diluted, loads are so high that the watercourse is almost always deoxygenated, and where there are reaches which lose water to the shallow aquifer by bed leakage, an unsaturated zone is likely to be absent. These conditions result in an early onset of reducing conditions in the saturated zone, whose extent is indicated by the redox potential (Eh).

5. CONTAMINANT TYPES: TRANSPORT BEHAVIOUR

The pollution risk to deep aquifers depends not only upon the vulnerability of the aquifer to pollution (Section 4) but also on the load and types of contaminants present.

Urban wastewaters are highly variable both in terms of the types of compound and their concentrations (Box 5.1) because they include both domestic sewage and industrial effluents. The latter may contribute significant quantities of metals (e.g. chromium salts from the tanning industry) and organic wastes (e.g. high Biological Oxygen Demand effluents from sugar refining and other food processing industries) to wastewaters which are discharged to the ground or to surface water courses. In general, it appears that the organic content of urban effluent in many developing countries contains significantly lower levels of persistent synthetic compounds (actual and relative) than those in industrially developed countries in Europe and N. America (BGS et al 1996b). Climate and more particularly effective rainfall, also plays an important role in determining effluent concentrations; effluent in semi and arid climates having generally higher concentrations than in more humid climates.

Given the long travel times described in Section 4 and summarised in Table 4.1, it is clear that only the most mobile and persistent compounds are likely to penetrate to the deeper aquifers. Table 5.1 summarises the main contaminants found in urban wastewaters, their transport characteristics and probability of migrating to deeper aquifers. It is apparent that nitrogen (as NO3 or NH4) and chloride are the most likely inorganic contaminants to reach deeper aquifers and this was confirmed by the two case studies (Box 5.2). Chlorinated hydrocarbons (CHCs) are persistent and widespread contaminants of urban groundwater in N. America and W. Europe and are probably the most frequently occurring organic contaminants found in groundwater. These compounds however, have not been observed in groundwater beneath Santa Cruz or Hat Yai (Box 5.2). The reason for this is that CHCs are not commonly used in many developing countries; CHCs were not detected in urban wastewaters from Hat Yai; Santa Cruz; León, Mexico; or Amman, Jordan (BGS et al 1997a, b). However, CHCs have been detected in deep groundwaters beneath the city of Dhaka, Bangladesh; the source of these contaminants is believed to be effluent from tannery and chemical industries (Ahmed at al 1998 in press).

Whilst nitrogen has been observed in deeper aquifers at concentrations above the WHO drinking water guideline (BGS et al 1997a, b), the health risk associated with these concentrations is considered low (see Section 7). Chloride concentrations did not approach the drinking water taste guideline value in either case study city. It is not likely to be problematic in other cities except where arid climatic conditions prevail, producing high initial concentrations in the effluent (e.g. Sana'a Yemen, Alderwish and Dottridge 1998 in press), or as a result of groundwater problems not resulting from pollution (e.g. saline intrusion in coastal cities) or where industrial processes generate high loads (e.g. de-salting hides in leather processing, BGS et al 1996a).

BOX 5.1 COMPOSITION OF URBAN WASTEWATERS

Urban wastewater is mainly comprised of water (99.9%) together with relatively small concentrations of suspended and dissolved organic and inorganic solids. Among the organic substances present in sewage are carbohydrates, lignin, fats, soaps, synthetic detergents, proteins and their decomposition products, as well as various natural and synthetic organic chemicals from process industries. Concentrations of these synthetic compounds tend to be much higher in developed compared with developing countries, reflecting the nature and degree of industrialisation. The table below shows the concentrations of the major inorganic constituents of strong, medium and weak urban wastewaters. In arid and semi-arid countries, such as Jordan and Mexico, water use is often fairly low and sewage therefore tends to be very strong. In wetter countries such as Thailand and Bolivia, sewage tends to be weaker. The table demonstrates that nitrogen (mostly in the form of nitrate) is the contaminant most likely to exceed the WHO Drinking Water Guideline.

	CONCENTRATION (mg/l)				
		HUMID		SEMI ARID	
	WHO DWGL*	Hat Yai, Thailand	Santa Cruz Bolivia	León,[1] Mexico	Amman,[2] Jordan
Conductivity	-	345	890	2500	2350
Dissolved organic carbon	-	10	35	96	40
Sodium	200	35	73	111	235
Potassium	-	13.5	19.3	16.2	37
Bicarbonate	-	130	440	685	830
Sulphate	250	12	40	81	35
Phosphate	-	1.5	6.1	4.0	19
Chloride	250	45	57	310	530
Nitrogen	10	11	39	40	93
Boron	0.3	0.05	0.08	0.26	0.56

Inorganic components of urban wastewater from developing cities

[1] BGS et al 1996a
[2] BGS et al 1996b
* Drinking water guideline value, World Health Organisation, 1993.

Table 5.1 Transport characteristics of the common urban contaminants

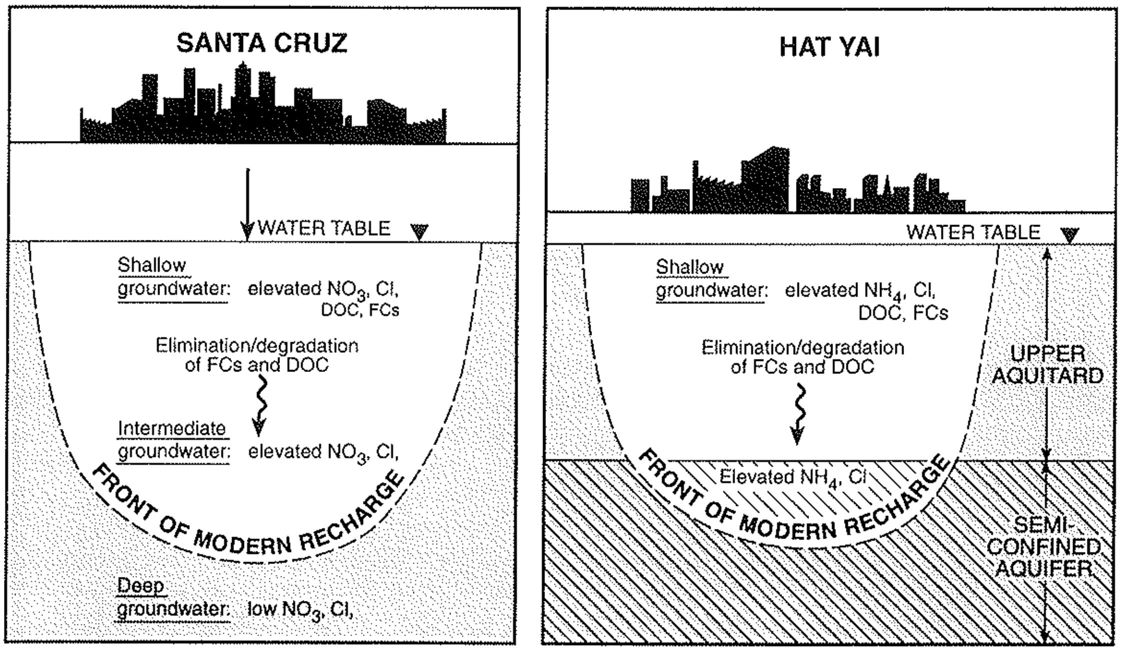
CONTAMINANT	SOURCE	ATTENUATION				PERMITTED CONCENTRATION IN DRINKING WATER	LIKELIHOOD OF MIGRATING TO DEEP AQUIFER	COMMENTS
		BIOCHEMICAL DEGRADATION	SORPTION	FILTRATION	PRECIPITATION			
NITROGEN (N)	sewage	*	* (ammonia is sorbed)	-	-	moderate (10-20 mg N/l)	probable	widespread, mobile and generally persistent. Can occur in deeper aquifers and at concentrations above permitted value
CHLORIDE (Cl)	sewage/industry	-	-	-	-	high	certain	widespread, mobile and persistent. Can reach deep aquifers but permitted concentrations are high and unlikely to be exceeded except in arid climate where input concentrations are high
FAECAL PATHOGENS (FCs)	sewage	***	**	***	-	v. low (<1 per 100ml)	unlikely	unlikely to penetrate to deep, granular aquifer
DISSOLVED ORGANIC CARBON (DOC)	sewage/industry (esp. food processing / textiles)	***	***	*	-	not controlled	unlikely	DOC, frequently present as long-chain fatty acids is generally easily degraded especially in aerobic aquifer systems
SULPHATE (SO_4)	road-runoff industry	*(can be reduced)	*	-	*	high	possible	persistent, likely to reach deep aquifer, unlikely to pose a quality problem
HEAVY METALS	industry	-	***	* (where occur as organic complexes)	*	low (variable)	unlikely	general non-mobile except under low pH-Eh conditions (except where metals form complex anions e.g. Cr^{6+})
CHLORINATED SOLVENTS (CHCs)	industry	*	*	-	-	low (10-30 μg/l)	probable	persistent compounds which may penetrate to deeper aquifers. Biodegradation potential low.
HYDROCARBONS (HCs)	spillages at petrol stations, industry	***	**	-	-	low (10-700 μg/l BTEX)	unlikely	spills of HC liquids likely to be retained at shallow depth. HCs degrade, especially in aerobic systems, and are amenable to biodegradation
OTHER SYNTHETIC ORGANICS	industry	variable	variable	-	-	low (variable)	possible	

KEY: *** major importance; ** significant attenuation; * some attenuation; - no attenuation

BOX 5.2 CONTAMINANT BEHAVIOUR IN SANTA CRUZ AND HAT YAI

The cities of Santa Cruz in Bolivia* and Hat Yai in Thailand are largely unsewered and so significant quantities of domestic and some industrial wastes are discharged to the subsurface. Principal contaminants entering the subsurface in this way include nitrogen, chloride, long-chain organics and faecal pathogens.

Shallow groundwater beneath both cities is contaminated, as indicated, by high concentrations of nitrogen (as NO_3 beneath Santa Cruz, and as NH_4 beneath Hat Yai), chloride, faecal coliforms and dissolved organic carbon (DOC). Metals were generally absent as the prevailing pH of the groundwaters in both cities was outside the range of predicted metal mobility. Synthetic organics were largely absent from the wastewater sources and thus not surprisingly were not observed in the shallow groundwater.



Groundwater settings in Santa Cruz and Hat Yai

Both cities are dependent on groundwater obtained from deeper semi-confined aquifers whose pumping has led to induced downward leakage. Elevated N^- and Cl^- groundwaters which have been observed beneath both cities indicate the depth of penetration of the front of modern recharge. However, faecal coliforms and elevated DOC were generally not observed in these deeper groundwaters, indicating attenuation and elimination at shallower depths. These processes are thus well illustrated in both cities, whose present deeper aquifer water quality is excellent.

