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UNCONSOLIDATED SEDIMENTARY AQUIFERS : REVIEW NO 12 - GROUNDWATER QUALITY MANAGEMENT IN UNCONSOLIDATED SEDIMENTARY AQUIFERS

P J Chilton, M E Stuart, W M Edmunds, H K Jones, A R Lawrence, P Shand and T R Shearer

Natural Environment Research Council

International Division
British Geological Survey
Keyworth
Nottingham
United Kingdom NG12 5GG

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Unconsolidated Sedimentary Aquifers

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UNCONSOLIDATED SEDIMENTARY AQUIFERS (UNSAs)

PREFACE

This review is one of a set of reports prepared as part of a project entitled "Groundwater Development in Alluvial Aquifers", Project R5561 (BGS 93/2), under the ODA/BGS Technology Development and Research (TDR) Programme of aid to the developing countries. The project addresses all unconsolidated sedimentary aquifers (UNSAs), not only alluviums.

The review is a compilation of existing knowledge. It is intended to be updated, as appropriate, following the results of research which will be carried out during the lifetime of the project, which is scheduled to finish in 1996. It addresses the management of groundwater quality, with particular reference to natural groundwater quality and the impacts on groundwater quality of urbanisation, agriculture, wastewater and salinisation.

The project is funded by the ODA as part of its research and development programme designed to improve living standards and conditions in the world's developing countries.

Project Manager: Dr R Herbert
Hydrogeological Adviser to ODA
British Geological Survey

INTRODUCTION

WHAT ARE UNSAs AND WHY IS IT IMPORTANT TO UNDERSTAND THEM?

UNSAs are unconsolidated sedimentary aquifers. These are the water-bearing strata within the swathes of unconsolidated sediment that mantle much of the earth's surface. There is no clear dividing line between UNSAs and aquifers in consolidated rocks, as lithification is a gradational process: deposits a hundred years old can be lithified, while some deposits 500 million years old are still essentially unlithified. However, for most purposes, UNSAs can be understood as deposits which have accumulated over the past few million years, that is during Quaternary and Neogene (late Tertiary) time. They are important sources of water in many parts of the world, and in particular constitute the only major sources of groundwater for vast areas throughout the developing world. In the influential text-book *Hydrogeology* by Davies and De Weist it says:

> "The search for ground water most commonly starts with an investigation of nonindurated sediments. There are sound reasons for this preference. First, the deposits are easy to drill or dig so that exploration is rapid and inexpensive. Second, the deposits are most likely to be found in valleys where ground-water levels are close to the surface and where, as a consequence, pumping lifts are small. Third, the deposits are commonly in a favourable location with respect to recharge from lakes and rivers. Fourth, nonindurated sediments have generally higher specific yields than other material. Fifth, and perhaps most important, permeabilities are much higher than other natural materials with the exception of some recent volcanic rocks and cavernous limestones."

To date, though, few attempts have been made to understand the detailed internal structure of unconsolidated aquifers even though such knowledge may be crucial to the long term success of any water development project. This shortcoming is probably the reason why the operational lives of many water boreholes are frequently much shorter than expected.

Understanding of the internal structure or "architecture" of many types of sedimentary deposit has, however, advanced greatly over the past couple of decades. Part of this research has been academic, but much has been sponsored by the oil industry, so as to better predict the possible location of oil within sedimentary traps. Oil, like water, is most profitably located within bodies of relatively coarse-grained and porous sediment. Thus, there is obvious scope for applying this recently gained understanding to hydrogeological problems. Advances have also been made in the understanding of the geometry of complex "soft-rock" deposits by the application of appropriate combinations of investigative techniques, including remote sending, rapid geophysical methods and new drilling techniques. The combination of these bodies of knowledge can provide a framework for locating and assessing UNSAs.

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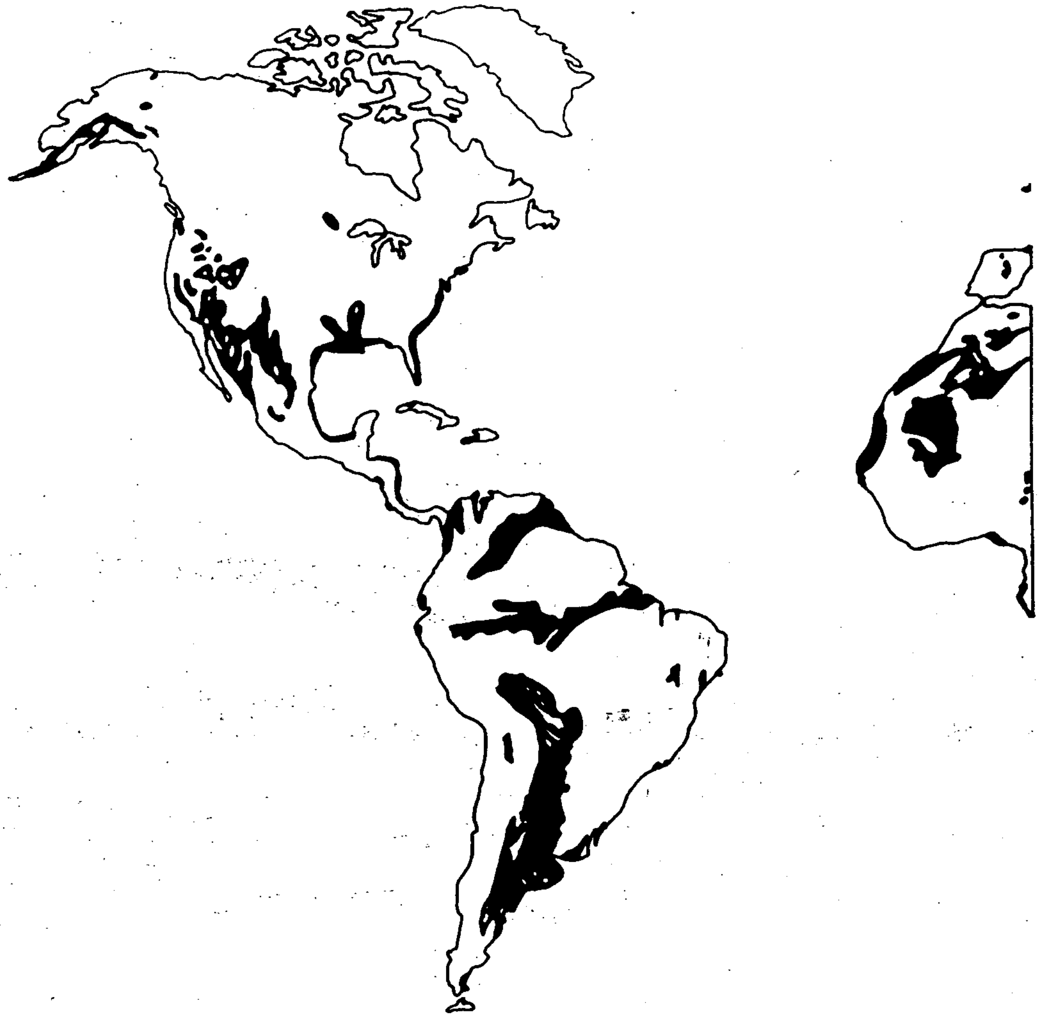


MAJOR AREAS OF UNCONSOLIDATED SEDIMENTARY AQUIFERS WORLDWIDE

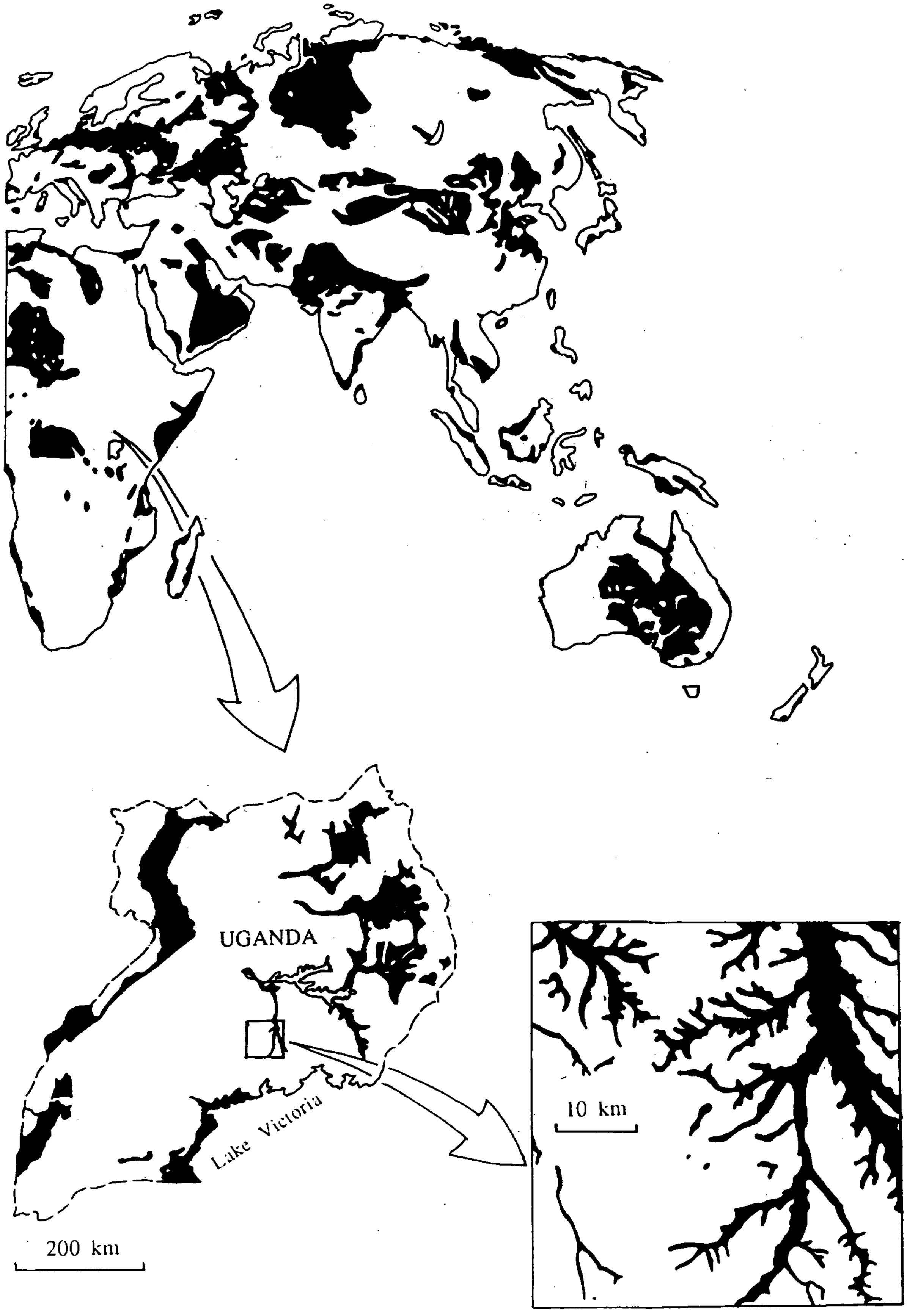
- The map shows the distribution of the thickest and most extensive Quaternary deposits in the world. The great majority of these are unconsolidated, and many include water-bearing deposits (UNSAs).

- A generalised world map such as this, though, severely under-estimates the true extent of UNSAs worldwide. This is because:

  - unconsolidated pre-Quaternary deposits are omitted; these too have a wide distribution, though are difficult to delineate (as they grade into consolidated deposits); they too can include significant UNSAs.

  - the simplification of linework necessary at this scale means that a large proportion of unconsolidated deposits have had to be omitted. The inset map shows the example of Uganda, which seems to have no unconsolidated sediments at the global scale, while significant and extensive deposits 'appear' once the country is looked at more closely. At a yet larger scale the unconsolidated sediments appear yet more widespread. The message is clear. *Unconsolidated sediments, and therefore UNSAs, are ubiquitous.*

Diagram data modified from various sources.


UGANDA
Lake Victoria
200 km
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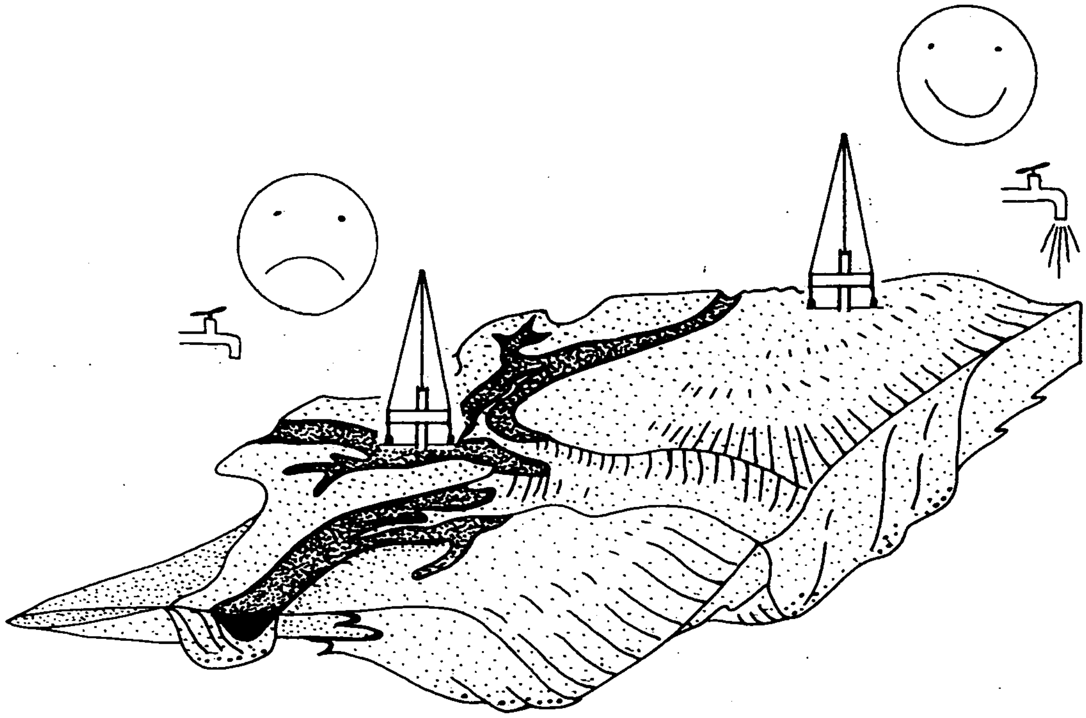

Sedimentary bodies are characterised by variably complex geometry and internal structure. These properties exert a strong internal control on the location, quantity and quality of groundwater. Diagram adapted from Galloway and Hobday (1983).

CONTENTS

1. SCOPE AND AIMS OF THIS REVIEW

1.1 Importance of groundwater in unconsolidated aquifers

Groundwater is an important resource providing a significant contribution to the potable, irrigation and industrial water needs of many countries of the world. It has been estimated that groundwater accounts for some 8 million km^3, or more than 95% of the world's usable reserves of fresh water, excluding water locked up in snow and ice. A very high proportion of this groundwater is stored in unconsolidated sedimentary aquifers (UNSAs). This is because, firstly UNSAs cover extensive areas of the world, as shown in the introduction, and are often of very great thickness. Secondly, the interconnected and drainable porosity of these sediments is significantly greater than that of other aquifer types, with the exception of some limestones and recent lavas.

UNSAs vary greatly and range from regionally extensive aquifers within alluvial coastal plains to thin river channel deposits of limited extent. The former may contain well in excess of 10,000 km^3 of groundwater and support the potable, irrigation and industrial needs of many tens of millions of people, whilst the latter may be capable of providing the water requirements of a small town only. However, even these limited groundwater resources may be important locally especially in semi-arid and arid regions where they may represent the only source of water supply.

UNSAs are rarely single, homogeneous aquifers but more typically are composed of laterally and vertically heterogeneous deposits such that aquifer characteristics can vary significantly over relatively short distances. It is not uncommon for several permeable horizons (aquifers) separated by less permeable layers (or aquitards) to be identified within a single UNSA. These permeable horizons are often regarded as separate aquifers although experience has shown that abstraction from an individual horizon will inevitably affect the water levels in the others (with subsequent leakage), albeit after a time-lag of days, weeks or even years.

In addition to differences in the extent of the aquifers and the volumes of groundwater stored, there is also considerable variation in the timescales of groundwater movement of thousands of years or more for deep regional flow systems (Figure 1.1). Such differences have a profound impact on groundwater quality and are discussed later. These may vary from a few tens of years for shallow, unconfined aquifers to many tens

The first review in this series classified UNSAs into broad sedimentary environments which include alluvial plains, coastal alluvium, intermontane basins, river valley deposits and wind-blown deposits. Table 1.1 takes some examples of these and indicates in a broad sense their importance as sources of groundwater. The first three environments are clearly the most important whether measured by volume of groundwater stored, total groundwater abstraction or by population dependency. The alluvial plains form the most extensive aquifers; the Indus valley deposits of Pakistan for example, cover some 320,000 km^2 whilst it is estimated that more than 100 million people are dependent upon groundwater for both water supply and for irrigation within the extensive Indo-Gangetic plain sediments.

Table 1.1 Examples of major UNSA types

Sedimentary Environment	Example Location	Climate[1]	Area[2]	Depth[3]	Aquifer Type	Groundwater Uses	Potential Pollution Sources	Known Water Quality Problems	Comments
COASTAL ALLUVIUM (includes deltaic alluvial and shallow marine deposits)	Cauvery Basin (India)	H	extensive	deep	multi-layered	extensive irrigation urban water supply	saline intrusion urbanisation agrochemicals	salinity, iron	>50-100 million dependent upon coastal alluvium in SE Asia. Volume of groundwater stored very large.
	Madras (India)	H	extensive	shallow-moderate	multi-layered	agriculture urban water supply			
	Jakarta (Java, Indonesia)	H	extensive	deep	multi-layered	urban water supply	urbanisation industry saline intrusion	salinity	
	Semarang (Indonesia)	H	extensive	deep	multi-layered	urban water supply			
	Calcutta (India)	H	v. extensive	deep	multi-layered	urban water supply (irrigation)	urbanisation industry saline intrusion	salinity	
	Cairo (Nile Delta, Egypt)	SA	v. extensive	deep	multi-layered	urban water supply (irrigation)	urbanisation industry saline intrusion agrochemicals	salinity	
ALLUVIAL PLAINS (includes alluvial fans, river alluvium)	Indus Valley (Pakistan)	SA	regional	deep	unconfined v. porous	agriculture urban/rural water supply	salinity, urban agriculture	salinisation of water table	>400 million dependent upon groundwater. Volume of groundwater stored is enormous.
	Indo-Gangetic Plain (India)	SA-H	regional	deep	unconfined-confined multi-layered	irrigation urban/rural water supply	urbanisation industry agriculture	high nitrate salinity metals, iron	
	Bramaputra Valley (Bangladesh)	H	regional	deep	unconfined-confined multi-layered	irrigation urban/rural water supply	urbanisation industry saline intrusion	high iron salinity	
	Central Plains (Thailand)	H	v. extensive	deep	unconfined-confined multi-layered	urban/rural water supply irrigation industry	urbanisation industry saline intrusion	salinity	
	Huang-Huai Hai Plain (China)	H	regional	deep	multi-layered	irrigation urban/rural water supply	urbanisation industry saline intrusion	salinity	

Sedimentary Environment	Example Location	Climate[1]	Area[2]	Depth[3]	Aquifer Type	Groundwater Uses	Potential Pollution Sources	Known Water Quality Problems	Comments
	Chiang Mai (Thailand)	H	extensive	deep	multi-layered	urban water supply	urbanisation industry		
	Mexico City	SA	extensive	deep	multi-layered	urban water supply	urbanisation industry	salinity industrial chems?	Considerable volumes of groundwater stored. Supports major concentration of population on valley floor.
	Bandung (Indonesia)	H	extensive	deep	unconfined-confined multi-layered	urban water supply	urbanisation industry	nitrate	
	Kathmandu (Nepal)	H	extensive	deep	unconfined-confined multi-layered	urban water supply	urbanisation industry	nitrate	
RIVER VALLEY DEPOSITS			limited	shallow	unconfined-semi-confined	urban/rural water supply	various		Locally important aquifers but volumes stored can be limited.
WIND-BLOWN DEPOSITS (a) Loess	China (Muanghe River)		extensive v. extensive	shallow-moderate	unconfined?	water supply	?	none	Population generally small.
(b) Aeolian sand dunes	Chinese Turkestan	A	v. extensive	moderate	?	?		poor quality	

1 H = Humid, SA = Semi-arid, A = Arid

2 Regional = >50,000 km^2, Very Extensive = 10-50,000 km^2, Extensive = <10,000 km^2, Limited = <100 km^2

3 Deep = >200 m, Moderate = >50 m <200 m, Shallow = <50 m

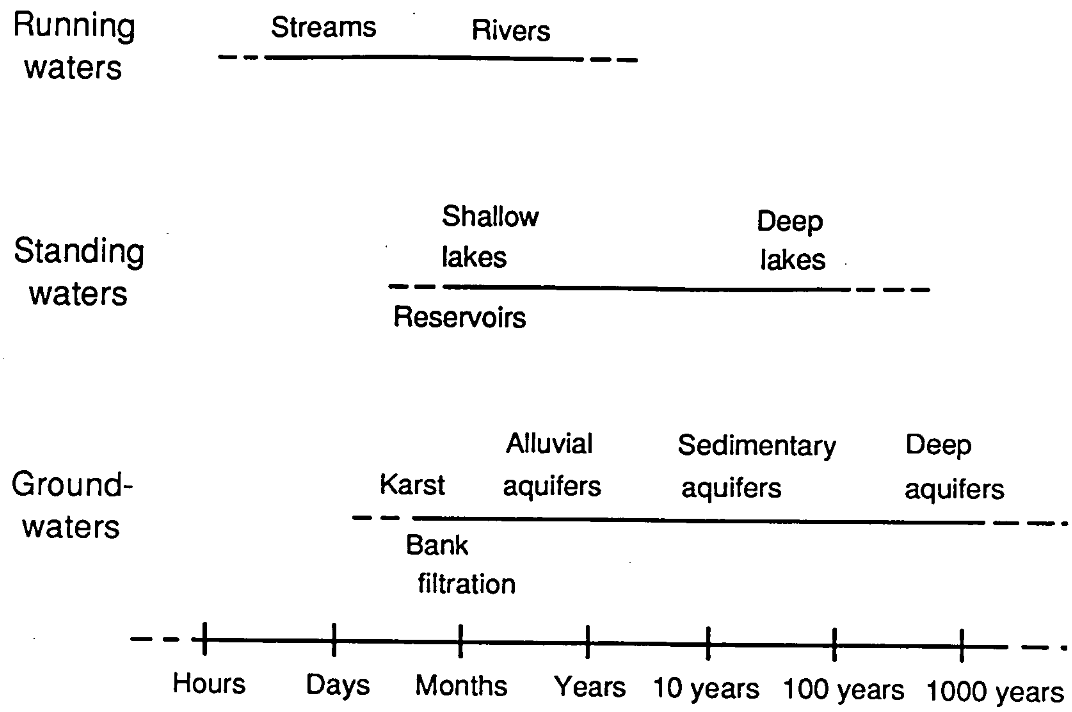

Figure 1.1 Water residence time in inland freshwater bodies (After Meybeck et al, 1989)

Aquifers in coastal alluvium whilst generally not so extensive as in the alluvial plains are however most important sources of water supply for many major population centres, i.e. Jakarta, Calcutta, Madras and Cairo. In South, East and South East Asia alone, it is estimated that aquifers in coastal alluvial sediments provide the urban water supply for more than 50 million people and probably in excess of 100 million.

The intermontane basins whilst often significantly less extensive than either the alluvial plains or coastal plains deposits nevertheless are important sources of water to many cities including Mexico City and many other large cities of the central mountain region of Mexico, Bandung, Jogjakarta, Kathmandu, and Chiang Mai.

In terms of the volumes of groundwater utilised and the population dependent upon groundwater, UNSAs represent the most important aquifer type. There is, however, widespread concern in many developing countries that the quality of groundwater resources is under threat, and in some instances is deteriorating. Governments have understandably devoted their finances to the development of new water supplies, and have often given less attention to the possible deterioration of existing sources. The deterioration is due largely to pollutants derived from activities on the land surface such as rapid urbanisation, industrialisation, intensification of agriculture and the disposal of sewage wastewater onto land. It is generally recognised that water quality is likely to become at least as important an issue as the quantity of available resources, and that water quality management must play a significant role in water policy in many countries in the 1990s and beyond.

1.2 Groundwater flow systems and residence times

Groundwater exists where there is sufficient rainfall to penetrate through the soil layer and where the underlying rocks are porous and permeable enough to store and transmit water. In the case of unconsolidated sediments, provided the deposits are mostly of silt grade, or coarser, they are likely to be sufficiently permeable to form useable water resources; this is especially true when providing potable supplies for rural communities where individual water supply requirements are usually low.

Water usually takes many months or years to move through the soil and unsaturated zone to the saturated zone of the aquifer. Once there, it can take many tens or hundreds of years to flow into a supply borehole. In some of the deeper alluvial basins, groundwater is likely to be thousands or even hundreds of thousands of years old. These time scales are an indication of the importance of aquifers as natural stores of water. The water cycle is often depicted as a simple system of circulation that takes water from the sea as vapour, deposits it on land as precipitation and returns it quickly to the sea by rivers. In this instance the ocean is the large store of water.

In fact, the cycle often transfers water between two stores, the ocean store (the primary and larger one) and the secondary store (the ice caps, glaciers and groundwater). UNSAs, particularly those in the alluvial and coastal plain sediments represent major stores of groundwater. The residence time of water in an aquifer can be up to many thousands of years (Figure 1.2). These long residence times and the subsequent slow release of water are important. Firstly, many rivers continue to flow during dry seasons because of slow release from groundwater (or from ice melt). Secondly, during prolonged periods of drought, aquifers can usually maintain supplies because of the considerable volumes of groundwater stored.

1.3 Factors affecting groundwater quality

Groundwater quality in UNSAs is generally good. The slow intergranular water movement ensures that microorganisms are generally removed during percolation to the water table. Further, the soil zone is effective in removing many contaminants as a result of sorption and degradation. Indeed it is the high quality of groundwater and the fact that no treatment (or minimal treatment) is normally required that makes groundwater so important and widespread a source of drinking water in many countries. However the rapid growth in population and its concentration in towns and cities, the increased use and disposal of chemicals on the surface and the widespread development of groundwater increasingly stretch the capacity of the soil and unsaturated zone to protect this valuable resource.

Groundwater quality is affected by many factors. Firstly, the climate type will largely influence the quality of the natural recharge. In arid regions, the salinity of the infiltration may be high as a result of concentration of the rainfall solutes by evapotranspiration. High nitrate concentrations have also been observed in some arid areas and these have been attributed to mineralisation and leaching of organic nitrogen during previous wetter cycles and limited dilution in subsequent low rainfall periods.

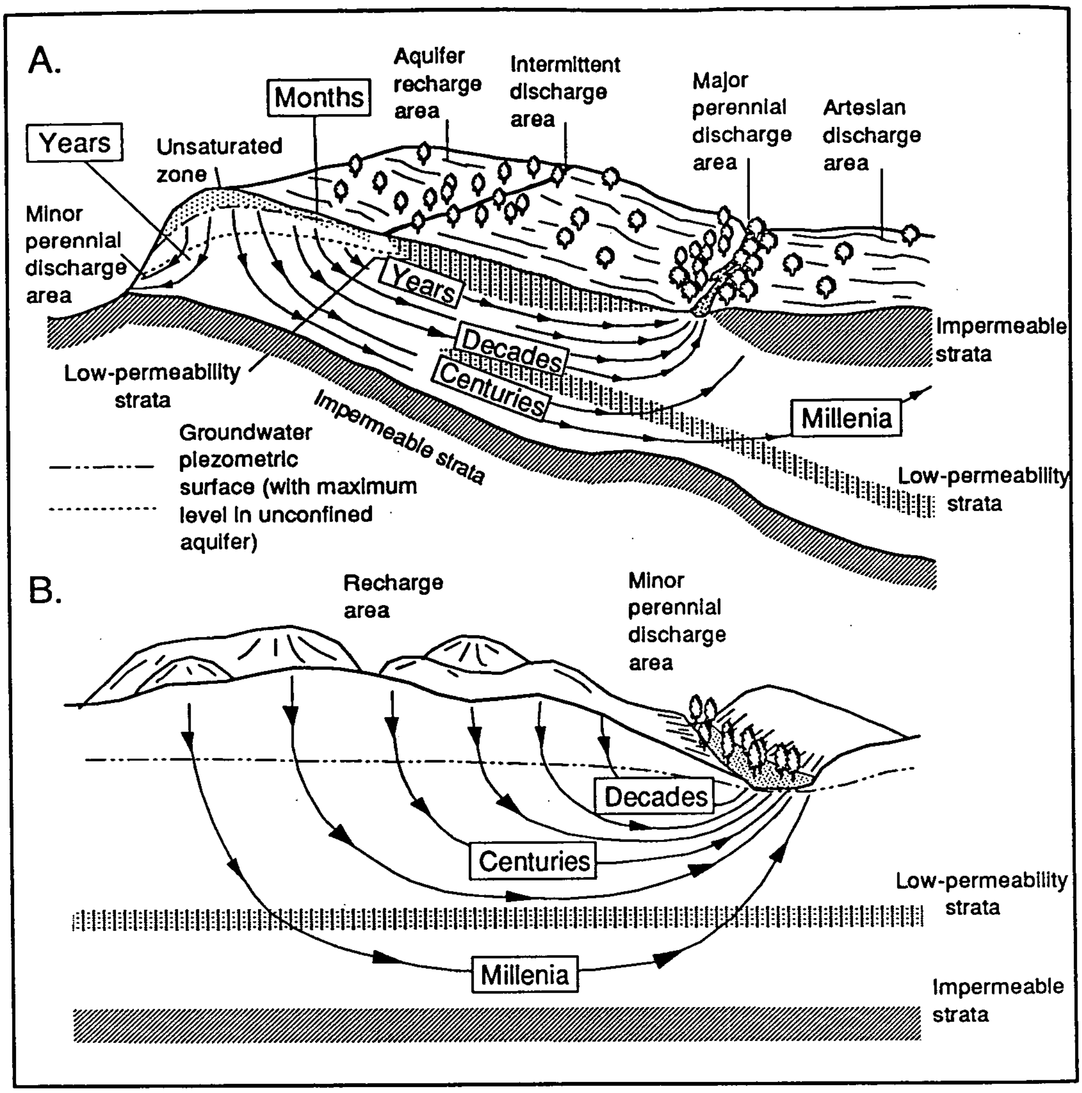

Figure 1.2 Groundwater flow systems A. Humid regions. B. Arid regions. The residence periods indicated are typical order-or-magnitude values from time of recharge to point of discharge (after Foster and Hirata, 1988)

Secondly, within the aquifer the quality of groundwater will evolve as a result of water-rock interaction, degradation, solution and precipitation processes. For example, in groundwater systems where oxygen has been depleted, iron may be reduced to the more soluble ferrous form, producing high dissolved concentrations. This problem has been frequently observed in deeper boreholes in alluvium.

Thirdly, groundwater quality is also strongly influenced by man as a result of both mismanagement of the aquifer and pollution from activities at the land surface. One serious problem is that of salinisation of groundwater. Overabstraction inducing seawater intrusion is common in many coastal aquifers whilst upconing of connate waters has also been widely reported. Salinisation of the shallow water table, caused by a combination of poor drainage and high evaporation rates is also a serious and widespread problem. In India alone, nearly a quarter of the irrigated land within the Ingo-Gangetic plain is currently affected by salinity problems.

Rapid development in some Asian and Latin American countries, including urbanisation, industrialisation and the intensification of agriculture all contribute to a deterioration in groundwater quality. This is exacerbated as the infrastructure to handle wastes and the regulations to control the use and disposal of chemicals and wastewater often lag far behind the development.

The long travel times in groundwater systems, the considerable dilution and mixing within UNSAs, and the lack of adequate monitoring ensure that many of these problems go unrecognised until large volumes of groundwater have been contaminated. Given that treatment and/or remediation of groundwater is nearly always expensive and is often not feasible, the implications of the deterioration of groundwater quality are serious.

1.4 Groundwater quality and pollution

While groundwater quality derived from unconsolidated sediments is generally of good quality, there are three categories of quality problem which are widely encountered:

- naturally-occurring quality problems, related to the hydrochemical evolution in certain types of strata
- anthropogenic pollution of vulnerable and inadequately protected aquifers
- saline intrusion in inadequately managed aquifers

These are described in turn in this review. A fourth and all too frequent cause of water quality problems and a constraint on groundwater use is inadequate design, construction, operation and maintenance of wells and boreholes themselves. Poor sanitary sealing of boreholes, which can allow the direct ingress of polluted water in the immediate surrounding of the well or borehole is a common cause of microbiological contamination. Where unconsolidated aquifers are permeable and have shallow water tables, this may in fact be a dominant cause of poor quality water. This issue is adequately covered in the literature (Lloyd and Helmer, 1991) and is not discussed in this review.

Most countries undertake at least some monitoring of groundwater quality, but existing quality information provides an inadequate overall picture of groundwater quality in UNSAs, or indeed in any aquifers. Monitoring of major ion chemistry is probably the most common. Information on organic pollutants such as industrial chemicals and pesticides is much less widely available. Because, as already described, they are often densely populated and support intensive agricultural activities, large urban areas and rapidly growing industries, widespread groundwater quality problems in UNSAs can be anticipated. Improvements to monitoring and assessment of groundwater quality are urgently required. This need is widely recognised, and useful sources of advice on improving monitoring are becoming available (Chapman, 1996).

The risk of pollution of groundwater by human activities depends on the interaction between the vulnerability of the aquifer and the contaminant load that is applied to the subsurface. The vulnerability depends on the intrinsic properties of the soil and aquifer, and hence cannot be modified. The pollution load can be controlled or removed, and management or control policies for protecting groundwater usually do this in various ways. The subject of aquifer vulnerability and groundwater protection policy is discussed in Review No 6.

1.5 Aims of review

The aim of this review of groundwater quality within UNSAs is to describe the processes, both natural and anthropogenic, controlling groundwater quality. This is done in the following chapters, which are reviews, in each case prepared by one or more authors as shown, of the existing state of knowledge rather than technical descriptions of detailed methods. Investigation of groundwater quality draws on many of the more detailed methods described in the other reviews, such as borehole construction, geophysics, stable isotopes and modelling. Where appropriate, however, the specific key issues and approaches to the investigation of groundwater quality are summarised from case studies as "boxes" within the text, and incorporated into the methods summary sheets at the end of each chapter. References are also provided at the end of each chapter. This review thus has a format which is considered to be suited to the subject matter, and therefore somewhat different from the more routine descriptions of methods in some of the companion reviews.

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2. NATURAL WATER QUALITY

2.1 Introduction and scope

The dominant properties of groundwater, especially its mineral content, chemical composition and taste, are determined by water-rock interactions during its circulation. The intergranular nature of unconsolidated sedimentary aquifers (UNSAs) has two important effects - it ensures intimate geochemical reaction between the circulating groundwater and the sediment, and greatly retards the movement of pollutants towards wells and natural discharge points. Therefore, in considering the development of the vast water resources locked up in tropical alluvial systems, it is the natural hydrogeochemical processes together with adequate recharge that will determine the main characteristics of the water for drinking and other uses. In the semi-arid tropics, UNSAs - as river valley sediments or aeolian deposits - often form a lifeline for small communities; here, the groundwater recharge may be limited and rather variable and the quality may be affected by the build up of salinity.

This chapter reviews the impact of natural chemical processes on groundwater quality with particular emphasis on unconsolidated sedimentary aquifers. The principal water quality issues are illustrated with examples and the methods which may be employed to study water quality are summarised.

2.2 Background

Geochemical investigation, using in particular inorganic, stable and radioisotope techniques, can assist in the investigation of unconsolidated aquifers in a number of ways. It is also necessary to characterise the solid phase in terms of its mineralogy. The UNSAs considered in this review are restricted to Quaternary sediments. The component detrital minerals will therefore be adjusting to the new sedimentary environment and undergoing a degree of diagenesis, allowing the formation of additional secondary minerals; alluvial sediments may also contain a significant amount of organic matter, enhancing their reactivity. As water moves along flow paths down hydraulic gradient a series of geochemical processes will take place (dissolution, ion exchange, oxidation-reduction and mixing for example). The application of geochemical methods is of considerable importance in helping to evaluate certain key physical parameters such as the amount of recharge, residence time of the groundwater and the degree of hydraulic continuity.

In this review, the emphasis is placed on methods that can be used for establishing the natural baseline geochemical conditions in an aquifer. This is essential in advance of any study of groundwater pollution since anomalous concentrations of inorganic constituents may build up as a result of normal chemical reactions. Such anomalies may be significant in relation to human health and may also serve to give important diagnostics on the condition of the aquifer. The main processes are illustrated with case histories which give an insight into the natural water quality in specific environments. Case studies have been reviewed, and it is clear that the most useful examples are mainly from developed countries. These are summarised and illustrated in the boxes. The range of unconsolidated sediments considered here is: (1) Alluvial

Plains and Deltas; (2) Coastal Alluvium; (3) Intermontane Basins; (4) River Valleys; (5) Wind Blown Sands.

2.3 Atmospheric inputs

The composition of rain and dry deposition has an important influence on groundwater quality in a number of ways. For a number of elements the atmosphere provides a major source of solutes to groundwater. Of particular importance are Cl and the other halogens Br, I and F but also the contribution of other marine derived ions. Mg and Na, for example, may also be significant. These sources are obviously important in coastal regions and their absence inland may have a subsequent impact on environmental problems such as iodine deficiency. In semi-arid and arid regions additional loading of aerosols may locally enhance the geochemical inputs from the atmosphere. The effect of the atmospheric contribution will be most noticeable (a) near coasts in the shallow aquifer system, and (b) in semi-arid or arid terrains where evapo(transpi)ration will concentrate the solutes. Knowledge of the atmospheric inputs is of additional value since the concentration of chloride and other non-reactive solutes may be used to estimate rates of recharge to the groundwater system (Box 2.1). Stable isotope measurements of rainfall ($\delta^{18}O$ and $\delta^{2}H$) also have an important application in following the hydrological evolution (Fontes and Edmunds 1989).

2.4 The unsaturated zone including soil

The unsaturated zone and especially the soil/bedrock interface is the most reactive environment in most aquifer systems and it is in this zone that most of the chemical properties of groundwaters are determined. Solutes are concentrated by evapotranspiration, acidic inputs (including normal rainfall at ca. pH 5) are neutralised and the concentrations of incoming solutes are modified by exchange processes on clays and other minerals. Microbiological processes are also likely to be of importance in the soil and near surface environment although conditions favourable for microbial growth (e.g. organic carbon source, nutrients - N,P,S) tend to decrease with depth. The unsaturated zone also provides a buffer zone of protection for the aquifer, since solutes may take several (or many) years to reach the water table, and may also be attenuated by adsorption processes. However, where the water table is shallow (e.g. less than 5 m) the unsaturated zone may act as a discharge route with the effect of concentrating solutes.

There is likely to be a fundamental difference between the unsaturated zone in different types of unconsolidated sediments and in different climatic zones. Two extreme types may be considered (1) semi-arid zone profiles and (2) alluvial sediments in tropical regions. Chloride profiles through the unsaturated zone are now used widely in semi-arid regions and especially in unconsolidated sediments to estimate the amount of groundwater recharge - in Australia (Allison and Hughes 1978), the Middle East (Edmunds and Walton 1980); the African Sahel (Edmunds et al 1988, 1994), and in India (Sukhija et al 1988). The chloride profile provides a reference timescale against which recharge history and the movement of other solutes can be assessed (Edmunds et al 1992a,1992b). Tritium has been used in some areas to study recharge but suffers

Box 2.1 River recharge and geochemical processes in Nile valley, Sudan

The interrelationships between modern recharge and palaeowaters are well illustrated by the situation in Central Sudan. Using a geochemical approach (chemical and isotopic) it has been possible to demonstrate exactly where present day recharge has been effective. Alluvial sediments overlie and are in hydraulic continuity with the Nubian Sandstone aquifer north east of Khartoum. There is clear evidence in this region that the river Nile acts as a recharge source and its chemically and isotopically distinct waters may be distinguished, superimposed on the palaeowaters.

The adjacent wadi system which may flow 10-20 times a year during heavy rains, acts as a lifeline in the region. Recharge enters the wadi bed and spreads laterally - as demonstrated by the presence of tritium up to 1 km from the bed. This provides the essential and vulnerable small water resource for both settled and nomadic people. Direct recharge through the unsaturated zone does not occur as shown by the chloride balance.

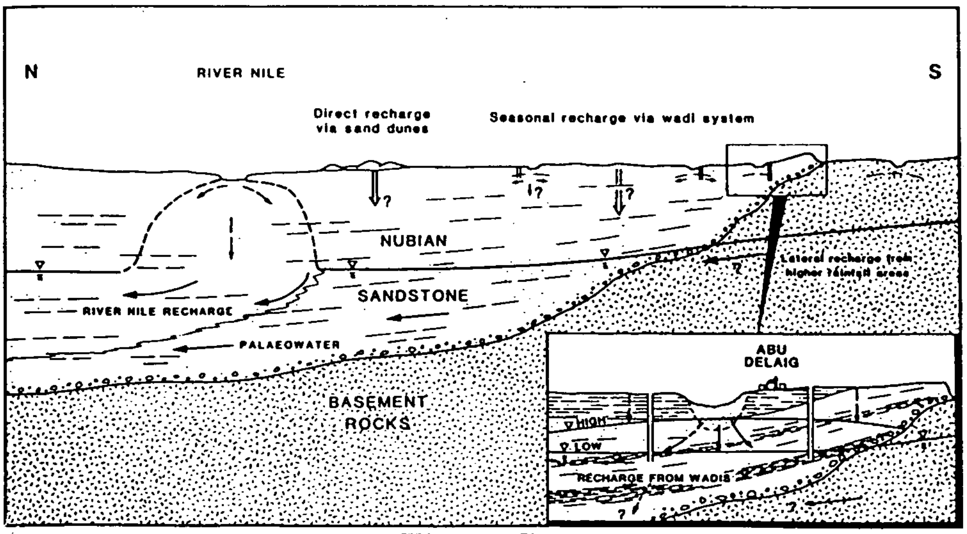
Recharge of river water can be traced chemically since its hydrochemistry reflects reactions with bedrock foreign to the area. The Nile has characteristically higher Mg/Ca and K/Na ratios than the local groundwaters and also has very low B. Whereas the regional groundwaters contain NO_3 and are always aerobic, the Nile recharge is anaerobic probably on account of its organic carbon content and increased microbiological activity. In consequence these groundwaters are NO_3 free yet do not contain troublesome concentrations of Fe.

Shallow groundwater in the wadi tributaries of the Nile is very variable in ionic composition but there is a distinct enrichment in Mg/Ca ratio and sulphate as compared with rain and wadi flood water. Most of the shallow groundwaters are saturated with respect to calcite and this demonstrates that carbonates and other minerals present in the soils and unsaturated zone from previous desiccation periods are reacting readily during the recharge process. The shallow alluvial groundwaters are also enriched in B (200-400 mg/l).

Groundwater in the Nile valley alluvium exhibits a range of isotopic compositions but mainly corresponds to recharge from the White Nile baseflow rather than the Blue Nile flood. The extreme enrichment in interstitial waters from the unsaturated zone confirms the strong evapotranspiration. Shallow groundwaters from dug wells are very slightly enriched in relation to the Khartoum average rain, indicating some evaporation has taken place. The palaeowaters are clearly distinguished by their light isotopic compositions.

Reference

Edmunds W M et al 1992. Sources of recharge at Abu Delaig, Sudan. Journal of Hydrology, 131, 1-24.



Sources of groundwater based on geochemical/isotopic data

several disadvantages. Unlike chloride, tritium is not conserved during the recharge and is lost by evaporation, making it difficult to carry out a mass balance. After a peak during the period when atmospheric nuclear testing was taking place, tritium is now close to background levels (15-20 TU) in the atmosphere and has a very low signal to noise ratio in modern groundwater. Low level sampling and analysis is also an expensive procedure. However tritium remains an unambiguous tracer of recent water and in certain environments may still be used to confirm recharge.

The hydrochemical processes operating in the arid zone are illustrated for the Quaternary Sand aquifer in Northern Senegal by Edmunds et al (1992). This region typifies the environment of much of the Sahelian zone of Africa and other areas where dune sands remain either as a result of Pleistocene or more recent aridity. The sandy aquifer is carbonate-free and the processes which have taken place mainly reflect the attenuation of acidity so that during unsaturated flow, reactions with the matrix largely neutralise the water. The chloride profile is calibrated against known recharge and residence time of the water is seen to be about 70 years. Another important feature illustrated here is the high nitrate concentrations, of the order of 30 mg/l NO_3-N. These high values are not the result of pollution but are entirely the result of natural processes, probably nitrogen fixation by leguminous plants and trees.

Whilst chloride and nitrate can be considered to be conservative in the profile, other components, such as exchangeable cations, provide a basis for estimating the reaction of silicate minerals in chemical weathering of the sands since deposition.

2.5 The saturated zone

It can be seen that the residence time of groundwater in the unsaturated zone of unconsolidated aquifers in semi-arid zones may be measured in decades or even centuries, whereas in the humid tropics, the water may reside in the unsaturated zone for only a few months. Within the saturated zone, groundwater will be subject to a series of processes as flow proceeds downgradient, notably further congruent and incongruent mineral dissolution, and redox reactions.

2.5.1 *Mineral solubility*

The main controls on the chemical evolution of groundwater are the mineral assemblages present, the relative solubilities of these minerals and the rates at which they may undergo various reactions. A reaction sequence may be considered based on relative mineral solubilities (Table 2.1).

Table 2.1 Relative solubilities of commonly occurring minerals

(Gypsum)	
(Calcite)	
Olivine	
Ca-plagioclase	
Augite	
Hornblende	Decreasing
Na-plagioclase	solubility
Biotite	
K-plagioclase	
Ca-Na-montmorillonite	
Quartz	
K-mica	▼
Gibbsite	
Kaolinite	
(Fe-oxides)	

A major distinction in hydrogeochemical evolution may be made based on whether the aquifer contains carbonate minerals or not. If present, carbonate minerals will neutralise the rainfall acidity and effectively buffer the water chemistry via the CO_2-H_2O system:

$$CaCO_3(s) + CO_2 + H_2O = Ca^{2+} + 2HCO_3^-$$

The progress in the acid neutralisation may best be followed by the increase in alkalinity (rather than by an increase in pH). Saturation with respect to calcite or another carbonate will eventually be reached. Calculation of the mineral saturation index ($SI_{mineral}$) provides another index of the degree of neutralisation of the water acidity and takes into account the whole water chemistry including the chemical speciation of the water; such speciation calculations are best made using one of a number of computer programmes, e.g. WATEQ, MINTEQA2.

An important concept to consider is the water-rock ratio and its influence on the hydrochemistry. Compared with the small mass of solute which is to be found in water the amount of rock available for reaction is vast. Therefore, small modal percentages of very soluble mineral phases such as calcite will exert a disproportionate influence on the water chemistry. In many types of Quaternary formations considered in this review, carbonate minerals may be sparse or absent. Under semi-arid conditions, soil carbonates, calcretes and freshwater (lacustrine) carbonates may be present in minor amounts.

Silicate mineral dissolution will be the primary process for the neutralisation of rainfall acidity in the absence of carbonates. The general form of such hydrolysis reactions may be expressed as:

Silicate mineral + Acid (H^+) + Water = Secondary mineral + Cation(s) + SiO_2

The source of this acidity will nearly always be dissolved soil CO_2 which hydrolyses as carbonic acid and dissociates to produce a proton and bicarbonate alkalinity. The soil microbiological activity therefore will be critical in determining the extent of weathering and the build up of solutes in the groundwater. Details of silicate mineral reactions in a sand and gravel aquifer are given in Box 2.2.

2.5.2 *Redox processes*

Rainwater infiltrating through the soil contains around 12 mg/l of dissolved oxygen (DO), which reflects that present in the atmosphere. As percolation and flow proceed the oxygen will progressively react with organic matter and with inorganic reduced species (e.g. Fe^{2+}) usually mediated by biological processes. In many aquifers the concentrations of DO are only slightly reduced on passage through the soil zone and it is common for the phreatic aquifer to be aerobic and to contain several mg/l DO (Edmunds et al, 1984). The consumption of oxygen will vary greatly depending mainly on the amount of organic matter in the sediment; in general oxygen will tend to be removed most rapidly in carbonate rather than non-carbonate aquifers. In many semi-arid regions oxygen may persist for timescales of 1000 or more years.

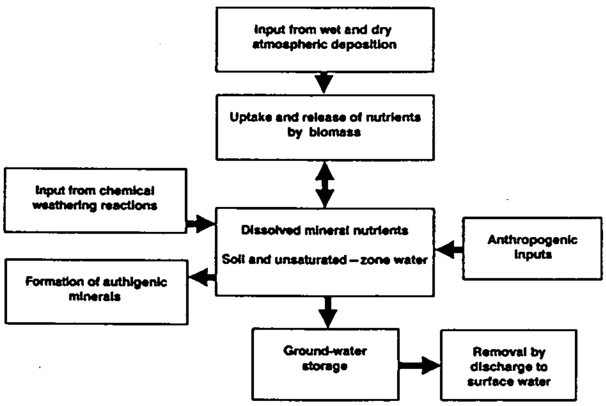
With the total reaction of oxygen in an aquifer which is closed to the atmosphere a marked change to reducing conditions takes place. This 'oxidation-reduction boundary', or 'redox barrier', is an important feature of most aquifers and can be easily defined not only by the complete reaction of oxygen but by a change of some 300 mV in the redox potential or Eh. A sequence of redox reactions can theoretically take place (Figure 2.1) such that following the consumption of oxygen, nitrate and nitrite will be reduced and so on. A series of such reactions are described by Jackson and Patterson (1982) in fluvial sediments in Ontario, Canada. The redox potential drops off rapidly in the shallow aquifer facilitated by large concentrations of organic carbon in the infiltrating water. In the lower aquifer there is a rise in pH associated with H^+ consumption during alteration of minerals such as biotite and carbonate dissolution. Here there are high concentrations of iron and manganese which are then lost down gradient, during sulphate reduction. The consequences to water quality of some of these reactions are described in Box 2.3.

A complementary sequence of reactions will also take place as oxidation occurs. Thus the aquifer has a natural capacity to attenuate pollutants and modify the overall composition, the most important of these being denitrification, which proceeds rapidly once oxygen has reacted. The redox reactions may be microbially mediated or may proceed chemically under favourable conditions. The presence of pyrite in anaerobic sediments (Box 2.4) may allow the reduction of oxygen and the denitrification to proceed even in the unsaturated zone.

Box 2.2 **Reactions of silicates in sands and gravels**

The principal source of water in NW Florida is an aquifer composed predominantly of quartz sand with smaller amounts of andesine, chlorite, calcite, kaolinite, and illite; muscovite and calcite are also present. Recharge to the aquifer is predominantly from precipitation (1405 mm/a). Water is either confined or unconfined depending on the absence or presence of local confining clay layers.

Water from the aquifer is characterised by low dissolved solids (median = 72 mg/l), and although there are no distinct hydrochemical facies, Mg and SO_4 tend to be lower than other major ions.



Conceptual model showing inputs to and outputs from a dissolved mineral reservoir

In order to calculate a detailed geochemical budget for the system, accurate chemical and water flux data are necessary. Since these are difficult to obtain, the composition of the two end members of the reaction sequence (precipitation and groundwater) are compared using mass-balance calculations to evaluate the relative importance of weathering (mineral dissolution and precipitation) reactions in controlling the composition of the water. Muscovite, andesine, calcite, and chlorite are the major reactive minerals - quartz is considered nonreactive. It was assumed that the only water entering the system is precipitation and the biomass is in dynamic equilibrium.