**Since the study began in 1991, piped sewage has been significantly extended in central districts of Santa Cruz.*

6. SECONDARY WATER QUALITY CHANGES

Both the industrial and the domestic components of urban wastewater have a high organic content. This organic matter is relatively easily oxidised under aerobic conditions. Where the water table is deep, oxygen and microorganisms in the unsaturated zone of the aquifer may remove (degrade) much of the organic content.

Below the water table, any further degradation (of organic matter) will consume the dissolved oxygen (dO_2) present in the groundwater:

$$2CH_2 + 3O_2 \rightleftharpoons 2H^+ + 2HCO_3^-$$

Oxygen in this reaction is termed the electron acceptor. The quantity of oxygen dissolved in groundwater is much less than that present in the unsaturated zone and is less rapidly replaced. Thus depletion of dO_2 is possible, whenever the oxygen demand for the degradation of organic matter exceeds supply.

When this happens the redox potential of the groundwater declines and further degradation of organic matter continues utilising other ions (electron acceptors) including NO_3^-, Mn^{4+}, Fe^{3+}, SO_4^{2-} (Figure 6.1). The utilisation of these ions produces significant water quality changes and these are termed, in this report, secondary changes (i.e. they are a result of changes to the redox potential rather than contaminants derived from the urban wastewater). Secondary water quality changes include increases in dissolved iron and manganese concentrations, and reduction in nitrate concentrations.

In Hat Yai, groundwaters within the upper aquitard have low redox potential (strongly reducing) and very high concentrations of iron and manganese are observed. There is also a positive correlation between dissolved iron and arsenic concentrations (Box 6.1) and it has been suggested that the mobility of arsenic is controlled by iron dissolution (BGS et al 1997a). Elevated iron and arsenic concentrations have also been observed in the main aquifer beneath the city as a result of downward leakage from the upper aquitard. The mobilisation of arsenic by redox changes to the groundwater is an important result for two reasons. Firstly because the health implications of excess arsenic in drinking water are serious (see 7.4) and secondly because arsenic is a relatively common element in many alluvial sediments; the mobilisation of arsenic by iron dissolution could occur more generally beneath unsewered cities wherever strongly reducing conditions (produced by the disposal of wastewaters to the subsurface) prevail.

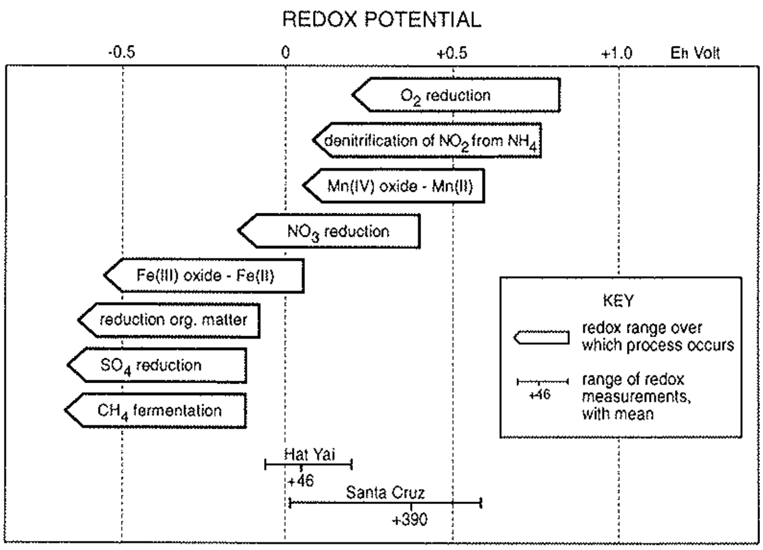
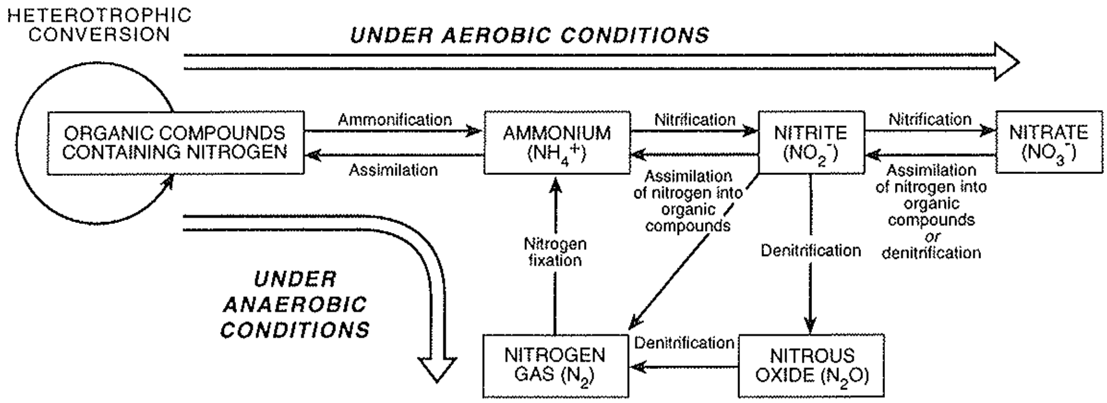


Figure 6.1 Sequence of Microbially Mediated Redox Processes

BOX 6.1 SECONDARY WATER QUALITY CHANGES IN SANTA CRUZ AND HAT YAI

Wastewater constituents in aquifers are able to react both with each other and with the subsurface gases and the porous medium of the aquifer matrix itself. Most geochemical changes in wastewater occur as a result of the reactions of a few major components that strongly affect the master variables of the system: the reduction/oxidation (redox) potential and pH.

Bacterially-mediated nitrification and denitrification processes can transform and volatilise a significant proportion of infiltrating wastewater derived nitrogen as it passes through zones of varying oxygen concentrations. Across much of the city centre in Santa Cruz, organic nitrogen entering the shallow aquifer has undergone nitrification via ammonia on its passage through the unsaturated zone, and so is mainly present as nitrate. This reaction generates acidity which, in the case of Santa Cruz, is buffered by the calcium carbonate present in the rock matrix and so no change in pH is detected. At a few sites where oxygen has been fully consumed in the breakdown of organic carbon, denitrification is occurring, and the nitrogenous leachate is converted to nitrogen gas.



Transformations of nitrogen from wastewater during groundwater recharge (modified from Madison and Brunett, 1985)

In Hat Yai the groundwater system has become more reducing than in Santa Cruz due in part to the very shallow water table. Little carbonate is present in the rock matrix, so during oxidation of the ammonia the tendency is for a fall in pH. At a lower pH, heavy metals have a greater mobility and are more likely to be transported to groundwater. In the city centre, all the nitrate has been consumed and naturally occurring microorganisms utilise manganese or iron present in the rock matrix in order to break down the organic food source. This leads to elevated concentrations of manganese and iron in solution as they are more soluble in their reduced form. In addition, any arsenic which was loosely bound to iron is also released into the groundwater and is found to be present at concentrations in excess of 1 mg/l in the most reducing areas of the aquifer. In general, high arsenic and iron concentrations are found to be associated.

In Santa Cruz, the redox potential of the groundwaters is higher than in Hat Yai (i.e. less reducing). In some areas dO_2 has been depleted and there is evidence for both nitrate reduction and manganese dissolution (Figure 6.1). The reason for the less reducing conditions in Santa Cruz (compared to Hat Yai) is attributed to the deeper water table and the greater capacity of the unsaturated zone to remove much of the organic load before it reaches the water table.

High iron and manganese concentrations although not a threat to health do represent serious water quality problems as they may be unacceptable for both domestic (because they impart an unpleasant taste and can stain laundry) and some industrial purposes. Removal of these ions by treatment is expensive.

7. RESULTS - SUMMARY

7.1 Recharge to deeper aquifers: importance of urban leakage and likely timescales

The results of this study confirm that leakage of shallow groundwater from beneath cities can make a significant contribution to recharge of deeper aquifers used for potable supplies.

The importance of vertical flow in groundwater systems is often overlooked, even though in many environments leakage probably contributes 20-70% of the overall recharge to the deeper aquifer. The reasons why vertical leakage is such an important component are firstly because vertical gradients can be very steep and secondly because the area over which vertical recharge occurs can be large (many km^2). Leakage is usually greatest close to major centres of pumping where steep vertical gradients are likely to be induced.

In Santa Cruz, modern (post 1960) recharge (identified by elevated chloride and nitrate) has migrated to depths of more than 70 m in places and provides evidence that urban derived leakage accounts for the bulk of the recharge to the deeper aquifer. Likewise in Hat Yai, modern (post 1960), water has migrated through the upper aquitard to the semi confined aquifer, beneath the city centre. The urban derived leakage provides more than 25% of the recharge to the deeper aquifer.

Timescales for leakage from shallow layers to deeper aquifers is typically in the range years-decades. Such timescales ensure that only the most persistent contaminants are likely to pose a threat to water quality.

7.2 Urban wastewaters: migration and behaviour of key contaminants

Urban wastewaters in many developing countries contain relatively high concentrations of nitrogen, chloride and organic matter but, when compared with wastewaters in N. America and Europe, low concentrations of metals and synthetic organics. This may change in the future as cities in developing countries industrialise. These wastewaters are frequently discharged to the ground via on-site sanitation systems or soakaway pits. Where the water table is shallow the above systems will not operate efficiently and disposal to surface water courses is more common. In either case seepage to shallow groundwater can occur, producing widespread chemical and microbiological contamination of the upper aquifer. The present work has confirmed that:

(i) Pathogenic microorganisms which probably represent the most serious health concern are not generally observed in deeper aquifers, although some deep boreholes may be contaminated. This is because microorganisms are effectively eliminated by filtration and degradation due to natural predation and die-off. Incidents of faecal contamination of deep boreholes almost certainly result from localised pollution caused by inadequate design and completion of boreholes (for example where long-screened boreholes interconnect shallow and deep aquifers).

(ii) Chloride and nitrogen (normally as nitrate) are widespread contaminants of shallow urban groundwaters. These contaminants are both mobile and persistent and will migrate to deeper aquifers, although in low oxygen groundwater systems some removal of nitrate by denitrification may occur (BGS et al 1997b). Concentrations of nitrogen in deeper groundwaters may exceed the drinking water guideline; precise concentrations depending primarily upon the input concentration in wastewater (which is itself a function of population density and per capita water supply) and the degree of mixing with low nitrate waters. Chloride values in deeper urban aquifers are unlikely to exceed drinking water guidelines except in semi arid climates where opportunities for dilution from rainfall recharge are low. Most heavy metals are likely to be retained within the upper layers as a result of sorption and precipitation processes.

(iii) Many of the long-chain organic compounds commonly associated with domestic wastewater and agro-industry effluents are likely to be rapidly degraded. These degradation reactions consume oxygen and change the redox potential of the groundwater which in turn can lead to mobilisation of metals that occur naturally within the matrix of the aquifer (see 7.3). Table 5.1 summarises the transport characteristics of the common urban contaminants.

7.3 Secondary water quality impact

An important outcome of the project has been the realisation that the mobilisation of elements present in the aquifer matrix by changes to the oxidising status (redox condition) of the groundwater may represent a more serious threat to water quality and have greater health and cost implications than contaminant migration.

Degradation of the organic content of wastewater consumes oxygen; producing a significant change in the redox potential of the groundwater. Where the unsaturated zone of the aquifer is deep, degradation of much of the organic content may occur above the water table because a plentiful supply of oxygen is available. However, for aquifers where the water table is within 5 m of the surface the degradation process will be incomplete. While degradation and specifically oxidisation will continue below the water table the dissolved oxygen content in groundwater is limited and is insufficient to satisfy the biochemical oxygen demand (BOD) of the wastewater. Depending on the load imposed, and the amount of throughflow, the groundwater chemistry is progressively altered and once the water is depleted in dO_2 significant and troublesome concentrations of iron and manganese can occur. Whilst iron and manganese do not pose a direct health risk in themselves, their removal from pumped groundwater can be costly.

Arsenic is often closely associated with iron and the mobilisation of iron can result in the release of arsenic into solution. A correlation between iron and arsenic mobility was demonstrated for the groundwater in the upper aquitard beneath Hat Yai (BGS et al 1997a). Concentrations of groundwater arsenic in the aquitard exceeded 1.0 mg/l (twenty times the WHO guideline value of 0.05 mg/l), whilst limited sampling in the underlying aquifer confirmed concentrations up to and beyond the guideline value. Since arsenic is a relatively common constituent in alluvial sediments, this mechanism of mobilising the element may be widespread wherever iron dissolution occurs. Arsenic is infrequently monitored. However, as this work has clearly shown that arsenic may be a significant problem in urban groundwaters whenever strongly reducing conditions occur, **it is strongly recommended that it should be more widely monitored (see 8.1) wherever excessive iron concentrations occur.**

It is apparent that the mobilisation of elements present naturally in the aquifer, as a result of changes to the groundwater geochemistry caused by contamination, has significant cost and health implications. Aquifers most at risk are those with very shallow water table (for example low-lying coastal plain aquifers) where the opportunity for degradation of organic matter above the water table is limited and where strongly reducing conditions develop as a consequence. These aquifers are often in other respects not especially vulnerable to pollution.

7.4 Health and cost implications

7.4.1 *Overview*

The social and economic implications of groundwater pollution in any situation will depend on:

a) the level and nature of threat posed (pollution risk);
b) the sensitivity of economic and social systems to such threats (societal vulnerability).

Elements of this risk assessment methodology are developed and described in greater detail in Section 8 below. In terms of societal vulnerability, however, it is clear that factors such as the level of dependency on groundwater, the nature of users and uses and the availability and affordability of alternatives will be crucial. For example, where groundwater is an important source of supply, where end uses are sensitive to changes in water quality, and where alternative sources of supply are only available at high cost, the broad consequences of degradation will be more severe. Impacts may include health problems associated with the use of poor quality drinking water, rising economic costs of water provision, and conflicts between different uses and users.

In the sub-section below, the health implications of groundwater contamination are discussed with reference to Hat Yai and Santa Cruz case studies. Potential economic impacts are discussed for Hat Yai, where a water use survey was commissioned. A different (non-project) case study - Gaza City in the Palestinian Territories - is then used to illustrate how economic impacts can be quantified where sufficient data are available.

7.4.2 *Health implications*

Clearly there are health implications whenever urban groundwater is used for drinking water supply. This is the case in both Santa Cruz and Hat Yai and many hundreds of cities worldwide. This research has shown that nitrogen and chloride are the contaminants most likely to reach the deeper aquifers. Chloride concentrations are unlikely to exceed drinking water guideline values except under extreme conditions (ie. high loading from industry and/or under semiarid climates where dilution by rainfall infiltration is very low).

The health risks associated with nitrogen are considered low; the main concerns of excess nitrate are methaemoglobinaemia in young infants and the possible, but still unproven, causal-link with stomach cancer. In addition to these concerns, ammonium may reduce the disinfection efficiency of chlorine. Full details of the health risks are presented in the companion report (BGS et al 1997a).

Secondary water quality changes are also of concern; manganese and iron concentrations too may exceed drinking water guideline values especially in anaerobic groundwater systems. These guideline values are based on aesthetic effects rather than health concerns. However, one negative consequence of groundwater containing excessive iron and manganese is that alternative, and possibly less wholesome sources of water (ie. shallow private wells) may be used instead to provide potable supplies. This can be a problem in low-income districts where alternatives like bottled water are economically unrealistic.

A more serious health concern is arsenic which may be mobilised as a result of iron dissolution under reducing conditions. For example the high arsenic concentrations observed beneath Hat Yai must represent a serious health risk. Arsenic ingestion is known to cause dermatological, cardiovascular, neurological and respiratory diseases (Gorby, 1994).

In summary, beneath unsewered cities deeper groundwaters are of much better quality than the corresponding shallow groundwaters. The latter are usually widely polluted by a range of chemical and, more importantly, microbiological contaminants and should be avoided for potable supplies. Deeper groundwater may, in time, have nitrogen concentrations that exceed drinking water guidelines but the health risks associated with this are considered low. Of greater concern is the risk of excess arsenic concentrations associated with iron dissolution in anaerobic groundwater systems. The health risks of excessive arsenic are potentially serious.

7.4.3 *Economic implications*

The economic impacts of groundwater pollution have been considered using Hat Yai and Gaza City (in the Gaza Strip) as case study examples. In the former case, impacts do not appear to be serious at present. This is not entirely unexpected, as the city is still of moderate size, and hitherto the semi-confined nature of the aquifer has afforded some protection from contamination. However, rapid urbanisation and industrialisation, and the subsequent movement of contaminants into deeper aquifers, may have serious economic consequences in future.

In Hat Yai, the results of the water user survey confirm that groundwater is an important source of supply for all sectors of the economy, and the principal source of supply for industrial users. In general, the survey revealed widespread satisfaction with groundwater supplies, and water quality did not appear to be a significant factor affecting choice between surface (piped) supplies and groundwater alternatives. Importantly, the survey highlighted significant use of 'private' groundwater from household boreholes and shallow wells, sometimes for potable use, and often in conjunction with other water sources.[1] This illustrates how complex urban water budgets can be where public and private, and surface and groundwater systems, coexist. Such budgets, and the incentives facing different groups of water users, need to be fully understood if effective, integrated, urban water management is to be achieved (see also Sections 8.2 and 9).

Against this background, it is interesting to speculate on the *potential* impacts of groundwater pollution in Hat Yai and other rapidly developing cities, assuming no action is taken to address degradation causes.[2] Economic impacts, at varying levels of scale, might include:

- Household/firm: increasing borehole investment and operating costs associated with borehole deepening; increasing treatment costs; abandonment of shallower boreholes and loss of investment; ultimately, the costs of securing alternative supplies (where not met from the public purse) in the eventuality of 'aquifer write-off' for sensitive uses. In terms of private groundwater use, it is important to note that use may continue even as quality deteriorates - whether or not it is deemed safe or acceptable - because of the difficulties associated with controlling and regulating private abstraction. In any given situation, much will clearly depend on factors such as access to and affordability of alternatives, and information on and attitudes to risk.

- City/region: as above for public supply boreholes, plus: costs associated with substitution of groundwater for alternative (public) supplies from a different source. Alternative water may be sourced from surface supplies or urban hinterland well fields.

In the case of Hat Yai, the abandonment of groundwater sources, which currently meet over 50% of total supply, would imply a massive increase in demand for public, piped supplies. However, while the Provincial

1 *Survey results indicated that roughly 60% of households had access to 'private' groundwater from boreholes and shallow wells, with around 15% of these households also receiving piped supplies from PWA. Bottled water was preferred for drinking, though groundwater was a significant source for those households with groundwater access.*

2 *In Hat Yai a major infrastructural investment programme aimed at improving the urban environment is underway, and includes investment in a wastewater treatment plant and wastewater canal lining. However, any positive impact on groundwater quality would seem more accidental than intended, as there is no mention of groundwater protection or pollution prevention in the development plans (GoT, 1990)*

BOX 7.1 ESTIMATING THE COST OF GROUNDWATER DEGRADATION

"A prerequisite for the sustainable management of water as a scarce vulnerable resource is the obligation to acknowledge in all planning and development its full costs." (UNCED, 1992)

Groundwater degradation reduces the amount and/or quality of water available for use in future periods, leading at some point to aquifer exhaustion. In this sense, current use of the resource has an opportunity cost which is the cost of use foregone in the future. This cost has been termed a user or depletion cost (ODA, 1988; Winpenny, 1995), and is equivalent to the present value of the cost of replacing a groundwater source at some future date. Where groundwater resources are being over-exploited or contaminated, as they are beneath many of the large cities of developing countries, user costs should, ideally, be factored into water tariffs and other controls (Winpenny, 1994).

For the cities of Hat Yai and Santa Cruz, insufficient data was available to make this calculation. However, an estimate has been made for Gaza City, in the Gaza Strip (Calow et al., 1997). Here, pressure on the coastal aquifer is intense, degradation is severe, and potable supplies likely to be exhausted within 15-20 years. Moreover, alternative supply sources, in the form of desalinated water, are expensive at around $1.20 m^3, compared with a marginal cost of supply estimate from the existing groundwater source of about $0.20/m^3 (1994 prices). These factors indicate that the user cost will be high. Calow et al. (1997) estimate that in 1994, the user cost was equivalent to almost $0.20/m^3, indicating that the full economic cost of water - including the discounted cost of a future replacement - was nearly $0.40/m^3, and considerably more than the existing (subsidised) tariff. Over time the user cost increases as aquifer exhaustion draws closer.

In policy terms, user costs help signal the future costs of replacement water supply in *current* prices, highlighting the growing economic scarcity of water over time. Decision-makers may find such 'degradation costs', expressed in monetary terms, more difficult to ignore than scientific data and exhortation. In theory, user costs can be added to municipal tariffs to ensure that diminishing stocks are preserved for higher value users, and to encourage conservation, recycling and reuse of treated effluent. They are also useful in setting a cost ceiling, or benchmark, beneath which the costs and benefits of alternative management strategies (including pollution controls) can be compared.

User cost principles, estimation procedures and application are discussed in more depth in Appendix A.

Water Authority (PWA) continue to expand their supply capacity, the intention may be to extend coverage to the growing peri-urban areas of Hat Yai, rather than provide a substitute for contaminated groundwater supplies within the existing municipal boundary. Although no PWA cost data could be obtained, any unplanned expansion of piped supplies might be extremely costly. In the near future though, groundwater will continue to provide an important source of supply for all economic sectors of the city, and ultimate aquifer 'write-off', itself very uncertain, would seem a long way off.

From the above, it is clear that the ultimate cost of groundwater degradation is the cost of eventual supply substitution. Where they can be made, replacement cost estimates can provide powerful signals to policy makers on the benefits of pollution control. They can also be used as a pricing and valuation tool, to ensure that diminishing stocks are used in ways that make most valuable use of them. Replacement cost calculations could not be made for Hat Yai and Santa Cruz because of insufficient data. However, Box 7.1, drawing on material discussed further in Appendix A, discusses theory and application in more detail with an illustration from the Gaza Strip.