The data for precipitation were corrected for evapotranspiration and subtracted from the median composition of the groundwater. Any excess Cl is assumed to result from saline water sources because there are no chloride-bearing minerals present in the aquifer. These may include aerosols (dry deposition), connate seawater or mixing with saline water. Subtracting the Cl, along with the relative proportions of other major ions in seawater leaves only those released from or consumed in weathering reactions. The geochemical mass-balance reactions can be written as a series of equations, solved using the computer programme BALANCE. The only mineral sources for Na, Mg and K are andesine, chlorite and muscovite. Ca and Si have multiple mineral sources: calcite, andesine, chlorite and muscovite. The amount of these minerals that react is however constrained by the water chemistry Δm_c.

The results show that wet deposition and sea water each account for 13% of the concentration of dissolved solids in groundwater. Dissolution of feldspar, chlorite and muscovite account for 94% of neutralisation of the total hydrogen ion input from wet deposition and from carbonic-acid weathering reactions. The excellent mass balance composition implies that incongruent dissolution of silicates and the resulting formation of kaolinite are the most important processes and others such as cycling by biomass or cation exchange appear to be of minor importance in influencing the chemistry of the water in the aquifer. Cation exchange, however may affect the chemistry in some local areas where Ca was replaced by Na from the movement of water through clay lenses. Silica comes from the breakdown of silicates and not from the dissolution of quartz - the more easily weathered minerals in a rock contribute disproportionately to the groundwater chemistry. In conclusion it was found that the background water chemistry is strongly influenced by mineral solution-precipitation reactions.

Reference

Katz B G and Choquette A F 1991. Aqueous geochemistry of the sand-and-gravel aquifer, northwest Florida. Groundwater, 29, 1, 47-55.

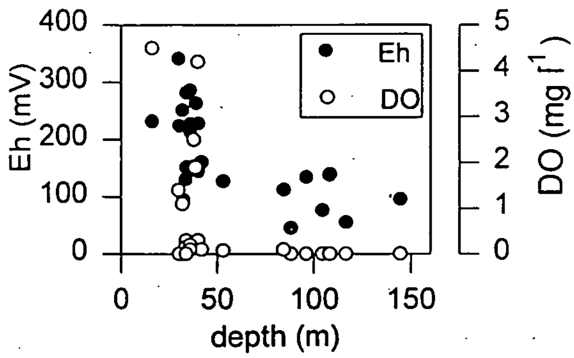
Box 2.3 **Groundwater quality problems in quaternary alluvium, coastal Orissa, India**

The coastal belt of tropical Orissa comprises a thick sequence (up to 300 m) of Tertiary and Quaternary alluvial and shallow marine sediments comprising heterogeneous clay, silt, sand, gravel, pebble and calcareous beds with prevalent lateritic deposits. Groundwater flow is principally towards the coast, as indicated by the flow of the major rivers. The alluvial aquifer in Delang is predominantly confined, probably as a result of impermeable clay horizons and some wells are artesian. Groundwater at depth is predominantly reducing and reduced forms of sulphur have been observed at depths of c. 30 m.

Monitoring of pH, total Fe and Cl showed that a significant proportion of the total Fe was in suspended solids. During pumping, some groundwaters showed decreasing ferrous iron concentrations but constant pH and Cl, suggesting that much of the high iron during initial pumping is derived from metalliferous components in the well which are purged after prolonged pumping. However, high iron is found in wells after prolonged pumping and also in artesian wells and this is likely to represent true aquifer concentrations. The correlation between Fe and Mn also suggests that both are derived from the aquifer matrix.

Water quality in the Delang Block is controlled by water-rock interaction, redox processes (probably including microbiological influences) and saline intrusion. The depth of boreholes has a great effect on redox conditions in the aquifer and is further highlighted by strong negative correlations of both Eh and DO with depth. The steep negative slope of DO with depth indicates that groundwater becomes readily depleted in oxygen and at depths of 50 m or more very low or no oxygen should be expected. Sulphate and nitrate reduction appear to be taking place which may be microbially mediated.

Correlations of major ions in Delang Block groundwaters strongly indicate mixing between fresh groundwater and modern marine water for example major ions plotted against Cl. With the exception of Ca, the more saline Delang groundwaters have concentrations corresponding with a seawater dilution curve indicating mixing of marine and fresh water. The more dilute groundwaters have cation concentrations offset to higher values, probably as a response to ion-exchange processes during mixing in the aquifer.



Correlations of Eh and DO with depth in Delang

The stable isotopes δ^2H and $\delta^{18}O$ show a positive correlation, and the trend points directly towards Bay of Bengal seawater suggesting that mixing with marine water has been important. Mixing is also indicated by correlations of stable isotopes and depth, SEC and log Cl concentrations.

Reference

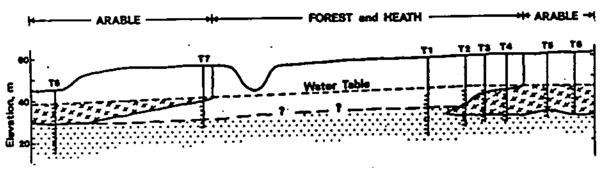
Smedley P L 1991. Groundwater quality problems in coastal Orissa, India: reconnaissance hydrochemical survey and proposal for future research. British Geological Survey Technical Report WD/91/41R.

Box 2.4 Nitrate reduction and pyrite oxidation in a sandy aquifer: the Rabis aquifer, Denmark

The Rabis aquifer is located in the Western part of Denmark and consists of Pleisocene deposits with a thickness of at least 35 m. These were laid down by braided rivers and are underlain by Miocene deposits. Some of the underlying material has been reworked into the Pleistocene deposits. The sediments consist mainly of quartz sand but at depth they contain small amounts of pyrite. Weathering has removed $CaCO_3$ from at least the top 30 m of the deposits.

The aquifer is unconfined with the water table at about 15 m below the surface. Most of the flow is in the horizontal direction with only a small vertical component.

A series of multi-level samplers was used to determine the 3-D variation of water quality within the aquifer. The main feature was the presence of a sharp redox front at about 15 m below ground level. Above this front, there were O_2 and NO_3-rich waters while below this zone were NO_3 free waters frequently with high concentrations of dissolved Fe. In the saturated zone plumes of nitrate rich water could be traced moving downgradient from areas of fertilised arable land.

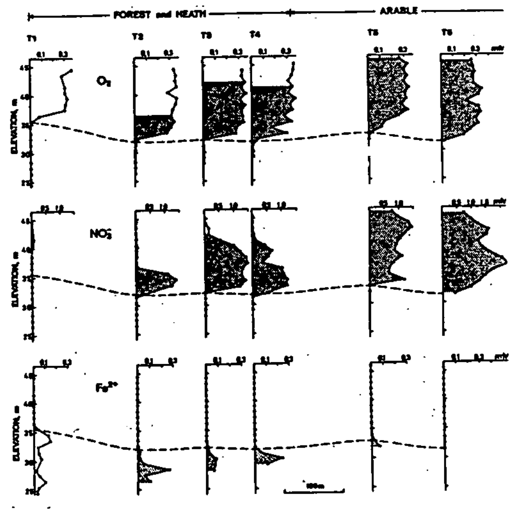


Nitrate pollution plumes entering the aquifer from agricultural fields. T1-8 refer to locations of multilevel samplers

The presence of pyrite in the sediment was the principal control on the position of the redox front. As dissolved O_2 and NO_3 penetrated the aquifer, they oxidised this pyrite, leading to the gradual downward movement of the front with time. Detailed modelling using the PHREEQM geochemical modelling package indicated that dissolved O_2 in the groundwater would move the redox front downwards at the rate of 0.3 cm/year but the presence of high concentrations of NO_3 (typically 17 mg N/l) would increase this rate by a factor of 5. The limiting factor in predicting this was the variability of the pyrite content of the sediment.

Pyrite seemed to be a more important reducing agent than organic matter, even though brown coal fragments were abundant. Elsewhere, particularly in the nitrate contaminated alluvial aquifers of Germany and Canada, where pyrite is absent, it is the supply of readily degradable organic carbon which limits the rate of denitrification.

The major effects which the nitrate contaminated waters have had on the deeper, reduced part of the Rabis aquifer has been to increase the SO_4 and Fe concentrations in groundwater. Therefore the benefits of in-situ denitrification are to some extent offset by the accumulation of other undesirable solutes in the groundwater.



Groundwater chemistry profiles indicating the depth where O_2 disappears

Reference

Postma D K, Boesen C, Kristiansen H and Larsen F 1991. Nitrate reduction in an unconfined sandy aquifer: water chemistry, reduction processes and geochemical modelling. Water Resources Research, 27, 2027-2045.

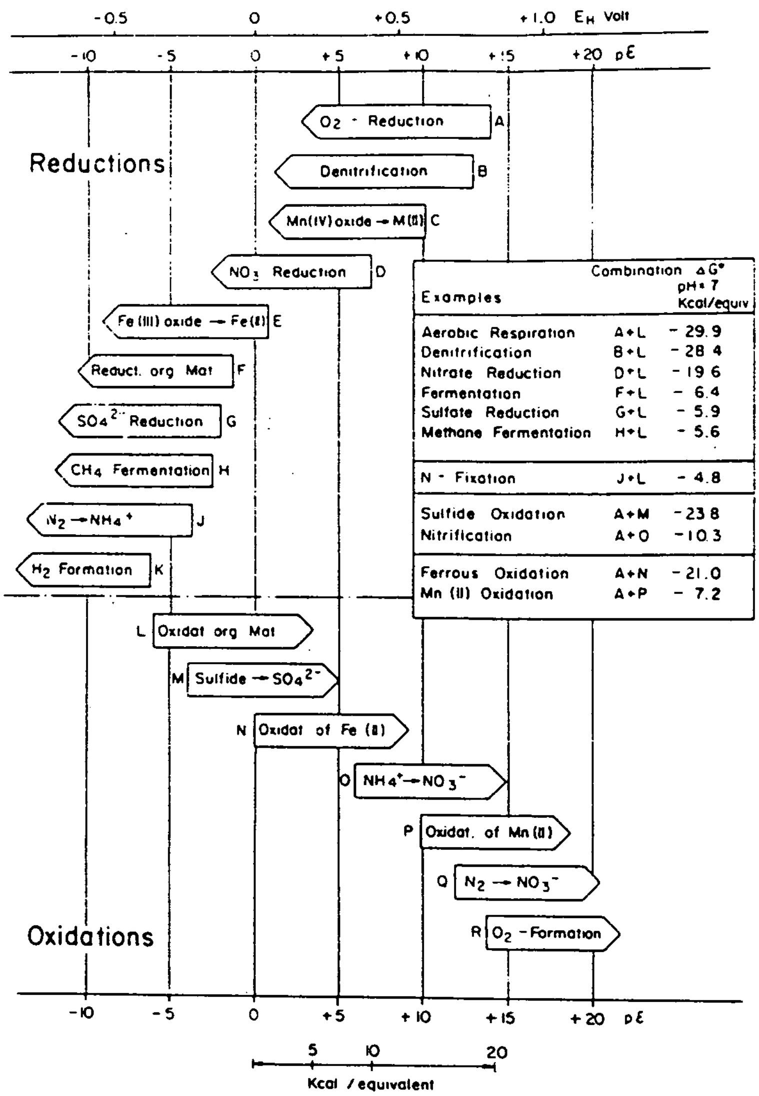


Figure 2.1 **Theoretical sequence of redox processes in reducing and oxidising environments (after Stumm and Morgan, 1981)**

2.6 Natural baseline conditions

The reactions taking place between water and unconsolidated sediment give rise to a characteristic natural baseline chemistry (Box 2.5). There are however relatively few readily accessible groundwater resources which can be considered uninfluenced by the activities of man - as evidenced for example by the presence of thermonuclear tritium, excess nitrate or synthetic organic compounds in deep groundwaters.

In order to recognise pollution it is first necessary to determine the natural baseline or background conditions. In the case of synthetic organics this is easily defined as zero, or near to zero in the case of tritium. For inorganic constituents this is less straightforward since geochemical inputs from the atmosphere, soil, vegetation and bedrock may give rise to naturally variable quantities for different elements. In fact it is impossible to be sure that pollution is occurring unless the baseline is first defined.

Box 2.5 **Regional hydrogeochemistry of the Ganges Delta System**

Bangladesh covers an area of approximately 144,000 km^2 encompassing the combined Ganges, Brahmaputra and Meghna delta system. The low-lying central parts of the system are regularly inundated by flood waters during the annual monsoon period. This pilot project was undertaken to assess the hydrochemical character of the main aquifer units of central and north-eastern Bangladesh and possible toxicity of groundwater to fish and humans.

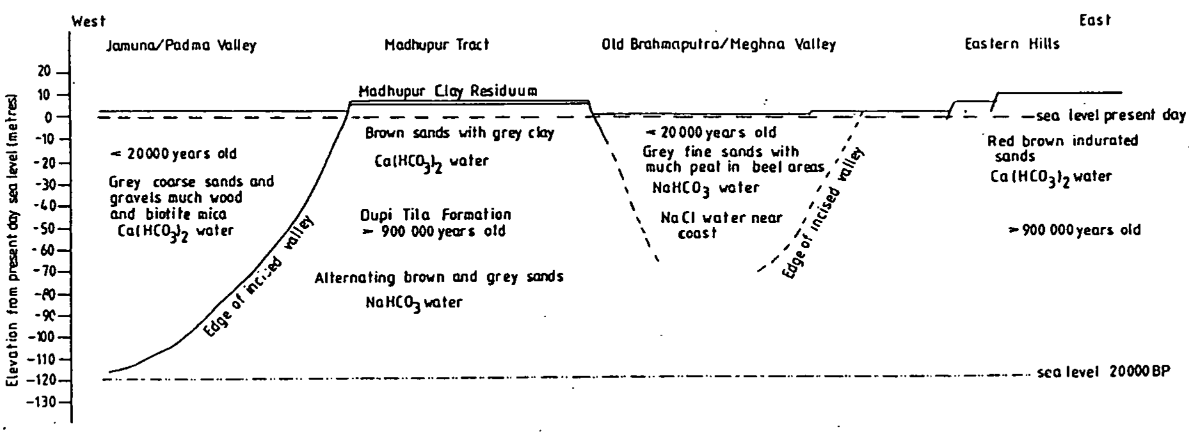
Two main groups of aquifers are present in the study area: an older group of aquifers of Pleistocene age comprising predominantly red-brown alluvial fine to coarse grained sands and gravels which are highly weathered with mafic minerals oxidised to form red brown ferric hydroxide cement and secondary clays that reduce intergranular pore space; and a younger group of Late Quaternary age along the valleys of mainly non-indurated grey alluvial sands and gravels with large quantities of unweathered mafic minerals, as well as wood fragments, and have a high intrinsic porosity and permeability.

The alluvial sediments that form the Pleistocene to Late Quaternary aquifers are composed of sediments derived from plutonic and metamorphic rocks of the Himalayas. The Pleistocene rocks underwent several periods of weathering when sea levels were depressed, and during these periods organic carbon and ferromagnesian rich minerals underwent decomposition and oxidation. Due to fixing of ferric hydroxides and removal of organics, these aquifers have become relatively inert in character with the groundwaters becoming acidic due to the lack of buffering. As a result, the shallow recharged waters are of $Ca(HCO_3)_2$ type whereas the deeper older waters are of $NaHCO_3$ type.

In contrast the Quaternary sediments, of $Ca(HCO_3)_2$ type, have been maintained in a water-logged state; there has been no oxidation of minerals because organic carbon in the form of wood fragments and peat deposits degrades slowly under anaerobic conditions, resulting in reducing conditions. Under these conditions iron is leached from biotite, passing into solution as the soluble ferrous state. Therefore, the nature of the alluvial sediment and its weathering experience control the chemical character of the groundwaters. However, infrequent inundation of low lying coastal areas with brackish waters results in mixing or replacement of the expected water type to give NaCl type waters.

Reference

Davies J and Exley C 1992. Short term BGS pilot project to asses the hydrochemical character of the main aquifer units of Central and North-eastern Bangladesh and possible toxicity to fish and humans. British Geological Survey Technical Report WD/92/43R.



Cross section of the main Bangladesh aquifers

Nearly all natural waters contain most elements in the periodic table but often at extremely low and unquantifiable concentrations. In Figure 2.2 the typical abundance of the elements in natural potable water at pH 7 is summarised. Nine major species (HCO_3, Na, Ca, SO_4, Cl, NO_3, Mg, K and Si) invariably make up 99% or more of the total solute content by weight with the minor and trace elements constituting the remaining 1%. This abundance table may be significantly altered as a result of geochemical or anthropogenic factors. For example, the suite of trace elements found naturally in a carbonate groundwater may differ from that in sandstone waters because of a different geochemical abundance in the parent rock. Increases in concentration of around one order of magnitude may occur for a number of trace metals with each unit decrease in pH.

Those elements are indicated which are currently considered to be essential to human health or metabolism and those which are considered to be toxic according to the present EEC guidelines for drinking waters (CEC 1980) are also shown in Figure 2.2. For a number of decades the significance of minor and trace elements in human epidemiology has been recognised. In some cases, such as the incidence of high fluoride with fluorosis the correlations are well established; fluoride deficiency is now also well established as a contributory factor to dental decay. This example provides a good example of the dual role of trace elements in relation to health. A further review of groundwater geochemistry and health is given by Edmunds and Smedley (1996).

2.7 Groundwater residence time

The extent of hydrogeochemical reactions will depend upon the residence time of the water in the aquifer and it is useful therefore to consider the evolution along flow lines where distance is roughly proportional to time. There are several methods for the measurement of absolute groundwater age (Table 2.2) using radioisotopes. Tritium has a useful role in determining the extent of penetration of post 1960's recharge into the system (Box 2.1) although the environmental background is now rather low and sensitive techniques are needed for measurement. Carbon-14 provides the only relatively easy method of interpreting ages of groundwater in the range 1000 - 50000 yr range. It is necessary to have full supporting information on the carbonate geochemistry of the water as well as the sediment in order to be able to correct/interpret the carbon 14 results. It is also possible to obtain valuable supporting information on residence time from a study of stable isotopes, especially to identify likely palaeowaters or waters of different recharge sources (with different relative ages).

2.8 Techniques for assessment of natural quality

The investigation of groundwater chemistry requires the application of a range of techniques. It is often the case that one measurement or a single approach to a problem will be indeterminate. The range of methods that may be used is summarised in Table 2.2. It should be noted that study of oxygen and hydrogen stable isotopes provide a direct way of following the physical processes in hydrogeological systems (evapotranspiration, mixing, recharge for example).

TRACE ELEMENTS → | ← MAJOR ELEMENTS

INCREASING CONCENTRATION →

1 ng l^{-1} — 1 PART IN TEN THOUSAND MILLION — 1 μg l — 1 mg l^{-1}

<0.00001	0.00001–0.0001	0.0001–0.001	0.001–0.01	0.01–0.1	0.1–1.0	1.0–10	10–100	>100
Nb	Cs	Rb	Li	P	Sr	[1] Mg	[1] Na	HCO_3
Ru	Zr	La	[2] Ba	[2] B	[1] F	[1] K	[2] Ca	
Rh	Mo	[2] V	[2] Cu	Br		[2] Si	[1] SO_4	
Pd	[1] Ag	[1] Se	[1] Mn	[1] Fe			[2] Cl	
In	[2] Be	[1] As	U	[2] Zn			[1] NO_3	
Hf	Th	[1] Cd	I					
Ta	Ce and REE	[2] Co						
Re	[1] Hg	[1] Ni						
Os	Sc	[1] Cr						
Ir	[1] Sb	[1] Pb						
Pt	Sn	[1] Al						
Po	Te	Y						
Au	Tl							
	Bi							
	W							
	Ga							
	Ge							

ESSENTIAL ELEMENTS

Cu — Elements considered essential for human or animal health.

Sr — Elements probably essential for health.

Pb — Non-essential elements.

TOXIC ELEMENTS

[1] Cd — Elements considered to be toxic or undesirable in excessive amounts and for which maximum admissible concentrations (MAC) have been set by the CEC.

[2] V — Other elements considered undesirable in excess but for which no statutory limit has been set by the CEC.

Figure 2.2 Concentrations of the elements in dilute oxygenated groundwater at pH7, with indication of essential and toxic elements in terms of human health

Table 2.2 Range of parameters, with examples, that can be measured in the investigation of water quality of unconsolidated aquifers

Parameter	Application in groundwater quality studies
SOLVENT (water molecule)	
Stable isotopes $\delta^{18}O$, $\delta^{2}H$	Tracing of waters with different origins, resulting from deposition at different latitudes, altitudes or having undergone evaporation of water-rock interaction. Distinguishing palaeowaters.
Radioisotopes ^{3}H (T)	Special tracing of pulses of water or recent origin, using tritium HT_2O originating from thermonuclear testing since 1950s. Half life 12.3 Yr.
SOLUTES	
Major ions Na, Cl, SO_4, Ca, Mg, K, HCO_3, NO_3	Of prime significant in understanding natural geochemical processes, characteristics of aquifer lithology. Na/Cl used to determine extent of water-rock interaction together with Mg/Ca. Alkalinity (HCO_3) indicates extent of acid neutralisation. Na/K, NO_3 useful indicators of pollution.
Minor elements F, Br, Sr, B, Li, Ba	Support major element interpretations. Often very specific information on evolution of groundwater and residence times given by mobile minor element. However, F, Ba may be limited by mineral solubilities.
Trace elements Fe, Mn, Al, Zn, As	Trace metals not usually mobile except under acidic or reducing conditions. Some elements (Fe, As for example used to assess redox conditions.
Stable isotopes $\delta^{13}C$, $\delta^{15}N$, $\delta^{34}S$	Useful in tracing origins of dissolved bicarbonate, nitrate or sulphate.
Radioisotopes ^{14}C, $^{238}U/^{234}U$ series	Potential value in determining age. Care needed in interpretation, especially use of other geochemical evidence.
Dissolved gases N_2/Ar, inert gas ratios	N_2/Ar ratios used to study nitrogen transformations in aquifer. Inert gas ratios useful to establish temperature of recharge. Palaeohyrological studies.
PHYSICAL/MEASUREMENTS	
Temperature	Important tracer of surface groundwater interaction flow. Indicator of depth of origin (geothermal gradients).
Specific electrical conductance (SEC_{25})	Important tool for establishing water quality trends with time and in space. Applied value increased if related to major element behaviour.
pH	Measure of acidity in waters. A difficult measurement however and reproducibility a problem. Alkalinity measurement a better measurement of acid status.
Eh	Valuable qualitative check of redox environment. Very sensitive indicator of disappearance of dissolved O_2.
PARTICLES	
Bacteria	Measurements in relation to potability and contamination.
Suspended matter	Examination of filter residues to give clues on redox environment, corrosion products, and dislodged clays and other minerals. Microscopic examination desirable.

Chloride behaviour also closely follows that of water, not being involved in reactions, and is also of use in studies of the physical hydrological cycle. Major ions and especially major ion ratios are probably the most useful way to determine the overall characteristics of the system since they provide the evidence of the main reactions that have taken place along the flow path. Trace elements and especially the minor elements indicated may often give important supporting evidence of the evolution and especially the residence time.

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METHOD SUMMARY SHEET (WQM 1)

TITLE: Investigating natural water quality

Scope and use of method

Groundwater movement within most UNSAs is dominantly intergranular. This provides opportunity for intimate contact between the circulating groundwater and the sediments, and gives a large contact surface area for hydrochemical reactions. The relatively slow movement of groundwater through most types of UNSA also enhances the opportunity for reactions to take place. Natural hydrochemical processes play a vital role on determining the characteristics of groundwater in UNSAs.

Determining natural water quality is essential for establishing baseline geochemical conditions in aquifers. Anomalous concentrations of inorganic constituents may build up as a result of natural hydrochemical reactions in the aquifer. These can present water quality problems for various uses, especially potable supply, and must be properly characterised and distinguished from quality changes resulting from the impact of human activities. Investigation of natural water quality can also provide important information about the physical conditions of aquifers in terms of sources and amounts of recharge, groundwater residence times and degrees of hydraulic continuity within and between aquifer sequences.

Method

The investigation of natural water quality requires a range of techniques, as it is common that one type of measurement or a single approach provides indeterminate results. The range of inorganic chemical, stable isotope and physical measurements that can be employed are listed, with their applications, in Table 2.2 of chapter 2 of the review. Major ions and major ion ratios provide the most useful and economical way of determining the overall characteristics of the natural water quality, since they provide evidence of the main reactions which have taken place along the groundwater flow path. Trace elements provide supporting evidence concerning hydrochemical evolution, and especially of residence time.

Some parameters are relatively unstable (pH, HCO_3, and should be measured in the field. Methods of sampling and analysis are given in the references provided. The use of isotopic methods is described in more detail in review no 8.

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3. IMPACT OF URBANISATION ON GROUNDWATER QUALITY

3.1 Introduction and scope

Urban populations in developing countries are growing rapidly, and are largely concentrated in the marginal slum housing districts where access to sanitation and piped water supply are often limited. Many of these cities are dependent on groundwater for a significant proportion of their water supply. Even in areas where piped water is mainly derived from surface water, groundwater may provide the balance to areas without mains coverage. Further, many industries and commercial premises obtain their supply from private boreholes because this is often cheaper and more reliable.

Most of the cities which are dependent on groundwater abstract supplies from large unconsolidated sedimentary aquifers (UNSAs). Historically, they were sited and have grown and flourished because of the availability of abundant shallow groundwater or surface water of good quality for domestic use and irrigated farming. The very rapid urban growth of the last few decades has produced increasing demands for potable water. Surface resources are either fully utilised or now of poor quality, and groundwater resources are becoming more important.

This chapter reviews the present state of knowledge concerning the impact of urbanisation on groundwater movement and quality with particular emphasis on unconsolidated sedimentary aquifers. The principal groundwater quality issues associated with urbanisation are illustrated by several case histories. The methods which may be employed to investigate the impact of cities on groundwater quality are summarised.

3.2 Background - Urban growth

The two factors which have dominated world demographic trends in the 20th century are an accelerating rate of population growth and continued emigration from rural areas. The result has been a constant rise in the proportion of the world's population living in urban areas, from less than 15% in 1920 to over 40% in 1990. It is estimated that by the turn of the century nearly half of the world's population will live in towns or cities (Figure 3.1). Much of this increase is concentrated in the developing world, where it is projected that 85% of the growth in the world's urban population between 1980 and 2000 will take place. The result is that by the year 2000, about twice as many people will be living in cities in developing countries (1900 million) as in the developed nations (950 million).

Such urbanisation in developing countries has been rapid and is generally unplanned. This has meant that the provision of mains water and, more significantly, waterborne mains sewerage has lagged markedly behind population growth. A large part of the urban population increase is concentrated in marginal settlements which normally have

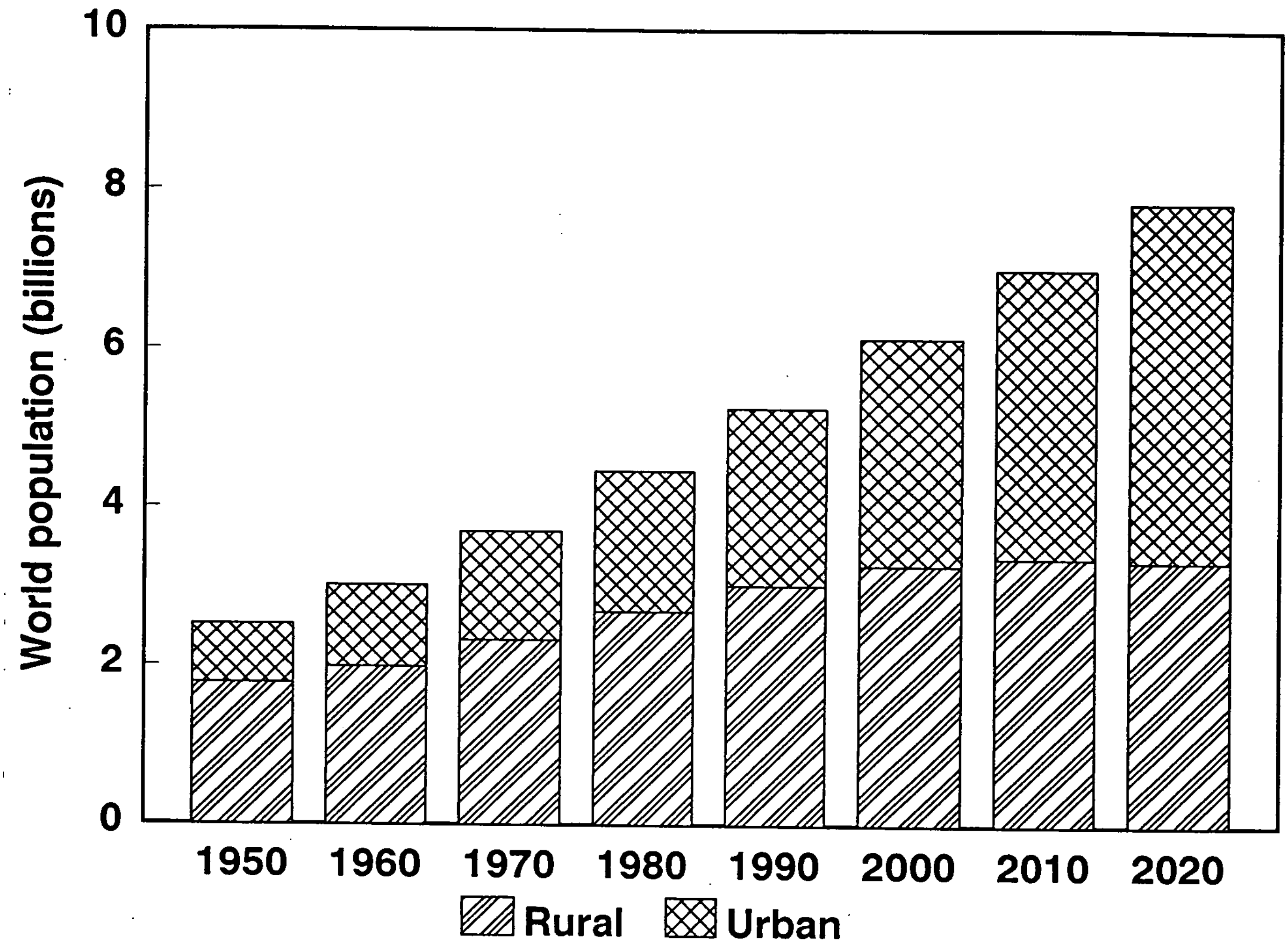

Figure 3.1 Growth in world urban population

only limited access to public water and sanitation (Table 3.1). These marginal settlements are growing at an alarming rate, in some countries 25-50% of the population is believed to be living in informal marginal settlements (Lea and Courtney, 1986).

Many cities in developing countries are dependent on groundwater obtained from unconsolidated sediments for part, or even all, of their water supply. Indeed many of these cities developed because of the availability of groundwater of good quality to provide potable supplies since surface water sources are either non-existent or of doubtful quality. Even in those cities where the piped water supply is largely derived from surface water sources, groundwater may still make a very significant contribution because a large proportion of the non-piped supply is obtained from groundwater. For some cities piped water coverage is as low as 30-40% (Table 3.1).

Table 3.1 Urban population with access to water supply and sanitation

Indicator	City			
	Manila	Jakarta	Calcutta	Madras
Total population (millions)	6.4	8.0	9.2	5.0
Area (km^2)	646	550	800	1170
Urban density (cap/ha)	98	200	115	43
% population in substandard housing	45	40	33	60
% living in illegal settlements	30	-	-	25
% with piped water to house	43	47	48	40
% garbage collected	70	25	55	78
% access to human waste disposal	60	42	45	58

Source Lea and Courtney (1986)

It has long been recognised that urbanisation results in important changes to the groundwater balance both by replacing and modifying groundwater mechanisms and by introducing new discharge patterns due to abstraction from wells (Foster, 1988). In particular, mains water and sanitation systems can have a significant impact on shallow aquifers that underlie a city and they may become major components of the urban hydrologic cycle as a result of leakage and/or seepage. Where a city relies on aquifers located within, or close to, urban areas for a significant component of its water supply requirements, these factors may lead to a deterioration in quality and a depletion of the resource of the underlying aquifer.

While pollution of surface water is more obvious than that of groundwater, the latter is more difficult to remedy. Restoration of a seriously contaminated aquifer to drinking water standards is costly and may not be possible. Furthermore, where aquifers are tapped by a very large number of private drinking water wells, treatment of all the wells is not a practical option. Protection of this valuable resource must always be the preferred policy.

3.3 Urbanisation and groundwater recharge

It is often thought that urbanisation reduces infiltration to groundwater due to the impermeabilisation of the catchment by paved areas, buildings and roads, however the reverse is often true and recharge beneath cities is usually substantially greater than the pre-urban values (Foster et al, 1993). This increase in deep percolation to groundwater is attributed to the importing of large volumes of water (from peri-urban well-fields or from surface water) and its subsequent infiltration to the subsurface as a result of leaking water mains, soakaway drainage, on-site sanitation systems and highway drainage soakaways.

The effects of urbanisation on groundwater recharge can be illustrated by two case studies. In Santa Cruz (Bolivia), the major components of urban recharge are contributed both by the disposal of wastewater to the ground via in-situ sanitation systems and by leaking water mains (Box 3.1). On the other hand in Hat Yai, in southern Thailand, where much of the domestic wastewater is discharged into canals, seepage from these surface water courses to the underlying aquifer is the principal source of recharge beneath the city (Box 3.2). The increase in recharge beneath urban centres can be considerable, particularly in semi-arid and arid climates (Figure 3.2). The city of Lima (Peru) provides a good, if rather extreme, example. Prior to urbanisation, rainfall recharge to the underlying sand and gravel aquifer was considered negligible due to high seasonal temperatures and near absence of rainfall. However, as a consequence of urbanisation groundwater recharge now exceeds 700 mm/a, largely as a result of mains water leakage and over-irrigation of amenity areas, with seepage from canals and wastewater lagoons being important in some areas (Geake et al, 1986). A more detailed review of urban recharge processes in the cities referred to here is given by Foster et al (1993).

The impact of this modification of recharge mechanisms on groundwater quality is difficult to predict and will depend on the relative importance of the various recharge sources (Table 3.2), the geological and hydrogeological characteristics of the subsurface layers, the climate type, the types of industry and mechanisms of waste disposal.

Table 3.2 Sources of recharge in urban areas - Implications for groundwater quality

Recharge source	Importance	Water quality	Pollution indicators
Leaking water mains	Major	Excellent	No general indicators
On-site sanitation	Major	Poor	NO_3, B, Cl, FC, DOC
Leaking sewers	Minor	Poor	NO_3, B, Cl, FC, SO_4 (industrial chemicals)
Pluvial soakaway drainage	Minor-major	Variable	NO_3, Cl, FC, HC, DOC, variable
Seepage from canals-rivers	Minor major	Moderate-poor	NO_3, B, Cl, SO_4, FC, DOC,(industrial chemicals)

FC = faecal coliforms; DOC = dissolved organic carbon; HC = hydrocarbons

Box 3.1 **Impact of urbanisation on groundwater recharge and flow: a case study from Santa Cruz, Bolivia**

The city of Santa Cruz, Bolivia is a low-rise, relatively low-density city, and one of the fastest growing in all the Americas. It is unusual in that, up to the present, all of its water supply is derived from wellfields within the city limits, abstracting from a semi-confined (outwash plain) alluvial aquifer. A total of 78 Ml/d is derived mainly from 18 production boreholes.

The city has developed relatively good coverage of mains-water supply, considering the very rapid rate of population growth, but only the older central area has mains sewerage. It has few parks or squares but all houses, including those close to the city centre, have substantial gardens and thus the proportion of the land surface impermeabilised by roofs and pavements is relatively low. Despite seasonally high-intensity rainfall, stormwater drainage to lined canals is only just being developed locally. For the most part, surface run-off infiltrates around the margins of impermeable areas into the sandy subsoil. Shallow flooding is relatively frequent but rarely persists for more than two days.

The largest additional components of groundwater recharge beneath the city are: (1) the in-situ disposal of wastewater to soakaways and septic tanks, and (2) leakage from the mains water supply. Seepage to groundwater from surface water is also believed to be significant, but is difficult to quantify precisely.

Private boreholes are generally less than 90 m deep and abstract groundwater from the shallow aquifers only. However the main public supply boreholes pump from deeper aquifers (90-315 m) and as a consequence induce significant leakage from the overlying aquifer.

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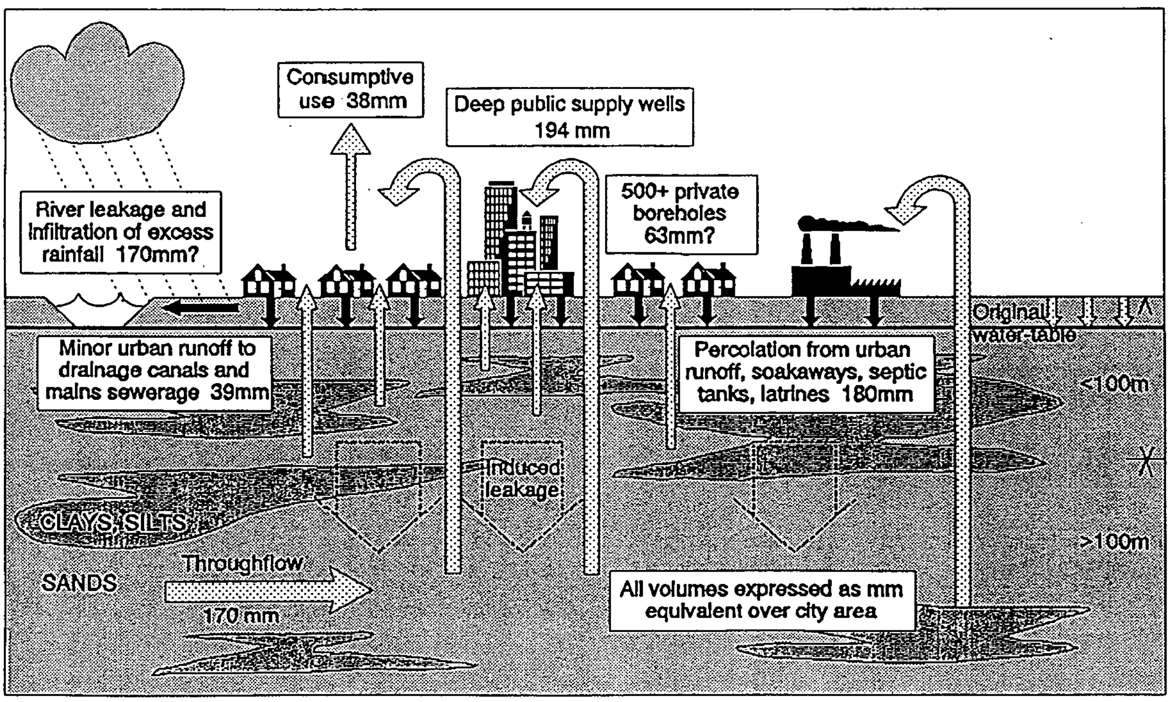

Box 3.2 Impact of urbanisation on groundwater recharge and flow: a case study from Hat Yai, Thailand

The city of Hat Yai, in southern Thailand, has a population of about 140,000 and is a rapidly developing commercial and industrial centre. It has a high population density, in excess of 200 c/ha within the city centre and, as a result, a high proportion, probably over 60% of the city area is impermeabilised by roofs and pavements.

The water supply situation is complex with most of the domestic mains water being imported from outside of the urban limits and derived from surface sources. Local groundwater resources however provide for the industrial and commercial users as well as a substantial component of the private domestic demand. As a consequence, groundwater represents as much as 60% of the total city water supply.

The city is situated on low-lying coastal alluvial deposits and in consequence experiences problems with wastewater and stormwater disposal. It is estimated that about 20% of the wastewater disposal is directly to the ground via unsewered sanitation units, the remainder being connected to open drains which discharge into larger drainage canals, which also receive stormwater runoff. However, as a result of local overexploitation of groundwater within the urban area and piezometric lowering in the semi-confined aquifer, canal seepage now represents the single most important component of groundwater recharge. It has been estimated that for some wells in the city centre up to 80% of the water is derived from induced seepage from canals.

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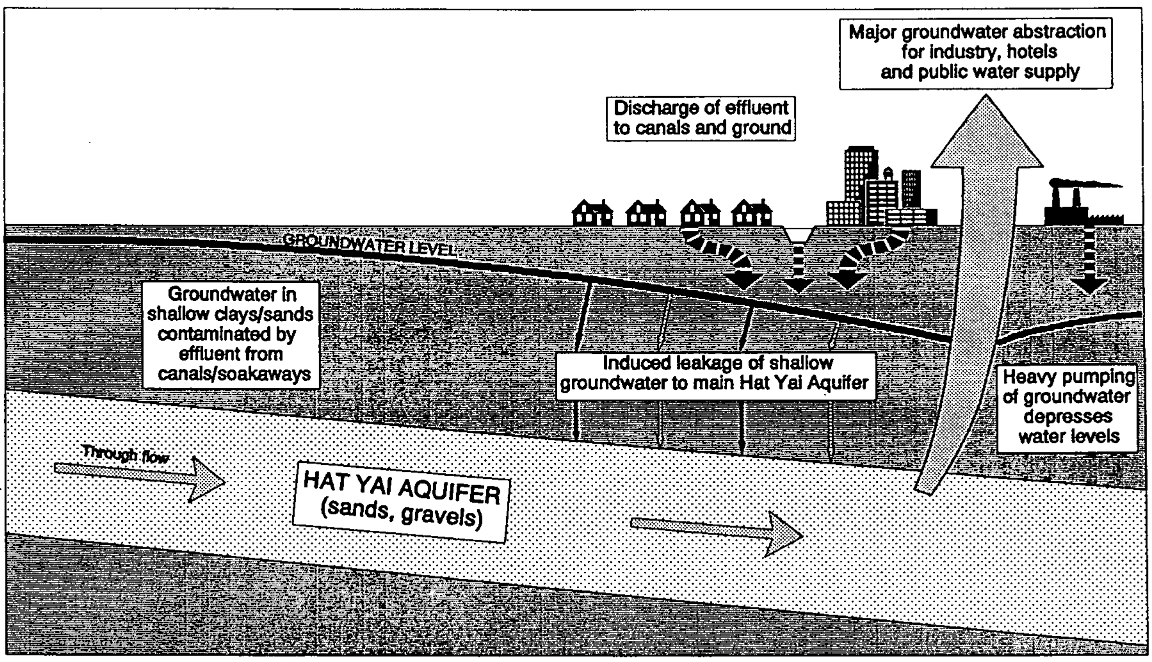
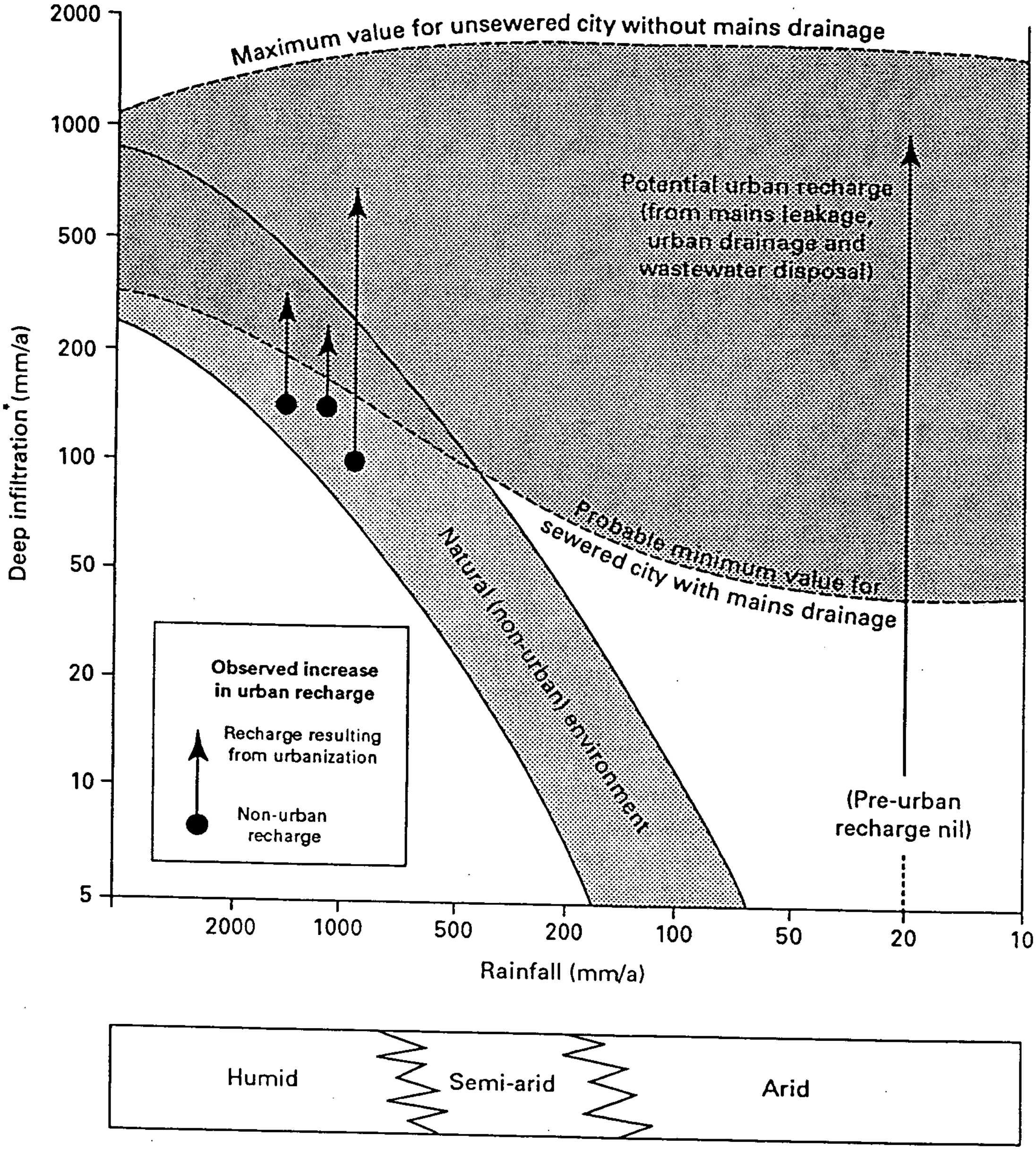
Foster S S D, Morris B L and Lawrence A R 1993. Effects of urbanization on groundwater recharge. In Groundwater Problems in Urban areas, ICE International Conference, London.



Pathway of pollutants from surface to main water-supply aquifer



* to unconfined or semi-confined aquifer recognising that surface runoff becomes more frequent in high rainfall situations, but not making any allowance for phreatic evapo-transpiration.

Figure 3.2 Increase in groundwater recharge due to urbanisation

In addition to the deterioration of groundwater quality caused by seepage and leakage to the subsurface of urban and industrial wastes, the considerable, and often uncontrolled, groundwater abstraction can produce significant lowering of water levels, despite an overall increase in groundwater recharge, leading to salt water encroachment (see Chapter 6). This is an especially important problem in south and east Asia as many cities are located on low-lying coastal aquifers and are therefore particularly vulnerable (e.g. Bangkok, Jakarta, Madras). In Bangkok, groundwater abstraction from deep, semi-confined aquifers in the underlying alluvial deposits has resulted in a significant depression of groundwater levels. Water levels declined by 40 m in the city centre between 1955 and 1980. This in turn resulted both in salt water intrusion and in subsidence (Ramnarong and Buapeng, 1991).

A similar pattern is observed in Jakarta; increased groundwater abstraction from deep, semi-confined aquifers in the alluvium to meet the rising demand for water by a growing urban population and by industry has resulted in declining groundwater levels producing in turn severe problems of seawater intrusion and subsidence (Figure 3.3). Furthermore, this lowering of water levels and reversal of groundwater gradients will limit the effectiveness of dilution of urban-derived contaminants by flushing with groundwater throughflow (Soefner et al, 1987).

In the thick UNSAs sequences which underlie these and other cities, groundwater abstraction from depth can result in groundwater flow regimes being dominated by vertical flow paths. Induced leakage from shallow aquifers, which are the first to feel the quality impact of urbanisation, can become the main component of recharge to deeper aquifers which may provide the main source of supply. Thus, while initially relatively well-protected, these deep aquifers may eventually suffer deterioration in groundwater quality. This modification of the groundwater flow pattern is observed in both Santa Cruz (Box 3.1) and Hat Yai (Box 3.2).

The use of private wells, which can account for more than 50% of the total urban water demand, presents particular problems to agencies responsible for managing urban groundwater resources since it is difficult to control their abstraction or monitor their quality. Further, as many private wells may have been constructed without adequate sealing of casing against the formation, or at the surface, this may directly result in serious contamination of the well itself or, even worse, of the aquifer.

3.4 Modification of groundwater quality as a result of urbanisation

Most urban areas present a complex array of human activities potentially polluting to groundwater. To attempt to evaluate the corresponding subsurface load, it is essential to subdivide such areas according to predominant activity and wastewater arrangements. In practice, boundaries will be gradational and to some extent arbitrary, making accurate assessments of contaminant load practically impossible.

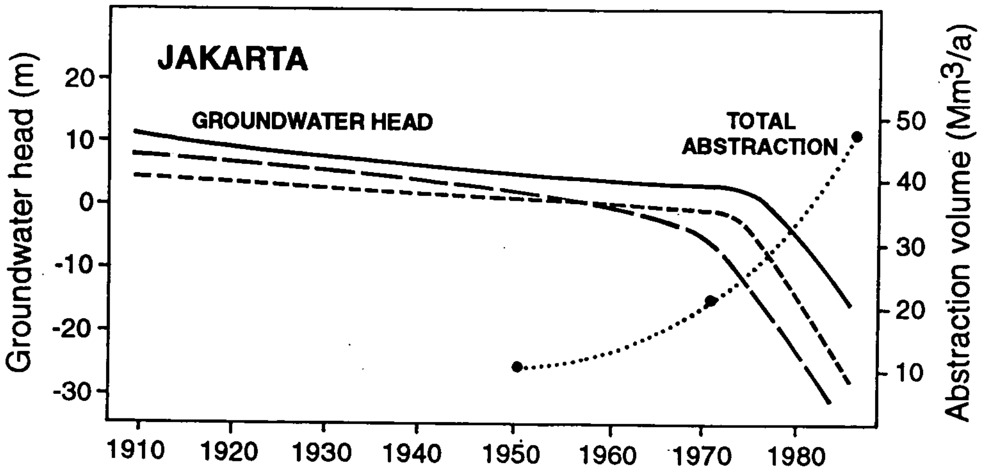


Figure 3.3 Consequences of overabstraction in Jakarta

3.4.1 *Urban residential areas*

In urban residential areas the principal concern is the subsurface load associated with unsewered sanitation units such as septic tanks, cesspits and latrines in residential areas without, or with incomplete coverage, by mains sewerage. Even areas which are essentially residential will normally include dispersed small-scale service industries whose potential contaminant load also has to be taken into consideration.

In areas which have a well designed and carefully operated sewerage system, this will greatly reduce the subsurface contaminant load associated with urbanisation, although local contamination may occur as a result of sewer ruptures and leaks. The growth in sanitation coverage has generally lagged behind that of water supply. Whilst a few cities have been able to implement master sewerage plans based on conventional sewerage systems, they often serve only a small minority of the population in the high income group and most sanitation coverage consists of low cost on-site disposal systems. These can provide adequate service levels for excreta disposal in villages, small towns, and even larger urban areas, at much lower cost than mains sewerage systems. Various types of installation may be used including septic tanks, cesspits, ventilated dry and pour-flush pit latrines (Figure 3.4). Since improvements in sanitation are still widely and urgently needed, a continued expansion of excreta disposal to the ground is likely to occur.

It is important to recognise that there are significant differences between septic tanks and other on-site excreta disposal units (Lewis et al, 1982). The former are likely to pose a less serious threat to groundwater since:

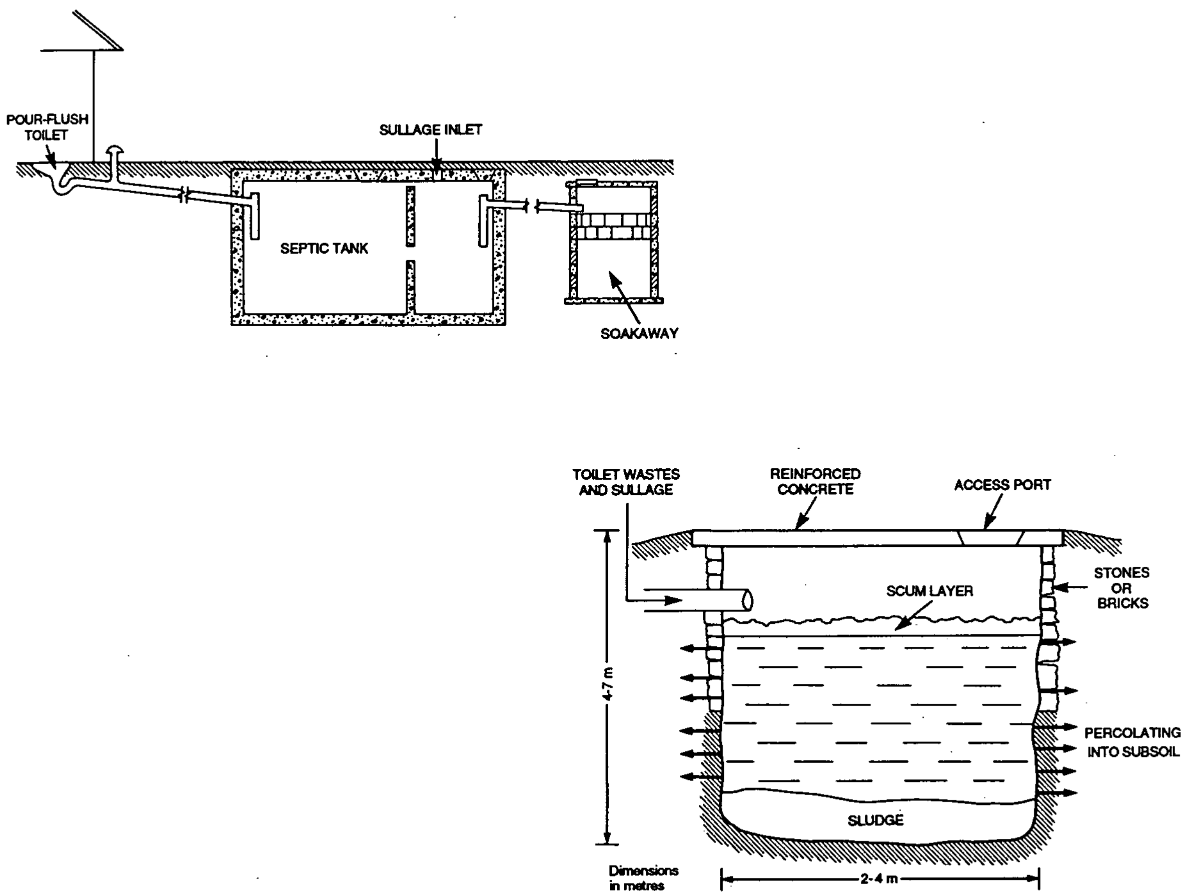


Figure 3.4 Construction of a pour flush latrine with connected septic tank and soakaway and a pit latrine

a) they discharge at higher levels in the soil profile where conditions are more favourable for pathogen elimination,

b) the rate of water discharge is often less and

c) when efficiently operated a large proportion of the solids in the effluent is periodically removed

Even so, the use of septic tanks in areas of high population density with inadequate space for on-site disposal of effluent can result in serious pollution. For example, in Jakarta which has more than 900,000 septic tanks, the effluent is discharged into inland waterways because of inadequate soakaway systems and poor maintenance; this causes severe pollution of the surface water and poses a significant health risk (Soefner et al, 1987).

The immediate concern to groundwater from on-site sanitation systems is contamination by pathogenic organisms. In unconsolidated deposits, filtration and 'die-off' during migration through one or two metres of fine-grained strata normally reduced pathogen numbers to acceptable concentrations (Lewis et al, 1982). Problems usually only arise where the water table is so shallow that on-site sanitation systems discharge directly into the saturated zone. However, in coarser-grained deposits, pathogens can migrate significantly deeper through the unsaturated zone. In Lima, microorganisms were observed to have migrated through up to 20 m of coarse sands beneath cultivated land irrigated with wastewater (Geake et al, 1986). Karstic limestones are particularly vulnerable to rapid pathogen transport from unsewered sanitation.

Once in the saturated zone, pathogen transport though generally more rapid than in the unsaturated zone is still likely to result in significant attenuation except in the coarsest and most permeable deposits. The risk to groundwater supplies may be significantly enhanced by the persistence of the organism in the subsurface and considerable uncertainty about the persistence of some pathogens (especially viruses) in groundwater systems still exists. Whilst bacterial contamination of shallow wells is generally believed to be widespread, it is in many cases likely to be more a consequence of improper design and construction of the well rather than of aquifer contamination.

In a study of public handpump tubewells in Thailand, Lloyd and Boonyakarnkul (1992) demonstrated that well contamination, assessed by the number of faecal coliforms found, was most clearly correlated to the proximity of latrines and to poor maintenance. They calculated a sanitary hazard index for each district for each of 10 identified problems from the above set. A good correlation was obtained between the hazard index and the degree of contamination of the well (Figure 3.5). Overall the highest ranking indices were the close proximity of a latrine, loose handpump and latrine up gradient of the handpump. In cases where poor maintenance or inadequate drainage were shown to be a problem then remedial action was able to return 94% of the wells previously with intermediate to high coliform counts to an acceptable quality (no or low risk).

	Sanitary Inspection Hazard Score	0	1	2	3	4	5	6	7	8	9	Risk based on faecal coliform count
Faecal coliforms (counts / 100 ml)	101-1000							•		•	•	Very high
	11-100					•		••	••••	•		Intermediate to high
	1-10			•		•••		••				Low
	0			••••	••••	•••		•				None
		No hazard No action		Low hazard Low action priority		Intermediate to high hazard Higher action priority			Very high hazard Urgent action			

Figure 3.5 Correlation of hazard index and faecal coliform counts in tubewells, Khon Kaen Province, Thailand (Lloyd and Boonyakarnkal, 1992)

Estimation of pollutant loading

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

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

where:

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

Greatest uncertainty surrounds the proportion of the deposited nitrogen (N) that will be oxidised and leached in groundwater recharge. A range of 20-60% (0.2-0.6) is considered possible (Walker et al, 1973; Kimmel, 1984; Thomson and Foster, 1986) and the actual proportion will depend upon the per capita water use, the proportion of volatile losses of N compounds and the amount of N removed during cleaning, which will vary with the type of installations involved. Considerable uncertainty may also surround the estimation of natural infiltration rates from excess rainfall.

Nevertheless, it is evident from the above equation that troublesome nitrate levels are often likely to develop except where water use is very high and/or population densities are very low. Especially high concentrations are likely to occur in those arid regions with low per capita water usage.

Surveys of groundwater quality in two Indian cities, Madras and Lucknow, indicate widespread contamination of the shallow groundwater by nitrate (Table 3.3). Both cities are underlain by unconsolidated sediments and are largely unsewered. Mean nitrate concentrations are in the range 10-25 mg NO_3-N/l, although locally much higher concentrations, in excess of 70 mg NO_3-N/l, occur (Sahgal et al, 1989). Assuming that the population density and urban recharge rate are in the range 100-200 c/ha and 50-200 mm/a and non-consumptive water use is 50 l/c/d, application of the above equation indicates that about 20% of the nitrogen from sanitation is being leached to the water table (Morris et al, 1994).

The city of Santa Cruz (Bolivia) is also largely unsewered and is dependent upon groundwater taken from the underlying outwash plain deposits which extend to considerable depth (>1500 m). Beneath the city, groundwater nitrate concentrations in the shallow aquifer (<45 m) are in the range 10-40 mg NO_3-N/l. The deeper aquifers, up to 200 m, also show a significant increase in groundwater nitrate concentrations (Box 3.3).

Table 3.3 Groundwater nitrate concentrations beneath Indian cities

City	Population (million)	Population density (cap/ha)	Rainfall (mm/a)	Sewage disposal systems	Aquifer type	Groundwater nitrate concentrations ($mg\ NO_3$-N/l)	
						Max	Min
Madras	4.3	220	1100	Septic tanks and to canal	Shallow coastal alluvium (10-20 m)	>250	>10
Lucknow	1.1	125	1000	Septic tanks, latrine soakaways	Alluvium (0-100 m)	140	>10

After Somasundaram et al (1993) and Sahgal et al (1989)

Many cities, especially in south and east Asia are located on low-lying coastal alluvial plains underlain by a shallow water table. The disposal of excreta to the ground by on-site sanitation systems under these circumstances is often not possible (because of surfacing of the water table during the monsoon). Thus in many areas where sewerage systems are non-existent, human faeces and other wastes are discharged directly, or indirectly (for example, where septic tank overflows are directed into drains), into surface water courses. The rivers and drainage canals provide convenient, if not generally acceptable, means of sewage and garbage disposal (Figure 3.6). As a consequence, they receive heavy loads of untreated effluent which exceed the natural purification capacity for many kilometres downstream. Such sections of water courses can become major line sources of groundwater pollution under certain hydrogeological conditions.

These conditions prevail in Hat Yai, a city in south Thailand, where a significant deterioration of groundwater quality beneath the city is observed. High concentrations of ammonium, nitrate and chloride occur close to, and as a direct result of, leakage from the canals which carry the bulk of the city's domestic and industrial effluent (Box 3.4). However given the high population density of the city, the mean total nitrogen concentrations in the groundwater (about 2 mg NO_3-N/l) are significantly lower than might be anticipated for an unsewered city and suggests that disposal of domestic effluent to surface water, whilst unacceptable for other reasons, does reduce groundwater nitrate concentrations when compared with on-site disposal systems (Table 3.3).

The presence of ammonium in groundwater is uncommon as it is not normally soluble and is usually oxidised relatively rapidly to nitrate. However the groundwaters beneath Hat Yai are of low pH (acidic) and probably of low redox potential and this stabilises the ammonium ion. Ammonium is also a widespread groundwater contaminant in the alluvial aquifer beneath the city of Hanoi. The discharge of industrial and community wastewaters to lakes and canals and their subsequent leakage to groundwater was considered responsible for the high ammonium concentrations (Do Trong Su and Trong Hien, 1992).

In addition to elevated nitrate and chloride concentrations in groundwater; on-site sanitation systems can result in increased dissolved organic carbon, reduced dissolved oxygen and elevated iron concentrations, the latter being mobilised as a result of the reduction in redox potential induced by the degradation of organic matter. Likewise, bicarbonate concentrations can increase as carbon dioxide released by the degradation of organic matter can mobilise carbonate in the aquifer matrix to the bicarbonate ion:

$$CO_2 + CO_3^{2-} + H_2O \rightarrow 2HCO_3^-$$

In addition, the consumption of the available oxygen as organic matter is oxidised lowers the redox potential of the groundwater, which in turn transforms iron from the relatively insoluble ferric state to the more soluble ferrous state. Iron concentrations in groundwater may be increased by this process. Other ions associated with urban domestic wastewaters include boron and sulphate, which are derived from detergents.

Box 3.3 Contamination of deep groundwater: a case study from Santa Cruz, Bolivia

The city of Santa Cruz, which is located on plains to the east of the Andean Cordillera, obtains all of its water supply from deep semi-unconfined aquifers within underlying alluvial outwash plain deposits. The city is largely unsewered and most urban wastewaters (from on-site sanitation) are discharged to the ground. 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 unpolluted background. However at shallow depths (<45 m) elevated nitrate and chloride concentrations, typically in the range 10-40 mgN/l and 40-120 mg/l respectively are widespread beneath the more densely populated districts.

The source of the nitrate and chloride in the shallow groundwaters is believed to be unsewered sanitation. These groundwaters also exhibit higher bicarbonate, but lower dissolved oxygen concentrations than the shallow groundwater upgradient. The higher sulphate concentrations also encountered in the shallow groundwater are thought to be partly derived from detergents and highway runoff.

Groundwaters from aquifers of intermediate depths (45-100 m) also have elevated concentrations of nitrate, chloride, sulphate and bicarbonate, although generally less than in the shallow aquifer. Substantial leakage from the shallow aquifer in response to groundwater abstraction from depth is thought to be the mechanism causing this contamination. However the front of contaminated water does not appear to have penetrated beyond 90 m depth. Many boreholes drilled to depths less than 90 m show increasing solute concentrations with time, whereas deeper boreholes do not show this trend. Bacteriological problems were not encountered in groundwater from either the intermediate or deep aquifers.

Reference

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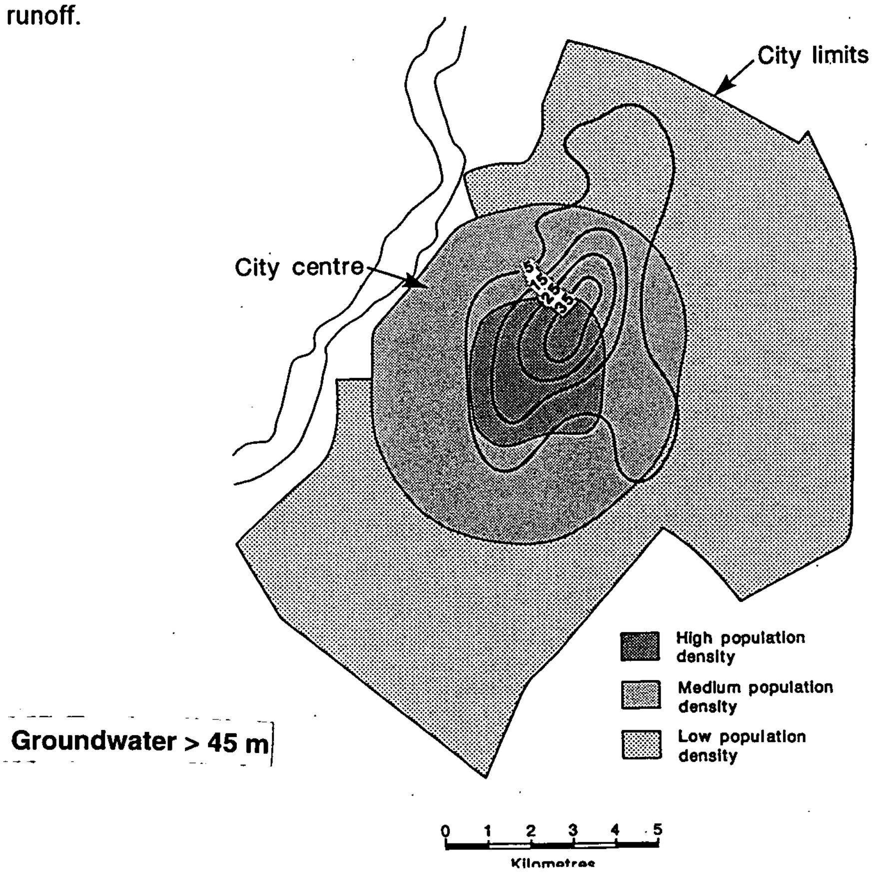
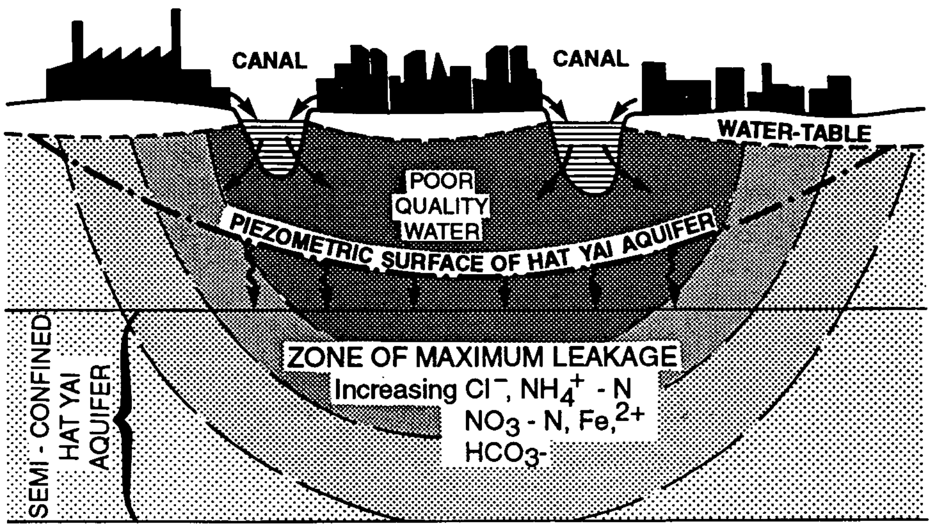


Contaminants from polluted shallow water penetrating the deeper layers of the aquifer

Box 3.4 Urban groundwater contamination from canal seepage, Hat Yai, Thailand

The city of Hat Yai, in southern Thailand, is situated on low-lying coastal alluvial deposits. The upper part of these deposits are of low permeability and have a shallow water table, as a consequence the city experiences problems with wastewater and stormwater disposal. It is estimated that about 20% of the wastewater disposal is directly to the ground via unsewered sanitation units, the remainder being connected to open drains which discharge into larger drainage canals, which also receive stormwater runoff. As a result of local heavy abstraction of groundwater within the urban area, the piezometric surface in the semi-confined main aquifer has been significantly lowered. Substantial leakage from the shallow phreatic aquifer to the semi-confined aquifer occurs beneath the city and canal seepage now represents the single most important component of groundwater recharge.

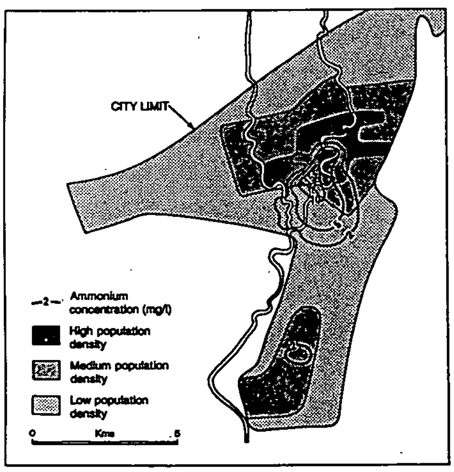


Schematic cross-section of water quality changes resulting from canal leakage

Elevated concentrations of ammonium, chloride and sulphate occur in the semi-confined aquifer beneath the city centre as a result of recharge by the poorer quality canal water. Where concentrations are highest they are equivalent to a mixed water containing some 60-80% canal water and 20-40% unpolluted groundwater.

Reference

Morris B L, Lawrence A R and Stuart M E 1994. The impact of urbanisation on groundwater quality. British Geological Survey Technical Report WC/94/56.



Degradation of groundwater in the city centre by ammonia

Figure 3.6 Canal receiving wastes from an informal settlement

3.4.2 *Other urban and industrial activities*

A number of developing countries are rapidly industrialising and there is a gradation from the major industrial powers, for example China, India, Brazil, Mexico and Korea, through to the emerging industrial countries, for example Indonesia, Thailand and Malaysia. Even in the less industrialised countries there are often numerous small factories processing agricultural products, such as food processing, textiles and leather.

In many of these countries extensive sectors of urban and fringe-urban areas remain unsewered, and it is often in these areas that increasing numbers of industries and activities (such as textiles, metal processing, vehicle maintenance, paper making, tanneries, vehicle filling stations) tend to be located, often on a small-scale and highly-dispersed basis. Most of these industries generate liquid effluents, such as spent oils and solvents. Contamination of groundwater by the chlorinated solvents, which are frequently used for cleaning and degreasing in a number of industries, is widespread in many of the developed countries. These solvents are serious groundwater pollutants both because they are toxic and their permitted concentration in drinking water is low (10-30 μg/l) and because they are known to be extremely persistent in the subsurface environment. They appear to be ubiquitous in urban groundwater and have been detected in almost all specific surveys. However, despite their increasing use in many developing countries, little monitoring for these solvents in groundwater is undertaken in these countries.

In the absence of either a sewage system or of on-site effluent treatment at the factory, disposal of industrial effluent will generally be to the ground or into a nearby watercourse. The former method will be favoured where the subsurface is permeable and the water table deep, allowing rapid and easy drainage away from the disposal point. In the low-lying alluvial aquifers, underlain by a shallow water-table, disposal via channels to canals is probably more common. Even in the latter case however, groundwater contamination can still occur where heavy abstraction can induce significant seepage from surface water to the ground. Improper disposal of industrial effluent has been cited as a cause of serious groundwater contamination in India and Mexico by various toxic metals.

There is often little if any information on sources of potential pollution from industry even though it should be possible to make very approximate estimates. To characterise fully subsurface contaminant load, information is needed on two separate factors:

(i) the quantity of effluent disposed of, or reaching, the subsurface,

(ii) the quality of this effluent.

The volume of effluent generated by a given industrial activity can generally be estimated with adequate reliability from the quantity of water used, which normally can be obtained from metering of mains water-supply and/or from estimates of yield capacity of boreholes at the industrial site itself. In the case of the large majority of industries, other than those which manufacture liquid products, this will give a reliable estimate of total effluent volume, because consumptive use is small.

Illustrative examples of effluent generation are given in Table 3.4. It must be emphasised that, in the case of subsurface contaminant load, it is not necessarily the bigger and more sophisticated industries which generate the largest subsurface contaminant load and highest groundwater pollution risk. This is because the chemical handling and effluent disposal practices are, in many cases, more carefully controlled and monitored. Equal or greater concern is associated with small service industries, such as metal workshops, dry cleaners, photographic processors and printers, because they are widely disseminated, often use considerable quantities of potentially-toxic groundwater contaminants and their effluent disposal practices may not be subject to strict control.

The assessment of effluent quality, or that part of the process fluids or effluents likely to be discharged to the subsurface, presents considerable problems because of:

(i) the great variety of industrial activities,

(ii) the considerable variation in the technological level of any given industry,

Table 3.4 Summary of chemical characteristics and risk indices for common types of industrial activity

Industrial type	Mazurek Hazard Index (1-9)	Flow (m^3/T)	pH	Salinity load	Nutrient load	Organic load	Hydro-carbons	Faecal pathogens	Heavy metals	Synthetic organics	Groundwater pollution potential
Iron and steel	6	30	6	*	*	**	**	*	**	**	2
Metal processing	8	-	7-10	*	*	*	*	*	***	***	3
Mechanical engineering	5-8	-	-	*	*	*	***	*	***	**	3
Non-ferrous metal	7	-	-	*	*	*	*	*	***	*	2
Non-metallic minerals	3-4	30	-	***	*	*	*	*	*	*	1
Petrol and gas refineries	7-8	-	-	*	**	***	***	*	*	**	3
Plastic products	6-8	1-4	-	***	*	**	**	*	*	***	3
Rubber products	4-6	1	-	**	*	**	*	*	*	**	2
Organic chemicals	3-9	92	7	**	*	**	***	**	**	***	3
Inorganic chemicals	6-9	115	-	**	*	*	*	*	***	*	2
Pharmaceutical	6-9	4000	-	***	**	***	*	**	*	***	3
Woodwork	2-4	1	-	**	*	**	*	*	*	**	1
Pulp and paper	6	108^{+}	8	*	**	**	*	*	*	**	2
Soap and detergents	4-6	5	-	**	*	**	**	**	*	*	2
Textile mills	6	400	-	**	**	***	*	*	*	**	2
Leather tanning	3-8	37	-	***	**	**	*	*	**	***	3
Food and beverages	2-4	-	-	**	***	***	*	***	*	*	1
Pesticides	5-9	30	-	**	*	*	*	*	*	***	3
Fertilisers	7-8	6	-	***	***	*	**	*	*	**	2
Sugar and alcohol	2-4	62	-	***	***	***	**	*	*	*	2
Electric power	-	-	-	*	*	*	***	*	***	**	2
Electric and electronic	5-8	-	-	*	*	*	***	*	**	***	3

\+ maximum value of average
\- no data available
* low)
** moderate) probability of troublesome concentrations in process fluids and/or effluents
*** high)

(iii) the extreme and erratic temporal variation in concentrations of toxic constituents in industrial effluents,

(iv) the wide variation in the use and efficiency of treatment processes for industrial effluents and uncertainties in their effectiveness in removing potential groundwater contaminants,

(v) the lack of effluent quality control and of full chemical analyses of effluents, including concentrations of heavy metals and synthetic organic compounds,

(vi) the lack of adequate published information on effluent characteristics for representative industries, especially those operating in developing economies,

(vii) the wide variety of modes of handling and disposition of process liquids and effluents, including the frequent adoption of clandestine practices.

Despite these many limitations, it is believed that untreated effluents can be characterised in qualitative terms from published data (Lund, 1971; Nemerow, 1971) for 22 major categories of industry. The relative frequency of such industrial categories can be illustrated by data from a survey of industrial activity in Sao Paulo State, Brazil.

Although much has been published on industrial effluent treatment processes (Lund, 1971; Nemerow, 1971; Eckenfelder, 1976), it is not easy from existing information to indicate by how much concentrations will be reduced for a given level of treatment. It is probably realistic to assume that if advanced tertiary treatment is practised the concentration of potential groundwater contaminants in the final effluent will be minimal. However in all other instances, significant concentrations must be assumed to be present.

Bigger industrial plants, generating larger volumes of process water, will also commonly have surface impoundments for the handling or concentration of liquid effluents. Such installations, and also underground storage tanks and industrial sewer lines, are theoretically containable, but in practice frequently leak and can represent major hazards (Box 3.5). An inventory of all sites at which the more hazardous chemicals are manufactured, stored or used, should be required, with enforcement of appropriate controls to minimise the risk of direct soil discharge.

3.4.4 *Solid waste disposal*

The disposal to land of urban and industrial solid wastes also gives rise to groundwater pollution. The most serious risks occur where uncontrolled tipping, as opposed to controlled sanitary landfill, is practised, and where hazardous industrial wastes, including drums of liquid effluents, are disposed of at inappropriate sites. In many cases no record is kept of the nature and quantity of wastes disposed of at a given site and abandoned sites represent a potential hazard to groundwater for decades.

Box 3.5 Groundwater contamination by leaking storage tanks, Japan

Contamination of a shallow unconfined sand and gravel aquifer beneath an electrical factory was first indicated by the detection of the chlorinated solvent trichloroethene (TCE) in a nearby public supply well. A survey revealed that a plume of contaminated water extended for 3 km downgradient of the plant, in which TCE was present at concentrations of more than 30 μg/l (the drinking water guideline limit) The highest concentrations (40,000 μg/l) was detected in groundwater beneath storage tanks. The quantity of solvent leaked is not known, but given that up to 9 tons of solvent were consumed each month, it is likely to have been considerable.

Following discovery of the groundwater pollution much of the most contaminated soil from the unsaturated zone was removed. This had an immediate effect on the concentration of TCE in the shallow part of the aquifer, but concentrations in the deeper groundwater remained unchanged. This implies that a large quantity of solvent remained in the deeper aquifer. Removal of the solvent from aquifers is usually achieved by pumping and treating the contaminated groundwater. These solvents are of limited solubility and tend to remain as a separate dense phase in water, so a spillage of undiluted liquid may dissolve only over a very long period of time and will form a long-term source of pollution.

This is clearly demonstrated at this site where even after several years of extraction of contaminated water, the concentrations of TCE remains above the drinking water limit. Restoration of aquifers contaminated by solvents to concentrations even approaching drinking water guidelines is likely to be expensive, time consuming and difficult to achieve.

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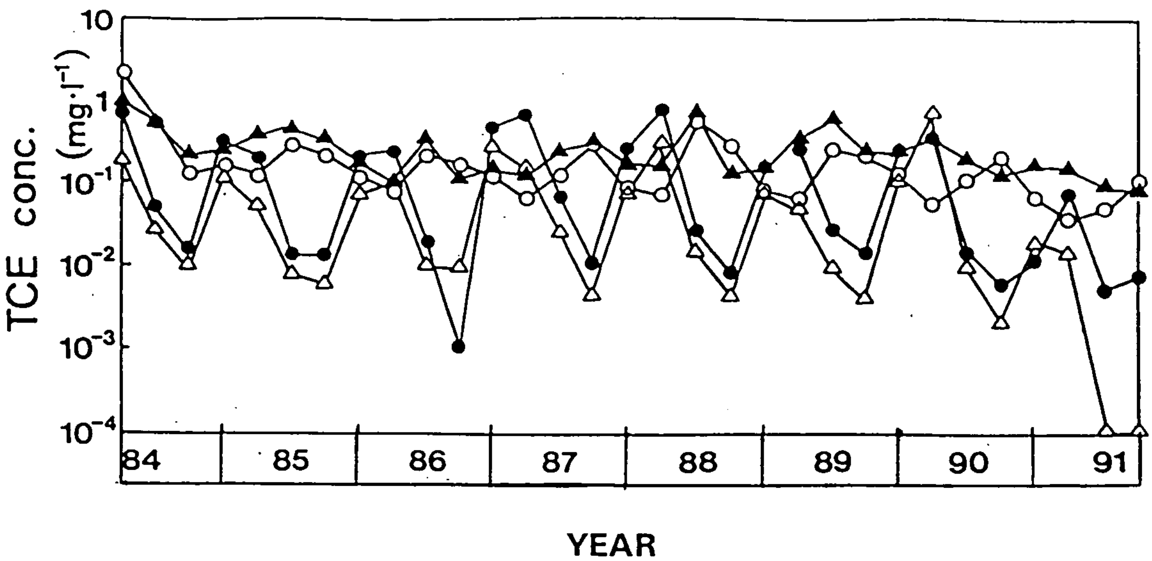
Hirata T, Nakasugi, Yoshioka M and Sumi 1992. Groundwater pollution by volatile organochlorines in Japan and related phenomena in the subsurface environment. Water Science & Technology, 25, 11, 9-16.



Long-term changes in TCE concentration following soil removal and groundwater extraction

Serious and persistent pollution of various water-supply boreholes with highly toxic hexavelant chromium on an industrial site situated on the outskirts of a Mexican city is believed to have resulted from the ground disposal of solid residues from a metal processing plant (Figure 3.7). The problem is often exacerbated because disposal is often to low-lying ground where the depth to water table is minimal and direct contamination of the shallow groundwater is possible. In areas where the shallow strata are highly permeable, elevated concentrations of contaminants derived from the waste disposal have been detected at significant distances from the waste disposal site.

A study around the city of Jaipur, India, where the aquifer is composed of highly permeable windblown sands and the water table is shallow, showed well defined contamination of groundwater from waste tips. These had been in use for less than 12 years, but poor water quality could be detected up to 500 m away.

The solid wastes in many developing countries are generally less toxic, having a greater content of water and decaying vegetable matter compared to typical solid wastes from developed countries, which often contain significant levels of heavy metals (cadmium, mercury, lead and chromium) and various synthetic organic compounds (solvents, phenols and PCBs).

Nevertheless, the resulting contaminant plume clearly can represent a serious health hazard. Further, as these cities expand, the urban sprawl is likely to encroach onto areas previously used for solid waste disposal. Informal or marginal housing is most likely to develop on or close to this land, as it is usually the least desirable and often the only land available. These settlements all too frequently have no provision for piped water and rely upon privately constructed wells. These supplies are largely unmonitored and potentially pose a serious health risk to the population.

3.4.5 *Effluent lagoons*

In those urban areas with mains sewerage systems, an economical method of sewage treatment, if any is considered or practised, is wastewater stabilisation by retention in shallow oxidation lagoons, prior to discharge into rivers, or onto land or to re-use for irrigation. Such lagoons are often unlined and may have high rates of seepage loss, especially after initial construction and subsequent cleaning. If so, they can have considerable impact on local groundwater quality, especially in relation to nitrogen and trace organic compounds.

3.5 Summary-Discussion

Urban populations in the developing countries are growing rapidly; a large proportion of this population increase is concentrated in the marginal (or informal) housing districts where access to sanitation and to piped water is often limited.

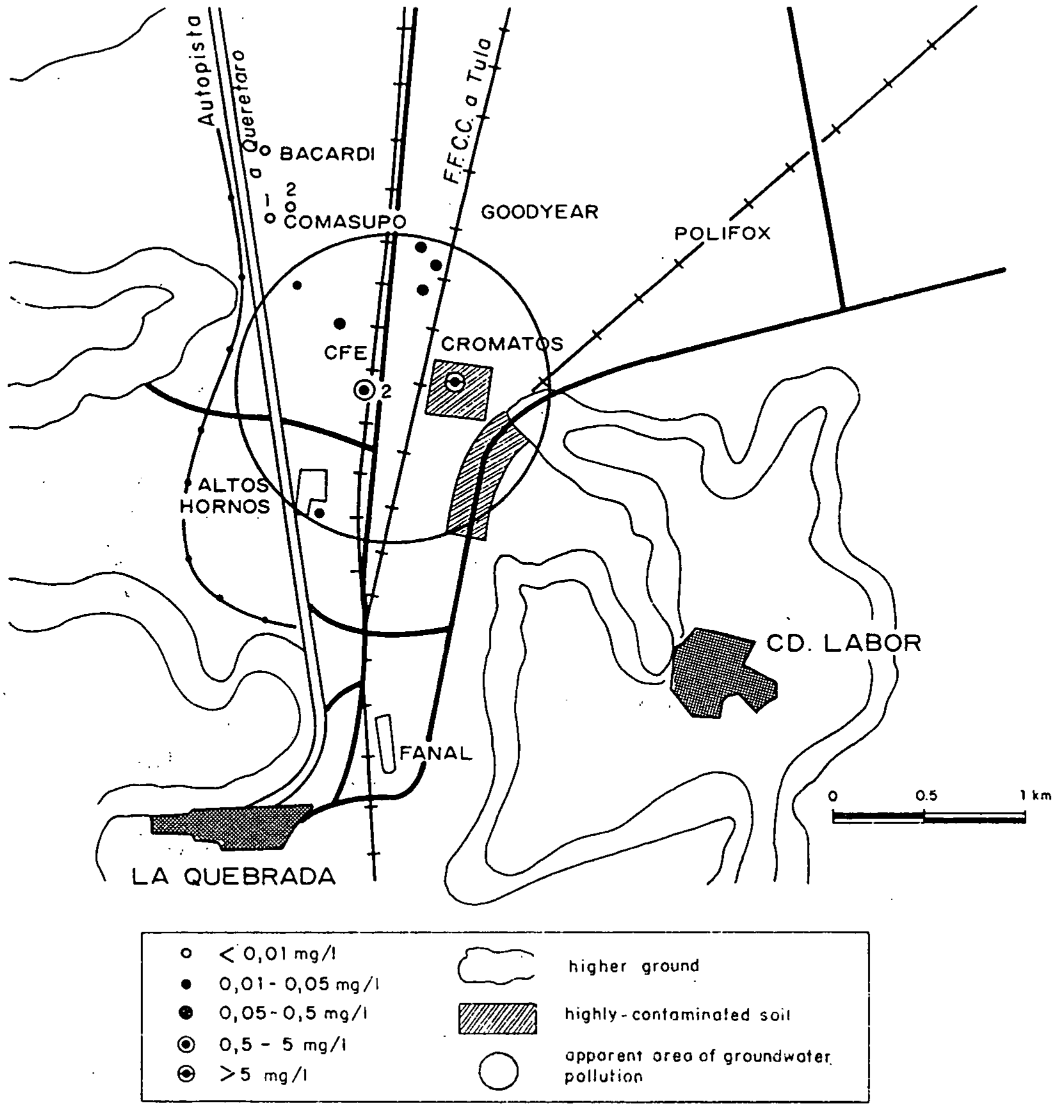


Figure 3.7 Water supply boreholes affected by chromium resulting from ground disposal of metal processing residues, Mexico

Many of these cities are dependent on groundwater obtained from unconsolidated aquifers for a significant proportion of their water supply and there is considerable concern that this rapid urbanisation may result in the deterioration of groundwater quality, especially in the longer term. Urbanisation has been shown to considerably increase groundwater recharge principally as a result of the introduction of new recharge sources. These include leaking water mains, seepage from on-site sanitation systems and induced leakage from surface water courses. This modification in the pattern of recharge and the introduction of new recharge sources does have a profound impact on the quality of groundwater in the underlying aquifers. However it is often not possible to predict precisely the full impact in the longer term as it is difficult to quantify the various recharge sources and the areas over which these recharge sources are likely to occur. Despite this, a number of general observations can be made:

(1) The presence of FC bacteria in deeper boreholes in unconsolidated aquifers almost certainly reflects local contamination of the borehole rather than of the aquifer.

(2) For most unsewered cities, urbanisation inevitably produces elevated groundwater nitrate concentrations in the longer term and, where this information is available, groundwater nitrate concentrations are typically in the range 10-50 mg N/l. The precise concentrations depend on (a) percentage of excreted nitrogen that is oxidised and leached, (b) the population density, (c) the non-consumptive water use, (d) the natural recharge, (e) types of on-site sanitation systems used and (f) dilution within the aquifer by groundwater throughflow.

Considerable uncertainty exists as to the percentage of nitrogen mobilised and leached to groundwater from on-site sanitation systems. For alluvial groundwater systems the percentage nitrogen mobilised is close to 10-20%. Where the sewage is directed into surface water courses, and subsequently recharges the shallow groundwater, a much smaller percentage of the nitrogen will enter the aquifer.

(3) In addition to nitrate, elevated chloride concentrations in groundwater also occur as a result of seepage from on-site sanitation systems. Where a good correlation exists between groundwater nitrate and chloride concentrations then it indicates that the major source of chloride is from unsewered sanitation. However, there are other sources of chloride (and nitrate) and in coastal aquifers especially a large part of the chloride in groundwater is contributed by seawater.

(4) Other parameters possibly indicative of urban contamination include ammonium, sulphate and bicarbonate. The reduction in dissolved oxygen, associated with the degradation of the organic load derived from wastewaters, can result in enhanced groundwater iron concentrations.

(5) Contaminants associated with industrial effluents, such as various metals and synthetic organic chemicals, do occur in groundwater beneath some industrial areas. Despite this evidence, there is little regular monitoring of groundwater for such contaminants in many developing countries. This probably reflects the costs and difficulties in sampling and analysing for these compounds. In addition, there is often considerable uncertainty as to which compounds should be monitored for.

Many cities face severe problems of water shortage which have often been compounded by inadequate management of their groundwater resources. This is because; firstly, a large proportion of the groundwater pumped is by private boreholes, used for both domestic and industrial supply, and it is difficult to control their abstraction. Secondly the problems are usually various, albeit that they are often inter-related, and include falling water levels, subsidence, sea water intrusion and contamination by domestic and industrial wastes. Thirdly the groundwater systems are often poorly understood and inadequately monitored largely because resources for such activities are usually limited. The available funds are normally allocated to increasing or improving water supply schemes.

The problems facing many cities are severe but should not be exaggerated. The subsurface is able to attenuate many contaminants; so for deep groundwater systems, serious groundwater problems may only occur in the longer term from the more persistent and mobile compounds. Reduction in groundwater contamination might be achieved by treating industrial wastes at source and by the introduction of waterborne sewerage systems in the most densely populated areas.

The use of urban wastewater for irrigation and subsequent groundwater recharge is likely to become increasingly important in the future (Chapter 5).

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METHOD SUMMARY SHEET (WQM 02)

TITLE: Investigating the impact of urbanisation

Scope and use of method

Many very large cities are dependent on groundwater drawn from UNSAs. They have often been sited and have grown and flourished because of the availability of shallow groundwater or surface water. The continuing rapid development of such cities places increasing stress on the underlying aquifers, both in terms of resources and quality. Large urban areas modify groundwater recharge mechanisms and discharge patterns, and also provide pollutants which can cause serious degradation of groundwater quality. For quality management, it is necessary to understand and quantify these processes if groundwater quality is to be effectively protected and maintained, or restored once polluted.

Method

The first step is to review and describe the hydrogeological conditions to identify the sources of recharge and discharge in the urban area of interest. Depending on the type and level of urban infrastucture, these could include leaking water mains and sewers, unsewered sanitation and infiltration of storm runoff from paved surfaces. The sources of groundwater discharge by municipal and private industrial or domestic boreholes must also be identified and quantified. A conceptual model of the hydrogeological regime and groundwater discharge and recharge patterns must be developed at an early stage of the investigation. Quantifying the sources of recharge may be difficult; real figures for leakage from sewers and water mains are likely to be particularly difficult to obtain. Groundwater discharge by abstraction can normally be arrived at by conventional inventory approaches. The balance derived must reflect trends in groundwater level; heavily exploited aquifers may show declining groundwater levels.

Most urban areas include a complex array of human activities, many of which have the potential to cause groundwater pollution. To evaluate contaminant load, urban areas need to characterised and subdivided according to their predominant human activity and wastewater disposal arrangements. Thus, for residential areas with unsewered sanitation, the pollutant load can be estimated as described in chapter 3 of this review, using the population density, non-consumptive water use, natural infiltration and composition of contaminants, chloride and nitrogen, in the wastewater. For sewered urban and industrial areas, detailed inventories of industrial activities will be needed. From the types of industry, the likely pollutants can be obtained, using the literature referred to and the material in chapter 3 of this review as a guide. To characterise fully the contaminant load, information is rquired on the quantity and quality of all industrial effluents which are disposed to or will reach the subsurface. This may be very difficult obtaini quantities may be reasonably estimated in many cases from water usage but monitoring data may be quite inadequate for characterising effluent quality. The types of pollutants, if not quantities, may be provided from the published literature.

Groundwater quality sampling to assess the impact of urbanisation also faces problems in determinand selection. If background quality is good, then electrical conductivity, chloride, sulphate, nitrate and bicarbonate may all provide a general indication of the impact of urban areas on the quality of the underlying groundwater. The chloride/nitrate ratio may provide a simple indication of urban impact.

Where the source of contaminants is likely to be leaking sewers or direct disposal to the ground of industrial effluents, the type of industries provide indication of which minor ions, trace elements and metals or organic compounds would be expected. This information can be obtained from the references cited below and in chapter 3.

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4. AGRICULTURE AND GROUNDWATER QUALITY

4.1 Introduction and scope

Intensification of agriculture, often by means of irrigation, can produce groundwater quality problems, particularly with respect to nitrate, salinity and pesticides. Ever-increasing demand for food means that intensification of cultivation will continue, and difficulty and conflicts will arise in maintaining a reasonable balance of interests between the need for increased crop production and the requirement to protect the quality of groundwater resources. This chapter reviews current knowledge of the impact of intensive agriculture on groundwater in unconsolidated sedimentary aquifers, summarises methods used to investigate the problem, and discusses some of the approaches which are being or could be employed to protect or restore groundwater quality. For convenience and ease of reference, the three principal groundwater quality issues of nutrients, pesticides and salinity are dealt with in turn. It is, however, recognised that they may occur together and may need to be studied and managed together within an integrated programme of investigations and control measures.

4.2 Background

In most regions of the world alluvial soils are the most productive and intensively cultivated. There are several reasons for this. First, they form extensive areas of fertile and easily worked soils. Second, sources of water for irrigation often exist close by, either as major rivers or as groundwater. Third, these areas are usually densely populated and provide ready markets for the food produced. Thus, some of the most important, extensive and intensively cultivated areas are in the lower flood plains and deltaic areas of large river valleys. These include many of the largest rivers in the world, such as the Nile, Yangtze, Indus, Ganges, Mekong and the Tigris/Euphrates. However, as the map accompanying the introduction graphically illustrates, smaller areas are locally important throughout the world. The underlying alluvial deposits contain considerable volumes of groundwater, often at relatively shallow depth, which are widely used for both potable and irrigation water supplies.

Environmental problems associated with the intensification of agriculture have been apparent in the northern hemisphere for some 20 years. The link between a combination of expanding cultivated areas and increased fertiliser use and nitrate concentrations in groundwater has been extensively researched in both Europe and North America (Foster et al, 1986; Hallberg and Keeney, 1993). Expensive measures to control nitrate pollution from agricultural sources are being implemented where nitrate concentrations in groundwater exceed or are approaching the EC or US EPA allowable concentrations of 11.3 and 10 mg NO_3-N/l respectively. Similarly, the link between nitrogen and phosphorus inputs from agriculture and eutrophication of freshwater bodies has been demonstrated. More recently, pesticides originating from intensive cultivation have been identified in groundwater used for potable supply in Europe and North America (Leistra and Boetsen, 1989; Cohen et al, 1986; Rao and Alley, 1993). While much of the work in temperate zones relates to cultivation without irrigation, nevertheless the same principles can be expected to apply in tropical regions and many of the same methods and approaches to investigating and alleviating the problems can be used.

Concern has been expressed about the likely impact on groundwater quality in tropical regions of the intensification of agriculture and the associated increases in fertiliser and pesticide use (Handa, 1983; 1987). In some parts of Latin America, the Middle East and Asian regions, rates of fertiliser application are high and up to three crops a year can be raised by intensive, irrigated cultivation. It is clearly important that the risks to groundwater quality posed by intensive agriculture should be assessed, so any control measures can be introduced.

In many rural areas in developing countries, potable water supplies are obtained from shallow groundwater. This is because groundwater is usually the cheapest and often the only source of water and has excellent natural quality. These shallow aquifers are however vulnerable to contamination derived from the land surface and soil and accordingly intensive agriculture represents a potentially serious and widespread pollution source. Rural groundwater supplies are not normally monitored and, moreover, even if the shallow groundwater was shown to be seriously polluted, treatment of such a large number of low yielding wells is not economically feasible. Further, many cities located on alluvial deposits obtain their water supply from wellfields outside the urban limits but within intensively cultivated areas. These sources of water supply may in the long term be vulnerable to agrochemicals leached from the cropped soils. Protection of these valuable resources must be a priority.

In many parts of the drier tropics much of the crop moisture requirement is provided by irrigation, and this too can have a profound impact on groundwater quality by increasing nutrient leaching and by the development of salinity problems. Salinisation of soil and groundwater is not a recent phenomenon, but an age-old problem which affected ancient civilisations. Six thousand years ago, on the Tigris-Euphrates floodplain of Mesopotamia, irrigation caused a build-up of salts in soil and water that inhibited food production and contributed to the decline of the Sumerian culture. The scope and origins of the problem are relatively well documented, but measures to prevent salinisation occurring and to restore affected soils and groundwater are difficult and costly. The large alluvial aquifers in some of the deltaic regions referred to above are among the most important locations of large-scale irrigation development in the world and, not surprisingly, also have severe problems of salinity of soils and groundwater. Even where the source of the irrigation water is surface water of very low salinity from rivers such as the Nile and Indus, the alluvial aquifers underlying the cultivated areas may be seriously affected by increasing salinity.

Increased food production during the past 30 years in the Latin America and Asia regions has been impressive, and has kept up with or even outstripped, population growth (Figure 4.1). A key issue for these regions is whether this situation can continue and, if so, how? In Latin America and Asia the future increased demand for food as the population rises cannot be met by an increase in the cropped area as additional land suitable for cultivation is simply not available (Figure 4.2). Indeed the cultivated area is more likely to be reduced as a result of both land degradation and pressure from urban expansion and road building. Further, the area under irrigation is unlikely to be increased significantly, at least in the near future, partly because water

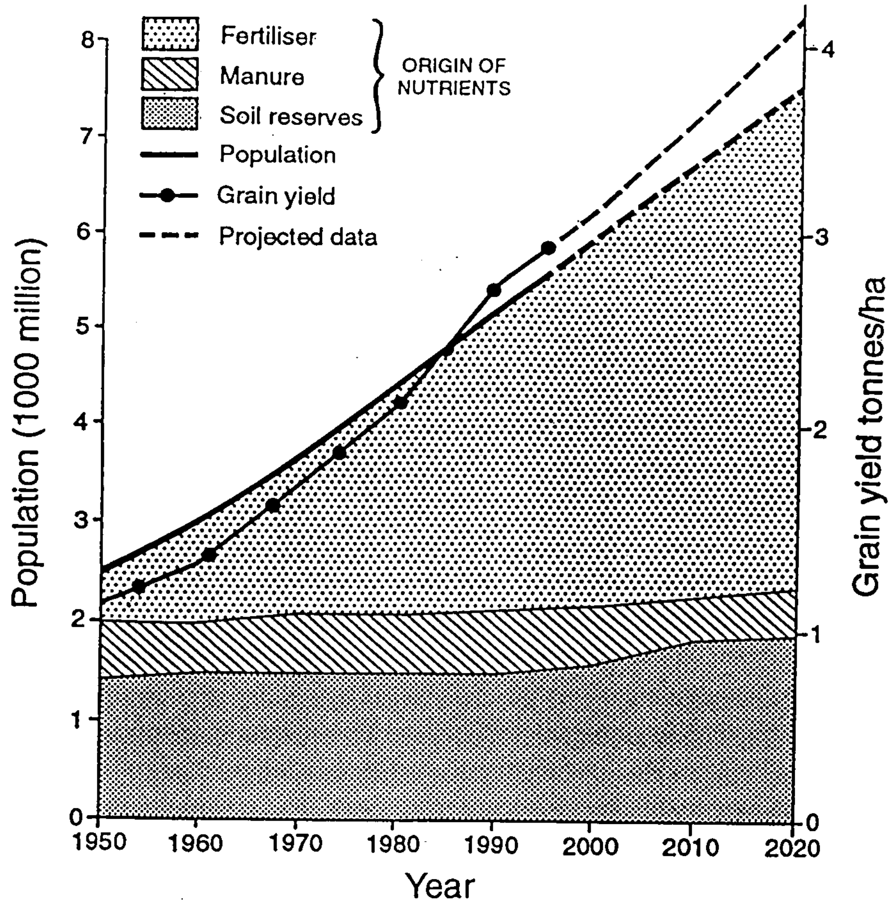


Figure 4.1 Trends in population growth, grain yield and origin of plant nutrients (ESCAP data)

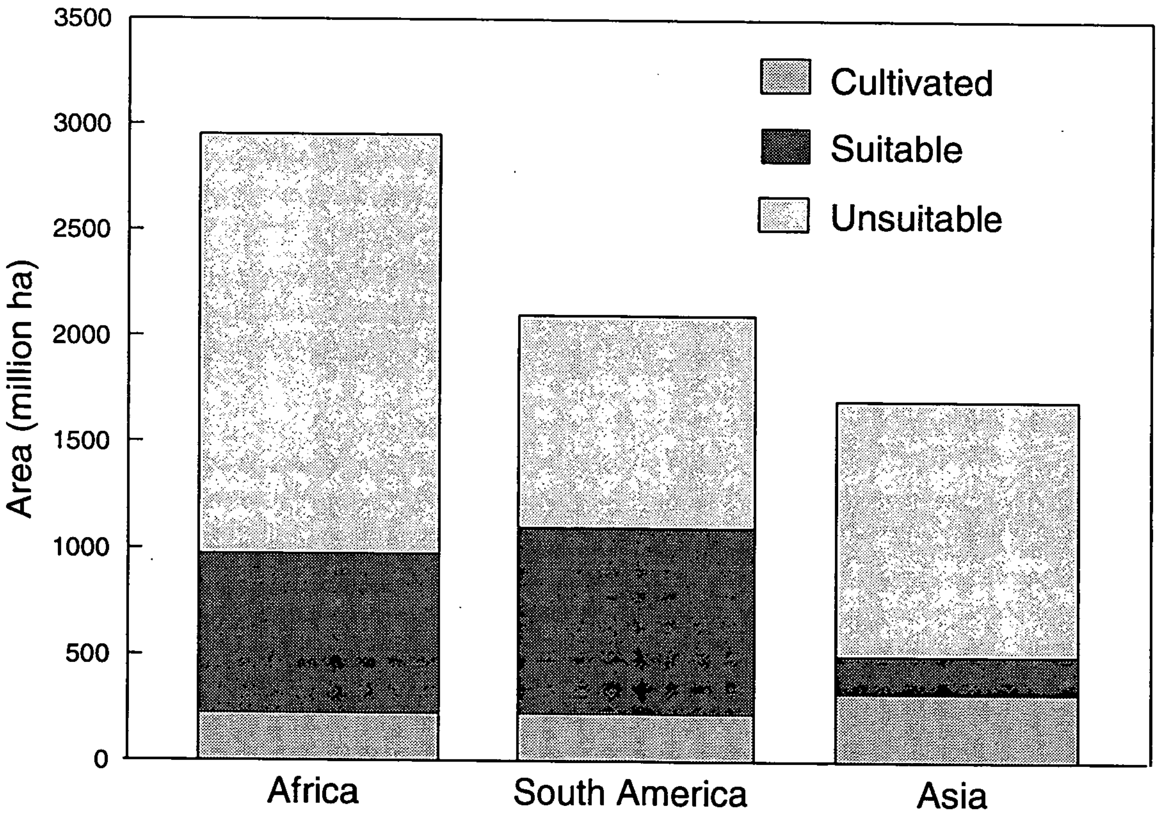


Figure 4.2 Landuse potential in Africa, South America and Asia (source FAO, 1990)

resources are often not available, and are competing with greater demands from potable supplies and industry, and partly because the costs of large scale irrigation are too high. Greater food production can only be achieved by increasing crop yields by a combination of more intensive cultivation, better crop-water management and/or improved cultivation techniques.