The notion of replacement may serve to remind policy makers of the ultimate cost of inaction. However, for comparing different mitigative options beneath this cost ceiling, information is needed on the relative costs and benefits of different policies and projects. For example, how would the costs of mains sewerage installation, or pollution control in different industries, compare with the costs of groundwater treatment, assuming that groundwater treatment is a realistic alternative? How, if at all, can the environmental benefits of pollution abatement be compared to its costs?

While such comparisons may sound straightforward, in practice they are difficult to make. From the perspective of environmental economics, the *theory* of pollution control is well rehearsed. According to conventional wisdom, a regulator would be able to quantify the increase in pollution damage cost as the pollution level rises (marginal social damage costs). Similarly, there would be sufficient information to quantify increases in pollution control cost as polluters reduce their discharges (marginal abatement cost).The regulator would then determine the point of optimal pollution, where the benefits and costs of control are equal, and introduce mandatory controls or economic inducements to attain it. In practice, however, such information is almost never available in developing countries[3] (Afsah et al, 1996), and the monitoring and enforcement of controls is often extremely difficult. In most developing country situations, therefore, the main goal is to confront polluters with at least some of the 'social damage costs' imposed on others by means of water tariffs and/or effluent controls (Bhatia and Falkenmark, 1991). Management issues are discussed further in Section 9 and Appendix A.

To summarise:

- The economic implications of groundwater pollution may manifest themselves in different ways, and at different levels of scale, according to situation. In Hat Yai and Santa Cruz, economic impacts are more potential than realised. This contrasts with the situation in Gaza City, where a separate study has shown that the economic and social impacts of groundwater degradation are already significant.

- In all situations, the ultimate cost of degradation is the forced substitution of degraded groundwater supplies by some alternative source of supply, typically at much higher cost.

- Comparing the costs and benefits of alternative prevention and treatment options is difficult, as the information needed is frequently unavailable. In most developing country situations, the challenge is to find workable institutional arrangements to confront polluters with at least some of the external costs they impose on other users who share the groundwater resource.

- Ultimately, cost-benefit analysis is one of many decision criteria in water resources management. Other factors influencing leverage over problems and policy choices include: institutional capacity; political feasibility; equity implications; and wider social and economic goals (eg for industrial development) which may conflict with water resource objectives. Some of these issues are discussed further in Section 9.3.

3 *In industrialised countries too, there is often considerable uncertainty about the costs and benefits of alternative policies. One result is the adoption of precautionary principles in groundwater management which urge caution in decision making.*

8. RISK ASSESSMENT

8.1 Contamination threat

A methodology has been developed to assess the risk posed to deeper groundwaters which is based on (a) the aquifer condition and hydrogeological setting and (b) the contaminant load and type. This assessment is summarised and presented in Table 8.1 and Figures 8.1 and 8.2. The water quality assessment presented in Table 8.1 helps to (i) identify the water quality problems likely to pose a threat to groundwater, (ii) guide where monitoring should be undertaken to provide an early warning of future water quality problems, and (iii) guide what water quality parameters should be monitored for. Where future problems are predicted by monitoring, management principles to reduce the scale of the problem are presented in Section 9.

It is apparent that many contaminants present in shallow groundwater will be removed by the time recharge reaches deeper aquifers and that only N, Cl, and persistent organic compounds pose the main contaminant threat to deeper groundwaters. N and Cl are likely to represent only a low health risk. Therefore, in some environments, the practice of disposing wastes to the subsurface from which groundwater is pumped for water supply can be sustainable. However, in densely populated urban areas water supplies may need to be blended or treated where nitrate exceeds permitted drinking water concentrations.

It is also apparent that in aquifer systems characterised by a thin unsaturated zone (typical of coastal alluvium) the groundwater becomes rapidly anaerobic and iron and manganese may appear at troublesome concentrations in the deeper aquifers. These elements although not a health hazard do cause an unpleasant taste and staining problems and their removal may be costly. Perhaps even more significantly **arsenic** can be mobilised with iron and, **as it is toxic at low concentrations, this does represent a significant health risk.**

It is important to realise that iron, manganese and arsenic are not necessarily 'natural' water quality problems but that beneath unsewered cities they can be a result of the changes to the geochemistry of groundwaters, caused directly by contamination. So monitoring in the deep aquifer itself does **not** provide an early warning of future quality problems. Once it is established that downward leakage is important, it is essential to monitor water quality in the shallow layers (including the aquitard) to confirm the position of the 'front' of modern recharge and to predict the impact that this front will have on groundwater quality in the deeper aquifer.

Key water quality parameters to monitor for include N, Cl, dO_2, Eh, HCO_3. Where persistent organic compounds (for example CHCs) are used within city then these compounds should also be included in the list of determinants for monitoring.

Finally it is also important to be aware that just as water quality deterioration is likely to occur over timescales stretching to decades so reversal of adverse water quality changes will also be protracted. It is therefore essential to use monitoring and surveillance to provide an early warning of likely problems before they fully develop.

8.2 Contamination impact

It is not enough to only indicate the risk of groundwater contamination. Of great importance is also the sensitivity of society and the economy to groundwater degradation. This is related to many different factors, but key issues include:

Table 8.1 Water quality assessment guide

AQUIFER TYPE	PREDICTED WATER QUALITY CONCERNS IN SHALLOW UPPER LAYER				PREDICTED WATER QUALITY CONCERNS IN DEEPER MAIN AQUIFER			COMMENTS
	UPPER AQUIFER	CONTAMINANTS	SECONDARY WATER QUALITY PROBLEMS	EARLY-WARNING MONITORING	CONTAMINANTS	SECONDARY WATER QUALITY PROBLEMS	DRINKING WATER-SUPPLY SURVEILLANCE MONITORING	
Multi Aquifer in unconsolidated sediments: shallow, upper aquifer overlies deep main aquifers present at depths > 30m	shallow layer with unsaturated zone <5m	NH_4-N Cl DOC FCs CHCs HCs (metals)	Fe, Mn, As and H_2S	monitor shallow groundwater for: NH_4N/NO_3 Cl (CHCs) Fe Mn Eh As HCO_3	NH_4 Cl CHCs	Fe, Mn, As	monitor aquifer for: NH_4N/NO_3 Cl CHCs Eh Fe Mn As	Front of modern polluted water likely to contain elevated NH_4 and Cl^- whilst absence of unsaturated zone produces strongly reducing conditions with consequent troublesome concentrations of Fe, Mn, As
	G R A D A T I O N							
	unsaturated zone >10m	NO_3 Cl (DOC) (FCs) CHCs HCs	(Mn)	monitor shallow groundwater for: NO_3 Cl Eh HCO_3 (CHCs) dO_2	NO_3 Cl CHCs	(Mn)	monitor aquifer for: Eh NO_3 Cl CHCs (Mn)	Front of modern polluted water likely to contain elevated NO_3 and Cl. The thick unsaturated zone removes bulk of organic load; Mn likely to be main secondary quality problem

( ) indicates lesser concern

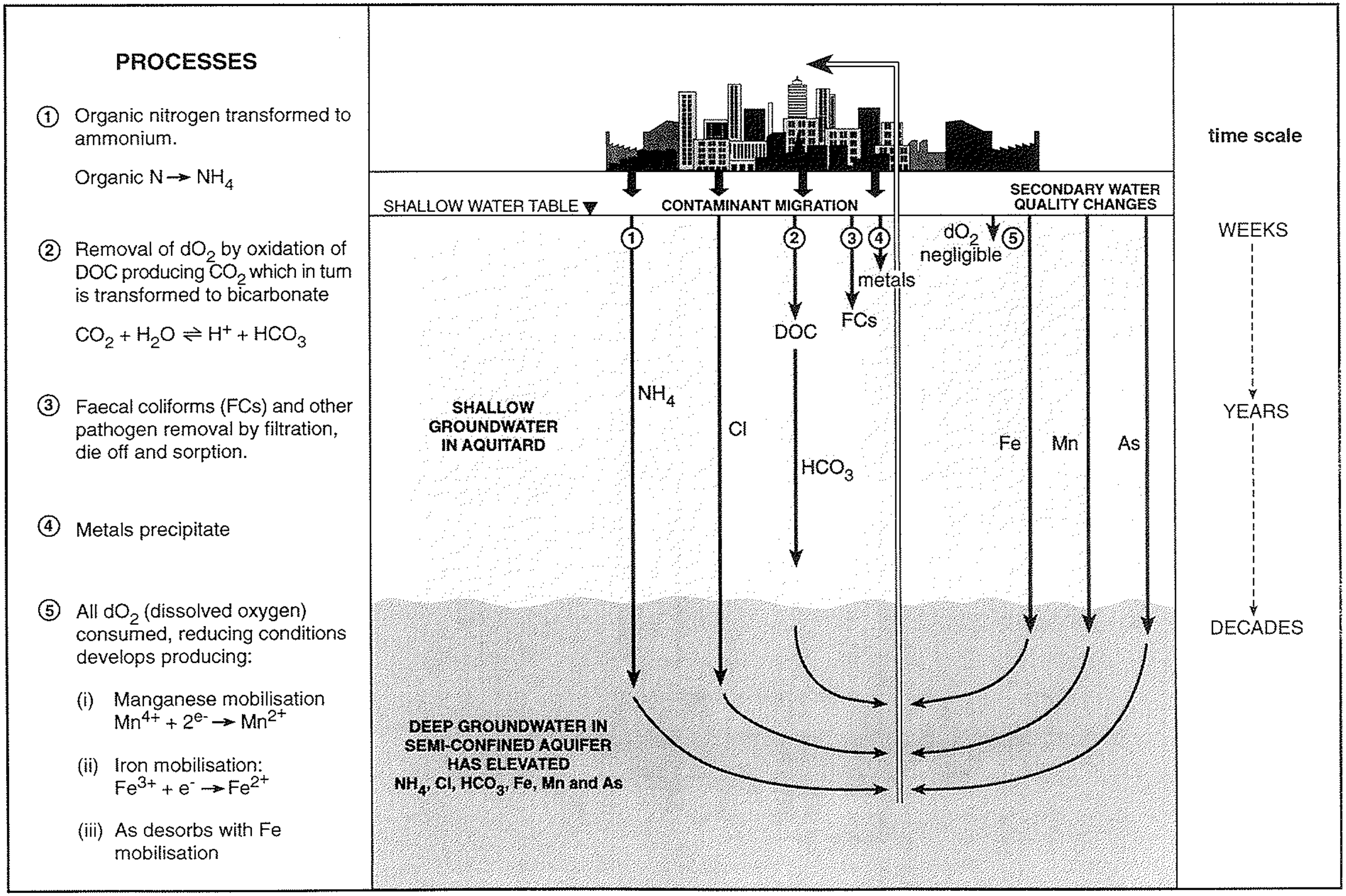


Figure 8.1 Assessment of urban water quality concerns and geochemical processes: anaerobic groundwater system