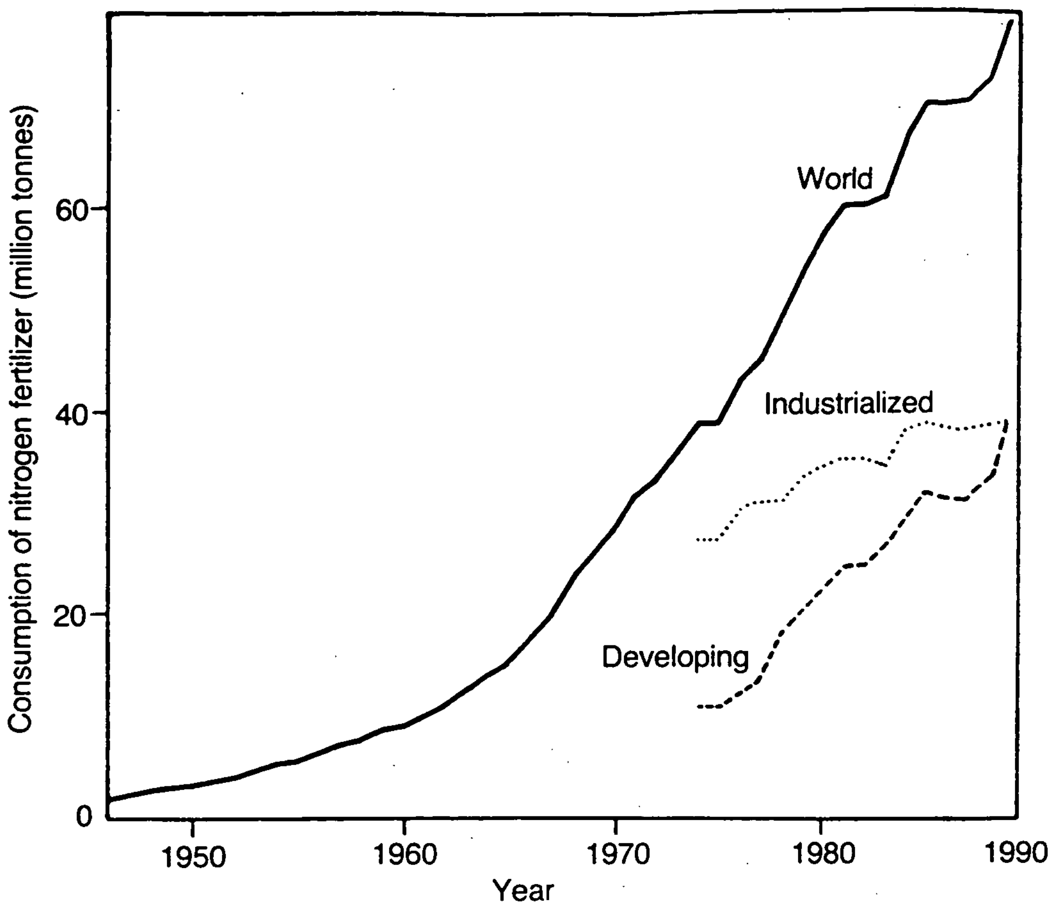
In tropical regions, where intensive cultivation is being carried out over shallow aquifers significant pollution of groundwater can be expected if soils are thin, well-drained and permeable. The results of the limited monitoring of groundwater nitrate concentrations in such areas (Handa, 1987) confirm that considerable leaching losses from the applied fertiliser can occur, which are an important economic loss to farmers. In most countries, however, there is inadequate monitoring of groundwater quality to establish the extent of nitrate pollution from agriculture. Analysis of nitrate in water is not costly; relatively simple methods can be performed on small sample volumes and, as national monitoring programmes become developed, further evidence of nitrate pollution of groundwater from intensive cultivation can be anticipated. There is a similar lack of information about the extent of pesticide pollution in tropical regions but, as suggested in Section 4.4 below, monitoring for pesticides in groundwater is much more problematic, and the lack of data will be much more difficult to remedy.

4.3 Nutrients

4.3.1 *Fertiliser usage*

The dramatic increase in fertiliser use since the 1940s occurred first in the farming areas of Western Europe and North America (Figure 4.3). In developing countries, average fertiliser applications remain low. The annual application of nitrogen fertilisers to arable crops is currently about 30 kg N/ha for Asia, 15 kg N/ha for Latin America and only 4 kg N/ha in Africa, and much of the cultivated land receives no inorganic fertiliser at all (Conway and Pretty, 1991). However, developing countries have shown the highest rates of increase in recent years (Figure 4.3); annual consumption of nitrogen fertiliser tripling since 1975 (Conway and Pretty, 1991) in response to the requirements for ever-increasing food production. Much of this increased intensification has been in areas underlain by alluvial soils and aquifers.

Recommended fertiliser applications for new, improved varieties of the staple food crops are typically in the range of 120-175 kg N/ha/a. National nitrogen fertiliser applications (Table 4.1),averaged over the total area of arable land thus mask much higher applications in areas of intensive cultivation. The increase in fertiliser usage has had enormous benefits in food production, although it is difficult to separate the relative impact on productivity of fertiliser use and other inputs. Studies have suggested that a quarter of the growth in output of rice production in Asia can be attributed to increased fertiliser use, and the remainder to new varieties, irrigation and other capital investment (Conway and Pretty, 1991). The most intensive cultivation in developing countries, with heavy use of inorganic fertilisers, can now produce cereal yields comparable to those of the developed countries. For staple food crops in tropical regions, the highest fertiliser applications are for rice, sugarcane and vegetables.



Figures 4.3 Consumption of nitrogen fertilisers, 1946-1989 (after Conway and Pretty, 1991)

Table 4.1 National average nitrogen fertiliser applications to arable land in selected countries in 1986

Country	Amount (kg N/ha/a)	Country	Amount (kg N/ha/a)
Netherlands	557	USA	75
Martinique	315	Indonesia	64
Egypt	260	Mexico	54
Belgium	247	Sri Lanka	53
UK	238	Colombia	40
Germany	212	India	39
Japan	145	Zimbabwe	30
China	138	Kenya	27
France	135	Chile	20
Cuba	99	Nigeria	6

(Summarised from Conway and Pretty, 1991)

However, there is some concern that this intensive agriculture, particularly in Asia, is not sustainable. Nitrogen fertilisers dominate at the expense of phosphate and potash. Trace nutrients are absent from many proprietary brands of fertiliser and deficiencies in compounds such as sulphur and zinc have been reported. The structure and fertility of the soil is being seriously affected by the use of inorganic nitrogen fertilisers with inadequate inputs of other nutrients and there is evidence that crop yields may decline. The response is often to apply even more nitrogen to the impoverished soils, producing greater risk of leaching and groundwater pollution.

A more balanced approach to fertiliser usage is recommended, with a greater proportion of the nitrogen being obtained from organic sources and from wider usage of K_2O, P_2O_5 and micronutrients.

4.3.2 *Sources, behaviour and fate of nitrogen in the subsurface*

Nitrogen fertilisers applied to the soil are subject to various complex processes. These include volatile losses, incorporation into the soil organic pool, uptake by crops, runoff and leaching below the soil (Madison and Brunett, 1985). Agricultural soils contain large quantities of nitrogen in an organic form. This nitrogen can be mineralised to soluble nitrate which can then be leached below the root zone. Thus, while the nitrate leached in a particular year does not necessarily come directly from the applied nitrogen, nevertheless the overall rate of mineralisation and leaching is related to fertiliser application rates.

The leaching of nitrate from agricultural soils is dependent on a complex interaction of soil and aquifer types, cropping regime and infiltration and is difficult to predict. Although it is convenient to express leaching losses as proportions of the applied mass of nitrogen, they do, however, come from the soil nitrogen pool as a whole. Only a very small proportion of the loss will be derived directly from nitrogen applied in any given year. Some leaching from the soil will occur when no nitrogen is applied and/or the land is fallow. In some semi-arid and arid climates the amount of nitrogen leached from beneath natural vegetation can be significant.

Whilst high nitrate concentrations in groundwater have been widely reported and the leaching of fertiliser nitrogen has in many cases been suggested as the probable cause (Handa, 1983; 1987), it is important to recognise that other sources of nitrate exist. These include geological sources, principally evaporite deposits, palaeo-recharge in semi-arid areas, where nitrate derived from pre-existing vegetation types can produce elevated nitrate concentrations in groundwater receiving little or no modern recharge, irrigation with wastewater, manure heaps, unsewered sanitation and atmospheric deposition. Contamination of groundwater by on-site sanitation systems is known to be especially widespread.

Thus in areas where intensive agriculture and unsewered sanitation occur together, determining the relative contributions of each of these to the eventual nitrate concentrations in groundwater is not easy (Gunasekaram, 1983). By itself, routine monitoring of nitrate concentrations in groundwater cannot be used to assess the impact of agriculture on groundwater quality. Evaluating the impact on groundwater needs to separate and quantify the two sources; approaches to this are described by Komor and Anderson (1993) and Böhlke and Denver (1995).

4.3.3 *Estimating nitrate concentrations*

Approximate and simplified estimates of the concentration of nitrate (or other soluble, mobile contaminants, for example chloride) in the groundwater recharge beneath intensively cultivated soils can be obtained from Figure 4.4. This requires a knowledge of the typical leaching losses from the soil and the quantity of recharge. The relationship is expressed by (Foster and Hirata, 1988):

$$C_F = \frac{F_f}{100I}$$

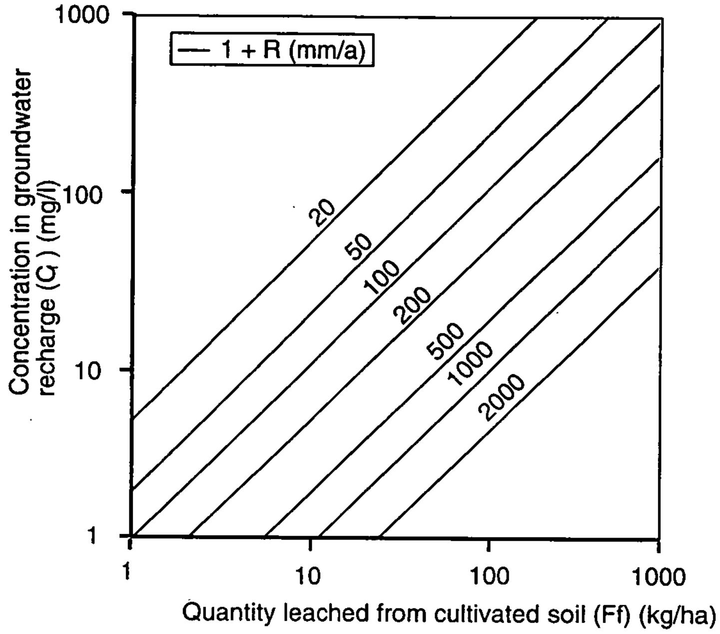
where C_F (mg/l) is the solute concentration
I (mm/a) is the infiltration resulting from excess rainfall and over-irrigation
F_f (kg/ha/a) is the rate of leaching loss

If different crops are mixed in the local area of interest, then a weighted mean may be used to provide an average leaching loss. The estimation of F_f causes significant problems as it is dependent on a number of factors including soil permeability and thickness, crop type, irrigation efficiency, type of fertilisers and the frequency and intensity of fertiliser application.

Agricultural production that involves continuous, albeit partial, crop cover of the soil (such as citrus, sugarcane and coffee plantations) is likely to result proportionally in much lower leaching losses than crop cultivation which involves regular fallow periods with subsequent disturbances of the soil. This is a consequence of the more continuous demand for nutrients and the lower soil oxygen status, tending to inhibit nitrification under continuous crop cover.

4.3.4 *Measurement of nitrate leaching*

As so many factors affect the leaching of nitrate from agricultural soils, generalised figures cannot be used and local, site-specific figures are required. Several methods have been developed for the measurement of nitrate leaching beneath soils. These include lysimeters and porous pot suction samplers (Addiscott et al, 1991), and core sampling in the unsaturated zone beneath the soil (Foster et al, 1986). Where direct measurements are difficult, numerical models have been used to simulate the nitrate leaching from agricultural soils (Addiscott et al, 1991). Such models may allow extrapolation to other soil types, cropping regimes and climatic conditions, and may permit the impacts of changing land-use practises, including pollution control measures, to be simulated.



-	soil permeability	+
+	soil thickness	-
-	excess rainfall	+
+	irrigation efficiency	-
+	continuity cultivation	-
-	grazing intensity	+
+	control of applications	-

0 0.20 0.40 0.60 0.80

Leaching index

Figure 4.4 Estimation of potential contaminant load in groundwater recharge beneath cultivated land (after Foster and Hirata, 1988)

A case study in a shallow coastal sand aquifer in Sri Lanka (Box 4.1) illustrates the use of several of these methods. The study showed that leaching losses equivalent to 12-25% of the applied nitrogen fertiliser are possible producing groundwater nitrate concentrations typically in the range 20-50 mg NO_3-N/l (Mubarak et al, 1993). Thus, where heavy applications of nitrogen fertilisers are made to permeable soils directly overlying shallow aquifers high leaching losses producing significant groundwater nitrate concentrations can be anticipated. Figure 4.5 shows the envelope of nitrate concentrations from dugwells in the shallow sand aquifer.

The precise nitrate concentration observed in groundwater supply boreholes will depend on dilution within the aquifer and the design of the boreholes. In the Lower Yaqui Valley area of Sonora, Mexico, groundwater generally has nitrate concentrations of less than 5 mg NO_3-N/l. This reflects the moderate nitrogen applications of 120-220 kg N/ha. Further, the alluvial aquifer in the area is thick, with a considerable volumes of groundwater storage. Most irrigation and water supply boreholes are screened at depths of 50 to 120 m. Thus, there is significant potential for dilution within the aquifer to reduce nitrate concentrations, and there is a considerable time lag between solutes leaching from the soil and arriving at the screened sections of the boreholes. Significant increases in nitrate concentrations may not be observed in such boreholes until many years or even decades later, even though, as in Sonora, there are indications of elevated nitrate concentrations in the upper part of the aquifer.

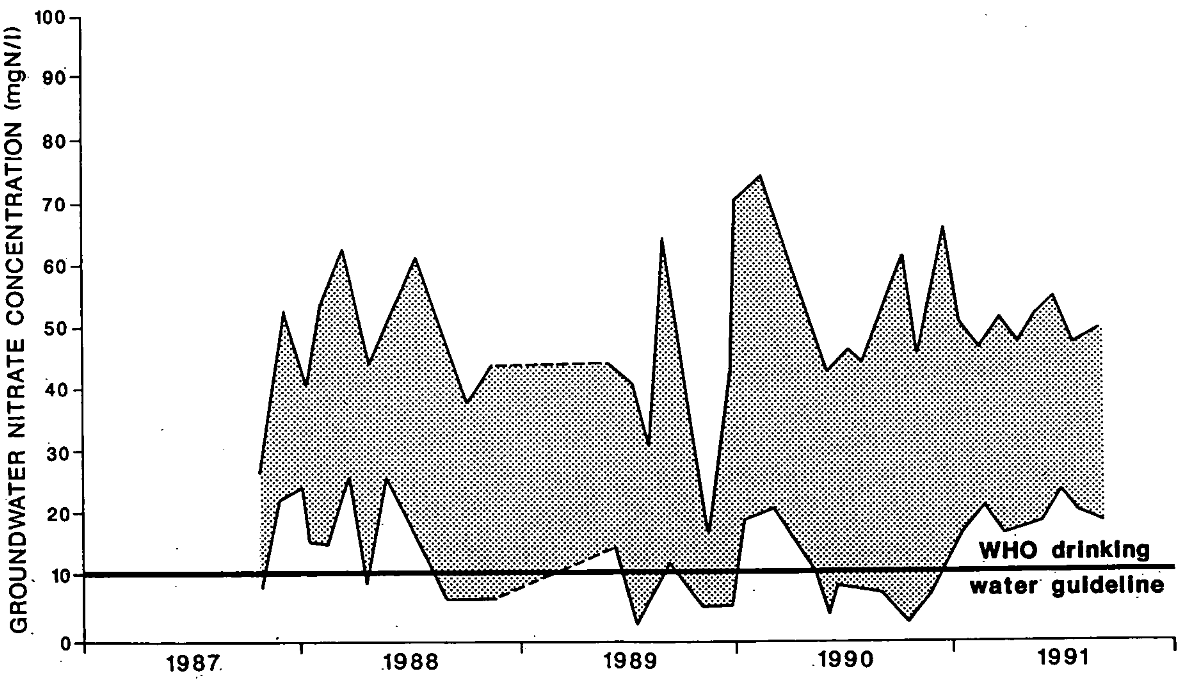


Figure 4.5 Nitrate concentrations in dug wells, Kalpitya Peninsula, Sri Lanka (Mubarak et al, 1992)

Box 4.1 **Nitrogen leaching from intensively cultivated permeable soils in Sri Lanka**

On the Kalpitiya Peninsula in the north-west coast of Sri Lanka, intensive horticulture is being carried out on permeable, well-drained sandy soils overlying a shallow sand aquifer. This type of cultivation has been progressively introduced, during the past 20-30 years, into an area where coconut plantations, with low nutrient inputs, were the traditional crop. Double and triple cropping of onion and chillies, with heavy applications of nitrogen fertilisers, is producing significant losses of nitrogen and high nitrate concentrations in groundwater (20-50 mg N/l) the only source of drinking water.

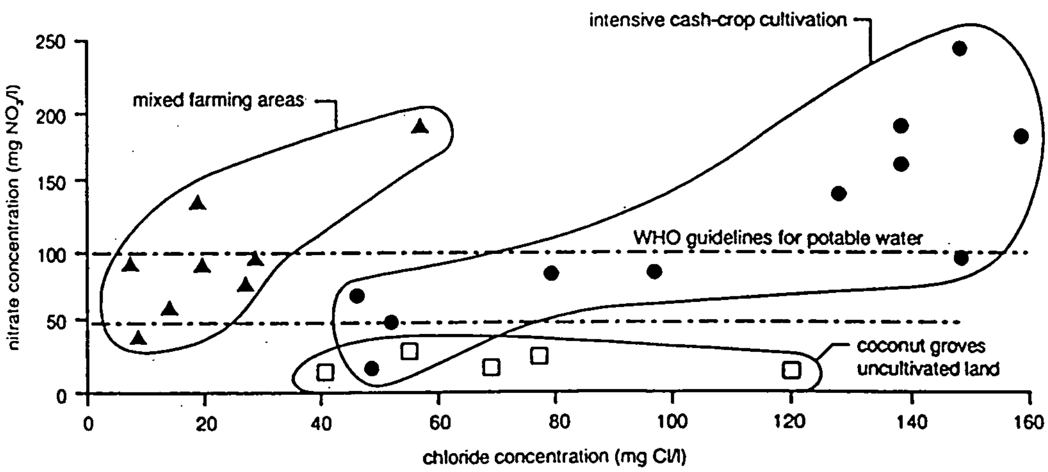
A good correlation was observed between land use and nitrate concentration (in groundwater) with land use is maintained because abstraction from the irrigation wells restricts groundwater flow to localised 'cells' and prevents mixing and dilution with groundwater from non-cultivated areas.

Monitoring of deep percolation, in lysimeters, from beneath an intensively cultivated field plot at the Agricultural Research Station, demonstrated nitrogen leaching losses (excluding nitrogen recycled in the irrigation water or about 180 kg N/ha/crop, or equivalent to about 70% of the applied nitrogen. However, losses in the farmer's fields based on the build-up of nitrate over a 20-30 year period suggest much lower average leaching losses (30-60 kg N/ha/crop of 12-25% of the applied nitrogen). These substantially lower rates were attributed to (a) lower nitrogen applications and deeper rooted crops when cultivation first started, and (b) greater volatile loss from ammonium fertilisers compared to urea, as used at the Research Station.

Nevertheless, the fertiliser leaching losses are clearly very consideration and represent a significant environmental hazard and financial loss to the farmer.

Reference

Mubarak and others 1992. Impact of agriculture on groundwater quality: Kalpitiya Peninsula, Sri Lanka. Final Report. British Geological Survey Technical Report WD/92/49.



Correlation between land-use and groundwater nitrate concentration

Rice is a major crop of many tropical flood-plain areas, particularly in south-east Asia, and often accounts for 20-40% of the cultivated land. Flooded paddy fields are likely to be a major source of groundwater recharge to the underlying UNSAs, and the quality of this infiltration has an important influence on the quality of the groundwater in these aquifers. Research on leaching losses beneath paddy cultivated soils suggests that nitrogen leaching losses are generally low (Box 4.2), even when nitrogen applications approach 300 kg N/ha/a (Krishnappa and Shinde, 1980; Krishnasamy et al, 1993). Denitrification within the anaerobic soils and volatile loss from the surface of the paddy field are thought to be largely responsible for these low leaching rates. However, where paddy cultivation is practised on relatively permeable soils then leaching losses could possibly be more significant.

4.3.5 *Implications for future groundwater quality*

An attempt has been made to indicate the leaching losses that might be anticipated with various crop types (Figure 4.6), based on the results of recent experience. This diagram is perhaps most useful in illustrating firstly the range in nitrogen application rates for various crops (and the approximate range in leaching losses that might be expected) and secondly the likelihood of increased risk of pollution associated with a change in cropping regime. The rapid development of intensive horticulture to meet expanding urban markets is a clear example which is likely to lead to increased risk of groundwater pollution by nitrate, and perhaps pesticides as well.

A wide variation in nitrate leaching losses from agriculture occurs, resulting from differences in soil and crop types, fertiliser application rates and irrigation practices. High rates of nitrogen leaching from the soil can be anticipated in areas where soils are permeable and aerobic, and high nitrogen applications are made to relatively short duration crops, i.e. wheat, vegetables. Continuous crop cover (coffee, sugarcane) tends to reduce nitrogen leaching losses.

Groundwater is most vulnerable to nitrate leaching where:

- the soil and unsaturated zone are thin and permeable
- several crops a year can be grown and fertiliser inputs are high
- excess irrigation can lead to rapid leaching of nutrients

Where these conditions come together, a rapid build-up in groundwater nitrate concentrations can be expected. In the field area in Sri Lanka described in Box 4.1, intensive cultivation began 10-20 years ago, and already nitrate concentrations exceed 20 mg NO_3-N/l in many of the irrigation wells. The overall volume of storage in the aquifer and hence scope for dilution is limited, and nitrate concentrations may be increasing at about 2 mg NO_3-N/l per year.

Over extensive areas underlain by fertile alluvial soils (i.e. North and East China plains, Indo-Gangetic plain) relatively high nitrogen applications (50-200 kg N/ha/a) are made to intensive wheat-based systems. Shallow groundwater in the alluvium may be at risk of nitrate contamination especially if the trend of increasing intensive cropping continues. Monitoring of the shallow aquifers to evaluate the scale and extent of the problem is recommended.

Box 4.2 Fertiliser leaching losses from paddy cultivated fields near Madras, India

The Araniar Korttalaiyer Basin, located to the north of Madras is typical of many of South India. Some 80% of the area is cultivated, with paddy being the most important crop. Irrigation with both surface water and groundwater is widespread and up to 2-3 crops per year can be grown. Nitrogen application rates are high - about 120 kgN/ha per crop (for paddy), with annual rates in excess of 300 kgN/ha for paddy-groundnut-paddy based systems.

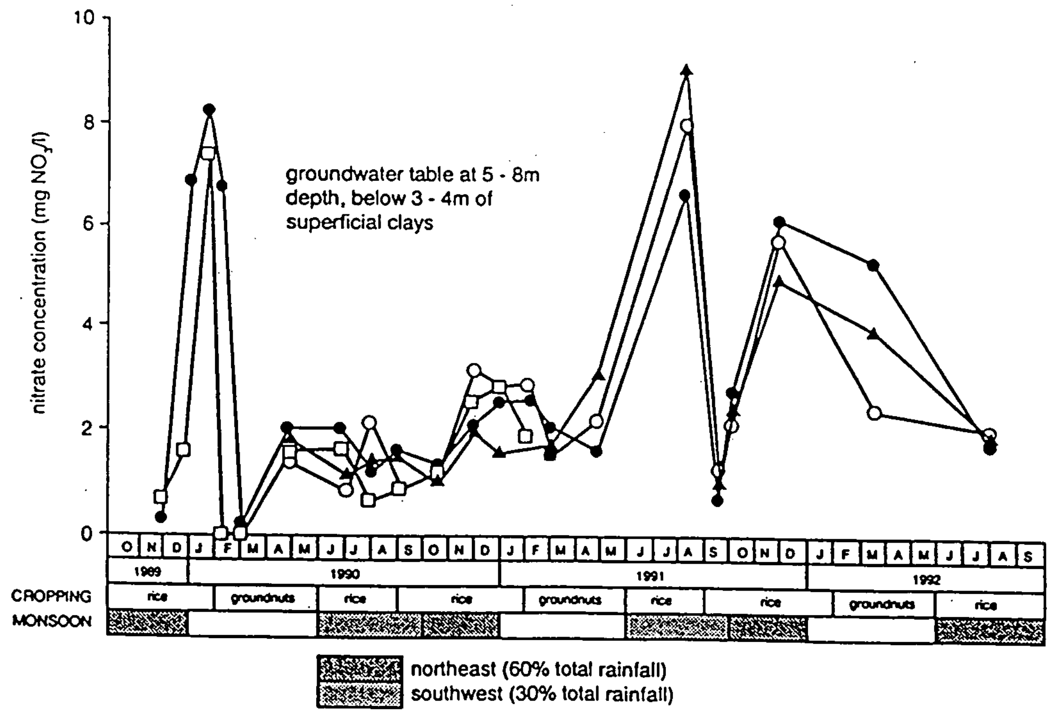
The main aquifer is sand-and-gravel layers within the alluvium, some 15 m below surface, and is overlain by less permeable sands, silts and clays. Recharge to the aquifer occurs largely as infiltration from paddy field, so that water movement in the upper, less permeable part of the aquifer is dominated by vertical seepage.

Monitoring of piezometers drilled to various depths in the upper part of the aquifer, immediately beneath a paddy field, confirmed the vertical head gradient was downwards, and that leakage from the water-table to the main aquifer occurred throughout the year. The rise in water level of the main aquifer during November-January was attributed to the reduction in pumping (for irrigation) during the rainy season, rather than to additional recharge.

Nitrate concentrations in all piezometers were low (1-3 mg N/l) throughout 3 years of monitoring, except for occasional 'highs'. The generally low nitrate concentrations demonstrated that little leaching of nitrogen from paddy soils was occurring. This is in agreement with nitrogen balance studies in paddy soils which indicate plant uptake, volatile losses and denitrification are the predominant processes.

Reference

Krishnasamy and others 1993. Impact of agriculture on groundwater quality in the alluvial aquifers of Madras, India. Final Report. British Geological Survey Technical Report WD/93/18.



Groundwater nitrate concentration in a shallow alluvial aquifer beneath paddy fields

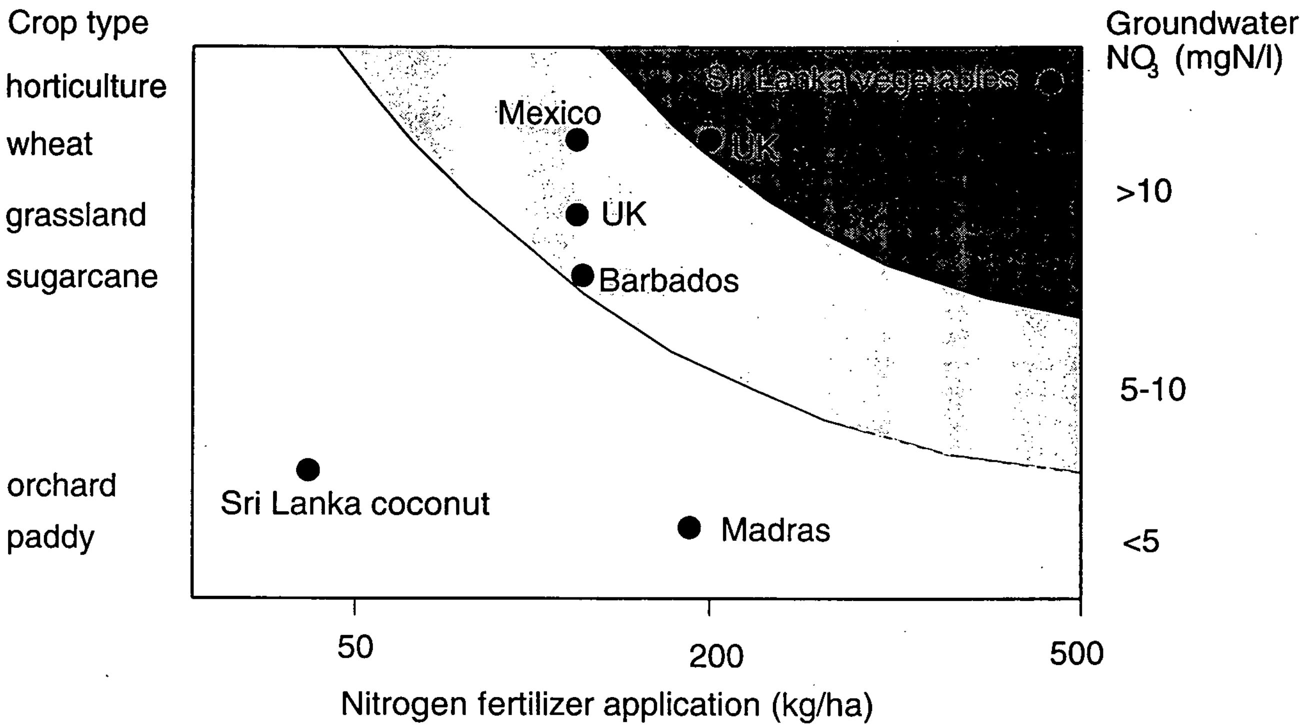


Figure 4.6 Groundwater nitrate concentrations for a range of crop types and nitrogen application rates

Nitrogen leaching losses beneath paddy cultivation are likely to be low as a result of volatile losses and denitrification in the waterlogged, anaerobic soil. This may not be the case where rice is cultivated on more permeable soils with heavy applications of nitrogen fertiliser.

The widespread use of muriate of potash (potassium chloride), as a source of potassium, in fertiliser mixtures in south and east Asia, results in the leaching of chloride to the water table. Unlike the other soluble ions in fertiliser mixtures chloride is neither taken up by the plant nor sorbed/degraded within the soil. Thus apart from any chloride which is removed by surface runoff, all the chloride will be leached to the water table. The resulting groundwater concentrations can be significant; it is estimated that on the Kalpitiya Peninsula of Sri Lanka chloride concentrations have practically tripled over a 30 year period due to leaching from the soil (Box 4.3). As a result, crop yields may begin to reduce. It is recommended that wherever possible an alternative source of potassium is used.

4.3.6 *Control measures*

In UNSAs in which groundwater nitrate concentrations are rising, stabilisation will not occur unless control measures are introduced. The options for control can be conveniently divided into those implemented at the point of abstraction and those designed to control the amount of nitrate entering the aquifer. Measures of both types have been implemented in Europe. The former include:

- blending high- and low-nitrate water from different sources
- closure of sources and development of alternatives
- treatment to remove nitrate
- drilling deeper to draw on low-nitrate groundwater

The last-mentioned, essentially engineering approach is of uncertain effectiveness and the benefits are likely to be short-term. The other measures listed have achieved the desired results, but often at great expense and in supply systems based on limited numbers of major abstraction sources. For technical and economic reasons, it is difficult to envisage their implementation for large numbers of relatively small abstractions, especially in rural areas. In these circumstances, effective control of rising nitrate concentrations means reducing the inputs by cutting nutrient losses from the soil. This would include restrictions on the timing and amounts of fertiliser application and using improved crop strains, cultivation practices and irrigation methods to promote more effective uptake of nutrients. If improved agricultural practices on their own are not sufficient, then the only approach is to contemplate radical changes in land use in designated vulnerable zones around abstraction sources, introducing land uses from which nitrate leaching losses are small. Any such agricultural control measures may have serious economic consequences for the farmers involved, and may in any case be difficult to monitor and enforce.

4.4 Pesticides

4.4.1 *Pesticide usage*

An ever-increasing number of pesticides are used agriculturally. The term 'pesticide' is used here to describe any compounds used as insecticides, herbicides and fungicides. Herbicides dominate pesticide use in temperate climates in Europe and North America, but insecticides are more commonly used elsewhere. The largest individual consumer of pesticides is the USA, followed by the countries of Western Europe. Japan is the most intensive user of pesticides per unit area of cultivated land, but other countries with important unconsolidated aquifers, notably Indonesia, Korea, India and China are also major users. Globally, pesticide use is concentrated on a small number of crops; more than 50% of the total is applied to wheat, maize, cotton and soya bean (Conway and Pretty, 1991). In developing countries, the highest applications are made to plantation crops such as sugarcane, coffee, cocoa, and tea and to rice and cotton, although usage on vegetables is becoming more important. Total consumption of pesticides continues to grow; the annual growth rate averages around 5%, having fallen from 10-12% in the 1960s (Conway and Pretty, 1991).

Box 4.3 Leaching of chloride from potassium fertiliser in Sri Lanka

Potassium chloride (muriate of potash) is widely used in Asia as a potassium fertiliser. Chloride is highly mobile in groundwater since it is not sorbed or degraded.

Potassium chloride is applied to onions and chillies at the research site in Sri Lanka at the rate of 150 kg/ha/crop. In addition the irrigation water also contains chloride which has accumulated from previous applications. It is difficult to estimate long-term groundwater quality trends from a few years data since there is considerable seasonal fluctuation. Chloride concentrations in those areas which have been cultivated for the longest period of time(20-30 years) are usually in the range 150-300 mg/l and compare with chloride concentrations of 50-100 mg/l in the non-cultivated areas.

It is anticipated that within 20 years, chloride concentrations in many irrigation wells will exceed 300 mg/l and be in excess of 400 mg/l in some areas for at least part of the year.

These high chloride concentrations could almost certainly adversely affect the growth of some crops. Excess salinity within the plant rootzone has a generally deleterious effect on plant growth, because it increases the energy that must be expended for the plant to acquire water. The salt tolerance is expressed in terms both of the threshold salinity in the soil root zone at which growth suppression is initiated and relative crop yield above the threshold. Many common horticultural crops, for example onions, are not particularly salt tolerant.

Chloride tolerance of selected agricultural crops

Crop	Threshold (mg/l)	Decrease in yield (% per mg/l above threshold)
Bean	350	0.050
Onion	350	0.046
Pepper	525	0.040
Potato	525	0.034
Alfalfa	700	0.020
Rice	1050	0.034
Wheat	2100	0.020
Sorghum	2450	0.046
Cotton	2625	0.010

Source FAO Irrigation and Drainage Paper 48

Developing countries together use only a small proportion of the total, but the rate of increase in usage is now more rapid in many developing countries than in the developed world. The most commonly used pesticides in the Asian region are listed in Table 4.2.

As for fertilisers, pesticide use has produced major benefits for agriculture. By reducing pest and disease attacks and weed competition, they have contributed significantly to improved crop yields and to the reduction in variable costs of production, including labour. For individual crops, the direct gains from pesticide use are often readily apparent, but it is much more difficult to arrive at estimates of national or regional benefit. Losses to pests in any tropical crops are put in the range 10-40% (Conway and Pretty, 1991), although it is difficult to separate losses in the field from losses during storage and distribution post-harvest. As for fertiliser use, it is difficult to isolate the benefits of pesticide use from other agronomic improvements, and estimates of direct benefits, often used to justify increased agrochemical use, must be treated with caution.

4.4.2 *Occurrence of pesticides in groundwater*

All pesticide compounds pose a significant environmental health hazard since they are designed to be toxic and persistent. Permitted concentrations in drinking water are low, in the range 0.1-100 ppb and dependent upon the individual compound and the regulating authority (Table 4.5). The stringent EC Drinking Water Directive has a maximum allowable concentration of 0.1 μg/l for any pesticide. This guideline is based not on toxicological evidence but on the detection limit of the organo-chlorine insecticides during the 1960s. The WHO guidelines (WHO, 1993) are specific for a few compounds only, whilst the US-EPA has a more comprehensive list, with maximum permitted concentrations based on toxicological data.

The triazines, together with various soil insecticides, especially the carbamates and chloropropanes, have been found in concentrations exceeding 1 μg/l in shallow aquifers in Europe and N America (Leistra and Boetsen, 1989; Cohen et al, 1986; Jones et al, 1987; Rao and Alley, 1993). Most pesticide concentrations observed in these studies have been in the range 0.1 to 100 μg/l, and it seems likely that concentrations significantly above this range can be attributed to local point source contamination close to the well or borehole, rather than conventional agricultural use (Cohen, 1990).

Evidence of significant pesticide occurrences in groundwater have only been available for a relatively short time from Europe and North America and the extent of contamination is far from fully evaluated. In the developing countries hardly any routine monitoring of pesticides in groundwater is currently undertaken.

Table 4.2 Major pesticide compounds in use in Asia (Data from ESCAP)

Compound	Classification	Usage (t/annum)	Area (000 ha)	Use	Persistence	Leaching hazard
Lindane (Gamma HCH)	organo-chloride insecticide	10,000	10,000	used for wide range of insect problems	very persistent	unlikely to be leached
Parathion methyl	organo phosphorus insecticide	6278	10429	non-systemic contact insecticide	non-persistent	unlikely to be leached
Parathion ethyl		3250	6500			
Paraquat	pyridine herbicide	5783	14060	non-selective herbicide - used widely for plantation crops	persistent	unlikely to be leached
Mancozeb	zinc compound	5182	4738	fungicide	uncertain	likely to be leached (very mobile)
Dimethoate	organo-phosphorous insecticide	5166	12,300	broad range of insect problems and wide range of crops	not persistent	likely to be leached (very mobile)
Dichlorovos	organo-phosphorous insecticide	4461	6622	broad range of insect problems	not persistent	likely to be leached
Monocrotophos	organo-phosphorous insecticide	4150	17,100	used for wide range of crops	not persistent	may be leached (moderate - low mobility)
Butachlor	unclassified herbicide	3877	3962	pre-emergent herbicide (esp. rice)	relatively persistent	unlikely to be leached
Malathion	organo-phosphorous insecticide	3824	6411	used on wide range of crops (incl. horticultural animal parasites and malarial control	uncertain	unlikely to be leached

4.4.3 *Pesticide transport and behaviour*

Most groundwater systems are characterised by relatively slow rates of groundwater flow and pollutant transport. The response time of deep water supply boreholes in unconfined aquifers to surface inputs of even mobile pollutants is often measured in years or decades. This has been clearly demonstrated by studies of nitrate pollution from agricultural activities (Foster et al, 1986). This slow response means that the analysis of pesticides from deep water supply boreholes may be an insensitive and tardy indicator of the state of quality deterioration in the groundwater system as a whole. In thick alluvial sequences, water tables may be shallow and groundwater may be vulnerable to pollution. Deterioration in quality at shallow depth may produce significant quality stratification, which is not detected by sampling the discharge from deep boreholes.

Problems and uncertainties in pesticide fate

To evaluate the current situation and to justify any required controls on pesticide use, data are needed on the three-dimensional sub-surface distribution of pesticide compounds beneath aquifer recharge areas, and especially in the unsaturated zone. Three important questions need to be answered:

(a) Which pesticide compounds are most likely to be leached to groundwater?

(b) What are the most probable pathways?

(c) Are pesticide concentrations currently detected in water supply boreholes likely to be approaching equilibrium with current pesticide applications and leaching?

However, the investigation of pesticides in groundwater systems presents substantial problems because (Foster et al, 1991):

(a) A wide range of compounds is in common agricultural use, many of which break down into toxic derivatives. Analytical scanning of water samples for all or many of these would be prohibitively expensive. Monitoring organisations require knowledge of local pesticide usage to select compounds for analysis.

(b) Very sophisticated analytical procedures and relatively large volumes of sample are required as some compounds are highly toxic at very low concentrations which are close to detection limits.

(c) Considerable care in sampling is required to avoid sample modification, contamination or volatile loss.

In addition to these important technical difficulties, significant scientific uncertainty remains in two key areas:

(a) Much of the available information on the physicochemical properties of pesticides originates from the trials which are carried out by manufacturers as part of the registration process. These are invariably performed on "standard, fertile, organic clay soils" from temperate regions. There is a lack of information generally on the properties of pesticides in groundwater and aquifers as opposed to soils, and an even greater scarcity of data pertaining to tropical environments.

(b) There are fundamental questions about the movement of pesticides through aquifers. Movement through an unconsolidated, granular aquifer or the matrix of a consolidated rock aquifer would allow time for attenuation processes to occur, albeit more slowly than in the soil. In fissured aquifers, preferential flow, effectively by-passing the rock matrix, could greatly reduce the opportunity for attenuation to occur, and permit rapid movement of relatively high concentrations of pesticides directly to the water table and thence to wells or boreholes.

Fortunately, in the case of unconsolidated aquifers groundwater flow is dominantly intergranular, and the issue of preferential flow is much less important than for consolidated, fractured aquifers. Uncertainty over pesticide behaviour is, however, important. Pesticide compounds leached from permeable soils into UNSAs enter an environment which contains less clay minerals and organic matter and has a greatly reduced indigenous microbial population. The attenuation processes which affect pesticides, particularly adsorption and degradation, are likely, therefore, to be much less active in the aquifer than in the soil zone, and the mobility and persistence of pesticides may be many times greater in aquifers than in soils (Lawrence and Foster, 1987).

Cavalier et al (1991) carried out studies of the persistence of several pesticides in groundwater. Long lag periods were observed before any degradation occurred, and very long half-lives were measured (Table 4.3). Although the study was a very simple one carried out on groundwater only without aquifer material, and the half-lives may be unrealistically long, they do support the general expectation that persistence is likely to be significantly greater in aquifers than in soils.

Pesticide leaching from the soil

The natural processes which govern the fate and transport of pesticides can be grouped into the following broad categories; sorption, leaching, volatilisation, degradation and plant uptake (Figure 4.7). Plant uptake is usually a small component. The mode of application and action of the pesticide are important factors in relation to soil leaching, since those targeted at plant roots and soil insects are usually much more mobile than those acting on the leaves. Volatile losses occur from the soil particles, from the plants and from soil moisture.

Two key factors determine whether pesticide residues will be leached below the soil zone. First the residues must persist in the soil for sufficient time to allow leaching to occur. Second the residues must be 'mobile', that is they preferentially dissolve and migrate with the soil water rather than sorb onto soil particles.

Table 4.3 Comparison of pesticide persistence in soil and groundwater

Pesticide	Half-life (days)		Ratio
	Soil	Groundwater	
Propanil	3	240	80
2,4-D	10	1200	120
Dichlorprop	10	900	90
Alachlor	20	1300	65
Metolachlor	40	800	20

Summarised from Cavalier et al (1991)

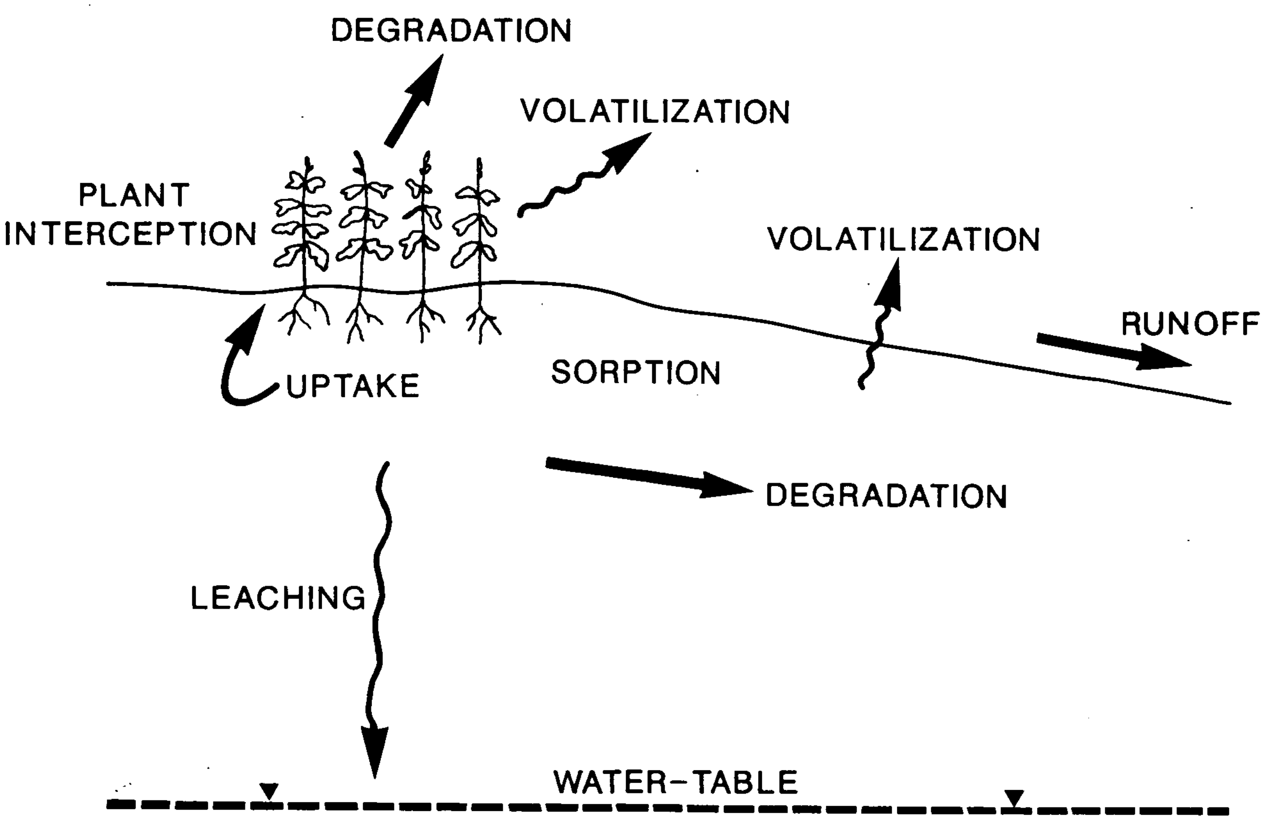


Figure 4.7 Natural processes which govern the fate and transport of pesticides applied to the soil

Pesticide compounds may degrade in the soil by microbial or chemical processes to produce metabolites and ultimately simple compounds such as ammonia and carbon dioxide. Soil half-lives for compounds in widespread use range from 10 days to years, but for the most mobile pesticides are normally less than 100 days. Many herbicides are applied to the soil before the weeds emerge and some insecticides are used for soil treatment. Given the timing of these applications, they are sufficiently persistent to remain in the soil for significant periods, when leaching may occur. Moreover, some derivatives from partial oxidation or hydrolysis may be as toxic and mobile as the parent compound.

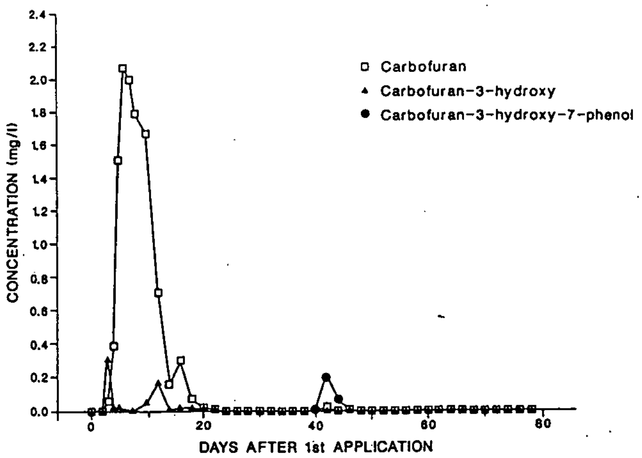
The importance of degradation in the soil in determining the proportion of pesticide residues that can be leached is well illustrated by a study in Sri Lanka where very rapid degradation by soil microbes of the second application of the soil insecticide, carbofuran reduced the amount of residues leached below the soil zone significantly (Figure 4.8). This process of more rapid degradation following the initial application is commonly termed 'enhanced biodegradation'.

Most pesticide compounds have water solubilities in excess of 10 mg/l (10,000 µg/l), and this is not a limiting factor in leaching from soils. The mobility of pesticides in soil solution will vary with affinity for organic matter and/or clay minerals. This is expressed by a partition coefficient, normally that for non-polar adsorption on organic carbon K_{oc}. Pesticides that are strongly sorbed onto organic matter or clay particles are likely to be retained in the soil rather than leaching to groundwater. Thus data on the mobility and persistence of pesticides in soil can be used to indicate which compounds are most likely to pose a risk to groundwater (Table 4.4). A possible exception to this might occur if a strongly adsorbed pesticide became transported in a fissured or very coarse-grained formation, in the sorbed phase on colloidal particles. Chemical reactivity of the compound with the soil matrix may also play an important role in reducing the risk of pesticide leaching, as a result of the generation of less soluble residues through, for example, neutralisation of acidic compounds in alkaline soils.

Knowing the likely degradation pathway is also important when evaluating the risk to groundwater supplies; since metabolites produced may pose a greater or lesser risk than the parent compound. Research undertaken in Sri Lanka and India on the movement and fate of carbofuran in the subsurface, indicated that whilst the parent compound is relatively mobile, the main metabolite, carbofuran phenol, was strongly sorbed. In a field study in Sri Lanka carbofuran phenol was not detected in samples in lysimeters or in shallow piezometers immediately beneath a field plot, despite the presence of the parent compound and minor metabolites (Box 4.4). Laboratory studies, however, confirmed carbofuran phenol as the main metabolite. Likewise near Madras India, monitoring of carbofuran residues in the soil beneath a paddy field confirmed that carbofuran phenol was significantly retarded with respect to the parent compound.

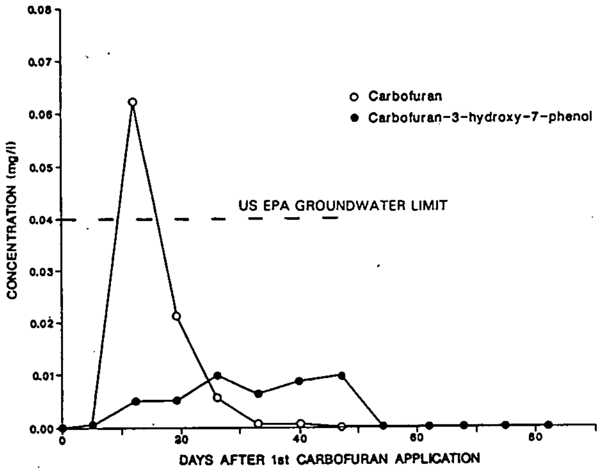
Box 4.4 The importance of degradation when assessing risk of pesticide leaching to groundwater

Research undertaken in Sri Lanka and India on the movement and fate of carbofuran in the subsurface, indicated that whilst the parent compound is relatively mobile, the main metabolite, carbofuran phenol, was strongly sorbed in soil and therefore unlikely to be leached to groundwater. In the field study in Sri Lanka carbofuran phenol was not detected in samples in the lysimeters or in shallow piezometers immediately beneath a field plot despite the presence of the parent compound and minor metabolites. Mass balance studies show that the 'disappearance' of carbofuran cannot be accounted for by the appearance of the minor metabolites.



Carbofuran residue concentrations in lysimeter at about 1 m depth

Laboratory studies confirmed the rapid degradation of 3-10 days for the parent compound. Field and laboratory data are therefore consistent with the rapid degradation of carbofuran, to carbofuran phenol, in the soil-unsaturated zone, with retention of the latter compound to the solid matrix.