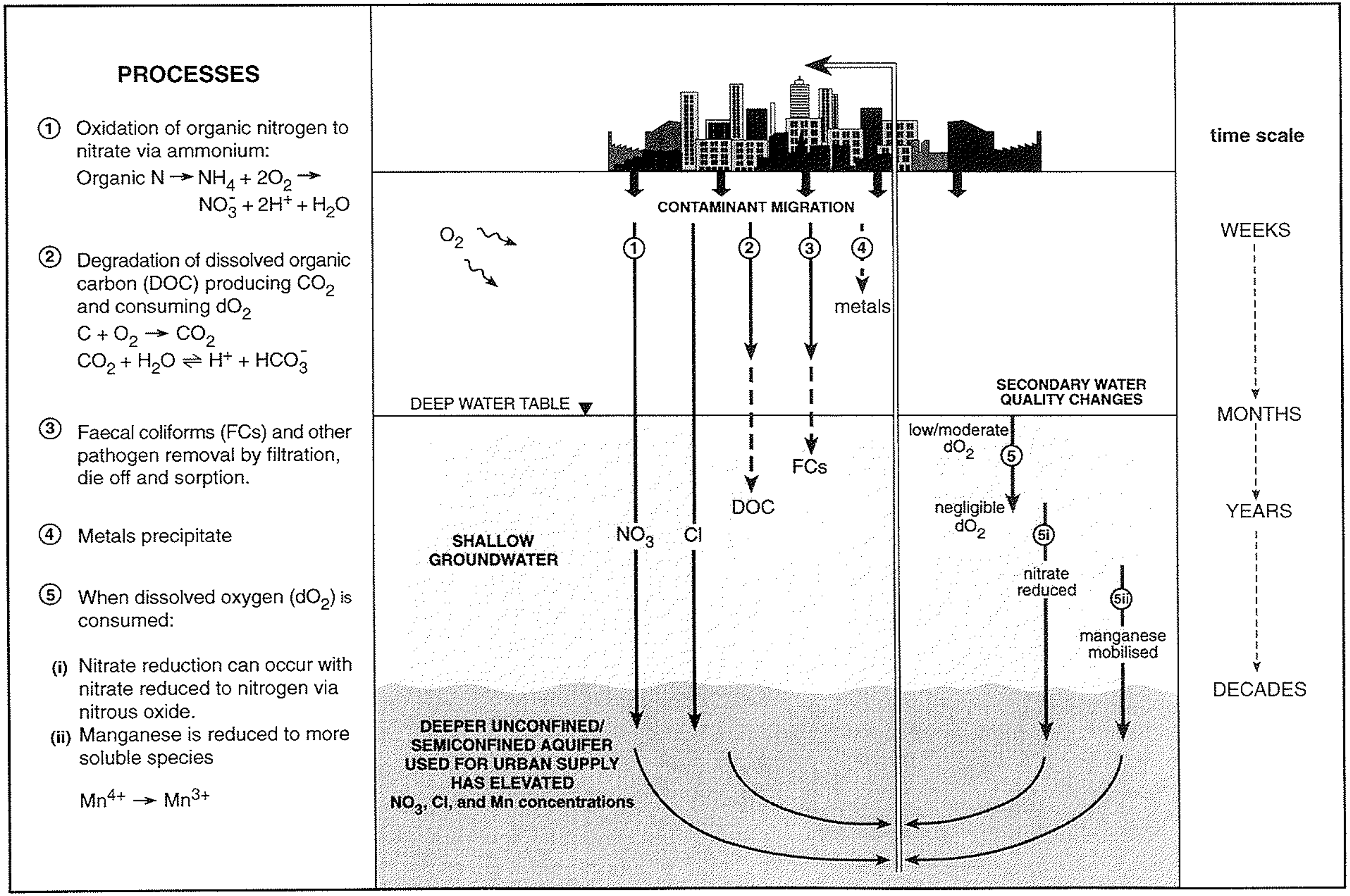


Figure 8.2 Assessment of urban water quality concerns and geochemical processes: aerobic groundwater system

a) Level of dependency on groundwater

If, in proportional terms, groundwater is little used within the economy, its pollution may not be as significant as where it is the bulk provider of water. However, much will depend on the **uses** to which groundwater is (or could be) put. Where water quality is of little concern - irrespective of groundwater dependency levels - pollution may pose little threat. Such a situation might arise where groundwater is exploited by large industrial users or municipality for non-sensitive uses, such as cooling or amenity area watering respectively. However, where groundwater dependency is high and where water uses are sensitive to changes in quality, the impacts of pollution are likely to be more severe.

In Hat Yai, where groundwater abstraction accounts for roughly 55% of total municipal water consumption, dependency is quite high, especially for larger users. However, while larger users may not be adversely affected by changes in water quality, smaller (household) users would be. One result could be a switch to piped (public) surface water supplies over the longer term, though probably at much higher cost. The situation in Hat Yai highlights the difficulty in drawing firm conclusions on the impact of pollution where much groundwater abstraction is in private hands, and where information on groundwater uses and users is sketchy. In circumstances such as these, there is a clear need for a comprehensive survey of groundwater uses and users, to inform water policy and avoid unintended outcomes (see section 9.3)

b) Opportunity cost of groundwater degradation

If alternatives to groundwater exist, then it follows that society and the economy are likely to be less vulnerable to potential groundwater degradation effects. This, clearly, is subject to the cost of such alternatives and their affordability to different uses and users. In a theoretical sense, alternative water will always be available, but the investment and operational costs of interbasin transfers and desalination are often prohibitively high, especially for lower value uses.

In the case of Hat Yai, no information was available on the cost of expanding supply capacity in the public (surface water) system should groundwater pollution render it unsuitable for sensitive uses. However, experience from other urban settings where water scarcity is increasing suggests that alternatives are typically 2-3 times more expensive (Bhatia and Falkenmark, 1991). Where groundwater abstraction is substituted by surface water development, the differential may be especially pronounced, particularly where groundwater is abstracted by private agents. In these circumstances, the cost and price differential will include the cost of providing a centralised reticulation system, as well as an alternative source of water.

c) Capacity to mitigate the problem

Sensitivity to a groundwater degradation threat is in large part a function of the capacity to mitigate its causes and/or effects. Financial and institutional capacity can be distinguished, though the former may help determine the latter.

Where the financial capacity to address causes and/or effects is high, groundwater degradation may be checked at an early stage or, alternatively, impacts can be mitigated more effectively. Thus, a sewerage system could be built to prevent seepage (address causes), or a treatment plant built to improve quality (address impacts). Alternative water sources will be more affordable, too. Thus in Hat Yai, a large capital programme is underway - with some external funding - to improve the urban environment. This includes measures to reduce pollutant flows in the city centre. In other, less well-off areas, such a scheme might not be possible. In general, large capital schemes prove popular with politicians keen to avoid potentially unpopular moves to curb water consumption and effluent discharges.

Institutional capacity encompasses many different factors. However, key issues revolve around (i) the existence of economic incentives and regulatory controls in the water sector; and (ii) their application. The distinction is an important one as, in many developing countries, it is clear that even where such measures exist - at least indicating a willingness on the part of the state to recognise problems - the measures are often ineffective. A fundamental problem in many areas is the monitoring and enforcement of restraints, especially where there are many private abstractors and polluters, and where low paid government officials are charged with controlling the activities of rich and politically powerful industrialists.

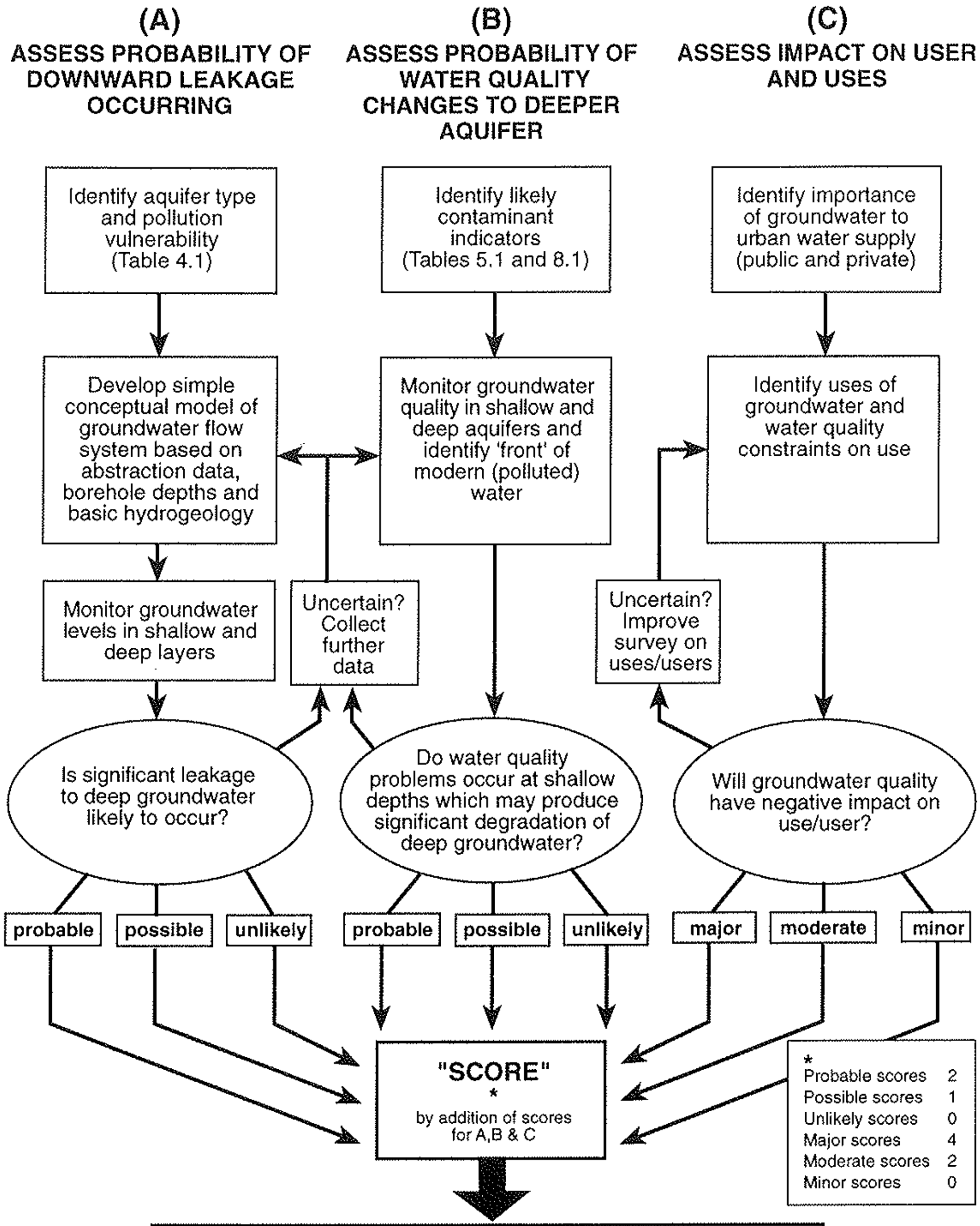
Institutional capacity can also be related to the notion of integrity. This refers to, firstly, the institutional separation of water supply and regulatory functions. This is increasingly being seen as indispensable for effective water management, but is still uncommon in the developing world. Secondly, integrity relates to the degree of institutional fragmentation within the water sector. Where, for example, groundwater and surface water, and abstraction and discharge management, are dealt with separately, piecemeal and uncoordinated decision-making may result. This may raise the threat of degradation by decreasing the ability to recognise causes and manage consequences.