Carbofuran residue concentrations in shallow groundwater beneath experimental field plot (Sri Lanka)

Likewise, monitoring of carbofuran residues in the soil beneath a paddy field research site near Madaras, India, confirmed that carbofuran phenol was the main metabolite and that it was retained in the soil layer for more than 80 days. By contrast, the parent compound, carbofuran, migrated rapidly through the soil but had largely disappeared within 15 days due to degradation.

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Krishnaswamy and others 1993 Impact of agriculture in the alluvial aquifers of Madras, India: Final Report. British Geological Survey Technical Report WD/93/18.

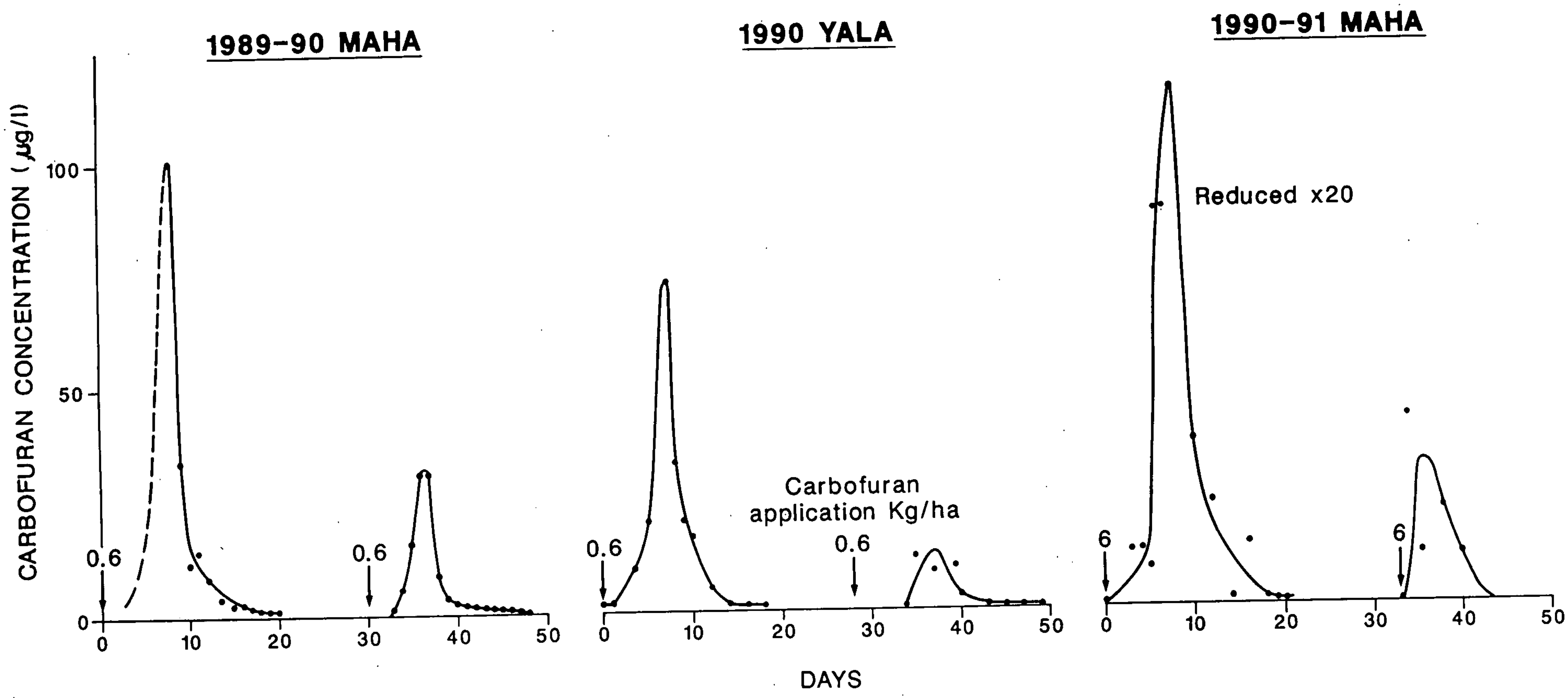


Figure 4.8 Carbofuran in lysimeter drainage over three cropping seasons, showing reduced leaching of second applications is a result of enhanced biodegradation (Mubarak et al, 1992)

Table 4.4 **Susceptibility to leaching of pesticides used in the Hampton Catchment, Barbados (Chilton and others, 1995)**

Active ingredient	Use	Type	Acute oral toxicity[1]	Mobility class[2]	Solubility[3]	Soil half-life[3]	Total application (kg/a)	Average application[4] (kg/ha/a)
Asulam	H	TC	IV (4000+)	?	3	?	7293	2.83
Ametryn	H	T	III (3080)	4	1	4	6570	3.59
Atrazine	H	T	IV (2000)	?	5	4	6334	4.25
Methylarsonic acid	H	-	IV (1800)	?	1	?	6127	3.39
2,4-D Amine	H	Ph	II (700)	?	4	2	5996	3.38
Glyphosate salt	H	-	IV (4000)	2	1	3	4753	2.86
Paraquat	H	Py	II (150)	9	1	2	4542	3.74
Ionoxynil octanoate	H	-	II (110)	?	6	1/3	1651	0.75
2,D-Isooctyl	H	Ph	II (700)	3	6	?	1651	0.75
Diazinon	I	OP	II (300)	5	5	4	6240	15.7
Acephate	I	OP	II (900)	?	1	?	3724	19.7
Benomyl	F	C	V (10000+)	?	6	?	2247	8.11
Chlorothalonil	F	-	V (10000+)	?	6	?	1638	8.71

[1] WHO classification 1988-89 on LD50 (mg/kg) rates but adjusted to include dermal toxicity and other factors
[2] Based on K_{oc} and K_{ow} partition coefficients (1 = most readily leached - 9 = unlikely to be leached (strongly sorbed))
[3] From Pesticide Manual, 8th Edition, 1987
[4] Average application rate for the plantations using the compound

Use
H = herbicide
I = insecticide
F = fungicide.

Type
T = triazine
Ph = phenoxyl acid
OP = organophosphorus
TC = thiocarbamate
C = carbamate.

Solubility classes
1 = >100 g/l
2 = 10-100 g/l
3 = 1-10 g/l
4 = 0.1-1 g/l
5 = 0.01-0.1 g/l
6 = <0.01 g/l

Soil half-life classes
1 = <10 d
2 = 10-30 d
3 = 30-100 d
4 = 100-300 d
5 = >300 d

Mechanisms of transport in the unsaturated zone

Preliminary estimates can be made of the possible transport of pesticides from soils into groundwater systems, based on the physico-chemical properties of the pesticides themselves and on knowledge of groundwater flow and aquifer properties gained from previous investigations. The retardation in transport of chemicals experiencing adsorption with respect to that of a conservative, non-reactive, mobile solute can be estimated from their physicochemical characteristics using the expression:

$$R_t = 1 + \frac{K_D \rho_B}{n}$$

where 'R_t' (the retardation factor) is the actual velocity of water flow divided by the transport velocity of the adsorbed species, and 'ρ_B' and 'n' are the density and porosity of the porous media respectively. 'K_D' (the partition coefficient) is the slope of the linear portion of the isotherm for adsorption onto organic matter, which can be approximately estimated from the corresponding coefficient for organic carbon (K_{oc}). An equation of this form was first applied to groundwater systems by Higgins (1959) and more rigorously proven in the early 1980s (Roberts et al, 1980; McCarty et al, 1981). It strictly applies to one-dimensional saturated flow. When considering unsaturated zone transport the equation can be re-expressed in terms of 'θ' the moisture content, and 'ρ_s' the grain density of the porous matrix such that:

$$R_t = 1 + \frac{K_D(1 - n)\rho_s}{\theta}$$

It is clear that in organic rich aquifers travel times to the water table, for some compounds, are likely to be substantially increased (over conservative mobile and non reactive solutes). Thus, in organic rich aquifers with deep water tables (>15 m) pesticide travel times are likely to be of the order of many years or even decades. These times should be sufficient for most compounds, but not all, to be significantly reduced by degradation. Conversely aquifers of low organic content and shallow depth to water table (<5 m) are vulnerable to contamination.

Saturated zone

Within the subsurface, it is the soil and unsaturated zone of the aquifer which provides the greatest protection to potable water supplies from pesticide contamination. However, some continuing attenuation of pesticide residues can be anticipated below the water table as a result of dilution and continuing degradation processes. The degree of attenuation depends principally on groundwater flow and storage characteristics of the aquifer, the properties of the pesticide and the mineralogical content of the aquifer (Box 4.5). Maximum attenuation is likely in fine-grained, organic-rich unconsolidated sediments in which groundwater velocities are low and aquifer storage relatively high. In such environments, the surface area of aquifer matrix in contact with the flowing groundwater will favour both retention by sorption and biodegradation. Further, the slow groundwater movement will ensure that the reduction in pesticide concentrations will occur within relatively short distances of the cultivated area.

Box 4.5 **Attenuation of pesticides in the saturated zone**

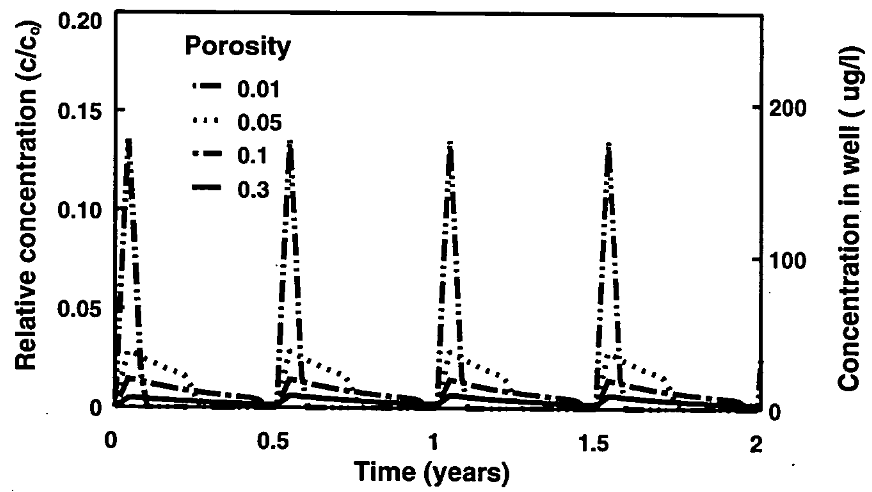
Dilution of pesticide residues by mixing with waters stored within the aquifer can significantly reduce pesticide concentrations but that this dilution can vary by 1-2 orders of magnitude depending upon the porosity of the aquifer and the depth of the groundwater flow paths. At one end of the scale the highly porous sand aquifer at the research site in Sri Lanka, which despite being only about 15 m thick, probably stores 5-8 times as much water as the average annual recharge. The potential for dilution in this aquifer is clearly considerable and even greater dilutions are possible for deep boreholes in thick granular aquifer systems. In contrast the fissured limestone aquifer of Barbados has storage representing only once or twice the annual recharge.

Dilution can be very effective in reducing pesticide residues to acceptable concentrations. Field observation in Sri Lanka demonstrated that carbofuran residue concentrations at the water-table, immediately beneath the cultivated plot, were reduced from 60 $\mu g/l$ to less than 5 $\mu g/l$ within four days.

Simple models can be used to indicate likely pesticide residue response in pumped monitoring wells for various pesticide half-lives and aquifer types (Barker and Lawrence 1993). The purpose of this modelling was to indicate how the shape of the residue concentration-time graphs varied for different aquifer/pesticide scenarios. It is clear that 'pulses' of relatively high pesticide concentrations might be expected in fissured aquifers of low storage. Whilst for the same compound (and the same quantity leached to the watertable), much lower concentrations (possibly below detection limits) over longer time intervals are predicted for porous granular aquifers.

Reference

Lawrence A R and Barker J A 1993. Simple groundwater model to predict pesticide concentration in pumped irrigation wells. British Geological Survey Technical Report WD/92/49.



Variation in pesticide concentration in pumped well with aquifer porosity

The importance of attenuation in the unsaturated zone is clearly shown in the Sri Lanka study. Sampling of groundwater in the sand aquifer beneath and around a cultivated plot showed that residues of alachlor had migrated only 2-3 m beyond the field boundary. A tracer experiment in which the behaviour of two pesticides was compared to that of the non-reactive compound lithium chloride (Figure 4.9) clearly indicates the importance of pesticide properties in saturated zone transport.

4.4.4 *Implications for groundwater quality*

Whilst there is concern that the intensive use of some agricultural pesticides may cause widespread contamination of shallow groundwater which is extensively used for potable supplies, there is little monitoring undertaken to provide evidence for this. The results of limited research in Sri Lanka and India suggest that whilst some compounds may be leached to the water table, dilution and degradation are likely to reduce concentrations at water supply boreholes to below drinking water guidelines for most compounds. Exceptions may occur where the highest risk compounds (mobile, persistent and with low drinking water guidelines) are used on soils overlying the most vulnerable aquifers (i.e. shallow depth to water table, permeable soils/unsaturated zone and limited aquifer thickness). Further the risk to groundwater needs to be balanced by the undoubted benefits pesticide usage has produced in terms of increased food production.

The presence in groundwater of some highly persistent but generally non-mobile organo-chlorine compounds such as DDT and its various isomers suggests that these compounds may migrate to the water table attached to particles of organic matter.

Both scientific investigation and routine monitoring of pesticides in groundwater present significant difficulties, as described in Section 4.4.3 above. It is not practicable for regulatory organisations to monitor for all of the large range of compounds in common agricultural use. This type of screening is prohibitively costly where potable supplies are drawn from a few high-discharge sources, and quite out of the question for large numbers of small, rural supplies. Choices have to be made, based on usage data, to focus on the high-risk compounds which are mobile and persistent, as shown in Table 4.4 for the Hampton catchment in Barbados.

Persistence of pesticide compounds in groundwater is largely unknown and more information on this aspect is required. The preliminary results of studies of pesticide degradation in aquifers suggest that it could be significantly slower than in soils, but much more information is required for both soils and aquifers in tropical environments. Degradation rates can vary within an aquifer for different pesticide compounds, and the observation of different metabolites of carbofuran in the aerobic field conditions of Sri Lanka and the anaerobic paddy cultivation in India indicate that degradation pathways can be dependent on environmental conditions.

Reported concentrations of pesticides in groundwater and the results of field and modelling studies referred to above are all in the range 0.1 to 100 µg/l, in the same general range as the guideline or allowable concentrations for most compounds.. The risks of pesticide concentrations in abstracted groundwater reaching 10 or 100 times these values as a result of normal agricultural use at recommended application rates are probably small. If higher concentrations are observed, they are likely to result from

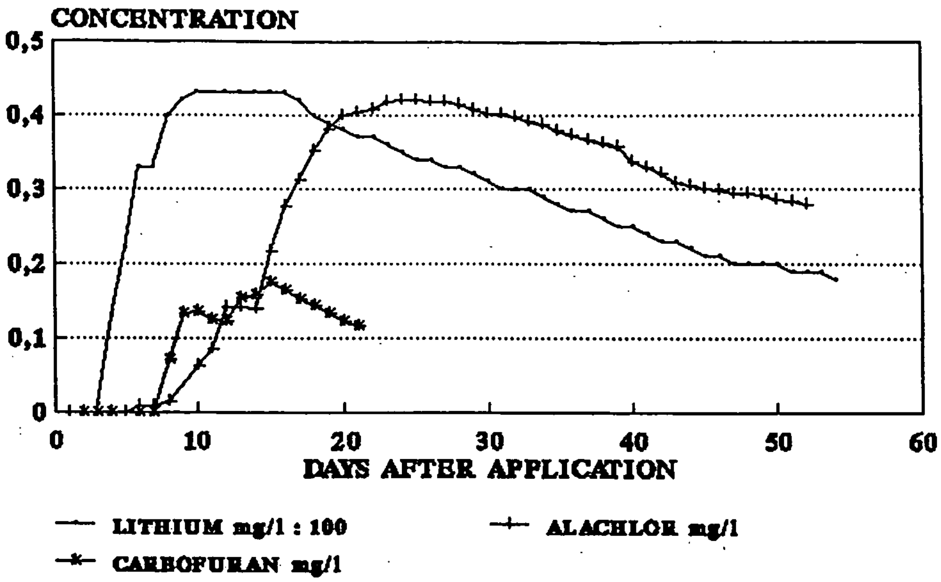


Figure 4.9 Relative retardation of the pesticides carbofuran and alachlor with respect to a conservative tracer (lithium)

point rather than diffuse sources, from such as spillages, poor disposal, soakaways and non-agricultural uses.

Modelling has been widely used to predict pesticide concentrations and to explore the sensitivity of pesticide residues in groundwater to aquifer conditions and pesticide properties. The soil and unsaturated zone provide significant scope for attenuation of pesticide residues. Both the field observations in the Sri Lanka study and the application of the simple model described in Box 4.5 indicate that pesticide residues may not be transported far and at high concentrations. It is possible that relatively small zones of restricted pesticide use around groundwater supplies may provide an adequate protection against the risk of high concentrations, especially in fine-grained unconsolidated sediments with slow intergranular groundwater flow.

4.5 Salinity

4.5.1 *Distribution and scale of the problem*

Agriculture, mainly irrigation, accounts for about three-quarters of global water use (L'vovich, 1979). Eighteen per cent of the world's cropland is irrigated, and this produces 30 per cent of the world's food. Thus, of a total commandable area of about 270 million hectares (WRI, 1987; FAO, 1990) some 235 million hectares are currently cropped (Figure 4.10), the balance being temporarily fallow or out of production for reclamation or other reasons. Of the total, 80% lies in arid and semi-arid tropical zones, mostly in Asia, and about 75% of the total is in developing countries. An even greater proportion lies on UNSAs. From 1950 to 1970 the gross irrigated area doubled, and continues to grow (Table 4.5), albeit at a somewhat slower rate. This is because a combination of rising costs and falling world prices makes irrigation less attractive economically and because the most suitable land and easily utilised water resources have already been developed.

In many areas especially in the semi-arid and arid regions, expansion of irrigation to improve agricultural output has led to land degradation due to the twin problems of waterlogging and salinity. Accurate estimates of the area affected are lacking but up to half of the world's irrigated land has been damaged to some extent by salinity, alkalinity and waterlogging. By one estimate, 20 to 30 million ha are seriously affected by salinity (FAO, 1990) and by another one-third of the world's irrigated lands are affected by soil salinity (Yadav, 1989). Some not very recent estimates of the worst areas are given in Table 4.6; countries containing some of the most extensive alluvial areas in the world. Thus, although the total irrigated area is still growing, one estimate suggests (Kovda, 1983) that salinisation at a rate of 1 to 1.5 million ha per means that irrigated land is going out of production at 30-50% of the rate at which new land is brought under irrigation.

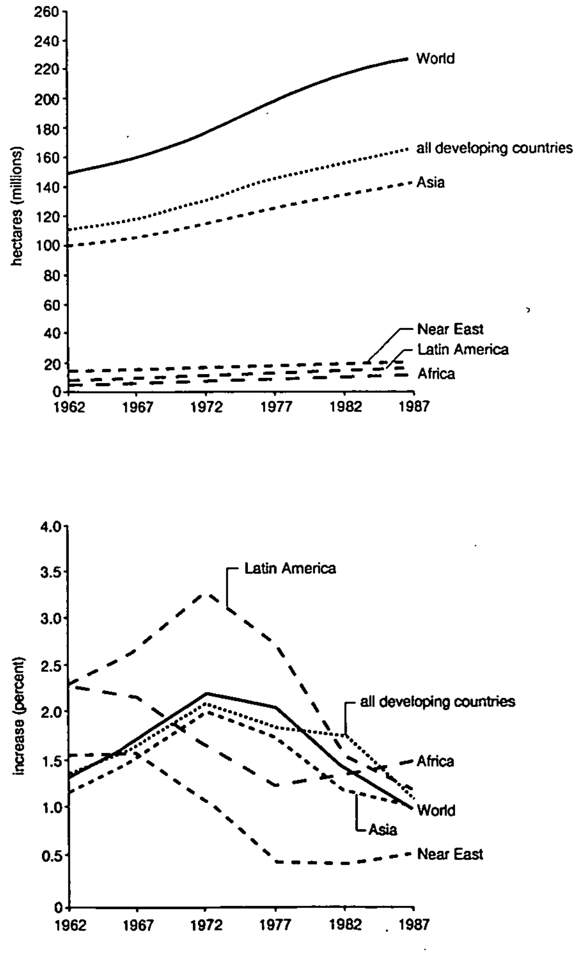


Figure 4.10 Area of irrigated land and percentage rates of increase (after FAO, 1989)

Table 4.5 **Growth of irrigated area by continent, 1950-85**

Region	Total irrigated area (10^6 ha) 1985	% Growth in irrigated area		
		1950-60	1960-70	1970-80
Africa	13	25	80	33
N. America	34	42	71	17
S. America	9	67	20	33
Asia*	184	52	32	34
Europe+	29	50	67	40
Oceania	2	0	100	0
World	271	49	41	32

* Includes the Asian portion of the former Soviet Union
+ Includes the European portion of the former Soviet Union

Source: WRI, 1987

Table 4.5 Areas affected by salinity and waterlogging from irrigation

Country	Total irrigated area		Area affected by salinity		Notes
	(10^6 ha)	(year)	(10^6 ha)	(year)	
China	45	1987	na		
India	42	1987	12	1977	
USA	1801	1987	4		
Former USSR	20.5	1987	na	1985	
Pakistan	16.1	1987	3.2	1987	80% of irrigated land is affected in Punjab
Iran	5.7		1.2	1977	
Iraq	4		0.45	1977	More than 50% of the land in the lower Rafidain Plain is affected
Egypt	2.9		0.8	1970	Mostly in north part of the Nile delta

na not available

Source: various, including FAO (1989) and WRI (1987)

4.5.2 *Processes of salinisation*

Deterioration of soil and water quality resulting from irrigation can usually be linked to poor planning and implementation of projects and subsequent mismanagement. The waterlogging that is a feature of so many irrigated areas often arises because the construction of proper drainage measures was postponed rather than being implemented at the same time as the water distribution system. Waterlogging and salinity are related to low irrigation efficiency; a measure of the proportion of the total water resource that meets the consumptive use of the crops. Thus defined, efficiencies average 30-50% for most schemes. There is excessive input of water into systems which have limited natural drainage capacities to deal with the water (FAO, 1990). Inefficient water management results in enormous water losses from canals, from infiltration and runoff because excessive water is applied to the land, and from evaporation from the irrigated land.

Water losses from canals and irrigated land produce extra infiltration that percolates down to the water table. If the extra infiltration is excessive, the result will be a rise in water levels. The rise in water table may be quite dramatic, reaching a metre or more per year in exceptional cases (Table 4.7). When water levels rise within 1.5 or 2 m of the surface, the rate of rise decreases as evaporation becomes effective, but this also increases salinity. In the Khairpur area of the Left Bank of the Lower Indus in Pakistan (Table 4.7), water levels began to rise after the completion of the Sukkur Barrage irrigation scheme in 1935. Large areas of the Left Bank irrigated area now have water levels within a metre of the ground surface. The importance of canal losses is clearly demonstrated in this region. Groundwater in this enormous alluvial aquifer is highly saline and seepage losses from the major canals over many years have built up substantial lenses of fresh groundwater which now form a valuable resource.

Table 4.7 Rise in water table due to irrigation

Irrigation project	Water table	
	Original depth (m)	Rise (m/a)
Nubariya, Egypt	15-20	2.0-3.0
Beni Amir, Morocco	15-30	1.5-3.0
Murray-Darling, Australia	30-40	0.5-1.5
Amibara, Ethiopia	10-15	1.0
State Farm 29, Xinjiang, China	5-10	0.3-0.5
Salt Valley, USA	15-30	0.3-0.5
SCARP I, Pakistan	40-50	0.4
Bhatinda, India	15	0.6
SCARP VI, Pakistan	10-15	0.2-0.4
Khairpur, Pakistan	4-10	0.1-0.3

Adapted from Smedema, 1990

When water evaporates from the ground, dissolved salts remain behind. The net result is to increase the amount of salts in the soil in direct proportion to the salt content of the irrigation water and the depth of water applied. As the water table rises, soluble salts, present in the lower part of the soil profile are dissolved and precipitated in the upper part of the soil zone following evaporation (Dent et al, 1992). The addition of more irrigation water temporarily dilutes the soil water, but evapotranspiration concentrates it further. At the same time, some of the excess irrigation water displaces soil water downward beyond the root zone, and this may in turn dissolve more salts from the subsoil and underlying aquifer material on its way down to the water table. Soil water is usually 2-3 times more concentrated than the applied irrigation water, and often 5-10 times. Management of irrigation schemes tries to ensure that the soil water salt content does not exceed the tolerance of the particular crops being grown. This can be accomplished by applying irrigation water in excess of that needed for consumptive use i.e. by leaching. This may, however, merely transfer the salinity problem from the soil to the underlying groundwater, and also contribute to the rise in water levels.

The waterlogging resulting from rising groundwater levels may produce severe salinity by the processes described above or because the groundwater is already more or less saline, as in the Lower Indus. In other instances, groundwater with naturally high salinity is used for irrigation especially in the semi-arid areas where groundwater may be the only source of water for irrigation. Prolonged use of such groundwater can produce a build-up of soil salinity. Particularly difficult problems to deal with are those of soda salinisation-alkalinization. This process is caused by alkaline groundwaters or dilute sodium-carrying irrigation waters, for example from the Indus or Nile, or a combination of the two. In the most severe cases the soil adsorption capacity becomes saturated with sodium to 60 or 70% of the cation exchange capacity and soil alkalinity rises to pH 9-11 (White, 1977). Soil degradation by compaction invariably results.

Local factors may combine with the overall mechanisms of waterlogging and salinisation to worsen the situation considerably. The dramatically increased infiltration which occurs as semi-arid land is brought under irrigation may leach out salts from desert soils ans subsoils and underlying geological strata. This may be a contributing factor to the increased salinity in the Lower Yaqui Valley in Sonora, Mexico, where initial commissioning of the irrigated land is said to have been accompanied by extremely heavy water applications.

4.5.3 *Control measures*

Improved irrigation efficiency

Waterlogging can be prevented or corrected by reducing excess water inputs and increasing natural drainage capacities. The former requires improved water application techniques and scheduling to reduce excess infiltration from the irrigated land, often combined with engineering measures to reduce losses in the water distribution system. Reducing excess water by increasing irrigation efficiency also offers scope for extending irrigated cultivation without greater water withdrawals.

Lining of major canals and distributaries is a straightforward but expensive engineering approach to improving irrigation efficiencies. Significant losses from canals have been measured in many studies (Wachyan and Rushton, 1987). In some, there was relatively little difference between lined and unlined canals, and this possibility was confirmed by the modelling studies performed by Wachyan and Rushton (1987). Perfect linings would prevent losses, but any imperfections greatly reduce the effectiveness of canal lining.

Considerable research has gone into improved irrigation efficiency, and many techniques are now employed. Furrow and basin irrigation are now being replaced by sprinklers and more recently by drip and trickle micro-irrigation techniques. Centre-pivot sprinklers can improve irrigation efficiency by up to 70% and micro-irrigation techniques by an additional 20-25% (WRI, 1987). A 99% efficiency has been claimed for the most recently-developed mobile drip units. Such results are, however, only achieved by extremely careful management, and there remains scope for improvement even with these sophisticated techniques as some technical problems remain. Less costly improvements to gravity flow systems, such as proper land levelling and better water distribution remain the most practical approaches to achieving better irrigation efficiencies in most instances.

Improved drainage

Adequate drainage to reduce or prevent waterlogging and salinity requires a general lowering of very shallow water tables to 2-3 m below ground. This can be achieved by pumping from boreholes or by horizontal tile drains, with a network of ditches and canals to remove the excess water. In very large irrigated areas with small topographic gradients, disposal of the drainage water can be a major problem. In Pakistan, over the last 30 years massive government schemes - Salinity Control and Reclamation Projects (SCARPs) - have been implemented in an attempt to halt and reverse water table rise to maintain and restore agricultural production. Many thousands of public boreholes have been constructed to lower groundwater levels, but operation and maintenance problems have limited their effectiveness. A large canal, the Left Bank Outfall Drain has been constructed to carry a design discharge of 400 m^3/s of saline drainage water to the sea.

Once soil salinity has reached levels that affect plant growth, salts must be leached from the soil and the leachate disposed of in an acceptable way. Soils affected by severe alkalinization can only be remedied by heavy applications of calcium-rich soil improvers.

Although the waterlogging and salinisation of irrigated land is an almost universal problem, there are few really encouraging examples of prevention and amelioration. Often a combination of political, financial and economic difficulties hamper the effectiveness of control measures. It is, for example, not always easy to decide who pays for remedial measures in relation to who benefits. Even if drainage measures are implemented, there is always the problem of disposal of the drainage water. In the Colorado River Basin, for example, 84% of the salt originating from irrigation returns reaches the river in the Upper Basin, and much of the damage occurs in the Lower Basin (Law and Hornsby, 1982).

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METHOD SUMMARY SHEET (WQM 03)

TITLE: Investigating the impact of agriculture - nitrate

Scope and use of method

Many of the most important unconsolidated sedimentary aquifers occur in areas of intensive agricultural activity, which may present threats to groundwater quality. Investigating the impact of intensive cultivation with agrochemical use is important for designing strategies for groundwater protection and in establishing groundwater quality monitoring systems. Such areas are often also densely populated, and distinguishing between several possible sources of nitrate in groundwater may become an important component of the investigation.

Method

The method is more of an overall approach which is likely to have the following components:

- collection of hydrogeological data to assess the general vulnerability of the aquifer (as described in Review no 6)
- agricultural surveys to define cropping regimes and fertiliser use (types and application rates), and general land-use and population surveys to identify other possible nitrate sources
- collection of existing groundwater quality data, with particular emphasis on nitrate
- estimation of the likely nitrogen loading and nitrate concentration in recharge, as described in chapter of 4 of this review

A preliminary appraisal can be made from the above. If more detailed studies are required, they would comprise:

- measurement of nitrate leaching by lysimeters, porous pots or unsaturated zone profiling
- investigation of other sources of nitrate from, for example, sanitation, by nitrogen isotopes, nitrogen/chloride ratios and other major ion ratios, trace elements and microbiological sampling

The field investigations would draw on the more specific methods described in the other reviews and in the key references cited below. For improved protection and management of groundwater quality, it is essential for the investigation to identify the source of nitrate pollution so that control measures can be correctly targeted. Establishing a groundwater quality monitoring programme to assess the impact of nitrogen originating from agriculture, and the effectiveness of control measures is relatively simple in terms of analytical requirements, although other nitrogen species such as nitrite and ammonia may need to be analysed. Monitoring for agricultural impacts may also have to take particular account of seasonal influences, especially in the shallower and smaller UNSAs.

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METHOD SUMMARY SHEET (WQM 04)

TITLE: Investigating the impact of agriculture - pesticides

Scope and Use of Method

Intensive agricultural activity on land overlying unconsolidated aquifers is increasingly accompanied by the heavy use of pesticides. In many situations, UNSAs are likely to be highly vulnerable to pollution from agrochemicals, because the aquifer material may be permeable and the water table shallow. Both general appraisals of the risk of pesticides reaching groundwater and detailed investigations of the behaviour of specific compounds and aquifers may be needed. Because of the very large numbers of compounds in common agricultural usage, indications of which compounds are in widespread use in the area of interest and the likelihood of leaching to groundwater are needed even for establishing monitoring programmes.

Method

A phased approach is recommended. The first stages of investigation are very similar to those for nitrate, and often go hand-in-hand as interest is likely to be directed to both fertiliser and pesticide risks. Much of the information collected serves both:

- collection of hydrogeological information on the general vulnerability of the aquifer, as described in Review 6, together with specific information related to pesticide transport and behaviour - content of clay and organic matter

- collection of existing data or carrying out of specific agricultural surveys of cultivation practices, cropping regimes, irrigation methods and pesticide usage and application rates, supplemented by data on the properties of the most important compounds related to the likelihood of their leaching to groundwater. These data include solubility, mobility (partition coefficients) and half-lives.

- collection for the area of interest or similar environments of any existing data on pesticide occurrence in groundwater

- simple modelling to test the sensitivity of pesticide transport to aquifer characteristics and pesticide properties

This information will allow a preliminary appraisal of the risk of pesticide leaching to groundwater, and selection of the compounds more likely to be leached for monitoring and for detailed investigation. Where significant risk is identified, or troublesome pesticide concentrations have already been detected in groundwater supplies, more detailed studies would comprise:

- field investigation by lysimeters, suction samplers, shallow piezometers and sampling of existing wells and boreholes to define the pathways of pesticide transport

- field and laboratory studies of degradation of the most important selected compounds

- further use of modelling using the new, locally-valid information about aquifer properties

The results of these studies can be interpreted to provide an indication of whether serious groundwater pollution by pesticides from normal agricultural use is likely, and whether pesticide concentrations in groundwater are likely to be at equilibrium. The results and subsequent interpretation can then be used to determine whether controls on pesticide usage are required, and what form those controls should take.

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METHOD SUMMARY SHEET (WQM 05)

TITLE: Investigating salinisation from irrigated agriculture

Scope and use of method

Deterioration of soil and water quality resulting from irrigation can usually be linked to poor planning and implementation of projects and subsequent mismanagement. Waterlogging often arises because the construction of proper drainage measures was postponed rather than being implemented at the same time as the water distribution system. Waterlogging and salinity are, therefore, linked to low efficiency of irrigation schemes, both of the distribution system and of application to the fields.

The excess infiltration from irrigated lands often brings water tables in UNSAs close to the ground surface, and allows direct evaporation and a build up of salts in the soil. Preventing waterlogging and salinity and restoring affected land requires improved drainage, the design of which depends on knowledge of the mechanisms and amounts of excessive infiltration, and the hydraulic properties of the soil and underlying UNSAs.

Method

A key component of investigating waterlogging and salinity is to determine the sources and amounts of recharge, and their impact on the underlying aquifer. Once salinisation has become established, the impact is clear from deteriorating crop yields and eventually from land going completely out of production, and the extent of such land is relatively easy to map. Warning of impending waterlogging comes from the observation of groundwater levels within and around the irrigated area. Consistently rising groundwater levels are an indication of problems to come, and when levels are within 1-2 m of the ground surface direct evaporative concentration of salts will occur.

For control and management purposes, it is often necessary to be able to distinguish between sources of recharge within the irrigation system, for example between distribution losses and losses from the fields themselves. Lining of canals is a simple but expensive engineering approach to improving efficiency, but there needs to be clear evidence that the canals are a major contribution to the problem before this expense can be justified. Estimation of losses can be made from differences between discharge measurements along the canals, from hydraulic measurements in piezometers close to the canals and by modelling.

Estimation of losses from the irrigated fields employs the normal techniques of recharge estimation, and requires accurate knowledge of the consumptive use by the range of crops which are grown, and application rates by the methods of irrigation employed.

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5. WASTEWATER REUSE AND GROUNDWATER QUALITY

5.1 Introduction and scope

Many of the world's largest cities are located on unconsolidated sediments in major river basins. Historically, they were sited and have grown and flourished because of the availability of abundant shallow groundwater or surface water of good quality for domestic use and irrigated farming. The very rapid urban growth of the last few decades has produced increasing demands for potable water and consequent increases in volumes of sewage effluent. As a consequence of growth and industrialisation, surface water resources are either fully utilised or now of poor quality, and groundwater resources are becoming more important.

The direct impacts of urbanisation on underlying aquifers are described in the previous chapter. The improved coverage in large cities of water-borne sewerage systems produces enormous volumes of wastewater for disposal. By reusing this wastewater in a carefully managed way water is conserved, groundwater quality is maintained and the overall resource situation improved, especially in arid areas. Increasing recognition is being given to the value of wastewater as an important resource, and strategies for reuse are likely to become more widely adopted.

This chapter reviews the present state of knowledge concerning the impact of wastewater reuse on groundwater quality, with particular emphasis on unconsolidated sedimentary aquifers. The principal groundwater quality issues arising from wastewater reuse are illustrated by several case histories. The methods which may be employed to investigate the impact of existing wastewater reuse systems on groundwater quality are summarised, and the ways in which wastewater reuse can be designed and operated to maintain quality in underlying aquifers are reviewed.

5.2 Background

The potential volume of the additional resource represented by wastewater is illustrated in Table 5.1. Many of these cities are in semi-arid or arid environments, and some of the largest are located on UNSAs.

The introduction of water-borne sewerage systems has often lagged behind the provision of potable supplies, and has been carried out in an intermittent, often piecemeal way over periods of many years. As a consequence, systems are not well integrated and rationalised. Effluent may be discharged at many different locations, often directly to the most convenient surface water courses, in most cases with no more than primary treatment or settlement.

Because cities deal with this disposal problem with varying degrees of sophistication, surface water courses often become highly polluted. In more arid regions, the wastewater may form a large proportion, if not all, of the flow in some rivers in the dry season. Until relatively recently, there has been little concern about the capacity of the receiving waters to assimilate the discharged effluent. In many cases the discharge points are now within the urban areas as a result of rapid population growth. In populated arid regions, this often highly polluted surface water is extensively used for agricultural irrigation, often in a largely uncontrolled way. Much of the wastewater irrigation has grown up gradually and informally. Such is the demand for additional water resources in some areas, that raw wastewater is indiscriminately or even illegally used directly from sewers or vegetables and fruit are cultivated with inadequately treated wastewater. These practices are responsible for many serious endemic health problems associated with pathogenic organisms, and have also been implicated in the spread of recent outbreaks of cholera.

Table 5.1 Projected annual effluent production by the year 2000

City	Population (million)	Effluent production (Mm^3)	Aquifer
Mexico City	31.0	2,602	Lacustrine clays, silts and underlying volcanics
São Paulo	25.8	2,166	
Cairo	13.1	1,100	Nile basin alluvium
Karachi	11.8	991	Indus basin alluvium
Teheran	11.3	949	
Lagos	6.9	579	Coastal alluvium
Addis Ababa,	5.6	470	

(After Pescod, 1984)

Both direct disposal and reuse of wastewater can have major impacts on groundwater. In some situations, the substantial volumes of additional recharge may completely alter the local hydrogeology. Perched aquifers and new groundwater flow regimes and discharge points may be created. Impacts may be simultaneously positive in terms of water resource conservation and negative in relation to groundwater quality. Improper disposal of untreated wastewater directly into aquifers used for water supply can cause serious pollution problems. On the other hand, controlled disposal of untreated or partially-treated wastewater at the ground surface to provide irrigation or artificial recharge can provide significant additional resources of good quality water. However, adequate knowledge of the hydrogeology, infiltration processes and the movement and natural attenuation of pollutants is required for effective design and management of such systems.

5.3 Wastewater disposal practices

Wastewater treatment and disposal practices are summarised in Table 5.2. Many of these practices give rise to significant risk of groundwater pollution. This risk must be put in context with other more immediate public health hazards of wastewater disposal, but nevertheless needs to be identified and characterised. The types of installations listed in Table 5.2 are intended to treat, reuse or dispose of wastewater. In the case of lagoons, trenches or land spreading, ease of infiltration to groundwater is an important feature of the design, because enhanced recharge is the overall objective. As a consequence, the quality of the infiltrating water is also a design consideration, as maintaining or augmenting water resources for a specific and planned use is the context within which the enhanced recharge is carried out. Installations which are primarily intended to treat or reuse wastewater (Table 5.2) may also produce significant recharge to groundwater which is, however, incidental to the primary objective. In many cases, therefore, the quality of this recharge was not considered at the outset, although it often becomes an important issue during operation. Similarly, the installations designed to dispose of wastewater (Table 5.2) almost invariably produce unintended, incidental recharge, the amount and quality of which were not initially considered.

Although the impacts on groundwater resources of allowing or encouraging wastewater to infiltrate into the subsurface can be both positive and negative, generally some degradation of groundwater quality can be anticipated from most wastewater handling practices. The quality deterioration may be in respect of any of the following:

- pathogenic viruses and bacteria
- nutrients
- inorganic contaminants
- organic contaminants

The degree of risk of pollution of groundwater will vary widely with the scale and method of wastewater use, the origins and quality of the wastewater itself, and the local hydrogeological conditions. In particular, the proportion of industrial effluent and the type of industries has a major influence on the composition of the wastewater and hence on the quality deterioration that can be anticipated. The potential impacts on quality are reviewed in turn, illustrated where possible by case studies, following a brief general discussion of the ways in which contaminants move from beneath wastewater reuse facilities to groundwater bodies.

Table 5.2 Summary of wastewater treatment, disposal and reuse practices

Installation/Process	Primary Objective	Level Treatment		Groundwater Recharge
		Normal	Required	
SEWERED				
Stabilisation/oxidation lagoons	Treat	P	P,S	u
Lagoon/pit/trench infiltration	Dispose, sometimes treat and reuse	P,S	P,S	i
Land drain infiltration		S	Si	i
Land spreading evaluation		R,P	R,S	u
Agricultural/amenity irrigation	Reuse	R,P	S,T	u
Seepage from polluted river beds	None	R,P,S	-	u
Leaking sewer lines	None	R	-	u
Deep injection wells	Dispose	S,T	T	i
UNSEWERED				
Septic tank soakaway	Dispose	P,S	S	u
Cesspits	Dispose	R	R	u
Latrines	Dispose	R	R	u
Illegal discharge to disused wells	Dispose	R	-	u

R = raw; P = primary; S = secondary; T = tertiary
i = intended, designed into scheme; u = unintended or incidental result of practice
(simplified from Foster et al, 1994)

5.4 General groundwater quality considerations

5.4.1 *Flow mechanisms and attenuation processes*

Movement of infiltrating wastewater through the soil, the unsaturated zone and the saturated aquifer improves its quality. Natural soils attenuate many, but not all, pollutants, and are considered to be a potentially effective treatment system for the disposal of wastewater. A number of projects which make use of the attenuation capacity in this way are described in the literature (Bouwer et al, 1984; Idelovitch and Michail, 1984). Unconsolidated sedimentary aquifers, with relatively slow, intergranular flow providing large surface areas and substantial time, provide better environments for this to be most effective than consolidated aquifers in which fissure-flow dominates.

The various methods of handling wastewater summarised in Table 5.2 permit recharge of groundwater to occur either deliberately or incidentally. In both cases the intensity of infiltration can vary greatly, from highly-concentrated localised, point recharge to diffuse recharge spread over a large land area. Many of the point sources produce recharge which is more or less continuous. In contrast, the more diffuse recharge may be intermittent, where irrigation becomes unecessary because of wet-season rainfall or where infiltration is designed in this way to allow the soil and aquifer system to recuperate through oxidation. Variations in recharge mechanisms and infiltration rates exert an important influence on the effectiveness of self-purification processes; the slower and more intermittent the infiltration, the more effective are the purification and attenuation processes.

Most of the processes which are important in attenuating pollutants are more active in the soil and unsaturated zone (Figure 5.1). The rates of most processes are highest in the biologically-active soil layer, as a result of its higher clay mineral and organic content and very much larger microbial populations. The soil is, however, removed in many of the wastewater handling practices listed in Table 5.2. The beneficial effects of the soil are thus lost, although the establishment of an organic mat at the infiltration surface produces a similar effect.

Below the soil, the unsaturated zone also occupies a strategic position between the land surface and the water table. In UNSAs, water movement in the unsaturated zone is normally slow and restricted to small pores with large specific surface areas. There is potential for:

- interception, sorption and elimination of of pathogenic bacteria and viruses
- attenuation of heavy metals and other inorganic chemicals through precipitation, sorption or cation exchange
- sorption and biodegradation of organic compounds.

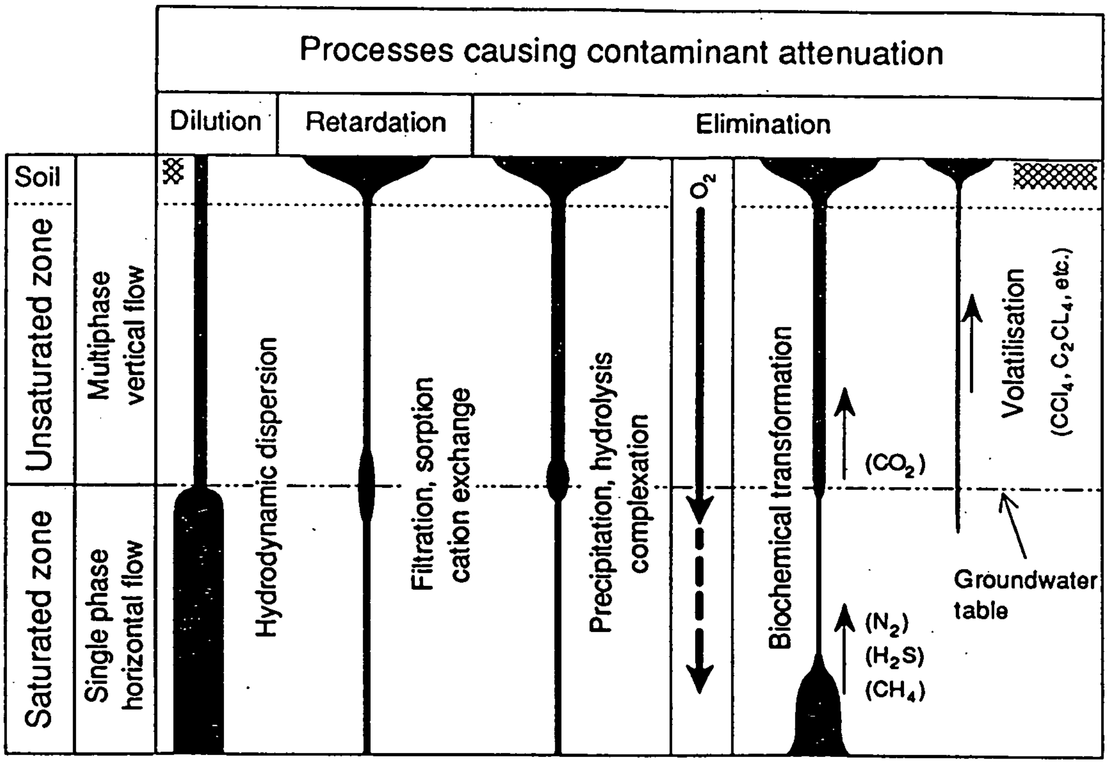


Figure 5.1 Processes causing contaminant attenuation (Foster and Hirata, 1988)

The processes continue, albeit less actively, at depth where they are more effective in UNSAs than in fissured rocks. The degree of contaminant attenuation will depend on the pollutant pathways and residence times, which in turn are controlled by the hydraulic characteristics of the strata. Moisture content and unsaturated vertical conductivity are both a function of matrix potential, which is largely determined by the pore-size distribution. Natural flow rates in the unsaturated zone of most sedimentary formations do not generally exceed 0.2 m/d in the short term, and are much less when averaged over long periods.

Several other factors have a bearing on contaminant attenuation in the unsaturated zone. The mineralogy of the sediment is important. Clay minerals have significant potential for the adsorption of the metals, phosphate, ammonium and viruses which may be contained in infiltrating wastewater. Organic matter within the aquifer will retain many hydrophobic organic contaminants. The ion exchange capacity of calcareous sands may result in the removal of sodium, potassium and ammonia in exchange for calcium and magnesium which then enter the infiltrating water.

Depth to water table is also an important consideration; many major UNSAs, especially those in large alluvial basins, have water tables very close to the ground surface. This leaves little scope for contaminant attenuation above the water table. Some of the wastewater disposal practices listed in Table 5.2 may result in direct communication with the saturated zone. In these circumstances, hydraulic conductivites may be much higher and pollutants may be transported much more rapidly away from the site of wastewater use. In some UNSAs, stratification within the sedimentary sequence can produce locally confined or semi-confined conditions or localised perched groundwater bodies. These may restrict downward recharge and encourage lateral movement instead.

The saturated zone is a more stable environment because of the lower oxygen levels and the relatively large volume of groundwater for dilution. Reaction processes are generally limited to adsorption, ion exchange and precipitation (Figure 5.1). Below the water table, the hydrodynamic dispersion which accompanies groundwater flow will also produce dilution of persistent and mobile pollutants (Figure 5.1).

5.4.2 *Wastewater composition*

Selected wastewater analyses are given in Table 5.3. The composition of wastewater can, however, be expected to exhibit wide variations from city to city, depending on the nature of the industries generating effluents which are discharged to the local sewerage system. Further, significant seasonal and even shorter-term variations in composition will occur where mixed wastewater and stormwater disposal systems are subject to dilution following heavy rainfall. Significant diurnal variations in composition have been reported from wastewater systems dominated by heavily-polluted industrial discharges, but sampling of sufficient frequency to observe this is rare. Such short-term variations in composition are unlikely to be important in relation to the movement of pollutants into aquifers, but they may need to be taken account of in establishing the average composition, which does describe the initial quality of the infiltrating wastewater.

5.4.3 *Quality criteria for reuse*

The suitability and effectiveness of wastewater reuse depends on the quality criteria for whatever use the reclaimed wastewater is to be put. These are principally potable supply, agricultural irrigation and amenity irrigation. The World Health Organisation (WHO, 1989) has produced quality guidelines for wastewater reuse in agriculture (Table 5.4). These are intended to improve public health protection of farm workers, consumers of produce and the public, especially in areas where crops eaten uncooked are being irrigated in an unregulated, often illegal manner with raw wastewater. If the recommended guidelines were adopted, they would achieve this improved protection and set technologically and economically feasible targets. The guidelines are intended as design targets for wastewater treatment systems, rather than standards requiring routine testing of effluents. Thus, treatment processes which can achieve the recommended microbiological quality consistently because of their intrinsic design characteristics, rather than by high standards of operational control, are to be preferred.

Guidelines have also been prepared which classify the suitability of water for optimum crop production (Ayers and Westcot, 1985). These were prepared for conventional sources of irrigation water, but can also be applied to wastewater use for crop production (Pescod, 1992). The scheme outlined in Table 5.5 must be considered as general in nature and applicable to "average" conditions as the suitability of water for irrigation depends on climatic conditions, physical and chemical properties of the soil, tolerance of the crops grown and irrigation management practices.

Table 5.3 Composition of domestic wastewater

Constituent	Concentration (mg/l)				
	Typical range[1]	Amman[2]	Giza[3]	Mexico City[4]	Lima[5]
Total solids	350-1200			1070	760
Dissolved solids	250-800	1170	1700	710	
Suspended solids	100-350	900		350	
Nitrogen (as N)	20-85	150		25	46
Phosphorus	6-20	25		5	4
Chloride	30-100		320	80	115
Alkalinity	50-100	850		380	
Sulphate		90	138	190	240
Grease	50-150			100	
Biological oxygen demand$_5$	100-300	770		130	
Total organic carbon		220			170

[1] UN Department of Technical Cooperation for Development (1985)
[2] Al-Salem (1987)
[3] Abdel-Ghaffar et al (1988)
[4] CNA and BGS (1995)
[5] Geake et al (1986)

Table 5.4 Recommended microbiological quality guidelines for wastewater use in agriculture

Category	Reuse condition	Exposed group	Intestinal nematodes[a] (mean/l)	Faecal coliforms[b] (mean/100 ml)	Wastewater treatment expected to achieve the required microbiological quality
A	Irrigation of crops likely to be eaten uncooked, sports fields, public parks[c]	Workers, consumers, public	≤ 1	≤ 1000	A series of stabilization ponds designed to achieve the microbiological quality indicated, or equivalent treatment
B	Irrigation of cereal crops, industrial crops, fodder crops, pasture and trees	Workers	≤ 1	No standard recommended	Retention in stabilization ponds for 8-10 days or equivalent helminth and coliform removal
C	Localized irrigation of crops in category B if exposure of workers and the public does not occur	None	Not applicable	Not applicable	Pretreatment as required by the irrigation technology, but not less than primary sedimentation

[a] *Ascaris* and *Trichuris* species and hookworms
[b] During the irrigation period
[c] More stringent for public lawns where public may come into direct contact
(WHO, 1989)

Table 5.5 Guidelines for interpretation of water quality for irrigation

Potential irrigation problems	Units	Degree of restriction on use		
		None	Slight to moderate	Severe
Salinity				
Electrical conductivity, EC	μS/cm	< 700	700 - 3000	> 3000
Total dissolved solids, TDS	mg/l	< 450	450 - 2000	> 2000
Infiltration				
Sodium adsorption ratio = 0 - 3 and EC=		> 700	700 - 200	< 200
3 - 6		> 1200	1200 - 300	< 30
6 -12		> 1900	1900 - 500	< 500
12 -20		> 2900	2900 - 1300	< 1300
20 -40		> 5000	5000 - 2900	< 2900
Specific ion toxicity				
Sodium, Na				
surface irrigation	SAR	< 3	3 - 9	> 9
sprinkler irrigation	meq/l	< 3	> 3	
Chloride, Cl				
surface irrigation	meq/l	< 4	4 - 10	> 10
sprinkler irrigation	meq/l	< 3	> 3	
Boron, B	mg/l	< 0.7	0.7 - 3.0	> 3.0
Trace elements[1]				
Miscellaneous effects				
Nitrogen, NO_3-N[2]	mg/l	< 5	5 - 30	> 30
Bicarbonate, HCO_3	meq/l	< 1.5	1.5 - 8.5	> 8.5
pH			Normal range 6.5 - 8.4	

[1] Varies, see original reference

5.5 Groundwater quality constraints

5.5.1 *Pathogenic microorganisms*

It has long been recognised that water can serve as a vector for disease transmission. Viruses, bacteria and protozoa are excreted by infected persons and raw wastewater may contain a whole range of pathogenic organisms. Feachem et al (1983) divided the infections caused by pathogens into five categories according to their environmental transmission characteristics. The risk of infection depends on the persistence in soil and aquifer of the pathogen concerned and the infective dose required.