8.3 Methodology

A simple methodology for assessing the risk posed to deep aquifers beneath unsewered cities has been developed by this project and is described below. This risk assessment is based on the probability of water quality changes occurring in deeper aquifers and the impact that these water quality changes have on water use and/or the user. For example a deep aquifer may be judged to be at only moderate risk even though the probability of water quality changes is considered high because the impact of these water quality changes on water use or the user is considered low.

The probability of water quality changes occurring in deeper aquifers is itself a function of both likelihood of significant leakage of shallow urban groundwater occurring (and the time scales) and the probability that this leakage may produce significant degradation of water quality in the deeper aquifer. Thus an aquifer may receive a large percentage of its recharge from shallow urban groundwater which is highly polluted but the impact on its water quality is judged low because the contaminants are non-mobile and/ or non persistent.

The risk assessment is produced as a flow chart (Figure 8.3). The risk assessment score is arbitrary and can be adjusted to suit local requirements. In the first instance it is probably useful to make a rapid assessment using existing information and Tables 4.1, 5.1 and 8.1 for guidance. If the risk assessment score is high (or moderate) it may be necessary to carry out further investigations (including monitoring) to improve the assessment. An example of how the risk assessment might be used is presented in Box 8.1 for the city of Dhaka, Bangladesh. The groundwater system and background data for Dhaka have been described elsewhere (Ahmed et al 1998 in press).



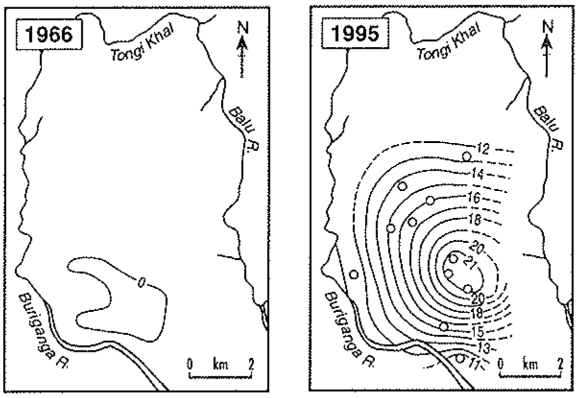
SCORE	RISK ASSESSMENT
HIGH (6 - 8)	High probability of quality degradation and significant negative impact on use/user.
MODERATE (3 - 5)	Moderate - high probability of quality degradation and/or moderate-major impact on use/user.
LOW (0 - 2)	Low probability of degradation and minor impact on use/user.

Figure 8.3 Risk Assessment Flow Chart

BOX 8.1 EXAMPLE OF RISK ASSESSMENT FOR DHAKA, BANGLADESH

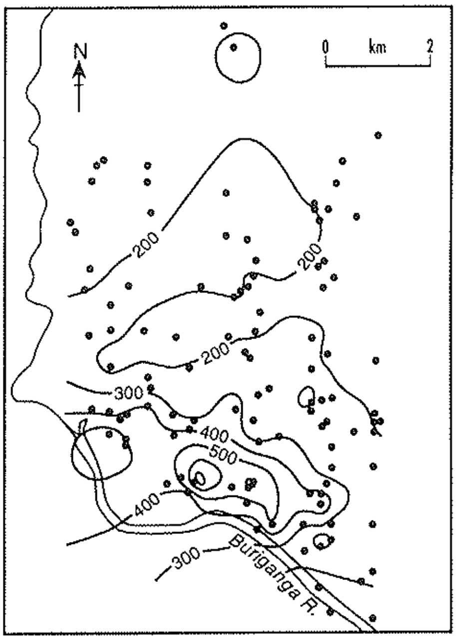
Dhaka, the capital city of Bangladesh, lies within the floodplain of the River Buriganga. The city, which has a population of 8.5 million, overlies thick unconsolidated alluvial sediments of sand, silt and clay. The city is dependent upon groundwater for water supply; more than 90% is provided by groundwater from the Dupi Tila aquifer. This aquifer is overlain by a 20-30 m fine-grained 'leaky' layer of silt and clay, known as the Madhupur clay. The city is largely unsewered and urban wastewaters are mostly disposed to the ground via septic tanks or discharged to surface water courses.

Recharge to the Dupi Tila aquifer occurs both as infiltration from the River Buriganga and as downward leakage through the Madhupur clay. Groundwater abstraction over the past 20 years has produced a significant depression of the piezometric surface.



Depression of piezometric surface of Dupi Tila aquifer, Dhaka

Electrical conductivities, representing chloride concentrations, have increased with time, especially close to the River Buriganga and where the depression of groundwater levels is greatest. The implication is that shallow (polluted) groundwater within the Madhupur clay, immediately beneath the city, is leaking into the aquifer in response to the steep vertical gradients induced by pumping (Ahmed et al, 1998 in press).



Electrical conductivity (μs/cm) of Dupi Tila aquifer, Dhaka, 1996

A preliminary assessment of the risk to the Dupi Tila aquifer can be made using Fig. 8.3. First, the probability of significant downward leakage of shallow urban groundwater must be considered high and should be given a score of 2. Second, the water quality problems that may occur within the shallow urban aquifer include high nitrogen and various synthetic organic compounds derived from industry. The probability that these water quality problems, given sufficient time, will produce a significant degradation of the water quality in the deeper aquifer must be considered high (score 1-2). Finally as the groundwater within the Dupi Tila aquifer is used for urban water supply, including for potable purposes, the degradation of water quality described above is likely to have a significant impact on groundwater use and/or user (score 2-4).

This initial assessment suggests the Dupi Tila aquifer beneath Dhaka has a high risk score (5-8) and that further action is recommended to confirm water quality problems within the leaky Madhupur clay layer and to predict with more accuracy, future water quality degradation in the aquifer.

9. GENERAL MANAGEMENT PRINCIPLES

It is apparent that water quality problems in deep groundwaters can be caused by the disposal of wastewaters to the ground, especially where steep (downward) vertical gradients are induced by pumping. As described in Foster et al (1997) strategies to achieve management objectives provide two principal options:

- reduce the steep vertical groundwater gradients beneath the city in order to reduce downward leakage. This can be more simply expressed as controlling groundwater levels
- moderate the contaminant load.

9.1 Means to achieve groundwater level control

The principal objective of controlling groundwater levels (and reducing vertical gradients beneath the city) is to reduce the leakage to deeper groundwaters from shallow contaminated aquifers. This can be achieved by:

(i) increasing shallow groundwater abstraction for non-sensitive uses
(ii) reducing pumping from the deeper aquifer and possibly restricting abstraction to sensitive (potable) uses only
(iii) redistribution of abstraction from within to outside of the urban area.

The choice of action will depend on the specific city problem. For example, increasing shallow groundwater abstraction may be most appropriate where the water table is very shallow and there is a significant demand for water for non-sensitive uses. Increasing the depth to the water table (a likely consequence of increased abstraction of shallow groundwater) may also have the beneficial effect of improving water quality by allowing greater oxidation and degradation within the unsaturated zone.

Reducing abstraction from the deeper aquifer will delay the arrival of urban-derived leakage. This may have some beneficial impact on water quality but it may only provide a useful time delay. Redistribution of abstraction for potable supplies from within the city to outside the urban area should provide a more satisfactory and long term solution, especially if combined with a policy of restricting land-use within and around the well field.

However, it is important to realise that the choice and scope for action will largely depend on whether groundwater abstraction is mostly controlled by a large number of private users or by a small number of water utilities. The difficulties in controlling groundwater levels in cities where abstraction is largely operated by many private owners are greater because:

(i) information on borehole construction, groundwater levels and rates of abstraction is often less readily available
(ii) the perspective of the private owner is focussed on obtaining sufficient water in the short term- the collective long-term good of the community is secondary
(iii) the small private owner has less opportunity (if any) for relocating their borehole outside of the city
(iv) even assuming resources exist, monitoring and enforcement is extremely difficult,whether controls use economic incentives or mandatory means, to manage abstraction. On this latter point, it should be noted that on a global scale, the establishment and **acceptance** of public ownership of groundwater is a rarity, and it is certainly more difficult to manage abstraction where private rights are strongly entrenched.

Furthermore, where large numbers of boreholes have been drilled, the risk of contamination caused by improper well construction or design is greatly increased.

9.2 Means to reduce contaminant load

Shallow groundwater is likely to be widely contaminated beneath many cities where mains sewerage is not universal and the soil layer and shallow subsurface are permeable. It is therefore accepted that it is not practicable to protect the shallowest aquifer from all pollution. However, it is essential to avoid excessive load of persistent pollutants, since even deeper aquifers, unless completely confined, are susceptible to the more persistent pollutants in the longer term, as this research has shown.

Controlling pollution load helps safeguard water quality by reducing the risk of unacceptable potable water quality problems and avoiding nuisance secondary effects caused by changes in the pH and redox potential of groundwaters. Industrial effluents containing persistent and mobile contaminants (e.g. CHCs) should be treated before disposal to avoid the risk of groundwater contamination.

Such control requires consideration of two separate but complementary aspects- the identification of those areas where the aquifer is vulnerable (to pollution) and the assessment of pollutant sources to include the types of contaminant and their associated loadings. The overriding objective would be to reduce or prohibit the potentially most-polluting activities in areas where the aquifer is vulnerable. However, where groundwater contamination is widespread and well advanced, for example beneath the predominantly unsewered districts of larger cities and especially in the arid regions where pluvial recharge (dilution) is limited, then it is important to recognise that the benefits of controlling contaminant loading are long term and may not be fully achievable.

Subsurface contaminant loading can be controlled by:

(i) restricting pollutant loading on vulnerable aquifers or sensitive areas (i.e. where steep vertical groundwater gradients exist). Specific measures might include; zoning land for different uses (specifically to concentrate certain types of development in low-vulnerability areas), selectively prioritising mains sewerage extension (to areas of high vulnerability or in the environs of source protection zones), restricting extension to, and density of, development in vulnerable areas

(ii) regulating treatment and disposal of industrial wastes. Specific measures might include; introduction of permits to dispose to ground, discriminatory charging for effluent disposal to favour pretreatment

(iii) regulating landfill design and location

(iv) stimulating waste reduction and recycling, by incentivising industrial users to treat their wastewater effluent or discharge in a regulated manner such that it can be used for a non-potable purpose, such as wastewater irrigation. This can be dovetailed with other integrated pollution control measures to effect a general improvement in industrial emissions standards and to reduce adverse environmental impacts

(v) spatially separating waste disposal from water supply functions.