Health concerns related to the reuse of wastewater have led to research into the ability of aquifers to remove pathogens during infiltration. Soil infiltration is an extremely effective method of removing pathogenic microorganisms from wastewater. Box 5.1 shows the results of microbiological sampling in the coastal plain aquifer of Lima beneath wastewater lagoons and land irrigated with wastewater. The formation of a biologically-active organic slime at the infiltration surface is an important factor in this removal, especially in facilities using untreated effluent. The ability to remove pathogens is a major benefit of wastewater treatment by infiltration, and is sometimes known as soil-aquifer treatment (SAT).

Protozoa and bacteria are retained by filtration and rendered ineffective, eliminating them completely. Viruses, which are much smaller than bacteria, are removed mainly by adsorption, but may be somewhat more persistent and mobile (Keswick and Gerba, 1980). There are much fewer published data from which to predict their behaviour, but of all the pathogens viruses are the most likely to find their way down to groundwater.

In all cases, the lower the overall hydraulic loading and the smaller the infiltration rate, the more effective is pathogen removal. Thus, most types of UNSAs, with their relatively slow intergranular flow, are particularly well-suited for SAT systems, and the risk of deterioration of microbiological quality is not the most important constraint in most hydrogeological conditions. More caution is required, however, in the more permeable alluvial aquifers if the water table is very close to the ground surface. In addition, if high rates of infiltration are employed over consolidated limestone aquifers, then even a relatively thick unsaturated zone may not effective in pathogen removal because of the possibility of rapid preferential flow developing.

5.5.2 *Nutrients*

The total nitrogen content of wastewater is high, and large-scale wastewater recharge can lead to unacceptable groundwater pollution by either ammonium or nitrate. The type and severity of the contamination depends on the level of effluent treatment, the method of recharge and soil conditions. Thus, if untreated effluent is discharged continuously to the ground, anaerobic conditions will be established. Nitrification is suppressed and ammonium may reach the water table, as in the case of the Lima study (Geake et al, 1986). If, however, more oxidised secondary effluent with lower organic content is applied at low rates, nitrification is encouraged and the recharge may have nitrate concentrations well above 10 mg NO_3-N/l (Box 5.1).

Nitrogen in wastewater may occur in several different forms - nitrate, nitrite, nitrous oxide, ammonia and nitrogen gas - depending on the conditions. Inorganic nitrogen usually occurs in wastewater as ammonia, which can then be oxidised to nitrate or reduced to nitrite and nitrogen gas by denitrifying bacteria in anaerobic conditions. Thus, the behaviour of nitrogen species associated with wastewater infiltration can be highly variable. In the Dutch sand dune schemes, for example, almost total nitrate removal occurred following the passage of recharging wastewater through anaerobic zones, whilst little removal occurred in other areas (van Puffelen, 1982).

Box 5.1 Stabilisation lagoons on a coastal alluvial aquifer, Lima, Peru

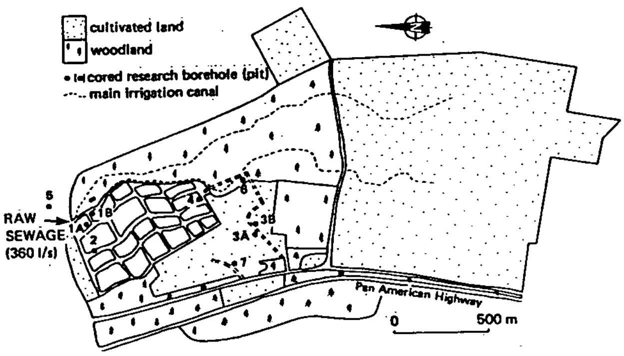
Lima is situated on the arid Pacific coast of Peru, where rainfall is negligible. Groundwater, which provides over 40% of municipal supply for the city, is derived from a large fan of alluvial and colluvial sediments, mainly highly permeable sands and gravels with occasional clay layers, which is more than 100 m thick. The aquifer is recharged directly by the Rimac and Chillon rivers which originate in the high Andes, and to a lesser extent indirectly from losses in the irrigation systems associated with these rivers. As a result of the heavy abstraction groundwater levels are falling over wide areas at 2-4 m per year.

Additional resources would therefore be of great benefit, and the availability of ample flat and potentially productive land close to the city led to early interest in wastewater reuse. Experimental stabilisation lagoons were established at San Juan de Miraflores in 1964, with the objective of improving wastewater quality for reuse in agriculture, forestry and fish production.

A series of 21 simple, unlined shallow excavations in sand, each less than 2 m deep occupies an area of about 20 ha. About 360 l/sec of mainly domestic raw sewage is delivered to the complex. About half passes through, with design retention times of 5-10 days. The effluent from the stabilisation lagoons, together with additional raw sewage is used to irrigate about 375 ha of trees, agricultural crops and recreational parks by furrow irrigation every 5-10 days at average rates of 5-10 mm/d.

The site is underlain by well-sorted, permeable, fine to medium aeolian sand and alluvial sands, with some beds of gravel and clay. Long-term average infiltration rates are estimated to be 10-20 mm/d. Infiltration rates increase dramatically following the draining and cleaning of lagoons, but fall off again rapidly with time. The groundwater table is 20-30 m below the surface at the site.

The lagoons function as facultative systems, producing significant reduction in BOD, elimination of most faecal parasites, some reduction in bacteria and viruses and total nitrogen load, but very low rates of nitrification.

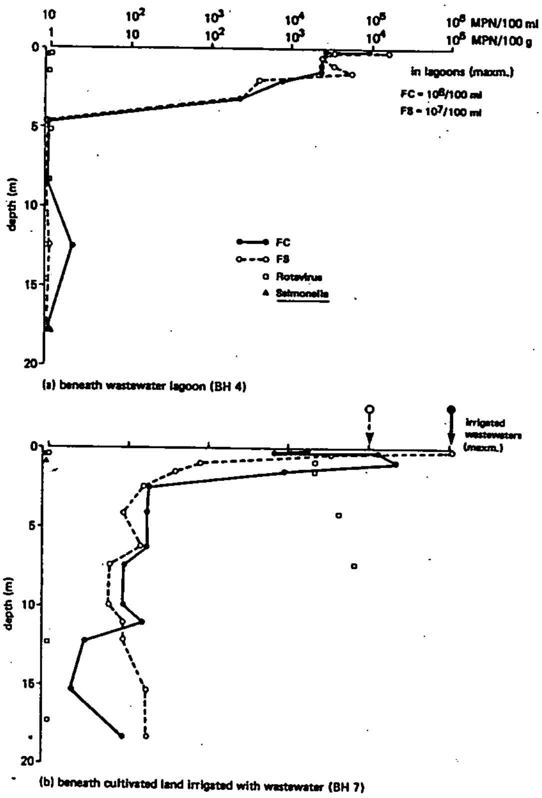


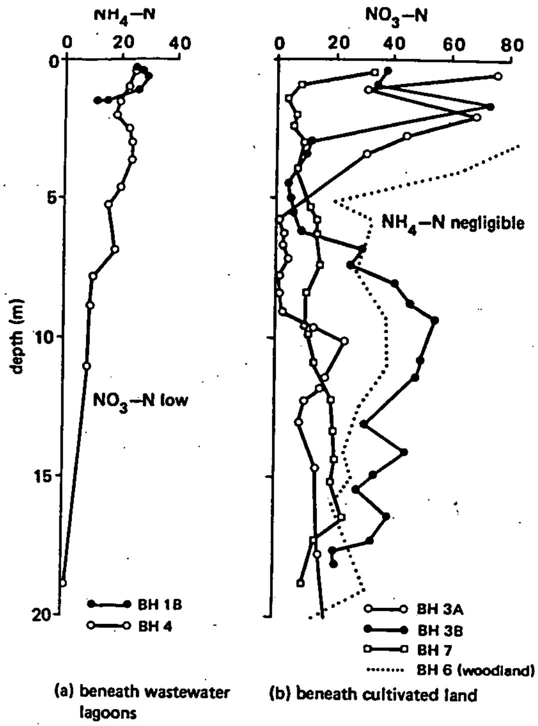
Plan of the San Juan de Miraflores Wastewater Reuse Complex

The impact of the scheme on the underlying groundwater has been investigated by the drilling of eight cored boreholes, some through the floors of drained lagoons and others in the irrigated land. A lightweight percussion rig was used to avoid sample contamination by drilling fluid. Porewater was extracted by centrifuge to provide vertical profiles of unsaturated zone water quality. Sand samples were assayed directly for bacterial populations by a modified MPN procedure, and presumptive positives confirmed by standard methods. Rotavirus densities were estimated by elution, physico-chemical flocculation of eluents and immunofluorescence. The methods are fully described by Geake et al (1986).

High concentrations of bacteria and viruses penetrate into the upper part of the unsaturated zone beneath the lagoons, but there is rapid elimination within the top 3 m. Beneath cultivated land, bacterial indicators also reduce rapidly in the first 3 m, but persist to greater depth. The reduced effectiveness of attenuation is attributed to the lack of a sludge mat.

Box 5.1 continued





Unsaturated zone profiles of nitrate and ammonia

Attenuation of faecal pathogens and indicator bacteria in the unsaturated zone beneath Wastewater Reuse Complex

The total nitrogen concentration in the sewage is 40-50 mg N/l, mostly as organic and ammonium forms. Beneath the lagoons, high NH_4 concentrations in the first few metres had decreased by 10 m. The presence of NH_4 rather than NO_3 confirms that conditions are anaerobic. In contrast, under the irrigated land aerobic conditions encourage nitrification, and NO_3 concentrations in recharge are well above the WHO guideline value.

TOC concentrations immediately beneath the lagoons were high but rapidly attenuated with depth to concentrations of 5-10 mg/l. Similar concentrations were observed beneath the cultivated land. These compare with values in excess of 100 mg/l in raw sewage and lagoon effluent.

In conclusion, the thick unsaturated zone attenuates some pollutants, but recharge quality remains poor in respect of salinity, nitrate, organic solvents and faecal bacteria. If the groundwater table had been shallower, as is commonly the case in many UNSAs, then the pollutant load would have been heavier.

Lining the stabilisation lagoons would reduce the risk of pollution by NH_4 and organic compounds. The use of more efficient irrigation techniques would reduce pollution by nutrients and pathogens, but salinity would increase. On overall public health grounds, primary treatment in lined oxidation ponds followed by infiltration from cyclically-filled spreading basins would be the preferred option.

Key Reference:

Geake A K, Foster S S D, Nakamatsu N, Valenzuela C F and Valverde M L 1986. Groundwater recharge and pollution mechanisms in urban aquifers of arid regions. British Geological Survey Hydrogeological Report 86/11.

Wastewater recharge to alluvial deposits in Phoenix, Arizona is long-established and well-described (Bouwer and Rice, 1984) (Box 5.2). The scheme at Flushing Meadows dates from 1967 and that at 23rd Avenue from 1975. Following soil-aquifer treatment, groundwater is used for irrigation. At Flushing Meadows, total nitrogen concentrations declined from 20 mg/l to about 7 mg/l if the recharge area was managed with alternate flooding and drying-out cycles to promote denitrification. At the 23rd Avenue site, a similar reduction was observed, with a suggestion of seasonality, NH_4-N only occasionally reaching the water table at significant concentrations (Bouwer and Rice, 1984). These results suggest that nitrification was complete but denitrification was partial and fluctuating. This was because the short flooding cycles favoured the active growth of aerobic microbial populations.

Experience of operating and monitoring the Phoenix sites suggests that it is possible to promote denitrification by allowing intermittent inflitration of primary effluent. With subsequent drying out of the soil profile aeration and nitrification occur. When anaerobic conditions are re-established, denitrification of this nitrate is encouraged. By careful control, this approach can reduce total nitrate concentrations in groundwater recharge to less than 5 mg/l.

After nitrogen, phosphorus is the second important nutrient in wastewater, usually in the form of inorganic phosphate. However, during infiltration, phosphate is either used in plant growth or adsorbed onto clay minerals, and does not usually pass beneath the soil to cause serious groundwater pollution. Thus, at the Flushing Meadows site, phosphate concentrations in the treated wastewater decreased from 13 to less than 2 mg/l following infiltration, and at the Dan Region project from 2 mg/l to less than 0.1 mg/l (Idelovitch and Michail, 1984).

5.5.3 *Inorganic constituents*

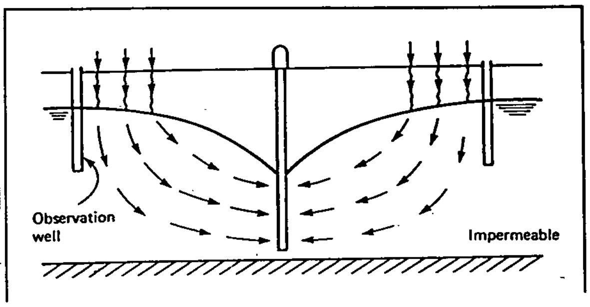
Wastewater frequently contains high concentrations of chloride (Table 5.3), originating from human wastes and some types of industrial activities (BGS and others, 1996). Because of its conservative nature, movement of chloride from wastewater into an aquifer is not attenuated in the soil or unsaturated zone, but may be significantly diluted beneath the water table. The impact of wastewater use for irrigation on the Mezquital aquifer, Mexico is described in Box 5.3.

Elevated levels of chloride in deep public supply boreholes may be the first sign of a negative groundwater quality impact arising from wastewater infiltration. The salinity is normally associated with the wastewater itself, rather than the leaching of soluble minerals from the aquifer matrix. An exception may occur when the oxidation of pyrite produces an increase in sulphate concentrations in groundwater.

Box 5.2 Soil aquifer treatment, Arizona, USA

The city of Phoenix, Arizona has been carrying out experimental soil-aquifer treatment (SAT) systems for many years. Part of the secondary effluent from two treatment plants is used to recharge a shallow alluvial aquifer at two sites, for subsequent abstraction to irrigate fodder crops, orchards and vineyards.

The scheme at Flushing Meadows dates from 1967. The soil at this site is a loamy sand, over a sand and gravel aquifer with water table at 3 m. Six long, parallel infiltration basins are operated on a schedule of 9 days flooding and 12 days drying.



SAT infiltration basin system

The larger 23rd Avenue site has been operated since 1975. The soils lack the loamy component of the Flushing Meadows site; sandy soils pass down into sandy and gravelly alluvium with water table at 15 m. Four 4 ha basins are used in an operating schedule of 14 days flooding with water depths of 0.15-0.2 m, and 14 days drying. Average infiltration rates are about 170-330 mm/d.

Renovated water was sampled from the aquifer below the basins and after lateral movement in the aquifer by means of observation wells and the total salt content of the renovated water was found to increase slightly due to evaporation and mobilisation of calcium carbonate.

Up to 70% of the nitrogen could be removed if the flooding/drying sequence was optimised; sufficient oxygen allowed to enter during drying to oxidise ammonia and flooding periods long enough for anaerobic conditions to develop to remove nitrate. Phosphate, fluoride and heavy metals were progressively removed with increasing depth by precipitation.

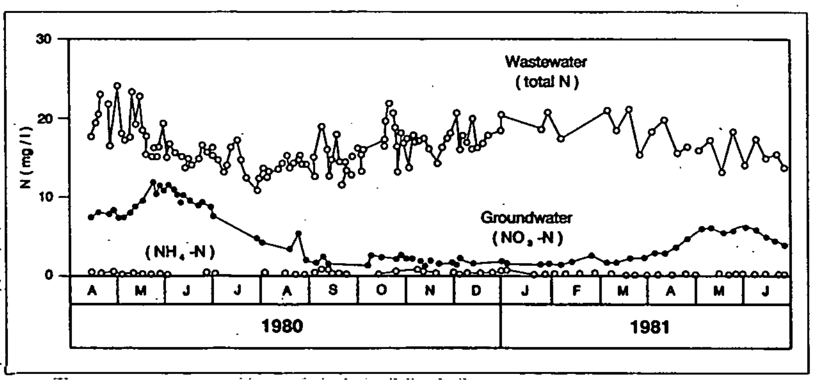
In both projects most improvements in effluent quality occurred in the unsaturated zone, but SAT was ineffective for the removal of boron due to a lack of clay.

Faecal coliforms were able to penetrate to the aquifer especially at the start of a new flooding period. Renovated water free from pathogenic bacteria and viruses could only be produced by a combination of prechlorination and SAT.

References

Bouwer H 1991. Groundwater recharge with sewage effluent. Water Science and Technology, 23, 2099-2108.

Bouwer H and Rice R C 1984. Renovation of wastewater at the 23rd Avenue rapid-infiltration project. Journal of Water Pollution Control Federation, 56, 76-83.



Effectiveness of SAT in the removal of nitrogen compounds

Box 5.3 **Use of untreated wastewater for irrigation, Mezquital valley, Mexico**

The large and rapidly growing urban areas of Mexico provide substantial volumes of wastewater which are used for irrigation. The Mezquital Valley in Hidalgo State, about 50 km north of Mexico City contains the largest and probably the oldest area of wastewater reuse for agriculture in the world. Effluent from the capital is the main source of water sustaining all development in the valley, and the enhanced groundwater resource from the recharge provides potable water supplies to about 500,000 people in the area.

The Mezquital Valley occupies a structurally-complex, elongated depression some 5-15 km wide and 15 km long. The valley floor at 1900-2100 mASL is composed of relatively thin lacustrine and alluvial deposits overlying basalts and tuffs, These form a complex regional aquifer of moderate to high transmissivity and varying degrees of confinement. At greater depths, and outcropping in the surrounding mountains are marine sediments containing highly permeable limestones and less permeable shales and sandstones. The area has a semi-arid climate with average rainfall of 500 mm/a.

The valley receives some 40 m^3/s of untreated wastewater via a canal and tunnel system. About 75% of this is empounded by the Endho Dam, and the remainder enters the Rio Salado, from where it is diverted for irrigation. A complex system of irrigation canals has evolved with the growing volume of wastewater, and now 575 km of main canals serve the valley. The Endho reservoir was designed primarily for storage purposes, but in practice provides a degree of partial treatment.

The area now comprises Irrigation District 03, covering 45,000 ha of irrigated land and 27,500 individual water users. The State authorities exert strict control over cropping to reduce the hazard of pathogen transmission to farm workers and by the marketed produce. Some 20,000 ha are cultivated with perennial alfalfa. Most of the rest is used for maize and other cereals.

The extensive and prolonged irrigation with wastewater has completely modified the groundwater recharge and flow regime in the area. Where only limited recharge occurred before, a new regional aquifer has been created by the excess infiltration from the dams, canals and irrigated land. Water levels have risen over the entire area, and major new groundwater discharge zones have been created. In these areas, drainage is now required to minimise waterlogging and salinity, and 20 m^3/s is discharged to the Tula and Actopan rivers.

The infiltration of wastewater has changed the hydrochemistry throughout the valley and the penetration of poor quality to considerable epths can be inferred.

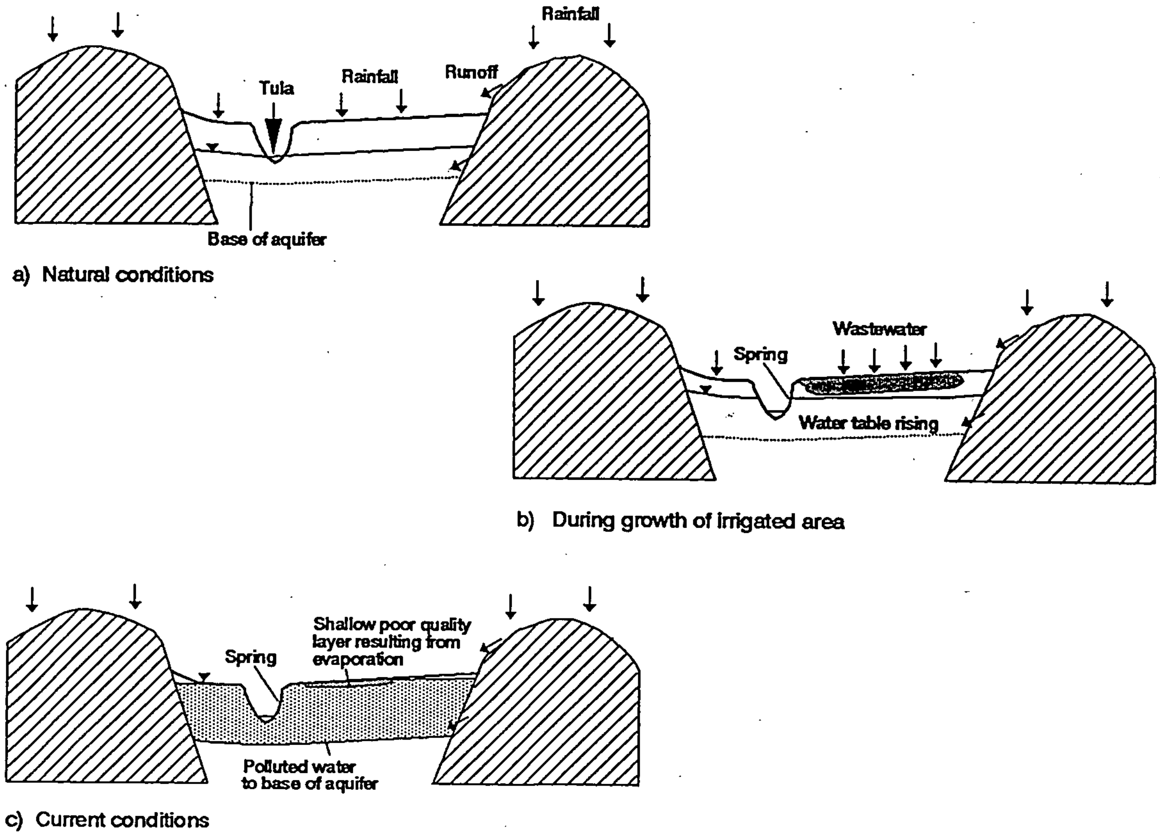
Key issues arising from these changes are the future security of public supplies and the likelihood of further land being affected by water logging and salinity.

Reference

CNA/BGS 1995. Impact of wastewater reuse on groundwater in the Mezquital valley, Hidalgo State, Mexico. British Geological Survey Technical Report WD/95/24

Evaporation at the land surface or in lagoons may produce an increase in dissolved solids before infiltration. At the Dan Region project (Box 5.4), the infiltrating water initially had sodium and calcium concentrations about 10 times greater than those of the underlying groundwater. The capacity of the unsaturated zone to remove boron, sodium and potassium by cation exchange with calcium and magnesium became exhausted, with the result that effluent and groundwater subsequently had similar concentrations (Box 5.4).

Box 5.3 continued



Conceptual model of hydrogeological evolution

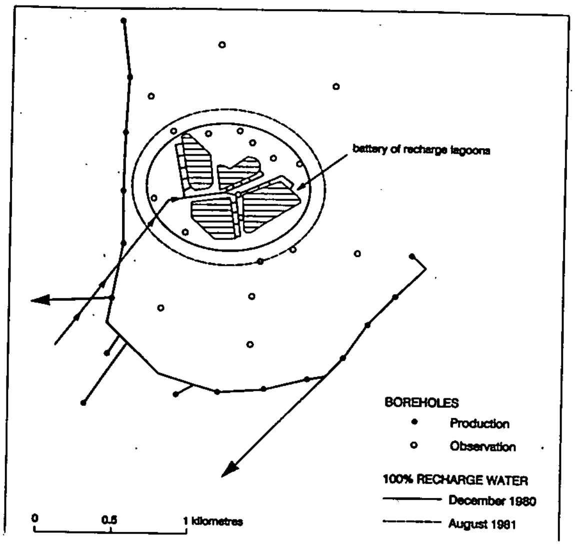
Most of the heavy metals found in wastewater originate from industrial processes. Several are considered to present health risks, being toxic at very low concentrations, and heavy metal contamination is recognised as potential problem of wastewater reuse schemes. The metals include cadmium, chromium, lead, mercury, together with selenium, arsenic and cyanide. Soils have significant capacity to remove metals from solution by cation exchange and adsorption. The mobility of heavy metals in wastewaters is a function of the pH - Eh conditions in the soil resulting from the infiltration. Where the soil pH is greater than 7, heavy metals generally form insoluble hydroxides and carbonates.

In the dune sand recharge schemes of the Netherlands, accumulation of heavy metals was observed in the soils, with concentrations falling to background levels within 1 m of infiltration (van Puffelen, 1982). Similar results were obtained from Leon, Mexico, where mixed industrial and domestic wastewater is used for agricultural irrigation (BGS et al, 1996). Leon is a prominent leather producing centre and wastewater contains high concentrations of chromium. Potable water is abstracted from lacustrine and

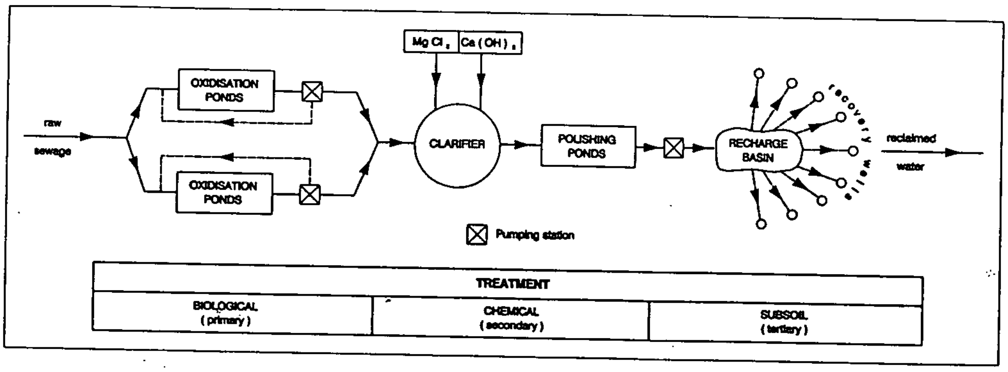
Box 5.4 Sewage reclamation project, Dan Region, Israel

The Dan Region sewage reclamation project in Israel uses the SAT principle to manage its limited water resources. Mandatory restrictions for agricultural use require the shortfall to be made up from wastewater reuse. The scheme recharges treated effluent from Tel Aviv, Jaffna and Holon to a Pleistocene coastal aquifer underlying sand dunes. The effluent is treated in oxidation ponds to improve quality through clarification, free ammonia stripping and natural recarbonation. The water infiltrates via a series of spreading basins and the renovated water recovered from a ring of boreholes around the site.

Recharge over a five year period has caused formation of a groundwater mound 8.5 m high. The salinity of the recharge water is ten times higher than the native groundwater so movement can be easily monitored.



Approximate spread of recharged wastewater



Wastewater flow diagram

Box 5.4 continued

Monitoring of groundwater showed that most heavy metals from the wastewater were reduced practically to background concentrations.

However the ability of the unsaturated zone to remove boron, sodium and potassium by exchange with calcium and magnesium was soon used up with the result that effluent and groundwater now have similar concentrations of these elements.

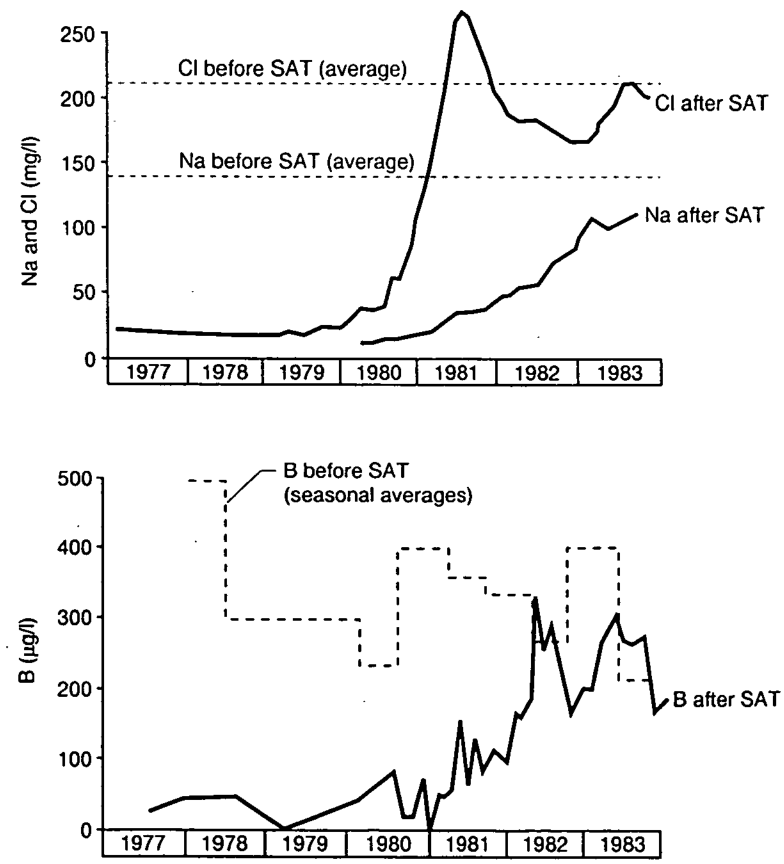
References

Idelovitch E and Michail M 1984. Soil-Aquifer Treatment- a new approach to an old method of wastewater reuse. Journal of Water Pollution Control Federation, 56, 936-943.

Shelef G 1991. The role of wastewater reuse in water resources management in Israel. Water Science and Technology, 23, 2081-2089.

Effectiveness of heavy metal removal by SAT

Parameter	Mean concentration before SAT	after SAT
cadmium	3	<1
chromium	10	<3
copper 18	5	
selenium	8	<2
nickel 38	10	



Behaviour of chloride, sodium and boron during SAT

alluvial deposits underlying the irrigated area. Other components of the wastewater, such as chloride and nitrate are now appearing in the public supply boreholes, but chromium is completely retained in the soil (Figure 5.2).

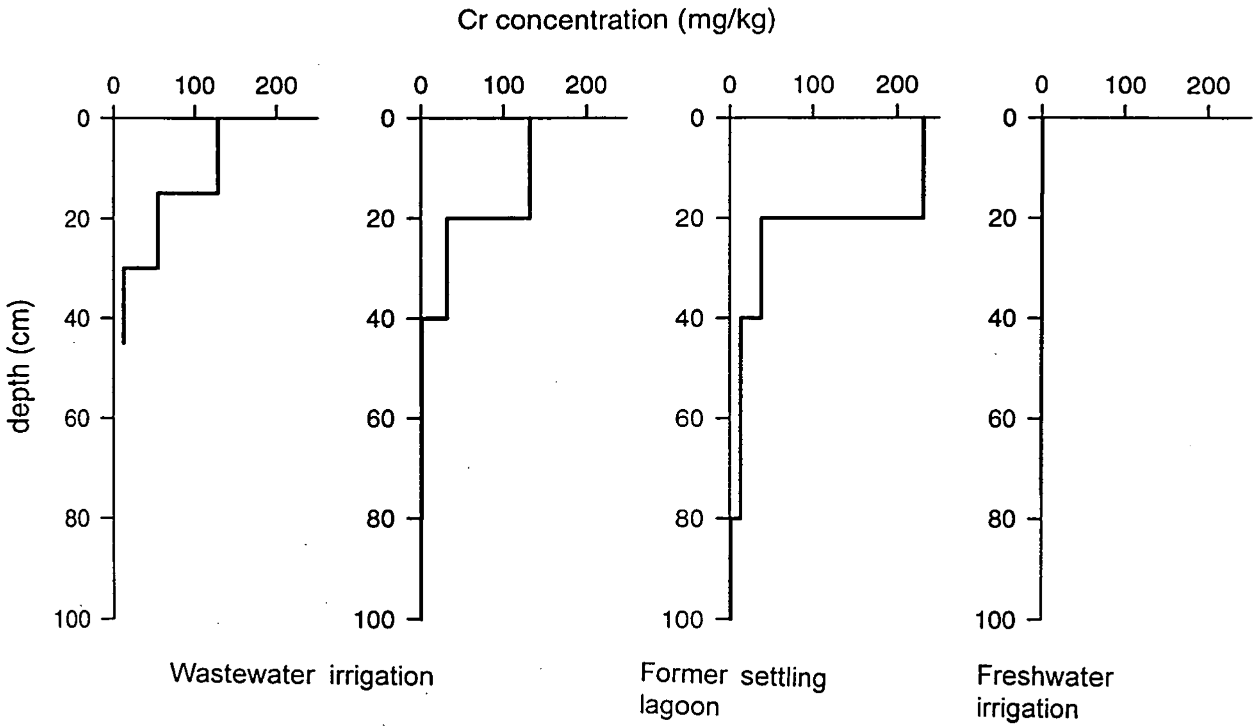


Figure 5.2 Profiles of chromium concentration in soils and former lagoon site (BGS and others, 1996)

5.5.4 *Organic constituents*

Wastewaters may contain a wide and highly variable range of natural and synthetic organic constituents, many of which are potentially toxic. Organic contaminants in wastewater include phenols, cresols, chlorinated solvents, pesticides, benzenes, surfactants and many others (Foster et al, 1994). Total and dissolved organic carbon (TOC and DOC) and biochemical oxygen demand (BOD) measurements provide an indication of the organic content of waters, and have been measured beneath a number of wastewater recharge sites. At Flushing Meadows, BOD was effectively eliminated after 3 m of infiltration through the unsaturated zone and 6 m through the saturated zone, all biodegradable carbon being readily utilised. TOC at this site was only reduced by 50 - 75% (Bouwer, 1991). In the nearby 23rd Avenue project, TOC reduced from 12-14 mg/l in wastewater to 3 mg/l at 5 m depth and 1.9 mg/l in a well drawing water from

30-54 m depth. Similar results were obtained from the alluvial sediments of the Dan Region project (Box 5.4).

The widespread association of TOC concentrations in groundwater above ambient levels with wastewater recharge suggests that a persistent fraction remains. A plume with TOC concentrations 5 mg/l or more above background is often present in the groundwater below and downstream of wastewater recharge areas (Bouwer et al, 1984). This gives cause for concern about the possible presence of the toxic trace organic compounds listed above, and the possiblity of trihalomethanes being produced following subsequent chlorination. There is little information about the health effects of the ingestion of trace organic compounds over long periods.

Analysis of specific organic constituents is required to identify the contaminants in each case. In the Netherlands, for example, 300 individual substances were identified beneath the sand dune recharge areas (van Puffelen, 1982). Studies of trace organic contaminants were also undertaken at the 23rd Avenue site in Tucson (Bouwer, 1991). Volatile organic compounds were subject to 30-70% removal in the infiltration basins, and soil infiltration removed 50-99% of non-halogenated organics, mostly by microbial decomposition. Concentrations of halogenated compounds did not decline as dramatically during infiltration, and they were found to be more mobile in the subsurface environment (Table 5.6). Bouwer (1991) concluded that SAT was effective in reducing the concentrations of many organic compounds, but the recharged water still contained sufficient trace organics to make additional treatment necessary for potable use.

At the San Juan de Miraflores site near Lima (Box 5.1), TOC concentrations in unsaturated zone porewaters are high beneath the lagoons and beneath irrigated cultivated land, but are rapidly attenuated with depth. These concentrations compare with values above 100 mg/l in the raw sewage and the effluent from the lagoon. The less-degradable detergents of the ABS type were also detected at concentrations of 1-5 mg/l beneath the lagoons.

5.6 Planning and management of wastewater reuse

The wide range of wastewater handling processes which produce recharge were described at the beginning of this chapter (Table 5.2). The selection of a particular method depends on many factors, including the availability of land, soil type, hydrogeology, financial resources, available technology, quality of wastewater and the quality requirements of any subsequent recovery of the recharged wastewater. Using methods which rely on infiltration through the soil, the nature of the soil and the required loading will determine the method (Figure 5.3). Further, Parizek (1973) classified saturated subsoils according to their suitability for wastewater treatment and recharge to aquifers. Very fine-grained soils provide good attenuation of pollutants but the rates of fluid percolation are unacceptably slow for effective infiltration. The coarsest soils permit rapid infiltration, but provide much less scope for the processes of pollutant attenuation. Highly treated wastewater will infiltrate much more readily than untreated wastewater, which may clog the pore spaces in the soil and underlying aquifer.

Planning and operating wastewater reuse schemes has to balance the two objectives of high infiltration rates and maximum quality improvement. High rates of infiltration are important to restrict the land area requirements of possible schemes, but residence times may be too short for the processes of renovation. At a site in Germany (Altman, 1982), rates of 10,000 mm/d resulted in bacteria penetrating to the aquifer in unacceptable concentrations. Published figures for long-term average infiltration rates in unconsolidated sedimentary aquifers are in the range 250-800 mm/d (Foster et al, 1994).

The wastewater handling processes listed in Table 5.2 can be considered as falling into three categories:

- infiltration basins, lagoons, ditches and drains
- irrigated land
- injection wells

The main factors controlling effective infiltration include the quality of the wastewater, the infiltration structure, the underlying aquifer and the management of the system. The hydraulic loading and periodicity of wetting and drying cycles are especially important in controlling the natural biological processes at any site. Most commonly, series of basins or ditches are used cyclically to permit restoration of infiltration capacity. At the Whittier Narrows scheme in California (Nellor et al, 1985), 279 ha is divided into 15 basins. About 750 Ml/d are disposed to one third of the basins operated in turn to give a 7-day filling, 7-day draining and 7-day drying regime. Groundwater in an alluvial aquifer is replenished in this way for local abstraction by a combination of treated wastewater, storm runoff and imported water, and no measurable impact has been detected on either groundwater quality or human health (Nellor et al, 1985).

At the Phoenix sites, secondary treated effluent is infiltrated with basins using a cyclical pattern of 9-14 days flooding and 12-14 days drying, to give average infiltration rates of 170-330 mm/d (Bouwer and Rice, 1984). Movement of the recharged wastewater in the aquifer is controlled by abstraction to irrigate fodder crops, orchards and vineyards. At the Dan Region project in Israel (Shelef, 1991), spreading basins are used with similar but shorter cycles of wetting and drying.

Injection wells have the advantage that they occupy very little land. They can be used where land is not available or where surface conditions are not suitable for infiltration. This method can be particularly suitable for coastal areas or islands, where the recharge has the dual benefit of wastewater disposal and creating a hydraulic barrier against saline intrusion. However, high quality treated wastewater is required to avoid reductions in acceptance rates caused by clogging in the injection well and/or in the aquifer close to the well.

Table 5.6 Effectiveness of soil-aquifer treatment (SAT) in removing micro-organic contaminants during wastewater recharge at Phoenix (Arizona), USA

Determinand	Mean concentration of secondary effluent (27 samples) (μg/l)	Decrease in groundwater recharge (6 samples) (%)
Aliphatic hydrocarbons		
5-2 methylpropyl nonane	0.35	>94
2,2,5-trimethylhexane	0.11	>82
2,2,3-trimethylhexane	0.21	76
2,3,7-trimethylhexane	0.12	50
Aromatic hydrocarbons		
o-xylene isomers	0.45-0.76	67-78
benzene isomers	0.48-0.56	84
styrene	0.26	>92
1,2,4-trimethyl benzene	0.80	78
ethylbenzene	0.19	53
naphthalene	0.22	68
Chlorinated aliphatic hydrocarbons		
chloroform	2.72	61
1,1,1-trichloroethane	2.94	34
carbon tetrachloride	0.91	*180
trichloroethane	2.63	*97
pentachloroanisole		
Chlorinated aromatics		
dichlorobenzene isomers	2.25-3.52	25-33
1,2,4-trichlorobenzene	0.19	42
trichlorophenol	0.01	0
pentachloroanisole	0.43	*150

*increase <u>not</u> decrease

(Derived from Bouwer et al, 1984)

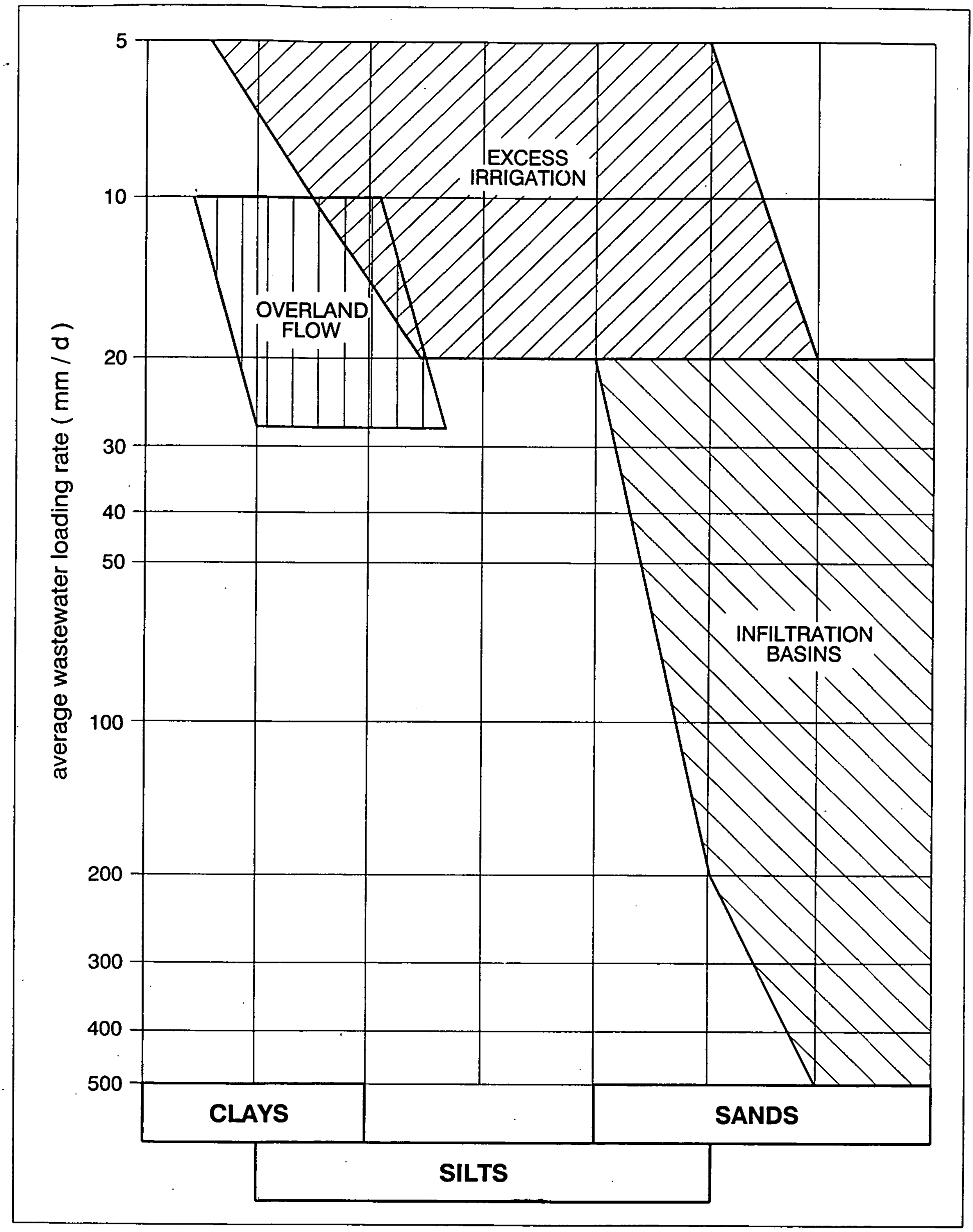


Figure 5.3 Relationship between wastewater infiltration method and soil type (Foster et al, 1994)

5.7 Investigating the impact of wastewater reuse on groundwater

Because of its value as a resource, especially in arid and semi-arid areas, reuse of wastewater is likely to be of increasing importance. The development of methods of both investigating and routinely monitoring the impact on groundwater of existing wastewater reuse schemes and evaluating or predicting the impact of new schemes is therefore necessary. An investigation of this nature will draw on many aspects of hydrogeological knowledge, and may also require the involvement of professionals from other disciplines.

Assuming the objective is to investigate the impact of wastewater reuse on groundwater, the scope is framed in the first instance by the type and quality of wastewater being reused and the use to which the underlying groundwater is put. If, as in the case of many of the schemes in Mexico, untreated urban wastewater is being spread above an aquifer from which groundwater is abstracted for potable supply, then the possibility of significant impact means extensive and detailed investigations. If treated wastewater is used, and the groundwater is abstracted for other uses with less stringent quality requirements, the potential impact is less and the investigative requirements correspondingly more modest.

5.7.1 *Characterising the wastewater reuse system*

Taking the first instance, an important first step is to characterise the wastewater by estimating the relative proportions of domestic and industrial effluents, and the types of industries involved, to assess the likely contaminants in the wastewater (Foster et al, 1994). This, together with any existing quality data, establishes the suites of determinands to be analysed and, if metals and organic pollutants are likely to be present, the degree of laboratory sophistication required for the investigation.

The next step is to gain an understanding of the way the system has grown up and operates. It is important to know the geometry of the scheme, whether there is short-term retention in lagoons, whether there are seasonal differences in quality if storm runoff also enters the system, which areas have been irrigated, for how long and by what methods. Finally, it is important to know whether any major changes are planned in the future.

5.7.2 *Characterising the hydrogeology*

The first step is to use existing information to describe the geological setting, types and characteristics of aquifers and the way in which groundwater occurs and moves. In thick UNSA sequences, more than one aquifer may occur, separated by less permeable material, and there is a possibility of perched bodies of poor quality groundwater being formed as a result of the enhanced recharge from wastewater infiltration. Water levels provide an indication of the thickness of the unsaturated zone available for pollutant attenuation and the directions of regional groundwater flow and, where time series are available, may provide an indication of the impact of the wastewater. Knowledge of historical, current and future groundwater abstraction is required.

In reuse schemes using lagoons, ditches or drains, it is clear how and where the recharge is occurring. For large wastewater irrigation schemes, this may not be so simple. There may be several sources of recharge, from canals, storage reservoirs and from the irrigated fields, and each may have somewhat different quality characteristics. Detailed studies of water levels to determine recharge sources and evaluations of canal losses may be required to quantify the various sources of recharge, and geophysical methods may be needed to assist in determining whether perched aquifers are present.

5.7.3 *Assessing the impact*

Sampling of existing wells and boreholes for major ions, metals, pathogens and organic compounds (as required according to the wastewater) may provide an overall picture of lateral variations in groundwater quality and the extent of the impact. Choosing wells or boreholes completed at different depths in the aquifer sequence may provide a preliminary indication of quality variations with depth and the penetration of poorer quality infiltrating water. In most situations the scope for existing sampling points to provide a picture of groundwater quality with depth is severely limited, and purpose-constructed investigation boreholes will be required to provide a more detailed representation of vertical quality.

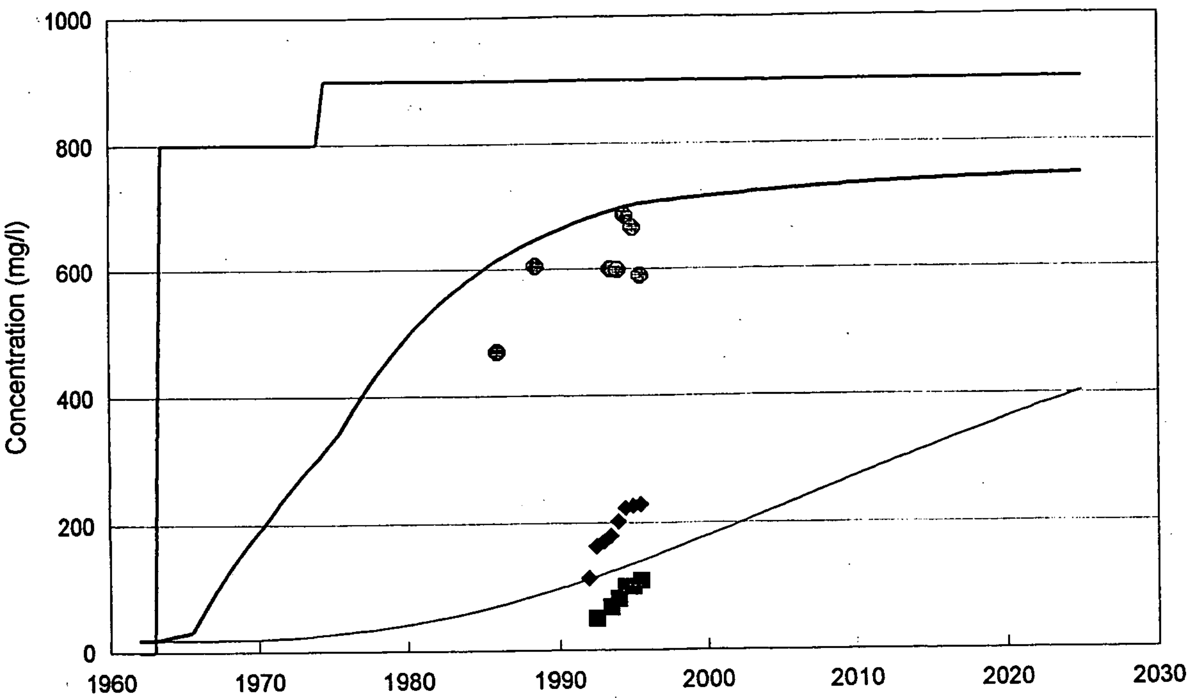
Application of flow modelling may give some insight into the rate of change of water quality, and can be extended to allow prediction of the impact of water management options (BGS et al, 1996). Figure 5.4 shows the prediction of chloride concentrations in the aquifers beneath Leon, Mexico using a spreadsheet model. The upper figure shows the water quality in the lower aquifer, which is abstracted for potable supply, continuing to decline if current practices are continued. The lower figure indicates the possible beneficial consequences of removal of high chloride industrial effluent from irrigation water combined with additional pumping from the shallow aquifer.