9.3 Demand management and pollution control: exploiting complementarities

There are clear links between water use (surface and groundwater) and wastewater production. As a result, efforts to control water use are likely to limit wastewater production, and vice versa. In terms of pollution threat, however, an important consideration is the extent to which risk is related to hydraulic load, as opposed to contaminant quantity and quality (Boland, 1991). As noted in Section 9.2, the location of discharges in relation to aquifer vulnerability and abstraction is also critical.

Experience from countries around the world indicates that constraints on water use, and/or controls on waste discharges, can lead to greatly increased recycling ratios and the adoption of other water conserving and waste reducing measures in urban areas (Boland, 1991; Bhatia and Falkenmark, 1991; Bhatia et al, 1995). Evidence from Chinese and Indian industry, for example, demonstrates that raising *water tariffs* to economic levels, or something approaching them, can encourage recycling, abandonment of 'once through' processes, and investment in wastewater treatment for reuse. In the latter case, treatment may improve the quality of discharges, as well as reduce final volume. In Egypt, similar links have been drawn. For example Kosmo (1989) suggests that higher water prices would facilitate the separation of toxic and non toxic waste for treatment and safe disposal by encouraging reuse of wastewater. In terms of the water conservation effects of demand management and pollution control measures, it is important to distinguish consumptive from non-consumptive use. Industries with a high consumptive use, extract water from the system and deny its use to others. Water savings in these industries can therefore liberate high quality water for use elsewhere.

Growing acceptance of the 'polluter pays' principle can ease the way for the introduction of *charges on effluent discharge*. In principle, an economic charge would be related to the environmental damage caused by the discharge, or the cost of prevention, treatment or restitution, whichever is least (Winpenny, 1994; Bhatia et al, 1995). In practice, it is empirically difficult to estimate 'damage functions' for water quality deterioration, as this requires quantifying the effects of various contaminants on large numbers of 'downstream' users. For groundwater systems, moreover, time lags between discharge, movement through the unsaturated zone, and water quality impact, make the 'polluter pays' principle especially difficult to operate. As a result, pollution charges are often set to recover the costs of monitoring, administration and, occasionally, treatment, rather than specific environmental damage (OECD, 1987).

Pollution charges, levied on a volumetric basis, can encourage firms to treat their effluent and wastewater and recycle or reuse it, realising the twin benefits of water conservation and environmental improvement. As noted above, however, the impact of discharges on surface and groundwater bodies depends on effluent quality and location, as well as its quantity. Consequently, pollution charges may need to be combined with *mandatory controls* to account for local factors (including aquifer vulnerability), using a 'carrot and stick' approach (Boland, 1991; Winpenny, 1994; Bhatia et al, 1995). Where it is not feasible to charge firms for using their own water sources, such as private boreholes, pollution charges/controls may be one of the few means of encouraging conservation and pollution control. *Direct government intervention* can also be effective, especially where the enforcement of controls is problematic (eg where the major polluters are state enterprises). For example, soft loans with low interest rates can be made available to provide investment support for water conservation and treatment measures.

Where government enforcement is difficult, *public disclosure programmes* may also be effective in curbing industrial water pollution, though these have been trialed in only a few countries (eg Indonesia) to date (Afsah et al, 1996). Under these schemes, the environmental performance of firms is published in local media using, in the Indonesian case, easy to understand grades. Poorly performing firms are then pressurised into cleaning up their acts to maintain image, share value and market position. Thus, the role of the regulator evolves from pure adoption and policing of rules, towards the empowerment of other agents through provision of appropriate information. Though still in their infancy, early trial results have been encouraging, especially where polluters are sensitive to, and internalise, reputation effects. However, their potential role in mitigating groundwater pollution - less obvious and immediate than surface water pollution - has yet to be evaluated.

From the discussion above, it is clear that a variety of policy instruments can be used to control groundwater abstraction and pollution and exploit complementarities. It is important to note, however, that *measures need to be carefully orchestrated and implemented* to achieve intended outcomes. For example, where effluent quality controls are introduced without water conservation incentives, firms may find it more convenient (and economic) to dilute pollutants, rather than treat them (Boland, 1991). The result may be more groundwater abstraction and pollution, not less. Similarly, where 'private' groundwater use coexists with public surface or groundwater supply, policies aimed at changing the incentives facing one group may modify those facing the other. Thus in Hat Yai, evidence suggests that price increases for public (surface) supply make unmonitored and unregulated groundwater use more attractive to households. Moreover, limits on deep abstraction, introduced in isolation, might increase the incentive to tap shallower, more polluted waters. The net result may be more shallow groundwater use by households, and greater health risks for those using groundwater as a source of potable supply. The message is clear: well-intended policies aimed at improving the management of urban water resources may have unintended and unwanted side-effects, unless all water users - both formal and informal - are considered.

Policy and management issues are discussed further in Appendix A. Some of the policy criteria which can be used to compare alternative management options are discussed in Box 9.1.

BOX 9.1 COMPARING MANAGEMENT OPTIONS

Pollutant control and demand management options can be compared in a number of different ways. Important evaluation criteria include:

Efficacy

- Impact on groundwater abstraction and pollutant loads - quantity and quality effects.
- Location of demand and load changes in relation to local hydrogeological conditions (eg aquifer vulnerability), abstraction points, and different water uses and users.
- Time lag between introduction of a measure and its effect. Some measures will bring improvements in groundwater quality sooner than others. In the short-medium term, it may be necessary to address the impacts of groundwater degradation (eg through treatment), before measures to control contaminant load take effect.

Costs and benefits

- Costs and benefits of alternative options. The key issue here is whether the benefits of pollution control, or treatment, outweigh its costs. Benefits (eg to health) may be difficult to measure, and centralised treatment may not be an option where unregulated, private groundwater abstraction is significant (see Section 7.4 and Appendix A for more details).

Equity

- Distribution of costs and benefits. Where do costs and benefits fall, in terms of polluters, different water users and uses, and between present and future generations? If groundwater degradation proceeds unchecked, the eventual 'losers' may be future generations forced to pay for more costly out-of-town supplies.

Political and public acceptability

- Policies that are acceptable to the parties affected have better prospects of being implemented than those likely to encounter vigorous resistance. Factors affecting acceptability include: severity of the problem, and the level of uncertainty over aquifer conditions, rates of change and causal factors and agents; and the distribution and timing of costs and benefits, especially where benefits are long term and political time horizons short (see above).

Administrative feasibility

- Operating a policy must be within the capacity of the government department or agency involved. This implies the ability to monitor and enforce compliance with controls.

10. CONCLUSIONS

(i) Most cities located on alluvial sediments utilise groundwater for urban water supply. However, many of these cities are also unsewered and untreated effluent is normally disposed to the ground or to surface water courses; this frequently produces widespread contamination of shallow groundwater. As a consequence the shallow unconfined aquifer is usually abandoned by monitored and regulated uses (eg. public supplies) and instead deeper, partly confined aquifers are developed for potable water supply.

(ii) It is important to recognise that alluvial systems often contain several aquifers at various depths and that they are hydraulically interconnected. Given time and a sufficiently steep vertical gradient, water movement from one aquifer to another will occur. Thus deeper partly confined aquifers can be vulnerable to pollution due to leakage of shallow water induced by deeper pumping. However, since the travel times for water to migrate from shallow layers to deeper aquifers is likely to be years if not decades, only the most persistent contaminants are likely to pose a risk to these deeper aquifers. For most unsewered cities, nitrogen (as NO_3 or NH_4) and chloride are likely to be the two most important persistent inorganic contaminants.

(iii) The organic content of the urban wastes in most developing countries is largely derived from domestic wastewater and from food processing industries and is usually composed of easily degraded compounds. The risk to groundwater is therefore much less than that associated with urban-industrial wastes in Europe and North America where persistent, synthetic compounds (for example chlorinated solvents), are relatively more important. However the rapid degradation of the organic compounds frequently found in the wastewater effluent of many developing countries does deplete the groundwaters of oxygen and can produce strongly reducing conditions. This can lead to serious secondary water quality problems.

(iv) The creation of strongly reducing conditions in groundwaters, by the degradation of their organic content, is observed beneath Hat Yai and is likely to be a widespread phenomenon beneath many unsewered cities which have a shallow watertable. This in turn can lead to increased concentrations of iron and manganese. Whilst these are often regarded as 'natural' water quality problems it is clear that they can be a direct result of contamination. In Hat Yai, a serious health consequence of the mobilisation of iron is the associated increase in arsenic concentrations. Since both iron and arsenic are relatively common elements in alluvial sediments, mobilisation of these elements by the above mechanism may be widespread beneath many unsewered cities.

(v) The practice of disposing wastes to the subsurface from which groundwater is also pumped for water-supply can be sustainable. However, given the potentially serious consequences to groundwater quality in some aquifer systems it is essential that the risk to the urban aquifer be assessed and that careful monitoring and surveillance of groundwater be routinely undertaken. Since the timescales for groundwater flow and contaminant migration in these aquifer systems are protracted, once water quality problems develop they are likely to require long term and costly remediation.

(vi) Monitoring of groundwater quality within the deeper aquifer only does not provide an effective early warning since contaminants once detected are likely to spread within the aquifer and persist for many years. Instead monitoring of groundwater quality in the shallow layer is recommended to provide an advance warning.

(vii) However, once pollutants have migrated through the shallow layers options to manage the problem are limited because downward pollutant migration continues in response to deep groundwater abstraction. Reducing the pollution loading at the surface will help in the longer term although the full beneficial effect may not be attained for decades. Short-term solutions include treating water supplies or restricting groundwater abstraction within the urban area to non-sensitive uses only. In cities where there are a large number of private abstractors, treatment is not a viable option.

(viii) It is also vital to assess societal and economic sensitivity to pollution threats. One of the first requirements is for information on groundwater use and users. Where groundwater abstraction is in mainly private hands, a water use survey may need to be undertaken to inform policy.

(ix) Where it is judged that a significant risk to groundwater exists, then it may be necessary to reduce the contaminant load in vulnerable areas of the aquifer, and minimise vertical groundwater gradients by encouraging abstraction from the shallow aquifer or both. Spatial separation of waste disposal and groundwater abstraction should where possible be encouraged. Options to tackle problems are more constrained where groundwater abstraction is controlled by many private owners. Large numbers of boreholes also increases the risk of contamination caused by inadequate well construction or design.

(x) In all cases, it is necessary to assess what leverage over problems is possible, and to look for ways in which complementarities between different policy measures can be exploited. Measures need to be carefully orchestrated and implemented to achieve intended outcomes.