5.8 Conclusions

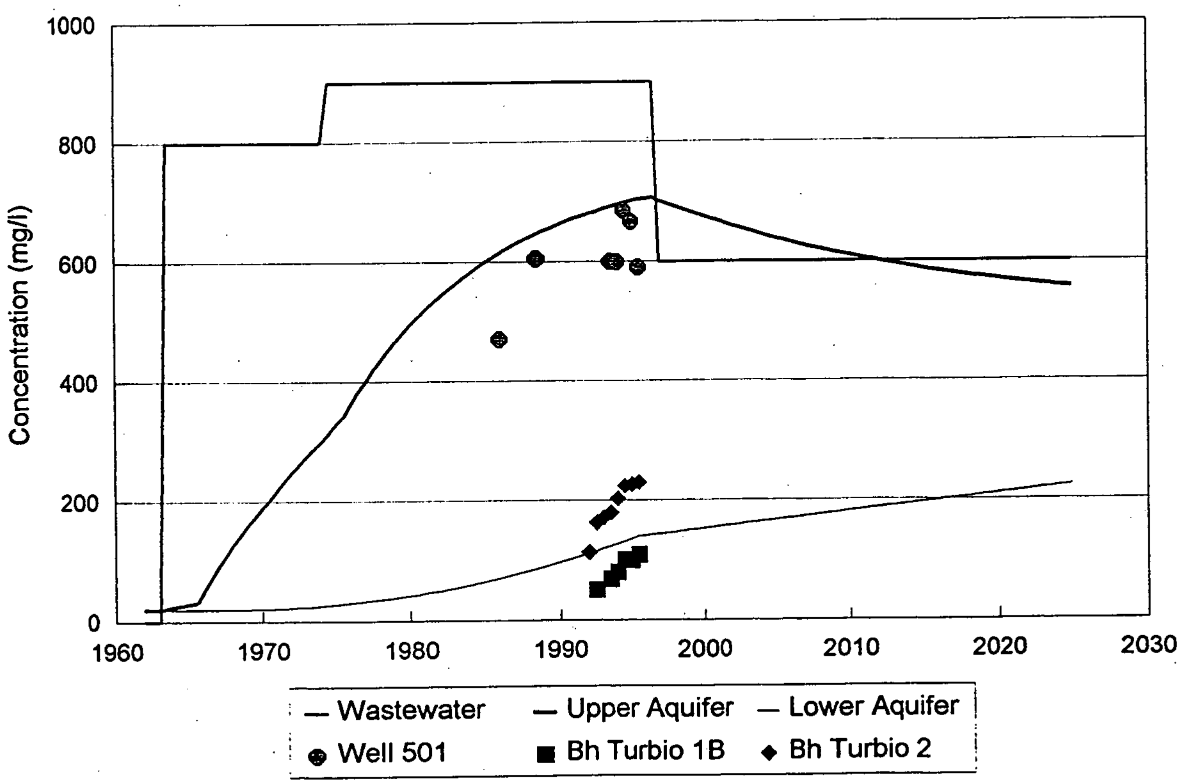
Wastewater reuse often has major impacts on groundwater, especially where relatively shallow phreatic aquifers are present. In some hydrogeological conditions, for example complex UNSA sequences, local perched aquifers may be created.

The impacts are simultaneously positive in terms of water resource conservation and negative from groundwater quality considerations. However, since aquifer recharge is an incidental consequence of many wastewater handling practices, their design and operation are not optimised with regard to maximum benefit from resource recovery or minimum groundwater quality deterioration. There are high public health risks associated with uncontrolled wastewater irrigation and many practical difficulties in controlling irrigation and cultivation practices. For these reasons, there are strong arguments for using aquifer recharge to improve microbiological quality before reuse for agricultural irrigation.

Infiltration has been demonstrated to be an effective method of treatment for wastewater. The slower and less continuous the infiltration, the better the quality of the recharge is likely to be. The soil and unsaturated zone are capable of removing some but not all contaminants. For pathogens, the lower the hydraulic loading and the smaller the infiltration rate, the more effective is the biopurification. Protozoa and bacteria are normally eliminated completely, but viruses may be more persistent and mobile. The risk of deterioration of microbiological quality is unlikely to be the main constraint on wastewater recharge.



Continuation of present abstraction and irrigation regimes



Additional pumping from upper aquifer and treatment of industrial effluents

Figure 5.4 Prediction of future water quality in the Leon aquifer, Mexico (BGS and others, 1996)

The total nitrogen content of wastewater is high, and large-scale wastewater recharge can produce unacceptable concentrations of nitrate or ammonium in groundwater. Continuous discharge of untreated wastewater produces anaerobic conditions permitting ammonium to reach the water table. If more oxidised secondary effluent is applied at low rates, nitrification occurs and nitrate is transported downwards with the recharge. Cyclic applications which allow alternate wetting and drying can be used to control and reduce nitrogen concentrations in recharge.

Heavy metals originating in the wastewater are likely to be adsorbed within the system itself or at shallow depths in the soil. The mobility of metals depends on the pH-Eh conditions imposed by the wastewater irrigation. Only in markedly acidic conditions is transport of heavy metals likely.

Wastewater infiltration usually causes some increase in groundwater salinity, and chloride may provide an early indication of the impact of wastewater reuse.

Infiltration through the soil is also very effective in removing organic carbon. Nevertheless, a plume of elevated organic carbon concentrations is often observed in groundwater below and down gradient of wastewater recharge areas. This suggests a persistent fraction remains, possibly containing toxic trace organic compounds.

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METHOD SUMMARY SHEET (WQM 06)

TITLE: Investigating the impact of wastewater reuse

Scope and use of method

Methods for the investigation and monitoring of the impact on groundwater of existing wastewater reuse schemes and the evaluation of new schemes draw on many aspects of hydrogeological knowledge, and may also require the involvement of professionals from other disciplines. The scope is framed by the type and quality of wastewater being reused and the use to which the underlying groundwater is put. If wastewater is being spread above an aquifer from which groundwater is abstracted for potable supply, then the possibility of significant impact means extensive and detailed investigations. In particular, the depth of penetration and the timescale of downward transport of the various pollutants contained in the wastewater are of critical importance in assessing the impact on deep potable supply boreholes. The complex hydrogeological conditions in many UNSA sequences are important in determining vertical movement of pollutants, and these conditions need to be well understood. If treated wastewater is used, and the groundwater is abstracted for other uses with less stringent quality requirements, the potential impact is less and the investigative requirements correspondingly more modest.

Method

A phased approach is recommended, in which the first stage is to characterise the reuse system:

- Collect information on wastewater quality and estimate the relative proportions of domestic and industrial effluents, and the types of industries involved, to assess the likely contaminants in the wastewater.

- Collect information to gain an understanding of the way the system has grown up and operate: the geometry of the scheme; short-term retention in lagoons; seasonal differences in quality; entry of storm runoff; areas which have been irrigated, for how long and by what methods; and finally if any major changes are planned in the future.

Then the investigation should characterise the hydrogeology:

- Collect information on the geological setting, aquifer types and characteristics and groundwater occurrence and movement. Water levels provide an indication of the thickness of the unsaturated zone available for pollutant attenuation and the directions of regional groundwater flow. Historical, current and future groundwater abstraction is also required.

- Detailed studies of water levels in large schemes to determine recharge sources and evaluations of canal losses may be required to quantify the various sources of recharge, and geophysical methods may be needed to assist in determining whether perched aquifers are present. The vertical distribution of lithologies and permeabilities indicates whether this is likely, and knowledge of these will be needed for assessing the risk of deep penetration of pollutants within the aquifer sequence.

At the same time the impact on groundwater can be assessed:

- Sample existing wells and boreholes for major ions, metals, pathogens and organic compounds (as required according to the nature of the wastewater) to provide an overall picture of lateral variations in groundwater quality and the extent of the impact. Choose wells or boreholes completed at different depths in the aquifer sequence to provide an indication of quality variations with depth and the penetration of poorer quality infiltrating water.

- In most situations purpose-constructed investigation boreholes will be required to provide a more detailed representation of vertical quality, and perhaps of vertical variations in piezometric head and vertical flow.

- The hydrochemical results are interpreted to provide and indication of current impact. High concentrations of major ions compared to background are likely to be indicative of impact, and chloride and bicarbonate may be particularly good indicators in this respect. Chloride/nitrate ratios may be illustrative in relation to sources of recharge and impacts of urban wastewater.

- Trends can be examined with a view to evaluating future evolution of groundwater quality. Modelling may be required to predict the outcome of a range of options for improved management of the wastewater irrigation.

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6. SALINITY AND SALINE INTRUSION

6.1 Introduction and scope

Increasing salinity is one of the most significant and widespread forms of groundwater pollution. This chapter reviews current knowledge of the impact of saline intrusion on groundwater quality by describing the causes of saline intrusion, summarising the approaches determining the extent of existing saline intrusion, and discussing appropriate techniques to identify the source of saline water. Management and control measures to reduce or prevent the problem are outlined. The increase of salinity in groundwater due to irrigated agriculture is discussed in Chapter 4 of this review.

6.2 Background

During the latter part of this century there has been a widespread increase in urbanisation. As many major cities in the developing world are situated on the coast, and many lie on unconsolidated aquifers, this has placed increasing importance on coastal UNSAs for water supply. As little as 2% seawater in freshwater can render the water unpotable, and saline water has been reported to be the most common pollutant in fresh groundwater (Todd, 1980). The problem of saline intrusion requires the application of specific management techniques.

The term saline intrusion specifically describes the situation where modern seawater displaces, or mixes with, freshwater within an aquifer in response to a change in the hydrogeological environment. The expression is, however, frequently used to describe any case where water bodies of differing salinities occupy the same aquifer system. The relationship between fresh and saline water is a complex physico-chemical one and, although the general principles of the fresh water/salt water relationship are known, there are numerous complicating factors.

Saline water may be derived from many sources:

- seawater intrusion into an aquifer
- upconing of ancient seawater (connate water)
- water concentrated by evaporation
- *in situ* mineralisation.

Saline intrusion, in the strict sense, occurs as a result of modern sea water encroaching into a coastal aquifer. If groundwater gradients are reduced (as may happen in coastal aquifers where excess pumping has disrupted the hydraulic equilibrium), the outflow of freshwater is reduced and denser saline water may displace the fresh water within the aquifer.

Examples of other sources of saline water are listed above. Under many UNSAs there are substantial stores of connate or ancient waters. These may date from the time of deposition, if the sedimentary environment was a marine one, and may become a problem if the overlying deposit is pumped and upconing occurs. Salinisation, due to the accumulation of minerals in the soil, can also result from climatic factors and agricultural practices. Firstly, irrigation water or rainfall in arid environments may be rapidly transpired, leaving most of the solutes in the soil. Secondly, if the water table is shallow, direct evaporation from the capillary fringe may lead to a build up of salts: this can result from removal of deep rooted vegetation, or excessive irrigation without sufficient drainage. These problems are covered in more detail in Chapter 4 on agricultural impacts on groundwater quality. Salinisation may also occur in deeply circulating meteoric waters which may increase in salinity by reaction with the rocks they are circulating through e.g. evaporites (Freeze and Cherry, 1979).

A further chemical reaction that may be important is caused by the mixing of chemically different types of water which may cause a change in hydraulic properties (Goldenberg et al, 1986). Goldenberg (1985) reported that a decrease in hydraulic conductivity of the interface zone may occur due to deflocculation and dispersion of small amounts of reactive clay minerals (montmorillonite) in the rock. A similar effect has also been reported when freshwater was injected into a salt water aquifer, and led to the failure of recharge wells (Custodio, 1985).

6.3 Nature and extent of saline intrusion

In order to manage a saline intrusion problem effectively, it is essential to understand the source of the saline water, the extent of the problem, and the hydraulics of the groundwater system. There are several approaches to identifying the source of the water and the extent of the problem; these are outlined in Section 6.4. Only the management of salinity problems induced by seawater intrusion will be covered in this chapter.

6.3.1 *Causes of seawater intrusion*

Seawater intrusion is caused primarily by the reduction of fresh water discharge to the sea as a result of groundwater abstraction. The water is derived from one of three sources: drainage from the water table; release from elastic storage; and drainage at the seawater/fresh water interface, where fresh water is replaced by salt water (Essaid, 1986). The salt water wedge consequently moves inland, flushing decreases, and the thickness of the wedge increases. If some fresh water flow is maintained at the coast, a new equilibrium eventually develops.

Groundwater abstraction above the interface causes upconing. Depending on aquifer characteristics, well penetration, and pumping rate, a stable situation may be attained, where salt water does not reach the well. However, when this critical state is exceeded, salt water enters the well, mixes with the fresh water, and the quality of the discharge decreases.

Abstraction is not the only control on the position of the saline water/fresh water interface. A number of other factors can affect its position including:

- seasonal changes in natural groundwater flow
- tidal effects
- barometric pressure
- seismic waves
- dispersion
- climate change - global warming and associated sea level rise.

Some of these have short term implications (tidal effects and barometric pressure) some are periodic (seasonal changes in groundwater flow) and others are long term (climate change and artificial influences).

6.3.2 *Extent of the problem*

The extent of the problem must be visualised in three dimensions - the depth from surface to the interface and the areal extent of the saline body. In the simplest case, the salt water forms a wedge, tapering to an apex. The length of the wedge depends on the permeability of the aquifer, rate of freshwater flow, hydraulic boundaries, and the relative densities of fresh and salt waters. In heterogeneous aquifers, such as many UNSAs, the situation is complicated by the existence of preferential flow paths and confining layers (Figure 6.1). Coastal UNSAs throughout the world are affected by saline intrusion. Some of these are highlighted in boxes 6.1-6.3.

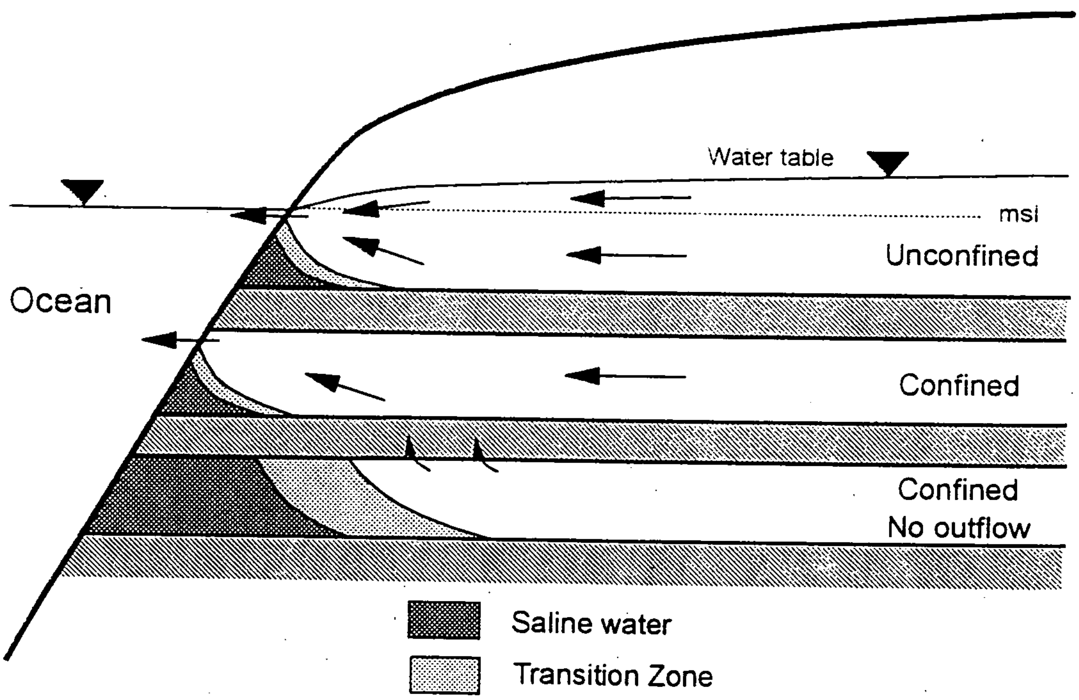

Figure 6.1 Saline intrusion in a multi-layered aquifer

6.3.3 *Management of the problem*

The control of saline intrusion demands knowledge of the hydraulic conditions within the aquifer; it also demands knowledge of the source of the saline water. It is therefore necessary to identify the extent of the problem, and to assess the behaviour of the saline water body under various conditions of recharge and discharge, such that efficient water resources management can be implemented.

The optimum solution to the problem of saline intrusion is prevention, with aquifer management being such that the encroachment of sea water is controlled to an acceptable degree. However, in many cases the problem is a legacy of the past, and management must concentrate on minimising further intrusion, and/or reducing the extent of the existing saline water. It may be that the aquifer in question is too badly polluted for its reclamation to be considered viable. In such cases, abandonment of the resource for potable water supply is the only option, although the water may still be utilised for certain industrial or agricultural applications.

In the case of UNSAs, many thick aquifers are multi-layered, and there may be alternating aquifers and aquicludes. In these situations, the deeper, confined aquifers are frequently developed after the upper aquifers have become polluted with saline water. However, there is then the risk of leakage occurring from upper aquifers as drawdown increases in the lower aquifer, and the head difference increases. There is also a potential threat to water quality in the lower aquifers if wells have not been completed satisfactorily. Such wells may provide a transmission path from the upper polluted aquifer to the lower non-polluted ones.

The problems of saline intrusion cannot be ignored. The implications in terms of public water supply include the abandonment of sources and/or reduction of abstraction, introduction of other alleviation measures, or desalination.

6.4 Methods of investigation

6.4.1 *Determination of extent of a saline water body*

The location of the sea water/freshwater interface is important to groundwater management in coastal regions. As will become apparent from the following sub-sections, the exact locations of the freshwater/saline water interface and the transitional zone are difficult to determine, either theoretically or by field testing.

There are many approaches to delineating the depth to, and extent of, a saline water body. First it is necessary to define what is meant by saline water. Terms relating to the degree of salinity were suggested by the USGS (see Table 6.1).

Saline and fresh water are miscible fluids. When water bodies with differing salinities are in contact, molecular diffusion causes mixing across the line of contact. Diffusion coefficients are relatively high in unconsolidated deposits, and therefore a wide zone of mixing would be encouraged. The effects of diffusion are exacerbated when groundwater and seawater are in motion, and the intergranular structure of the formation causes dispersion (Figure 6.2). External influences such as tides, recharge events and pumping will cause movement of the interface and encourage mixing, thus increasing the interface thickness. Thus the processes of diffusion and dispersion result in a transition zone where the salinity gradually changes from completely fresh to fully saline, the thickness of the zone depending on these two components. In permeable coastal aquifers which are subjected to heavy abstraction, the zone may attain a thickness of up to 100 m (Bowen, 1986).

Table 6.1 Terms describing degree of salinity as used by USGS

Description	TDS (mg/l)
Fresh	< 1000
Slightly saline	1000 - 3000
Moderately saline	3000 - 10000
Very saline	10000 - 35000
Brine	> 35000

After Hem (1985)

As, in reality, there is no sharp interface, a threshold level must be decided, at which water is considered saline. When investigating the use of geophysical and chemical techniques, Tellam et al (1986) used the horizon at which salinity started to rise near the saline water body.

There are no significant differences between water table and confined aquifers, and the same principles apply. However, in water table coastal aquifers, fresh water is able to discharge freely to the sea. Under confined conditions, a finite head of fresh water is needed to compensate for the denser sea water column at the submarine outlet (Custodio, 1985). If this situation occurred, no fresh water flow could occur, and the sea water would act as a plug; the mixing zone would therefore expand by diffusion and dispersion, aided by tidal and seasonal fluctuations, and a long, thick mixing zone would develop.

The presence of saline water in an aquifer is established either by the measurement of conductivity, or from chemical analysis. Historical data may be available in some areas. Such records may include partial chemical analyses or observations of the existence of saline water, indicating that a saline intrusion problem existed in the past, and may still exist. Other indications that a problem existed include records stating that a borehole was partially backfilled to improve quality, or abandoned because of deteriorating water quality.

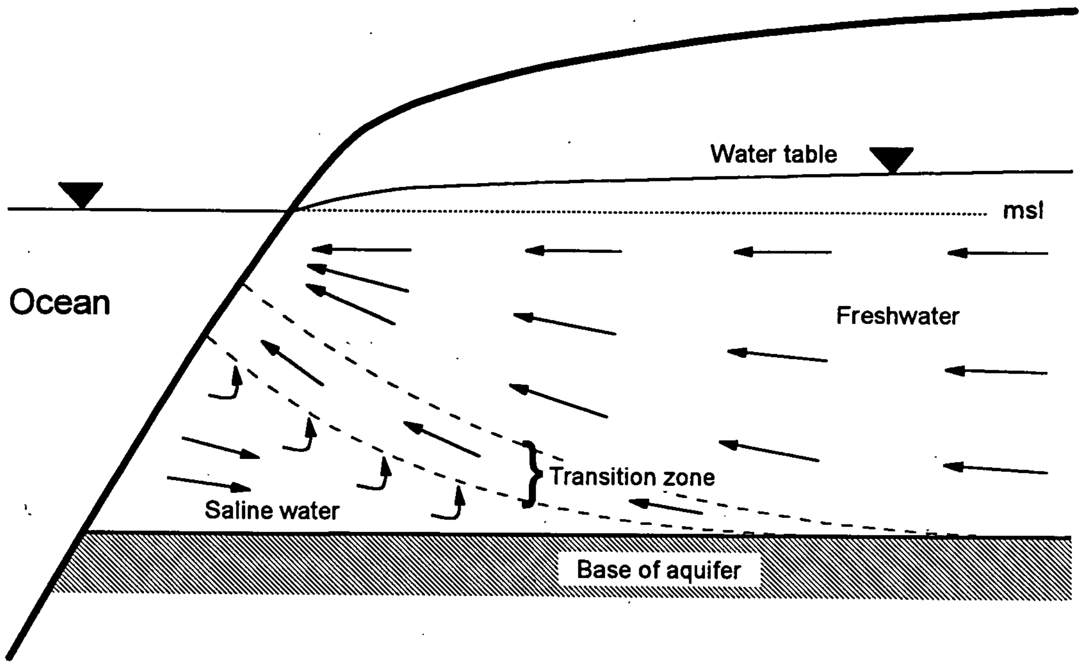

Figure 6.2 Development of a transition zone due to groundwater and saline water movement

6.4.2 *Sampling*

Samples of groundwater may be obtained by various means. In the interpretation of the results of chemical analysis particular attention should be paid to the method of acquisition of the samples. It should be stressed that aquifers are complex, three dimensional flow systems, and that the placement of boreholes within these systems can complicate the hydraulics even further. Methods and their problems are briefly discussed below and a summary given in Table 6.2. Although the methods will be briefly introduced here, they are is covered in greater detail in existing publications (Foster and Gomes, 1989; Nielsen, 1991).

Table 6.2 Summary of groundwater sampling techniques

Sample Method	Advantages	Drawbacks
Bailed	Easy operation, cheap	Only samples top of water column, making interpretation difficult or impossible
Depth sample	Some depth discrimination possible	Samples may not be representative of sample depth; may disturb water column
Pumped sample	Provides composite sample; cheap	Sample may not be an 'average' of the borehole
Pore water	Detailed, specific samples	Expensive and difficult
Packer	Detailed, specific samples	Good knowledge of vertical profile required for accurate siting; uncertainty about seal in rough boreholes; relatively expensive and difficult to use
Geochemical logging	Continuous monitoring of pH, SEC; relatively cheap	

Grab samples include bailed samples and depth samples. The former are obtained using a bucket or bottle attached to a line. Although very simple, the method only samples the top of the water column, and the results may be difficult or impossible to interpret. Depth samples are obtained from various depths in the water column using devices with valves that can be remotely closed at a certain depth. These provide some depth discrimination; however, it should be noted that this approach samples the artificial regime in the borehole. Rushton (1980) indicated that, if influenced by abstraction, the apparent interface within the borehole does not necessarily coincide with the actual interface within the formation, i.e. if there is vertical flow within the borehole (Foster and Gomes, 1989), the sample will not be representative of the aquifer at that depth. Therefore, boreholes will only reflect the true position of the interface under conditions of horizontal flow.

Pumped samples are generally thought to provide an average sample of the water in the borehole (Kashef, 1971). However, the composition of the sample is a function of the heads of the contributing aquifers, which is not therefore a true average of the contributing aquifers. In addition, such samples do not aid in detecting the transition between salt and fresh water. These problems can be overcome by using packers to obtain samples from precise intervals, although care should be taken in screened boreholes where water may bypass the packered interval (Andersen, 1978).

Porewater abstracted from cored samples probably provides the most accurate data (Tellam et al, 1986). However, there may be problems in obtaining such samples from unconsolidated formations.

6.4.3 *Chemical analysis for saline water*

The two most widely used approaches to detect the presence of saline water are measurement of conductivity and chloride concentration. Tellam et al (1986) suggested the use of simple measures of conductivity to identify horizons where salinity was increasing. However, natural waters are not simple solutions, but contain a variety of ionic and undissociated species, the amounts and proportions of which may range widely. Therefore conductivity cannot be simply related to ion concentrations or dissolved solids (Hem, 1985). However, in practice, when investigating salinity problems known to be a result of seawater intrusion conductivity is probably a reasonable indicator parameter.

Determination of chloride is also widely used. Chloride is a conservative element and (excepting deep groundwaters) is usually only found at low concentrations in groundwater, but is the dominant anion in seawater. In contrast, bicarbonate is the most abundant anion in groundwater and but not in saline water. Revelle (1941) proposed that the chloride-bicarbonate ratio ($Cl^- / \{CO_3^{2-} + HCO_3^-\}$) could be used to evaluate saline intrusion. Although this approach will distinguish saline from fresh water, it cannot differentiate between different sources of saline water (see Section 6.5).

Cation determination can also provide an indication of the dynamics of saline intrusion. Under conditions of saline intrusion, Na in seawater replaces Ca adsorbed onto the surface of clays. This results in relative depletion of Na in seawater, and Na concentrations plot below the seawater/freshwater mixing line. When seawater is flushed by groundwater, Ca expels adsorbed Na (and other cations), causing a relative surplus of Na (Geirnaert and Laeven, 1992). Therefore samples from the mixing zone may provide data to indicate whether the saline front is advancing or retreating.

6.4.4 *Theoretical methods*

The following calculation methods use formulae to describe the interface depth and saline wedge penetration under steady state conditions and simple geometric configurations. The principles are outlined below, and references given for further detail. Various workers have arrived at different solutions based on different assumptions. If the possible influence of dispersion, diffusion, tidal influences, barometric effects, evaporation, leakage from and into confining beds, earthquakes, non-steady flow, etc., are considered, it can be appreciated that any solution will be complex. Generally accuracies are acceptable for most practical purposes (Custodio, 1985), although errors may become significant close to the coast and under transient flow conditions when vertical flow components become important.

Sharp interface

Early work discovered that saline water occurred underground, not at sea level, but at a depth below sea level of approximately 40 times the height of fresh water above sea level (Drabbe and Baydon-Ghyben, 1888-89; Herzberg, 1901). The phenomenon is due to a hydrostatic equilibrium existing between the two fluids with different densities, and is described by the Ghyben-Herzberg equation:

$$h_s = \frac{\rho_f}{\rho_s - \rho_f} h_f \tag{1}$$

where ρ_s is the density of saline water, ρ_f is the density of fresh water, h_s is the depth to the interface below mean sea level, and h_f is the height of the potentiometric surface above mean sea level (Figure 6.3). In general, a specific gravity of 1.025 is used for sea water, which gives the following:

$$h_s = 40\ h_f \tag{2}$$

The method requires only one piezometer to measure the head of the freshwater zone with respect to sea level, and to obtain a measure of the density of the fresh water. If the density of the saline water within the formation is also known, the G-H formula can then be applied.

The above equation was developed for an unconfined, coastal aquifer, but may be extended to confined aquifers by substituting the piezometric surface for the water table. In both cases, the solution requires that the appropriate surface is above sea level, and is inclined towards the coast.

The above equations have limitations. The Ghyben-Herzberg principle assumes hydrostatic conditions, and only applies where there is no vertical component of flow, the saline water is stationary, and there is no zone of mixing. It has been recognised that the calculated position of the interface actually lies above the true curve, particularly where the groundwater velocities are relatively high (Kashef, 1977). The curves become close moving inland, and are virtually identical when the natural groundwater velocity is very small (Kashef, 1977; Adams, 1980) (Figure 6.4). It has also been recognised (Herzberg, 1901) that the equations are strongly affected by the grainsize of the sediments being studied.

Where fresh water flow to the sea takes place, the hydrostatic assumptions of the G-H relationship are not satisfied. In particular, it does not allow for a seepage face at the coast. Other approaches, for example Hubbert (1940), have attempted to allow for steady-state outflow to the sea.

This approach requires a second piezometer located in the saline water, in which the density and head of the saline body may be measured. In this way, allowance is made for movement of the saline water body. However, if the transition zone is thick and/or there is significant vertical flow, Hubbert's formula is not applicable.

More exact formulae to calculate the position of the interface exist. However, considering the number of factors, such as formation heterogeneities, that can affect the dynamic equilibrium between fresh and salt water, their use would not, in general, be justified.

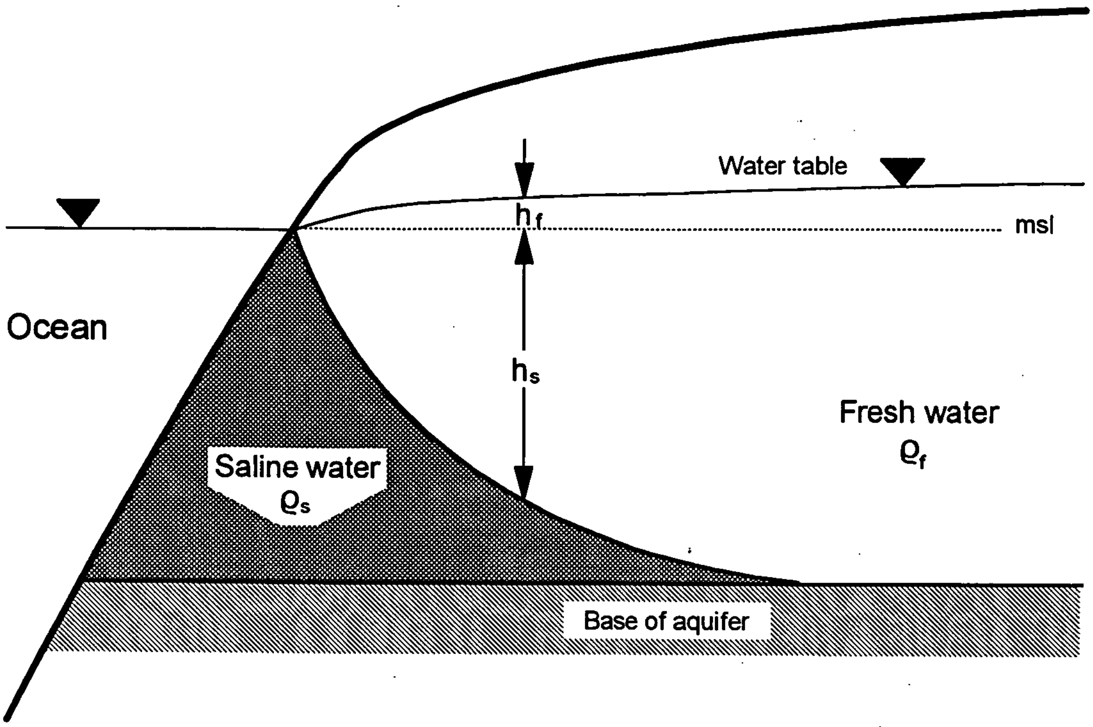

Figure 6.3 Parameters used in the Ghyben-Herberg equation

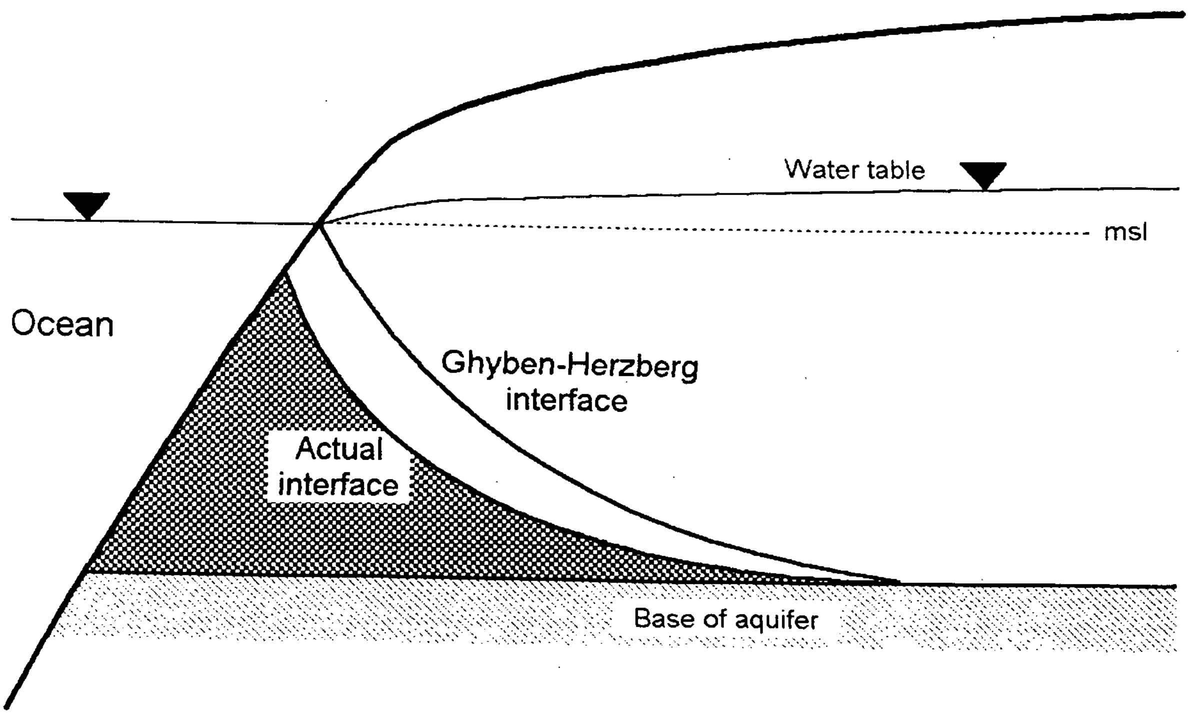

Figure 6.4 Actual and calculated positions of the saltwater/freshwater interface

Diffuse interface

The above approaches assume that the interface separating fresh and saline water is a sharp boundary. However, the interface is not distinct, and there is a brackish water transitional zone where there is mixing of the two types of water.

Where the zone is narrow, the above methods may provide a satisfactory solution; however, an extensive zone of diffusion may affect the flow pattern and the position of the interface, and should be taken into account. Henry (1960) presented a mathematical solution for a steady-state situation, which included the consideration of dispersion. Cooper et al (1964) summarise various analytical solutions. In terms of modelling, several codes have been developed to solve areal flow and transport equations, incorporating the effects of dispersion (e.g. Segol et al, 1975).

Most work has concentrated on determining the position of the interface under steady state conditions. The management and control of saline intrusion problems generally requires more detailed data than the location of the natural interface under idealised conditions. This can be particularly important for heterogeneous aquifers which exert significant influences on groundwater movement under transient conditions. Andrews (1981) developed a finite element dispersion model to solve the flow and transport equations in coastal aquifers that were being pumped; he applied this model to the Costa de Hermosillo aquifer in Mexico. This was felt to be of particular use in the study of areal intrusion when the effects of dispersion cannot be ignored.

Modelling

Modelling may provide a useful tool to assess the extent of saline intrusion, with many models being developed over recent years (see Appendix 1). It is beyond the scope of this review to discuss modelling in detail, and for greater detail, the reader is directed to one of the many references available (Bobba, 1993). The application of modelling techniques to the Bangkok aquifer is described in Box 6.1.

Saline intrusion problems are intrinsically complex due to the number of interacting phenomena. It is therefore difficult to acquire the detailed understanding needed to effectively manage such situations. Several quite common real-world occurrences do not lend themselves to conventional analytical techniques. Numerical models are therefore valuable tools for quantifying the hydraulic parameters and testing conceptual models of the aquifer system.

The general concepts of mathematical modelling are dealt with in the modelling review. As with most modelling exercises, saline intrusion models usually incorporate certain simplifying assumptions and approximations. These may prejudice the accuracy of the results but such simplifications are often necessary because of limitations in the available field data for the system. To be useful, the numerical model must be based on an adequate conceptual model that incorporates all the mechanisms relevant to the situation under study.

Box 6.1 Modelling of saline intrusion in Bangkok, Thailand

Bangkok is situated on the deltaic Central Plain of Thailand, which was formed on a fault/flexure depression which became infilled with up to 1800 m of clastic sediments. The Plain is relatively flat, and the land surface elevation in Bangkok is only 1 to 2 m above sea level. These sediments form a complex, multi-layered aquifer system comprising alternating clay, sand and gravel layers. Heavy groundwater abstraction, from both private and municipal wells, resulted in the potentiometric surface in all aquifers dropping from artesian to around 45 m below ground level by 1979. Due to this decline, water of poor quality was drawn into the main producing areas, and groundwater quality deteriorated.

From the early 1980s, work has been carried out to investigate the source of saline water in order that remediation could be instigated to reduce the extent of contamination. Potential sources considered were: sea water from the Gulf of Thailand, saline water trapped in sediments during deposition, and inundation of sediments with saline water at times of high sea level during past geological times. Chloride concentrations suggested that contamination was occurring on the whole from scattered sources to the west and south west, with limited contribution from the Gulf of Thailand.

The setting up a of monitoring network was considered to be too costly and probably impractical for an aquifer of this size; consequently a mathematical model was used to evaluate the regional extent of contamination in the lower aquifer and thus to plan an efficient monitoring network for critical areas. Results indicated that connate water bodies were more significant in the contamination of the aquifer than sea water. However, movement of saltwater in the model was less than that observed from field data due to infiltration from the overlying, brackish Bangkok aquifer, possibly via abandoned wells.

A solute transport model was developed to assess the effects of different pumping regimes on salt water encroachment. The model comprised two components: the first part dealing with groundwater flow, and the second using predicted head distributions to estimate the transport of solute. After calibration of the model, the analysis was extended to predict future advancement of the saline water under various groundwater abstraction schemes. The predictions indicated that phasing out of pumpage for public water supply, and reduction in abstraction from private wells would be required to halt the rate of groundwater level decline, and hence reduce the rate of water quality degradation. Without proper management of the problem, saline intrusion will become an increasing problem.

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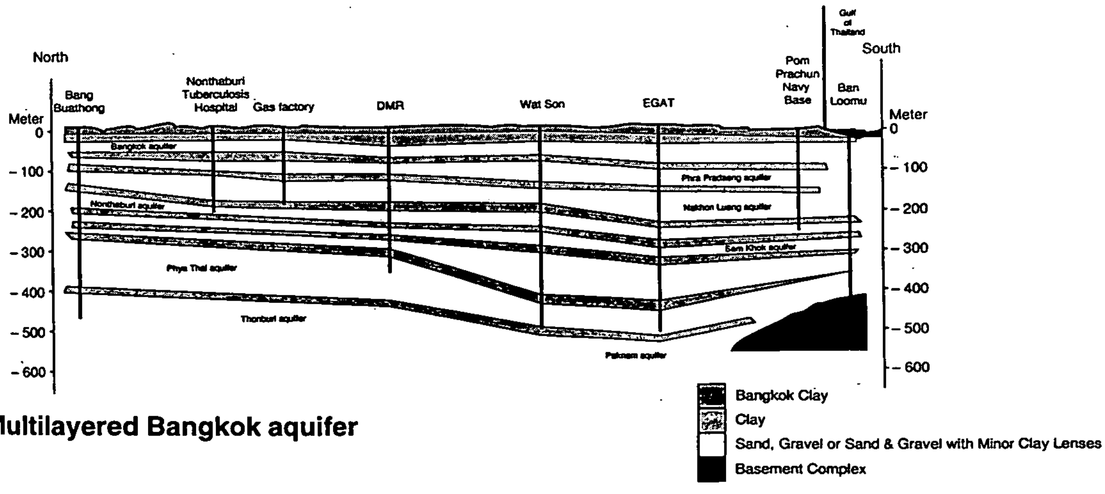


Multilayered Bangkok aquifer

The dominant mechanism of saline intrusion is mass transport: the salt in solution is carried with the water by advection. In addition to the normal flow equations, saline intrusion models usually take account of the effects of density variations within the groundwater due to the mass of salt in solution. These introduce vertical flow to the system and lead to stratification in the eventual salinity distribution. This is important because a number of common conceptual models assume groundwater flow is entirely horizontal. The transport is often regarded as conservative though at a higher level of sophistication solute production or decay and chemical evolution can be included.

The principle distinction between classes of saline intrusion model lies in their treatment of the interface between the separate bodies of water within the aquifer. As described earlier, saline and fresh water are miscible, and mixing occurs at the interface due to molecular diffusion and mechanical dispersion. The parameters that control diffusion and dispersion are difficult to quantify directly and in any case the width of the transition zone is often small compared to the dimensions of the saline water body. Consequently many models have included a sharp interface approximation, with the advance of one water body completely displacing the other body by piston flow.

The vertical flow components mentioned above means that saline intrusion problems are fully three-dimensional. However often some line of symmetry can be deduced (eg. perpendicular to a straight coastline) so that vertical strip, 2-d models can provide a useful insight to the situation while minimising the computational complexity. Horizontal 2-d models imply a vertical interface which can be misleading: wells that appear safely on the fresh side of the interface might be at risk of degradation by upconing at the toe of the intruding wedge.

6.4.5 *Geophysical approaches*

Although theoretical approaches allow determination of the position of the saline/fresh water interface, they can only be applied with confidence where the vertical component of flow is zero or close to zero i.e. inland and distant from the effects of abstraction. Various geophysical techniques have been used to identify the morphology of underground saline water bodies (Tellam et al, 1986).

Borehole and surface geophysical approaches are available to investigate the geometry of saline water bodies. They rely on the increase in conductivity between fresh and saline water. The actual methodologies are covered in greater detail in review numbers 10 and 3 on the application of surface and borehole geophysics respectively.

Borehole logging

Borehole fluid conductivity logs may be used to determine the position of the interface within the borehole. Providing there is not a major component of vertical flow within the borehole, such logs provide a reasonably accurate indication (to within 1 m) of the interface depth within the formation (Tellam et al, 1986). However, if the borehole and aquifer are not in equilibrium, errors may be significant. Correlating conductivity with flowmeter logs can identify strata that contribute saline water to the borehole. This is particularly useful if the logging is carried out during pumping when the normal flow regime of the well exists (Shearer, 1991).

Various types of resistivity logs can provide a direct indication of the salinity of the water within a formation. However, their applicability is limited by the fact that, in UNSAs, boreholes generally have to be screened. If the screening material is metal, resistivity methods are not applicable.

Surface geophysical methods

Electrical resistivity and electromagnetic methods have been investigated as a means of delineating saline water bodies (e.g. Arora and Rose, 1981; Prasad et al, 1983). The methods all rely on the fact that saline water has different electrical properties from freshwater. They have advantages in that: (i) geophysical surveys are low cost compared with drilling boreholes; (ii) data can be directly related to water quality.

Electrical resistivity

In areas of relatively homogeneous sediments, electrical resistivity surveys can be used to indicate anomalies attributable to saline water bodies. Depth probing can be used to estimate the thickness of the saline zone, and resistivity traversing to delineate the surface of the saline water body across an area.

Tellam et al (1986) predicted a slight overestimation of the depth to the saline interface using this approach. This would be due to the fact that the real resistivity change is gradual, but is represented by the model as a sharp interface, with a large resistivity step. They suggested prediction accuracies of within 10 m at 30 m depth.

The effectiveness of the technique in detecting saline intrusion may be limited because of the similarity in electrical conductivity between clay, salt water in sandy aquifers, and brackish water in sandy aquifers. The occurrence of clay layers may also confuse interpretation (Prasad et al, 1983). Realistic evaluation is generally only possible if the results of such surveys are complemented by geological or geochemical data.

Electromagnetic methods

Electromagnetic methods use two wire coils, one to generate an alternating magnetic field, and the other to detect the resultant magnetic field generated in the sub-surface. The method may be used to obtain general information at shallow depths over large areas (frequency domain electromagnetic profiling), or to collect more detailed quantitative data about the electrical profile of the subsurface (time domain electromagnetic sounding).

Frequency domain (FDEM) profiling is most suitable for shallow depths (up to 30-40 m). It does not require any contact between the transmitter and receiver antennae and the ground surface, and is therefore rapid. However, its main disadvantage results from the fact that the receiver measures the bulk conductivity of the subsurface, and results are therefore only qualitative.

Time domain electromagnetic (TDEM) sounding or profiling has advantages over both electrical resistivity and FDEM methods. It has good lateral and vertical resolution, it is minimally influenced by near surface inhomogeneities, and non-uniqueness of data interpretation is reduced compared with electrical resistivity (Goldman et al, 1991). The methods has been applied to the coastal aquifer of Israel (calcareous sandstone) and was found to be a feasible approach for detecting the saline/fresh water interface (Goldman et al, 1991).

6.5 Determination of source of saline water

The presence of saline water in an aquifer is established by chemical analysis or measurement of conductivity. The question is then posed as to how the water acquired the salinity, in order that remedial measures can be assessed. In many cases the recognition of modem seawater does not pose difficulties. However, the water may be mixed with water from other sources, or may have undergone modification within the aquifer. Certain chemical signatures can be identified which may indicate the origin of salinity. However, as most saline waters are NaCl type, major ion analysis is not always useful, and other parameters may be needed.

If the source of saline water is uncertain, an initial assessment may be made by examining geological, hydrogeological, and geographical information. For example, proximity to a source of modern saline water (i.e. to the coast), or existing knowledge of saline water at depth in an aquifer. If saline water is encountered in an inland area, and there is freshwater between the inland area and the coast, evidence points towards a source other than seawater.

The response of the saline water front to pumping may also provide an indication of the source of the water: if the saline zone is coincident with an area of intense abstraction, and salinity varies with pumping rate, this would indicate a deep source of saline water. Conversely, if the saline water was not restricted to areas of heavy pumping, and did not vary greatly with pumping rates, this would indicate a more likely source to be seawater.

Chemical analyses may aid in the identification of the source of saline waters. In particular, the Mg/Ca ratio, typically 4:1 in seawater, may help to confirm the source. However, such waters may undergo chemical modification within the aquifer, therefore altering their original chemistry (Bowen, 1986; Tellam and Lloyd, 1986; Apello and Geinaert, 1991). For example, ion exchange may occur between the water and the minerals in the aquifer, sulphate reduction, or carbonate solution and precipitation may occur. These reactions may take many years, and when investigating recent seawater intrusion, major ion analyses are useful. Other approaches may also be utilised.

6.5.1 *Isotopic analysis*

Stable isotopes may be used to aid in differentiating between the main types of saline water. Their use is described in detail in review no.8. If seawater intrusion is the primary cause of salinity in the aquifer, the deuterium and $\delta^{18}O$ values of the groundwater samples would fall on a mixing line between the stable isotopic composition of seawater and that of the least saline groundwater (Payne 1988). A similar positive correlation would be expected using either of the stable isotopes and chloride. Payne suggested that isotopic analysis would not necessarily be able to differentiate between seawater and fresh water. However, with high precision of measurement of $\pm 1‰$, and a difference between sea and fresh groundwater of approximately 6‰, this approach should be able to easily discriminate between the two types of water.

Where salinity is caused by the concentration of dissolved salts by evaporation, the regression of deuterium against $\delta^{18}O$ would be indicative of an evaporation process with a slope between 4 and 6 (Payne, 1988). Where salinity is caused by leaching of evaporitic salts, there would be no change in the stable isotopic composition of percolating water, and there would therefore be no correlation between stable isotopes and ionic species.

6.5.2 *Trace elements*

Several trace elements, such as lithium, strontium, and iodine, increase with residence time in groundwater, and are present in low concentrations in seawater. These may therefore be useful to discriminate sources of saline water. For example, iodine is present in many sedimentary rocks, in a form that easily leaches to groundwater in the form of iodide. If groundwater is resident in the aquifer for a long time period, it becomes enriched in iodide compared to modern seawater (Lloyd et al, 1982). The iodide content of saline waters may therefore be used to distinguish between saline water resident in the aquifer for a long time period (such as connate water), and modern saline intrusion (Box 6.2).

6.6 Management of saline intrusion

Recent increases in global population, together with enhanced standards of living, have created greater demand on water resources, requiring improved groundwater management. Any new groundwater development should take into account the possibilities of saline intrusion, and ensure adequate control, with prevention of saline intrusion being seen as the ideal. Any intrusion carries with it the risk that the matrix of the aquifer will become contaminated, causing a permanent loss of freshwater storage capacity. The impact of saline intrusion in the Nile delta is described in Box 6.3.

Where the possibility of intrusion exists, appropriate monitoring procedures should be routinely carried out. A network of sampling piezometers should be established to monitor heads and salinity changes along the coastal fringe.

Actual methods for controlling saline intrusion vary widely according to geology, extent of the problem, water use, and economics. They generally rely on the principle that, in order to limit sea water intrusion, some fresh water outflow above the saline wedge must be maintained. They can be broadly divided into methods relying on barriers and those dependent on aquifer management; some of these are discussed briefly below.

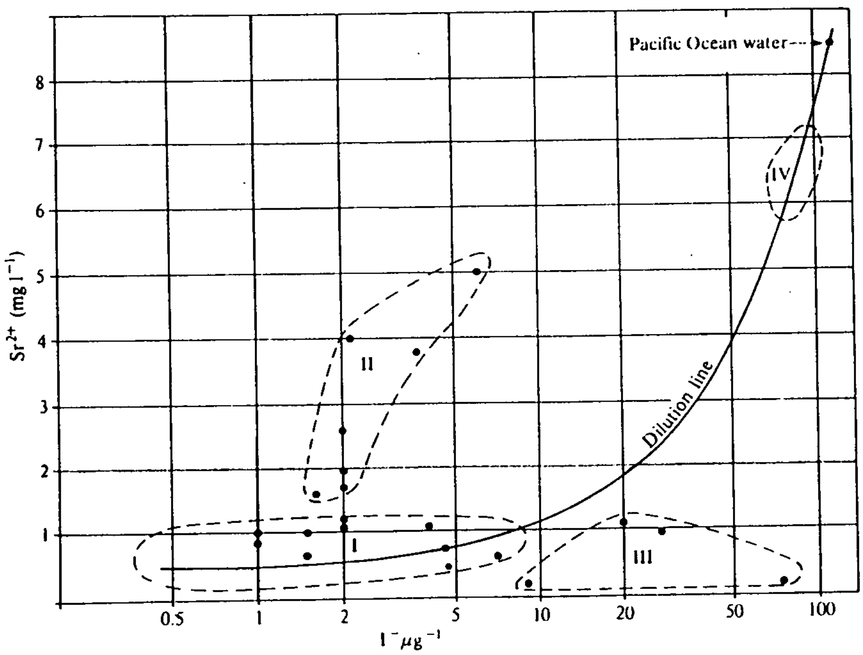
Box 6.2 **Identification of saline water source using trace elements in Lima, Peru**

The Lima Basin aquifer, Peru, is contained in a sequence of alluvial pediment sediments at the base of the Andes, up to 600 m in thickness. The main recharge to the aquifer comes from rivers flowing from the mountains, from potable supply losses and irrigation returns. Saline water occurs at the boundaries of the aquifer originating both from Teriary and older rocks and at the coast from seawater intrusion.

The iodine/strontium ratio was used to identify the sources of saline water within the aquifer and to differentiate water originating from the major peripheral rock types. The main source was shown to be modern Pacific Ocean seawater. The water associated with Jurassic sediments was Sr^{2+} enriched and that from the granodiorites I^- enriched. This iodine was attributed to leaching of hydrothermal deposits.

Reference

Lloyd J W, Howard K W F, Pacey N R and Tellam J H 1982. The value of iodine as a parameter in the chemical characterisation of groundwaters. Journal of Hyrology, 57, 247-265.



I: groundwaters only associated with alluvium
II: goundwaters entering alluvium from Jurassic sediments
III: groundwaters entering alluvium from granodiorites
IV: reverse ion-exchanged groundwaters from seawater intrusion

Strontium-iodide relationship in saline waters from the Lima Basin

Box 6.3 Saline intrusion in the Nile Delta, Egypt

The Nile Delta aquifer system has an estimated capacity of 280 x 10^9 m^3 making it a major groundwater reservoir. It consists of two water-bearing layers; the lower layer being very permeable Pleistocene sand and gravel, and the upper being relatively low permeability Holocene clay and silt. The aquifer is bounded by the Mediterranean Sea to the north, the Suez canal to the east, and the Western Desert to the west. To the south, the aquifer thickness reduces so that there is only a small window connecting with the Upper Egypt Nile Valley aquifer. It is separated from other aquifers by thick clay layers.

Saline intrusion has extended about 130 km inland from the coast leaving only a comparatively small triangle of freshwater in the south of the delta. In 1989 the annual abstraction was 1.6 x 10^9 m^3.