(xi) It is apparent that groundwater taken from deep aquifers is chemically and microbiologically of a much higher quality than that from shallow groundwater. The latter must be discouraged for potable use.

APPENDIX A. Some Cost and Policy Implications of Groundwater Degradation

Sustainable pricing and non-renewable use

The full economic cost of using a groundwater resource (of a given quality and at a given time and place), can be considered to have four separate components. If these costs can all be captured in its price, then consumption can be regarded as sustainable (Winpenny, 1995):

1. a *cost of provision,* or supply, in terms of the long term capital and recurrent costs of abstraction, treatment and distribution;

2. any *environmental costs* imposed on third parties, and not captured in (1) above (eg building damage costs arising from land subsidence);

3. a *current opportunity cost*, reflecting the potential scarcity value of water in that situation (value in alternative use), where it is feasible to transfer water between uses; and

4. a *future opportunity cost, or user cost* associated with non-renewable use, equivalent to the extra cost of a replacement source of supply, as compared with the current one. Groundwater use can be regarded as non-renewable when pumping rates exceed long term recharge, and/or contaminant loads exceed the assimilative capacity of the aquifer.

This 'sustainability formula' implies that, even where pumpers and polluters are confronted fully with the present economic costs of their actions (a rarity), groundwater usage may still be unsustainable in economic terms. This is because *future* scarcity, in terms of the cost of eventually having to replace the resource, is not signalled to present day users. This user cost can be estimated by calculating the present value of the cost of switching cost to an alternative at some future date. Substitution can be achieved by switching to another, 'backstop' technology (eg desalination or treatment of wastewater), or by switching to alternative stocks of the resource (eg a different aquifer, or surface water source).

The size of the user cost depends on several factors:

- the amount of groundwater of usable quality remaining;
- its rate of degradation;
- the cost and availability of substitute sources in the future; and
- the rate at which future consumption is discounted.

Thus, when groundwater reserves are abundant, rates of exploitation and pollution low, and current exploitation costs low, user costs are likely to be small or non-existent. These variables are often positively correlated. When stocks are low, degradation rapid and the cost of the 'next best' source is high, however, the user cost will be relatively high.

In many groundwater dependent cities in developing countries, demand and pollution pressures are intense, and alternative supply sources increasingly costly. In these circumstances, the principle of a degradation or user cost can be usefully applied (Winpenny, 1994).

Case study illustration: estimating the long term cost of groundwater degradation in Gaza City

In the Gaza Strip, groundwater abstraction greatly exceeds recharge, with the result that saline seawater is replacing freshwater in the coastal aquifer. At the same time, heavy pollution of the resource is affecting its quality, rendering it unsuitable for potable use in some areas. This means that current groundwater abstraction and pollution is reducing the amount of potable supply available for use in the future, bringing forward the time when more costly sources (desalination plants) have to be developed. In this context, a user cost attributable to degradation can be estimated, equivalent to the discounted extra cost, in today's prices, of having to use the more expensive substitute in the future.

The value per unit of resource J of the user cost UJ in any particular year t is indicated by the expression:

$$UJ_t = \frac{PA_{t+n} - CJ_{t+n}}{(1+r)^n}$$

Where:

- n is the number of remaining years over which the resource will be exploited. This is estimated at about 20 years for water of potable standard in Gaza City. After this, alternative domestic supplies are assumed to be provided by desalination.

- PA_{t+n} is the price of the equivalent quantity of the substitute resource which will be available in the time period n. Desalinated water is assumed to cost around $\$1.20/m^3$ for the quantity of domestic water needed in 20 years time. Current prices are used in the calculation for each successive year; this is reasonable if relative prices do not change.

- CJ_{t+n} is, in theory, the long run marginal cost (LRMC) of abstracting, treating and distributing a unit of groundwater at some future time, n, assuming that some still remains to be extracted. In municipal areas where water tariffs are levied, it is thought that prices do not reflect full production costs. In the absence of more data, LRMC is assumed to be around $\$0.20/m^3$, a little above the average cost of water paid by a typical household using 30 m^3/month in Gaza City.

- r is the opportunity cost discount rate used to convert future costs into present day prices. Here,it is assumed to be 10 %.

In Figure 1, user cost estimates obtained using this formula have been plotted for the 20 year 'exhaustion' period. This illustrates how the user cost increases as depletion of high quality groundwater stocks approaches, rising from just less than \$0.20 in 1994, to \$1.00 20 years hence. Thus in Year 20 (2014), the full economic costs of groundwater and desalination sources are equalised at \$1.20 (user cost plus cost of provision, and ignoring present opportunity and environmental costs for the moment), and users are assumed to have fully switched from one source to the other.

The size of the user cost in the 1994 base year, over and above the cost of production, indicates that present groundwater users pay much less than full economic costs for their water. Indeed, full economic costs may be higher than so far indicated, as environmental and present opportunity costs (items 2 and 3 above) have thus far been ignored. In Gaza City, one environmental cost is the impact of municipal pumping on private well operators in surrounding areas, affected by saline intrusion. Other unquantified environmental costs can also be identified: for example, groundwater pollution may be causing health problems in the city. The current opportunity cost of municipal water use (item 3 above) is ignored, as the irrigation value of water in the surrounding area (the next best alternative use of urban groundwater) is assumed to be less than existing municipal values.

Before examining the policy implications of this discussion, some major caveats should be noted. In this illustration, there are some uncertainties about the data used. In addition, the example highlights some wider concerns with user cost calculations, and the assumptions on which they are based. For example, the calculation assumes constant production costs, and independence between increasing user costs, groundwater demand, and aquifer exhaustion. In reality, if the user cost premium is incorporated in a groundwater tariff, or pollution charge, the increasing cost to pumpers and polluters will reduce groundwater demand and pollution. The value of the user charge will therefore have an influence on parameter n. To estimate n at all also involves taking a view of the shape and behaviour of demand for the resource over the life period of the stock (ODA, 1988). Ideally, an iterative series of calculations would be the best way to estimate demand each year, and thus determine the period n. This would seem especially important for those groundwater resources which can be exploited in both renewable and non-renewable ways, and where a transition may occur from non-renewable to renewable use.

The Gaza City illustration highlights a further issue: sustainable use of a groundwater resource does not *necessarily* imply that some particular quantity and/or quality of resource need be maintained intact. In Gaza, degradation of limited urban groundwater would seem inevitable, because demand and load pressures are so great. The key issue emerging from discussion above is whether present users can be confronted with present and future costs of scarcity; if they can, degradation is still likely, but pricing and use could be considered sustainable .

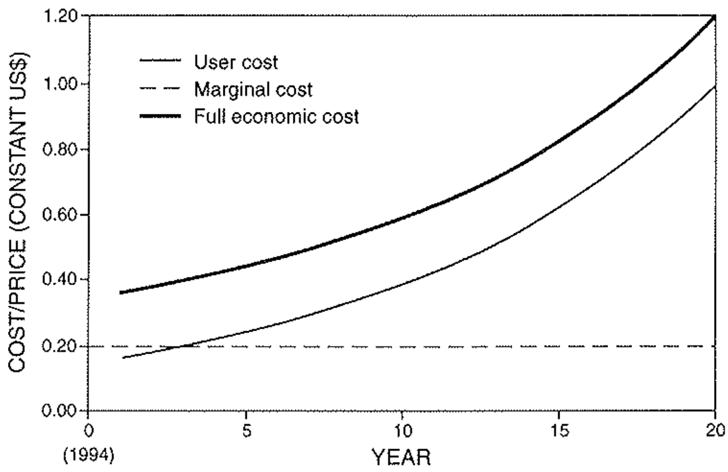


Figure 1 The long term cost of water supply in Gaza City

Policy implications: some general issues and conclusions

- Groundwater degradation threatens the water supplies of many developing country cities, imposing direct and indirect costs on water utilities, private well owners, consumers in general and society at large. In addition, unchecked degradation imposes a cost on future generations, in terms of the (often) much higher cost of a replacement source. Some megacities (eg Mexico City) are facing these costs now. For the many fast growing medium-sized cities in developing countries, the challenge is to make decision-makers aware of the costs of inaction. User cost estimates help in this respect, as they signal the future costs of replacement water supply in current prices, highlighting the growing economic scarcity of water.

- In theory, user costs can be factored into water tariffs and effluent charges so that pumpers and polluters are confronted with future scarcity costs, over and above existing water supply, opportunity and environmental costs. For example, a user charge could be added to a municipal water tariff to ensure that dwindling reserves are used as efficiently as possible, with lower value users giving up water to higher value ones as exhaustion approaches. This would ideally be an iterative process to reflect the fact that, for groundwater, demand is not given, and final depletion not a foregone conclusion.

- There are clear complementarities between water pricing and pollution control, and in many situations the setting of economic charges for water may be the best way of discouraging industrial water pollution (Bhatia and Falkenmark, 1991; Winpenny, 1992). Where it is not feasible to charge firms for using groundwater (eg when they operate their own private wells), effluent charges can be effective in reducing both water demand and pollution (Winpenny, 1994; Bhatia et al, 1995). In theory, an economic effluent charge would be related to the environmental damage - including the cost of future supply substitution - caused by the discharge (a function of its quality, quantity and location), or the cost of prevention, treatment or restitution, whichever is least. In practice, it is empirically difficult to estimate 'damage functions' for water quality deterioration, as this would require quantifying the effects of various contaminants on large numbers of 'downstream' water users. Application of the 'polluter pays' principle is especially difficult for groundwater, where polluters discharge into a common pool. Nevertheless, it is important to recognise that a water tariff, and/or effluent charge, should include some damage cost estimate, including user costs.

- The discussion above hints at the gulf which often exists between 'textbook' economics and the practical realities of groundwater management. As Briscoe (1997) notes, conventional economics may tell us that users and polluters should pay full economic costs, but in most developing country situations, aiming for economic perfection is not practical. Where does this leave us? Briscoe (1997) argues that the first requirement for reform in the water sector generally is *demand* for reform, and that scarcity and pollution often provide a powerful impetus. A problem with groundwater, however, is that degradation is typically slow and insidious. In this context, it would seem important to make the costs of degradation, including the increasing *economic scarcity of water in future*, explicit. Placing a dollar sign against degradation outcomes may therefore have the political effect of making *mitigation* more persuasive.

- Having demonstrated a need for change, the next challenge is to devise a set of *workable* institutional arrangements that confront pumpers and polluters with at least some of the costs of their actions. In this sense, fine tuning the policy mix to incorporate, for example, the four components of sustainable pricing described above, is a long way down the reform road. For many cities, a first step is monetary charging which fully recovers production costs, rather than full economic costs, and legal and institutional reforms which give some leverage over private abstractions and polluting industries. Direct government interventions can also be effective, especially where the monitoring and enforcement of controls is administratively difficult. For example, soft loans with low interest rates can be made available to provide investment support for water conservation and effluent treatment. From an economic perspective, the costs of direct intervention may be considerably less than the costs of finding additional water supplies (Bhatia and Falkenmark, 1991).

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