Following the construction of the High Aswan Dam in 1964, the potentiometric head has increased on average 1.5 m throughout the delta and as much as 2.0 m at the southern end. This has caused a reversal in the intrusion: the 250 mg/l salinity contour moved approximately 2.5 km seaward in the ten years between 1968 and 1978 giving an improvement in water quality in the south. However elsewhere the raised waterlevels have caused a deterioration in water quality because of evaporation of groundwater and also waterlogging problems. 33% of the 2.69 Mha of irrigated land in the delta is now classified as salt-affected.

To counteract these problems an extensive pumped drainage system is being constructed to lower the watertable and allow flushing of salts from the root zone. In 1987 there were 129 pumping stations with a combined capacity of 2500 m^3 s^{-1}. It is also planned to almost double the freshwater abstraction by the year 2000.

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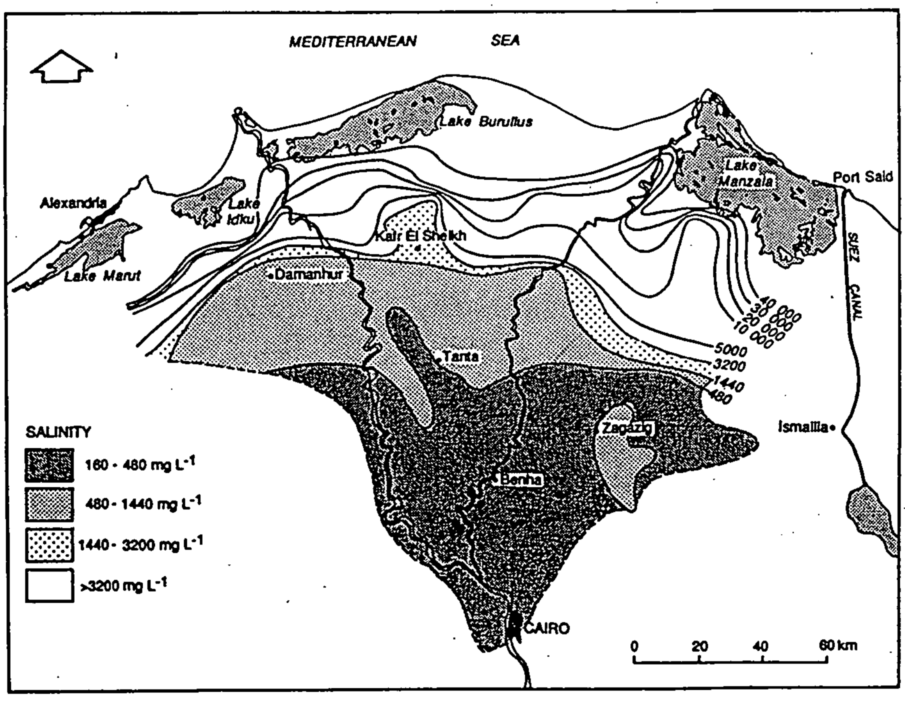

Salinity map of the Nile Delta aquifer for 1978

Barriers to saline intrusion include recharge mounds, abstraction troughs, and physical barriers.

- Abstraction troughs: An abstraction barrier is created by maintaining abstraction along a line of wells close to the coast. This creates a pumping trough, with seawater flowing inland to the trough, and fresh water flowing seaward (Figure 6.5a). The fresh water can then be utilised by inland wells.

- Recharge mounds (injection barrier): This approach uses recharge wells to maintain a pressure ridge along the coast. The injected water flows both landward and seaward (see Figure 6.5b). Ideally high quality fresh water is required for recharge, necessitating the development of a supplemental source.

- Physical barriers: An impermeable subsurface barrier may be created through the vertical extent of the aquifer, parallel to the coast. The barrier may be constructed of various materials including sheet piling, puddled clay, cement grout, or bentonite. The approach is best applied to small scale problems.

Aquifer management i.e. modification of pumping, is generally carried out on a regional scale. Modification requires changes in operational practices:

- Control of pumping: in the case where an aquifer is underlain by saline fluid, upconing can be limited by proper design and operation of wells (Bowen, 1986). For example, wells should be as shallow as is feasible, and should be pumped at a low, uniform rate. Riddel (1933) suggested that a multiple well system with small individual pumping rates was preferable to a high capacity single well. Another alternative is an infiltration gallery (i.e. horizontal well) which has been reported to help reduce the upconing that can result from heavy pumping by a vertical well (Das Gupta, 1983). However, these are expensive to install, and similar benefits may be obtained by utilising several shallow wells.

- Redistribution of pumping: Relocating pumping wells inland may help to re-establish fresh water outflow. Some schemes to reduce the effects of saline intrusion in the Chalk of Great Britain have utilised two sets of boreholes, one inland and one coastal (Headworth and Fox, 1986). The coastal wells are pumped during periods of high groundwater levels when outflow is large, with the inland wells being used during periods of low water levels.

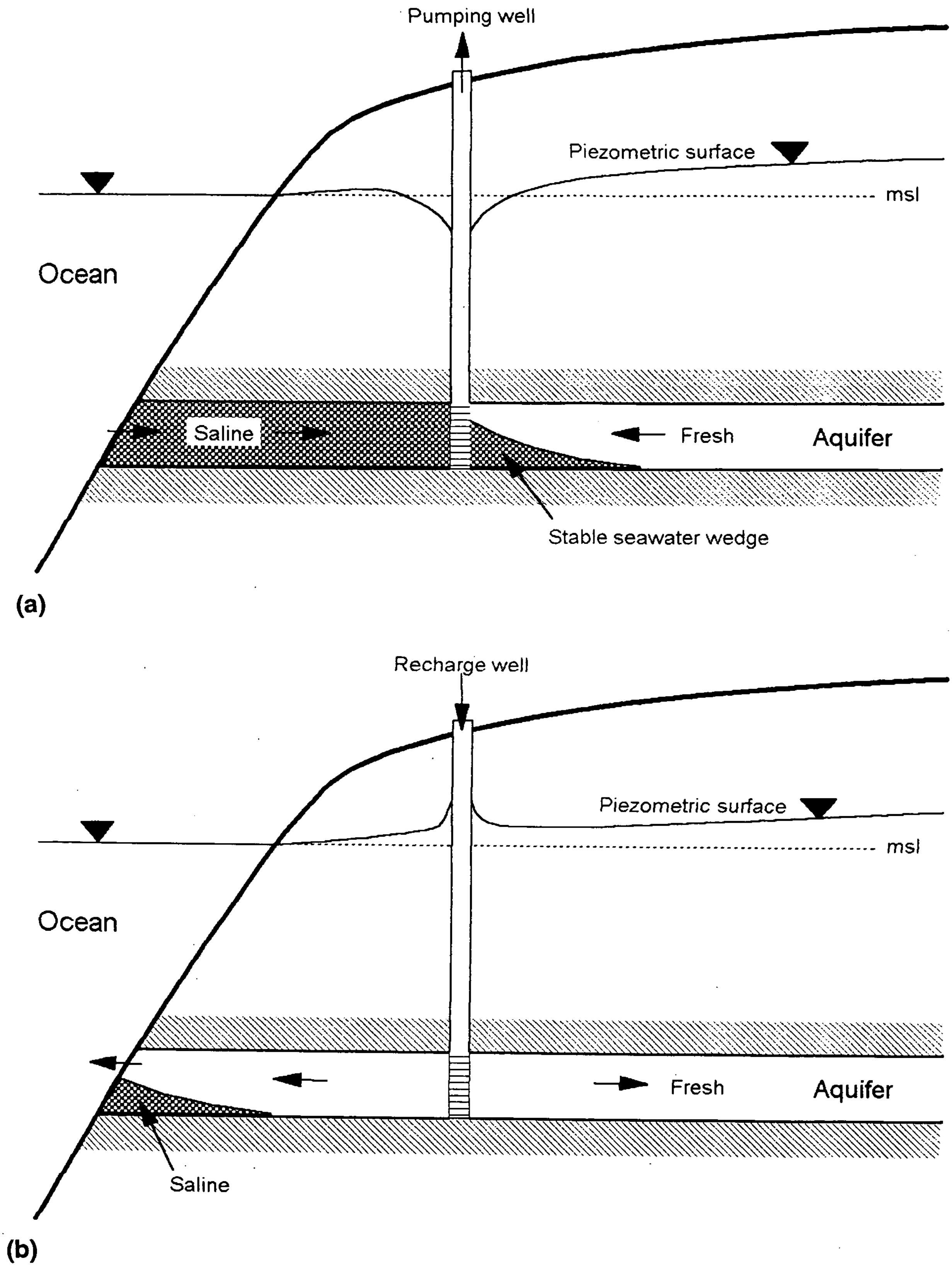

Figure 6.5 Management of saline intrusion (a) Abstraction through (b) Recharge mound

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METHOD SUMMARY SHEET (WQM 07)

TITLE: Investigating saline intrusion

Scope and use of method

Saline intrusion is a major groundwater quality management issue, and is one of the most significant and widespread forms of groundwater quality deterioration. Saline intrusion, in its strictest sense, occurs as a result of saline water encroaching into coastal aquifers, usually as a consequence of excessive abstraction of groundwater and reduction or reversal of hydraulic gradients in the coastal region. Encroachment and mixing of fresh and saline water in the same aquifer can also result from upconing of older, connate waters and it may be necessary to distinguish between the two so that control measures are correctly targeted.

Method

The most important task is to determine the extent of saline intrusion laterally and the depth and nature of the interface and transitional zone. Having done this, the characterisation of saline intrusion needs to be improved to take account of seasonal and other temporal influences, so that the processes controlling the saline intrusion can be established.

The most common way in which saline intrusion is detected and mapped is by sampling groundwater from production boreholes. From these, measurements of electrical conductivity or routine chemical analyses may provide a reasonable indication of lateral extent and change with time. They will not, however, usually define precisely the position and movement of the interface. Once saline intrusion is anticipated or suspected, purpose-built observation boreholes may be used to refine the 3-dimensional picture, allowing regular sampling, including from different depths, and also repeated electrical conductivity logging to define the position and movement of the interface.

The position of the interface can also be estimated numerically from the Ghyben-Herzberg relationship and by mathematical modelling. A large number of numerical models have been developed, and the characteristics and particular uses of some of them are summarised in Appendix 1.

Although theoretical approaches may allow the determination of the position of the saline/fresh water interface, they can only be applied with confidence when there are very limited vertical components of flow. Surface and borehole geophysical methods, described in more details in reviews no 10 and 3 respectively, can be used to investigate the three-dimensional geometry of saline intrusion incidents. Borehole geophysics applied to saline intrusion problems comprises the use of fluid conductivity logs to determine vertical distribution of salinity and hence the position of the interface. Surface geophysical methods which can be used are the electrical resistivity and electromagnetic methods, which are summarised in chapter 6 of this review and described in more detail in review no 10.

Once the distribution and evolution of salinity have been characterised, the origin of the saline water may need to be further investigated. Stable isotopes and trace elements may be used to differentiate between types of saline water, particularly between older, connate water trapped in sediments and modern seawater which has entered the aquifer as a result of recent saline intrusion.

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APPENDIX 1 Computer models for saline intrusion simulations

AQSIM

Author(s): D A Blank

Address: Tahel Consulting Engineers Limited
P O Box 11170
Tel Aviv
Israel

Description: A two-dimensional finite-difference model to solve transient horizontal groundwater problems in isotropic, heterogeneous, confined or phreatic aquifers connected with a stream; optional simulation of salt-fresh water interface.

BEAVERSOFT

Author(s): J Bear
A Verruijt

Address: IGWMC
Holcomb Research Institute
4600 Sunset Avenue
Indianapolis, IN 46208 USA

Description: A package of analytical and numerical solutions for groundwater flow and solute transport. The package includes models for steady and unsteady two-dimensional flow in heterogeneous aquifers, for flow through dams, contaminant transport by advection and dispersion and for salt water intrusion.

FEGM

Author(s): M Kawanishi

Address: Abiko Research Laboratory
Central Research Institute of Electric Power Industry
1646 Abiko
Abiko-City
Chiba-Pre, 270-11
JAPAN

Description: FEGM is a computer program to analyse the groundwater flow in saturated-unsaturated media by the fine-element method and is compatible to the radionuclide transport model FERM. This model has been modified from the FEMWATER code. The modifications enable quasi-three and three-dimensional problems to be analysed. A salt intrusion model has also been included.

INTERFACE

Author(s): R H Page

Address: Water Resources Program
Department of Civil Engineering
Princeton University
Princeton, NJ 08540 USA

Description: A finite-element model to simulate transient flow of fresh and saline water as immiscible fluids separated by an interface in an isotropic, heterogeneous, water-table aquifer.

NEWVAR

Author(s): E Ledoux

Address: Centre de'Informatique Geologique
Ecole des mines de Paris
35 rue Saint-Honore 77305
Fontainebleau
Paris, France

Description: NEWVAR is designed to allow the analysis of a regional two-dimensional, multi-layered groundwater flow in coastal aquifers. The numerical solution permits the prediction of both regional fresh water levels and two-dimensional fresh water/salt water interface by using nested square meshes. the numerical solution is based on the finite difference method: the Gauss-Jordan direct method is used for solving steady and unsteady state linear equations. Different procedures are used to avoid numerical difficulties in the transient position of the interface for two-dimensional area flow.

SWIFT (Verruijt)

Author(s): A Verruijt
J B S Gan

Address: Department of Civil Engineering
Delft Technical University
Sterinweg 1
2628 CN Delft
THE NETHERLANDS

Description: A cross-sectional fine-element model for transient horizontal flow of salt and fresh water and analysis of upconing of an interface in a homogeneous aquifer. Includes buoyancy and leakage.

SWIFT (Dillon et al.)

Author(s): R T Dillon
R M Cranwell
R B Lantz
S B Pahwa
M Reeves

Address: National Energy Software Center
Argonne National Laboratory
9700 S Cass Avenue
Argonne, IL 60439 USA

Description: A three-dimensional finite-difference model for simulation of coupled, transient, density-dependent flow and transport of heat, brine, tracers or radionuclides in anisotropic, heterogeneous saturated porous media. Includes. Advection, dispersion, diffusion adsorption, ion exchange, decay, chemical reactions.

SWIGS2D

Author(s): D N Contractor

Address: Water and Energy Research Institute of the Western Pacific
University of Guam
College Station
Mangilao, Guam 96013

Description: A two-dimensional finite-element model to simulate transient, horizontal salt and fresh water flow separated by a sharp interface in an anisotropic, heterogeneous, confined, semi-confined or water table aquifer.

SWIM

Author(s) A A G Sa da Costa
J L Wilson

Address: R M Parsons Laboratory for Water Resources and Hydrodynamics
Department of Civil Engineering
MIT
Cambridge, MA 02139 USA

Description: A versatile finite-element model to simulate transient, horizontal salt and fresh water flow in porous media, separated by a sharp interface.

SWSOR

Author(s): J W Mercer
C R Faust

Address: Geotrans, Inc.
250 Exchange Place
Suite A
Herdon, VA 22070 USA

Description: A finite-difference model to simulate the areal, unsteady flow of saltwater and freshwater separated by an interface in anisotropic, heterogeneous porous media. Includes leakage and infiltration.

SEAWTR/SEACONF

Author(s): R I Allayla

Address: Civil Engineering Department
Colorado State University
Fort Collins, CO 80523 USA

Description: A two-dimensional finite-difference model for horizontal simulation of simultaneous flow of salt and fresh water in a confined or water-table aquifer with anisotropic and heterogenous properties, including effects of capillary flow. Includes capillary forces and influence of capillary region on specific yield.

SUTRA

Author(s): C I Voss

Address: US Geological Survey
National Center
12201 Sunrise Valley Drive
Reston, VA 22092 USA

Description: SUTRA (Saturated-Unsaturated TRAnsport) is a computer program which simulates fluid movement and the transport of either energy or dissolved substances in a subsurface environment. the model employs a two-dimensional hybrid finite-element and integrated-finite-difference method to approximate the governing equations. The two interdependent processes that are simulated are:

(1) fluid density-dependent saturated groundwater flow, and either
(2) transport of a solute in the groundwater, in which the solute may be subject to: equilibrium adsorption on the porous matrix, and both first-order and zero-order production or decay,

or,

(3) transport of thermal energy in the groundwater and solid matrix of the aquifer.

SUTRA provides, as the primary calculated results, fluid pressures and either solute concentrations or temperatures, as they vary with time, everywhere in the simulated subsurface system. SUTRA may also be used to simulate simpler subsets of the above process. SUTRA flow simulation may be employed for areal and cross-sectional modeling of saturated groundwater flow systems, and for cross-sectional modeling of unsaturated zone flow. Boundary conditions, sources, and sinks may be time-dependent. An option is available for storing intermediate results and restarting at the intermediate time. Options are also available to print fluid velocities in the system, to print fluid mass and solute mass or energy budgets for the system, and to make temporal observations at points in the system.
SUTRA has been modified by IGWMC to include three options for the moisture-characteristic curve during simulation unsaturated flow. These are: van Genuchten (1980), Brooks and Corey (1964), and input of tabular data for pressure, relative hydraulic conductivity, and moisture content.

A graphical post-processor SUTRA-PLOT is distributed with SUTRA, SUTRA-PLOT can draw the finite element mesh, the model region boundary, and fluid velocity vectors. Can plot contours of pressure, saturation, concentration and temperature within the model region. To run SUTRA-PLOT requires only SUTRA input and output files.

SWICHA

Author(s): B Lester

Address: GeoTrans, Inc
46060 Manekin Plaza
Sterling, VA 22170, USA

Description: SWICHA is a three-dimensional finite element code for analyzing seawater intrusion in coastal aquifers. The model simulates variable density fluid flow and solute transport processes in fully-saturated porous media. it can solve the flow and transport equations independently or concurrently in the same computer run. Transport mechanisms considered include: advection, hydrodynamic dispersion, adsorption, and first-order decay.

The flow and transport equations are discretized using the finite element method. Simple rectangular and triangular prism elements are used. the combination of such elements enables flow regions of complex geometry to be modeled accurately. In addition, element matrices can be computed efficiently without having to perform numerical integration. Matrix assembly is performed slice by slice, and the matrix solution is achieved by using a slice successive over-relaxation (SSOR) scheme. This scheme permits a fairly large number of unknowns to be handled cost effectively. For a coupled flow and transport problem, the nonlinearity is handled by picard iterations. At the user's option, artificial dispersion can be added to the transport equation stiffness matrix to prevent exceeding a critical Peclet number.

SWICHA is capable of performing several types of analysis, performed in an areal plane, vertical cross-section, axisymmetric configuration, or a fully three-dimensional mode. Because of its special design features, SWICHA is capable of handling a wide range of complex three-dimensional, steady-state or transient field problems. Several Test Problems have been used to verify the code and check the accuracy of the numerical solution and the solution algorithms. These problems range from simple one-dimensional to complex three-dimension coupled flow and transport.

The SWICHA code may be used in many types of practical situations. These include: 1) groundwater resource evaluations; 2) assessments of well performance and pumping test analysis; 3) groundwater contamination investigations; 4) hazardous waste subsurface storage programs; and 5) seawater intrusion studies.

Technical features of models

Model Name	Medium	Homogeneous	Isotropic	Confined	Dimensions	Layers	Var. in Space	T-dep. Input	Transient	Solution Type	Pathline	Conservative	Advection	Dispersion	Multiphase	Saturation	Capillority	Dens.-depend.	Salt/Fresh	Diffusion	Adsorption	Ion-exchange	Retardation	Decay	Reactions	Degradation	Mass Balance	Mass Transfer	Redox	Recharge	Delayed Yield	Leakage	GW/SW	Evapotrans.	Plant Wat.	Infiltration	Consolidation	Conduction
AQSIM	P	B	T	B	2				Y	D	N	-	-	-	N		N	Y	Y	-	-	-	-	-	-	-	-	-	-		N	N	Y	N	N	Y	N	N
BEAVERSOFT	P											C	Y	Y	N			Y	Y						N				N						N			
FEGM	P				3					E					Y													Y										
INTERFACE	P	N	Y						Y	E		-	-	-	Y	S	N	Y	Y	-	-	-	-	-	-	-	-	-	-		N	N		N	N	N	N	N
NEWVAR																																						
SEAWATR/SEACONF	P	N	N	B	2					D		-	-	-	N	S	Y	Y	Y	-	-	-	-	-	-	-	-	-	-		N	N		N	N	N	N	N
SUTRA	P				2				Y	E		N	Y	Y	N	B	Y	Y	Y	Y	Y	N	N	N	Y	N	N	N	N									
SWICHA	P				3					E		N	Y	Y	N	S		Y	Y	Y	Y				Y													
SWIFT	P	N	N		3				Y	D	N	N	Y	Y	Y	S	Y	Y	N	Y	Y	Y	N	Y	Y	N	N	N	N		N	N		N	N	N	N	N
SWIFT/11	D	N	N	C	3	M	Y	Y	Y	D	N	N	Y	Y	N			Y	N	Y	Y	N	N	N	Y	N	N	N	N	Y	N	N	N		N		N	
SWIGS2D	P	N	N	A	2				Y	E	N	-	-	-	N	S	N	Y	Y	-	-	-	-	-	-	-	-	-	-		N	Y		N	N	N	N	N
SWIM	P								Y	E		-	-	-	T					-	-	-	-	-	-	-								Y	Y			
SWSOR																																						

Key to entries in the Technical Features table.

FIELD	EXPLANATION (where necessary)
Model Name	
Medium	P - Porous, F - Fractured, D - Dual Porosity, S - Soils
Homogeneous	Y - Yes (Homogeneous) N - No (heterogeneous)
Isotropic	Y - Yes (isotropic), N - No (anisotropic)
Confined	C - Confined, U - Unconfined, S - Semi-confined, B- Both confined and unconfined (not semi-confined), A - All. does not define whether you can have these conditions combined within the same model
Dimensions	1, 2, 3
Layers	1, 2.. M - Many
Var. in Space	Variation of properties in space
T-dep. Input	Time-dependent input
Transient	Y - Yes (may deal with steady-state as well)
Solution Type	A - Analytical, D - F.Difference, E - F.Element, B - Boundary Integral, I - Integrated Finite Difference, C - Method of Characteristics. Problems may exist here where more than one method is used, the flow is quoted as priority if applicable.
Pathline	
Conservative	C - Conservative, N - Non-conservative, production/reduction of pollutants etc
Advection	
Dispersion	
Multiphase	
Saturation	S - Saturated, U - Unsaturated, B - Both
Capillarity	Capillary forces or capillarity
Dens.-depend.	Density-dependent flow
Salt/Fresh	Saltwater/freshwater interaction

FIELD	EXPLANATION (where necessary)
Diffusion	
Adsorption	
Ion-exchange	
Retardation	
Decay	
Reactions	
Degradation	
Mass Balance	
Mass Transfer	
Redox	Redox reactions
Recharge	Recharge to the aquifer
Delayed Yield	
Leakage	
GW/SW	Interaction between groundwater and surface water
Evapotrans.	Evapotranspiration
Plant Wat.	Plant water uptake
Infiltration	
Consolidation	
Conduction	

INTERFACE

Author(s): R H Page

Address: Water Resources Program
Department of Civil Engineering
Princeton University
Princeton, NJ 08540 USA

Description: A finite-element model to simulate transient flow of fresh and saline water as immiscible fluids separated by an interface in an isotropic, heterogeneous, water-table aquifer.

NEWVAR

Author(s): E Ledoux

Address: Centre de'Informatique Geologique
Ecole des mines de Paris
35 rue Saint-Honore 77305
Fontainebleau
Paris, France

Description: NEWVAR is designed to allow the analysis of a regional two-dimensional, multi-layered groundwater flow in coastal aquifers. The numerical solution permits the prediction of both regional fresh water levels and two-dimensional fresh water/salt water interface by using nested square meshes. the numerical solution is based on the finite difference method: the Gauss-Jordan direct method is used for solving steady and unsteady state linear equations. Different procedures are used to avoid numerical difficulties in the transient position of the interface for two-dimensional area flow.

SWIFT (Verruijt)

Author(s): A Verruijt
J B S Gan

Address: Department of Civil Engineering
Delft Technical University
Sterinweg 1
2628 CN Delft
THE NETHERLANDS

Description: A cross-sectional fine-element model for transient horizontal flow of salt and fresh water and analysis of upconing of an interface in a homogeneous aquifer. Includes buoyancy and leakage.

SWIFT (Dillon et al.)

Author(s): R T Dillon
R M Cranwell
R B Lantz
S B Pahwa
M Reeves

Address: National Energy Software Center
Argonne National Laboratory
9700 S Cass Avenue
Argonne, IL 60439 USA

Description: A three-dimensional finite-difference model for simulation of coupled, transient, density-dependent flow and transport of heat, brine, tracers or radionuclides in anisotropic, heterogeneous saturated porous media. Includes. Advection, dispersion, diffusion adsorption, ion exchange, decay, chemical reactions.

SWIGS2D

Author(s): D N Contractor

Address: Water and Energy Research Institute of the Western Pacific
University of Guam
College Station
Mangilao, Guam 96013

Description: A two-dimensional finite-element model to simulate transient, horizontal salt and fresh water flow separated by a sharp interface in an anisotropic, heterogeneous, confined, semi-confined or water table aquifer.

SWIM

Author(s) A A G Sa da Costa
J L Wilson

Address: R M Parsons Laboratory for Water Resources and Hydrodynamics
Department of Civil Engineering
MIT
Cambridge, MA 02139 USA

Description: A versatile finite-element model to simulate transient, horizontal salt and fresh water flow in porous media, separated by a sharp interface.

SWSOR

Author(s): J W Mercer
C R Faust

Address: Geotrans, Inc.
250 Exchange Place
Suite A
Herdon, VA 22070 USA

Description: A finite-difference model to simulate the areal, unsteady flow of saltwater and freshwater separated by an interface in anisotropic, heterogeneous porous media. Includes leakage and infiltration.

SEAWTR/SEACONF

Author(s): R I Allayla

Address: Civil Engineering Department
Colorado State University
Fort Collins, CO 80523 USA

Description: A two-dimensional finite-difference model for horizontal simulation of simultaneous flow of salt and fresh water in a confined or water-table aquifer with anisotropic and heterogenous properties, including effects of capillary flow. Includes capillary forces and influence of capillary region on specific yield.

SUTRA

Author(s): C I Voss

Address: US Geological Survey
National Center
12201 Sunrise Valley Drive
Reston, VA 22092 USA

Description: SUTRA (Saturated-Unsaturated TRAnsport) is a computer program which simulates fluid movement and the transport of either energy or dissolved substances in a subsurface environment. the model employs a two-dimensional hybrid finite-element and integrated-finite-difference method to approximate the governing equations. The two interdependent processes that are simulated are:

(1) fluid density-dependent saturated groundwater flow, and either
(2) transport of a solute in the groundwater, in which the solute may be subject to: equilibrium adsorption on the porous matrix, and both first-order and zero-order production or decay,

or,

(3) transport of thermal energy in the groundwater and solid matrix of the aquifer.

SUTRA provides, as the primary calculated results, fluid pressures and either solute concentrations or temperatures, as they vary with time, everywhere in the simulated subsurface system. SUTRA may also be used to simulate simpler subsets of the above process. SUTRA flow simulation may be employed for areal and cross-sectional modeling of saturated groundwater flow systems, and for cross-sectional modeling of unsaturated zone flow. Boundary conditions, sources, and sinks may be time-dependent. An option is available for storing intermediate results and restarting at the intermediate time. Options are also available to print fluid velocities in the system, to print fluid mass and solute mass or energy budgets for the system, and to make temporal observations at points in the system.
SUTRA has been modified by IGWMC to include three options for the moisture-characteristic curve during simulation unsaturated flow. These are: van Genuchten (1980), Brooks and Corey (1964), and input of tabular data for pressure, relative hydraulic conductivity, and moisture content.

A graphical post-processor SUTRA-PLOT is distributed with SUTRA, SUTRA-PLOT can draw the finite element mesh, the model region boundary, and fluid velocity vectors. Can plot contours of pressure, saturation, concentration and temperature within the model region. To run SUTRA-PLOT requires only SUTRA input and output files.

SWICHA

Author(s): B Lester

Address: GeoTrans, Inc
46060 Manekin Plaza
Sterling, VA 22170, USA

Description: SWICHA is a three-dimensional finite element code for analyzing seawater intrusion in coastal aquifers. The model simulates variable density fluid flow and solute transport processes in fully-saturated porous media. it can solve the flow and transport equations independently or concurrently in the same computer run. Transport mechanisms considered include: advection, hydrodynamic dispersion, adsorption, and first-order decay.

The flow and transport equations are discretized using the finite element method. Simple rectangular and triangular prism elements are used. the combination of such elements enables flow regions of complex geometry to be modeled accurately. In addition, element matrices can be computed efficiently without having to perform numerical integration. Matrix assembly is performed slice by slice, and the matrix solution is achieved by using a slice successive over-relaxation (SSOR) scheme. This scheme permits a fairly large number of unknowns to be handled cost effectively. For a coupled flow and transport problem, the nonlinearity is handled by picard iterations. At the user's option, artificial dispersion can be added to the transport equation stiffness matrix to prevent exceeding a critical Peclet number.

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Technical features of models

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BEAVERSOFT	P											C	Y	Y	N			Y	Y						N				N						N			
FEGM	P				3					E					Y													Y										
INTERFACE	P	N	Y						Y	E		-	-	-	Y	S	N	Y	Y	-	-	-	-	-	-	-	-	-	-		N	N		N	N	N	N	N
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SUTRA	P				2				Y	E		N	Y	Y	N	B	Y	Y	Y	Y	Y	N	N	N	Y	N	N	N	N									
SWICHA	P				3					E		N	Y	Y	N	S		Y	Y	Y	Y				Y													
SWIFT	P	N	N		3				Y	D	N	N	Y	Y	Y	S	Y	Y	N	Y	Y	Y	N	Y	Y	N	N	N	N		N	N		N	N	N	N	N
SWIFT/11	D	N	N	C	3	M	Y	Y	Y	D	N	N	Y	Y	N			Y	N	Y	Y	N	N	N	Y	N	N	N	N	Y	N	N	N		N		N	
SWIGS2D	P	N	N	A	2				Y	E	N	-	-	-	N	S	N	Y	Y	-	-	-	-	-	-	-	-	-	-		N	Y		N	N	N	N	N
SWIM	P								Y	E		-	-	-	T					-	-	-	-	-	-	-								Y	Y			
SWSOR																																						

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Evapotrans.	Evapotranspiration
Plant Wat.	Plant water uptake
Infiltration	
Consolidation	
Conduction	

E82K 063204

WORK ORDER/ESTIMATE - COPYING

BIN No ______ JOB No est 2811

CUSTOMER: British Geological CONTACT:

ADDRESS: Keyworth

TEL: FAX: ORDER NO: DATE: 21/8/96 TAKEN BY: PC

DATE PROOF REQUIRED ____ **DATE TO COMPLETE JOB** 23/8

REPEAT JOB: NO ☐ YES ☐ A/W STORED BY: US ☐ CUSTOMER ☑ CHARGE ☑ CASH ☐

DESIGN

DESCRIPTION: 1

LAZER BRIEF: (I.E. Font, Pt. size, etc.)

OUTPUT:

SIZE/No UP:

PASTE-UP BRIEF: SEE OVERLEAF

DESCRIPTION: 2

LAZER BRIEF: (I.E. Font, Pt. size, etc.)

OUTPUT:

SIZE/No UP:

PASTE-UP BRIEF: SEE OVERLEAF

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MACHINE: Colour ☐ 5090 ☑ Plan ☐

FINISHED SIZE: A4

TOTAL QUANTITY OF COPIES: 35

RUN:

	DESCRIPTION	NO. OF COPIES	NO. OF ORIGINALS	NO. UP	SIZE	S/S	D/S
S-S	a) Black/White	35	172	1	A4	✓	
Yellow	b) Black/White	35	5	1	A4	✓	
	c)						

PAPER: Yellow Tint

WEIGHT/COLOUR 80 GSM

SPECIAL INSTRUCTIONS:

2

DESCRIPTION: Colour

MACHINE: Colour ☑ 5090 ☐ Plan ☐

FINISHED SIZE: A4

TOTAL QUANTITY OF COPIES: 35

RUN:

	DESCRIPTION	NO. OF COPIES	NO. OF ORIGINALS	NO. UP	SIZE	S/S	D/S
Front	a) Cover Colour	35	1	1	A4	✓	
Colour	b) copy	35	2	1	A4	✓	
	c)						

PAPER: 100 GSM FYNE. or Colotech

WEIGHT/COLOUR

SPECIAL INSTRUCTIONS:

FINISHING

FOLD ☐

COLLATE ☐

STAPLE ☐

TRIM ☐

SCORE ☐

PERF ☐

NUMBER ☐

PAD ☐

FAN APART SETS ☐

LAMINATING ☐ 80GSM BOND

other WIRO BIND - WHITE

FOLD ☐

COLLATE ☐

STAPLE ☐

TRIM ☐

SCORE ☐ Put on 120gsm

PERF ☐ Colour copy stock

NUMBER ☐ or thickest we

PAD ☐ have

FAN APART SETS ☐

LAMINATING ☐

other

BINDING

TYPE PVC FRONT

COLOUR White/Red/Blue/Black 170 GSM CARD BACK

COVERS

other

TYPE

COLOUR White/Red/Blue/Black

COVERS

other

PRICING

	TOTAL		DEPOSIT ☐ PAID		TO PAY	
COPYING:						
PRINTING:						
BINDING:						
FINISHING:						
COLOUR:						
LAZER:						
PLAN:						
SPECIAL SERVICES:						
OTHER:						
SUB TOTAL:						
V.A.T. @ %						
TOTAL:						

AG08V1

Customers Signature .... **Print Name** ....

British Geological Survey

Natural Environment Research Council

Official Order **E82K** 063204

Invoice address

British Geological Survey
Keyworth Nottingham NG12 5GG

For the attention of the Accounts Section

☎ 0115 936 3100 Telex 378173 BGSKEY G Fax 0115 936 3200
VAT Reg. No. 618 3673 25

Name and address of supplier

Alphagraphics
6 Angel Row
Nottingham NG1 6HL

Delivery address, if different from invoice address

For the attention of J D. Bennett

Quantity	Description of goods or services	Spec. No.
35	35 copies of Report WC/96/39 (UNAs Review 12),	
	as per spec. sheet attached.	

Delivery is required by: ASAP

The British Geological Survey requests you to supply the items specified above in accordance with the Natural Environment Research Council's standard conditions of contract for the supply of goods and services.

Please complete and return the acknowledgement copy of this order for the attention of the delivery addressee.

New goods receiving times
Goods will be received
between the hours of:
8.00 am–12 noon + 12.30–3.30 pm
Monday–Thursday
8.00 am–12 noon + 12.30–3.00 pm
Friday

Please quote the above order number on all correspondence relating to this order

All orders from the British Geological Survey will be made or confirmed on an official order form

Authorised signatory (Finance)

J Haslam

Date

21/8/96

CONDITIONS OF CONTRACT

1 Specifications, etc.

The Articles shall be of qualities and sorts described and equal in all respects to the Samples, Patterns Specifications, Plans, Drawings or any other documents individually or collectively which form part of the Contract. Except insofar as may otherwise be indicated by a Sample, Pattern, Specification, Plan, Drawing or other document the Articles shall be strictly in accordance with the latest relevant British Standard Specification where such exists, published before the date of the Contract, or otherwise shall be to the satisfaction of the Authority

2 Issues of Council Property

(1) All Council Property issued in connection with the Contract (hereinafter called "Issued Property") shall remain the property of the Authority whether paid for by or charged against the Contractor or not, and shall be used in the execution of the Contract and for no other purpose whatsoever, without the prior approval in writing of the Authority. If requested, the Authority will notify the Contractor, within a reasonable time, of the current value of Issued Property.

(2) (a) Upon receipt of Issued Property the Contractor shall subject it to:

(i) A reasonable visual inspection, and

(ii) such additional inspection and testing as may be necessary and practicable to check that the issued Property is not defective or deficient for the purpose for which it has been provided and as can reasonably be carried out within the undermentioned period;

and shall notify the Authority within 14 days of receipt, or such longer period as may be specified in the Contract of any defects or deficiencies thereby discovered, provided that items issued in a "preserved, identified and packaged condition" shall not be unpacked earlier than is necessary and for such items the said 14 days or longer period shall count from the date on which packages are opened.

(b) Where the contractor cannot reasonably carry out the additional inspection and testing within the prescribed period as required by paragraph (a) of Clause (2) of this Condition, whether after receipt or unpacking as the case may be, he shall inform the Authority promptly of the position and shall carry out such inspection and testing as soon as is practicable thereafter and shall notify the Authority within 7 days of completion of such inspection and testing of any defects or deficiencies thereby discovered.

(3) The Authority shall within a reasonable time of receipt of any notice under Clause (2) of this Condition replace, reissue or authorise repair of Issued Property agreed to be defective or deficient, and if appropriate in the circumstances the Authority shall revise the Contract Price and/or the time specified in the Contract for delivery of the Articles. Should the Authority fail to replace, reissue or authorise repair of defective or deficient Issued Property within a reasonable time (of receipt of notice under Clause (2) of this Condition) such revisions of the Contract Price and/or of the time specified in the Contract for delivery of the Articles shall be made as may be appropriate provided that the Contractor shall have taken all reasonable measures to mitigate the consequences of any delay.

(4) The Contractor shall be responsible for the safe custody and, subject to Clause (5) of this Condition, due return of Issued Property, whether or not incorporated in the Articles, and shall be responsible for all loss thereof or damage thereto from whatever cause (except as provided below) until redelivered in accordance with the Authority's instructions. For the purposes of this Condition, defects or deficiencies notified to the Authority in accordance with Clause (2) of this Condition or deterioration in Issued Property resulting from its normal and proper use in the execution of the Contract shall not be deemed to be loss or damage (except insofar as the deterioration is contributed to by any misuse, lack of care or want of maintenance by the Contractor). Except as hereinafter provided the Contractor shall not be liable for loss or damage to the Issued Property arising from

(a) aircraft or other aerial devices or objects dropped therefrom including pressure waves caused by aircraft or such devices whether travelling at sonic or supersonic speeds

(b) ionising radiations or contamination by radioactivity from any nuclear fuel or from nuclear waste from the combustion of nuclear fuel.

(c) The radioactive, toxic, explosive or other hazardous properties of any nuclear assembly or nuclear component thereof.

(d) riot, civil commotion, civil war, rebellion, revolution, insurrection, military or usurped power or King's enemy risks (within the definition of that expression contained in section 15(1)(a) of the War Risks Insurance Act 1939 as for the time being in force):

Providing that the Contractor shall be so liable to the extent that any of the aforementioned risks are covered by his existing insurance

(5) Instructions for the return or disposal of defective or deficient issued Property shall be issued by the Authority and such property shall not be at the risk of the Contractor once it has been delivered in accordance with the Authority's instruction.

(6) If an Article in which issued Property has been incorporated is damaged or rejected or is the subject of additional costs by reason of a defect or deficiency in the Issued Property which was not and could not reasonably have been discovered by the Contractor and notified to the Authority in accordance with the provisions of Clause (2) of this Condition, and provided that such defect or deficiency shall not be attributable to any misuse, lack of care or want of maintenance by the Contractor, the Authority shall replace, reissue or authorise repair of the Issued Property and shall make such revision of the Contract Price and/or of the time specified in the Contract for delivery of the Article as may be appropriate provided that the Contractor shall have taken all reasonable measures to mitigate the consequences of any delay.

(7) Neither the Contractor, nor any subcontractor, nor any other person, shall have a lien on Issued Property, whether paid for by or charged against the Contractor or not, for any sum due to the Contractor, subcontractor or other person, and the Contractor shall take all such steps as may be reasonably necessary to ensure that the title of the Authority, and the exclusion of any such lien, are brought to the notice of all subcontractors and other person dealing with any Issued Property.

3 Loss of or Damage to the Articles

(1) The Contractor is responsible for the Articles and any materials, equipment, fittings or things acquired or allocated by him for incorporation therein until delivery has been effected in accordance with the Condition No 6 and shall make good any loss or damage to the Articles or any such materials, equipment, fittings or things however occasioned which may occur before such delivery

(2) The provisions of Clause (1) of this Condition shall apply notwithstanding that the Articles concerned may have been inspected in accordance with the Contract or that the property therein may in accordance with the provisions of the Contract Vesting Condition where applicable or other provisions specifically made in the Contract have passed from the Contractor to the Authority or his agent earlier than upon delivery

(3) Unless the Contract specifically otherwise provides the Contractor is not responsible for the Articles after delivery save that he shall become responsible in all respects for any Article which under Condition No 13 the Authority rejects after delivery and such responsibility shall take effect upon the Contractor removing the Article in accordance with Clause (3) of Condition No 13 or upon the returning of the Article to the Contractor in accordance with Clause (4) of Condition No. 13 or, if he fails so to remove the Article, or if the Authority does not exercise the right to return the Article, on the expiry of the 8th working day from receipt of notification of rejection of the Article.

4 Inspection

(1) The Authority may inspect or arrange for the inspection of the Articles or any of them in course of production, at the Contractor's premises, at any reasonable time

(2) Without prejudice to the Authority's right of inspection under Clause (1) of this Condition, the Authority may inspect or arrange for the inspection of the completed Articles, or any of them, at the Contractor's premises where the Articles have been produced, or after delivery, or as otherwise provided in the Contract.

(3) When the Authority wishes to exercise its right of inspection under this Condition, the Contractor shall give to the representative of the Authority full and free access to the said premises as and when required for that purpose and shall provide at his own expense all such accommodation and facilities in connection with the inspection as the Authority may reasonably require and all appliances, materials and labour required for inspection purposes.

5 Packages

(1) Unless otherwise provided by the Contract all containers (including packing cases, boxes, tins, drums and wrappings) supplied by the Contractor shall be considered as non-returnable and their cost as having been included in the Contract Price.

(2) If the Schedule provides for returnable containers they shall be separately priced, and shall be so invoiced by the Contractor who shall give full credit on their return, at the expense of the Authority, to the Contractor in clean and good condition within a reasonable time after delivery in accordance with the terms stated in the Schedule. The containers shall be clearly marked "Returnable".

6 Delivery

(1) The Contractor shall hand over the Articles to the Authority, or the agent of the Authority at the time or times and at the place or places and in the manner specified in the Contract or in orders (written or printed out by computer) issued under the Contract.

(2) When handing over the Articles in accordance with this Condition the Contractor shall

(a) ensure that the Articles are properly packed and secured as may be stipulated in the Contract and

(b) comply with any additional instructions which from time to time the Authority may give with regard to the transportation of the Articles, provided that any extra cost necessarily incurred in so doing shall be borne by the Authority as an addition to the Contract Price.

(3) When the Contract or any order issued under the Contract specifies that the Articles shall be handed over ex-works or despatched f.o.r., f.a.s., f.o.b., or f.d.d. the Contractor shall hand over for despatch the Articles accordingly, consign ing them to such destinations as the Authority may require.

(4) When the Articles are handed over in accordance with Clause (1) of this Condition, delivery of the Articles shall occur on their being so handed over. When the Articles are handed over or despatched in accordance with Clause (3) of this Condition, delivery of the Articles shall occur on their going into the possession of the Authority or his agent.

(5) Unless the Contract specifically otherwise provides and subject to the provisions of the Contract Vesting Condition where applicable, the property in the Articles passes from the Contractor to the Authority upon delivery in accordance with Clause (4) of this Condition.

(6) When after delivery an Article is rejected under Condition No 13 that Article shall for the purposes of the Contract be considered as not having been delivered under the Contract and the Property in that Article shall return to the Contractor from the Authority provided that this Clause shall have effect only when the Contractor has received notice of rejection.

7 Default

(1) Should the Article or any portion thereof not be delivered within the time or times specified in the Contract, or in a Warrant or Order where used, the Authority may without prejudice to or any other remedies, by notice to the Contractor determine the Contract either as respects the Articles which have not been delivered in accordance with the Contract at the time of such determination, or as respects all the Articles to which the Contract relates other than those delivered in accordance with the Contract before that time

(2) Where the Authority has determined the Contract under Clause (1) hereof and without prejudice as aforesaid the Authority may replace all or any of the Articles as respects which the Contract is so determined by purchasing or manufacturing other Articles of the same or similar description, or by allocating other Articles of the same or similar description in the possession or control of the Authority to the purposes for which the Articles replaced are required and there shall be recoverable from the contractor the amount by which the aggregate of the cost of purchasing and of manufacturing articles in this way and of the value of any Articles allocated as aforesaid exceeds the amount which would have been payable to the Contractor in respect of all the Articles so replaced if they had been delivered in accordance with the Contract.

8 Law (English)

The Contract shall be considered as a contract made in England and subject to English Law.

9 Arbitration (English Law)

All disputes differences or questions between the parties to the Contract with respect to any matter or thing arising out of or relating to the Contract other than a matter or things as to which the decision of the Authority is under the Contract to be final and conclusive and except to the extent to which special provision for arbitration is made elsewhere in the Contract shall be referred to the arbitration of two persons, one to be appointed by the Authority and one by the Contractor or their Umpire, in accordance with the provisions of the Arbitration Act 1950 and the Arbitration Act 1979 or any statutory modification or reenactment thereof

10 Transfer and Subletting

The Contractor shall not give, bargain, sell, assign, sublet (except as is customary in the trade), or otherwise dispose of the Contract or any part thereof or the benefit or advantage of the Contractor or any part thereof without the previous consent in writing of the Authority.

11 Bankruptcy etc.

The Authority may at any time by notice in writing summarily determine the Contract without compensation to the Contractor in any of the following events:

(a) If the Contractor being an individual, or, where the Contractor is a firm, any partner in that firm, shall at any time become bankrupt, or shall have a receiving order or administration order made against him, or shall make any composition or arrangement with or for the benefit of his creditors, or shall make any conveyance or assignment for the benefit of his creditors, or shall purport to do so, or if in Scotland he shall become insolvent or notour bankrupt, or any application shall be made under any Bankruptcy Act for the time being in force for sequestration of his estate, or a trust deed shall be granted by him for behoof of his of his creditors, or

(b) if the Contractor being a company shall pass a resolution, or the Court shall make an order, that the company shall be wound up, or if a receiver or manager on behalf of a creditor shall be appointed, or if circumstances shall arise which entitle the Court or a creditor to appoint a receiver or manager or which entitle the Court to make a winding-up order;

Provided always that such determination shall not prejudice or affect any right of action or remedy which shall have accrued or shall accrue thereafter to the Authority

12 Decisions by the Authority

Any decision to be made by the Authority under the Contract may be made by any person or persons authorised to act for that purpose.

13 Rejection

(1) The Authority may reject any Article which on inspection in accordance with Condition No. 4 is found not to conform with the requirements of the Contract.

(2) The Authority may reject the whole of any consignment of the Articles if inspection in accordance with Condition No. 4 shows that:

(a) such proportion or percentage of the Articles in that consignment as the Contract may specify as being appropriate for the purposes of this Condition, or

(b) samples taken indiscriminately from that consignment, whether of the Articles or of the material in the Articles

do not conform with the requirements of the contract

(3) When under this Condition the Authority rejects any Article or consignment after delivery, the Contractor shall, subject to the provisions of Clause (6) of this Condition, at his own expense remove from the Authority each and every rejected Article and shall do so within such period as is provided by the Contract, or if the Contract makes no such provisions, within 8 working days from receipt of notification of rejection.

(4) If the Contractor shall fail to remove the Articles or any of them in accordance with Clause (3) of this Condition the Authority may return the rejected Articles or any of them to the Contractor at the Contractor's risk, the cost of carriage being recoverable from the Contractor

(5) When under this Condition the Authority rejects any Article or consignment after delivery, the Contractor shall at his own expense deliver in the place of each and every rejected Article an Article which conforms with the requirements of the Contract and shall do so within the period for delivery stipulated in the Contract or within such further reasonable period as the Authority may allow.

(6) If the Contractor considers himself aggrieved by a rejection under this Condition, he may give the Authority notice of objection. To be effective, such notice shall be given within 8 working days from receipt of notification of rejection and before removing the rejected Articles from the Authority The objection shall constitute a dispute between the parties which if not otherwise resolved between the parties within a reasonable time shall be dealt with in accordance with the provisions of the Contract relating to the settlement of disputes. If the Contractor gives notice of objection the Articles shall not be removed until the Authority directs

(7) If any of the Articles whether completed or in course of production is rejected on inspection by the Authority, the same shall if the Authority so requires, be marked in such a manner satisfactory to the Authority as to ensure its subsequent identification as a rejected Article

14 Acceptance

Acceptance of an Article shall take place when the Authority confirms acceptance of the Article in accordance with the procedure specified in the Contract, or if none is so specified then the Authority shall be deemed to have accepted an Article without prejudice to any other remedies, when as soon as any of the following events has occurred:

(a) The Authority has taken the Articles into use

(b) the Authority has not exercised its right of rejection of the Article under Condition No. 13 within any period specified for that purpose in the Contract.

(c) there being no period for exercising the right of rejection specified in the Contract, a reasonable time, all the circumstances having been taken into account, has elapsed since delivery of the Article was effected in accordance with Clause (4) of Condition No 6.

15 Payment

Bills for payment agreeing with the Contract Price shall be submitted after satisfactory completion of the Contract.

British Geological Survey
Natural Environment Research Council

Official Order **E82K** 063204

Name and address of supplier

Alphagraphics
6 Angel Row
Nottingham NG1 6HL

Invoice address

British Geological Survey
Keyworth Nottingham
For the attention of the Accounts Section

☎ 0115 936 3100 Telex 378173 BGSKEY G Fax 0115 936 3200
VAT Reg. No. 618 3673 25

Delivery address, if different from invoice address

For the attention of J. D. Bennett

Quantity	Description of goods or services	Spec. No.
35	35 copies of Report WC/96/39 (UNA Review 12),	
	as per spec. sheet attached.	

Delivery is required by: ASAP

Acknowledgement of order

We hereby accept this order from the British Geological Survey, and agree to supply the items requested in accordance with the Natural Environment Research Councils current standard conditions of contract no later than ....

Signature .... Date ....

Name in capitals .... ☎ ....

[redacted]

Please quote the above order number on all correspondence relating to this order

All orders from the British Geological Survey will be made or confirmed on an official order form

Authorised signatory (Finance)

J Haslam

Date

21/8/96

35

S.S.W — ~~#~~ 29 + 18 + 9 + 20 + 19 + 10 + 25 + 20 + 5 + 17

~~D.S.W~~ —

S.S. Colour sheet — Yellow |||| C Blank

D.S — " — ||

Colour Copy
S.S.C — || ✓
~~D.S.C~~

35 wiro's —

front sheet mass thick paper for colour copy X 35

35 Acetates

35 card white backs.

1007 2811

21 August 1997

Alphagraphics
6 Angel Row
Nottingham NG1 6HL

TDR REPORT SPECIFICATION: WC/96/39

Copying, collation and binding of above report (Unconsolidated sedimentary aquifers - Review 12 (groundwater quality management)):

- 35 copies

- ring-binding to allow full opening

- front cover to be white glossy card (or heavy weight copying paper plus clear acetate outer), rear cover likewise

- note 2 colour xerox (cover and text figure as flagged)

- single sided except for Introduction (as clipped, to allow world map halves to face each other) on **yellow** paper

- **yellow** divider sheets (blank, no text) as marked before Method Summary Sheets at end of each main chapter

J D Bennett
International Division

21/08/96 6:34 pm Wed

Job Ticket

Invoice # 12375
Job# 4 of 5
Ordered on: Wed, 21/08/96 at 6:28 pm
Wanted by: Fri, 23/08/96 at 12:00 am
Proof Due:

Customer: British Geological Survey
Contact: Accounts Dept (J.D.BENNETT)
Phone: 0115 936 3255
PO Number: E82K063204
Sales Rep: PC

Proofed by: ______

Quantity	Description	Actual	Done

Pre-Press

No Pre-Press operations on this job

Production

Description: COLOUR COPY
Stock: 105 GSM COLOTECH
Run Size: A4
Color: White

Vendor: XEROX
Parent Size: A4
Finish Size: A4

Sheets: 2 Up: 1
Copies: 35
Copier: Colour Copier A4
Sides: 1

Qty Ordered: 35
Parent Req'd: 70

	Front & Back	Actual	Done
Imp/Total:	70		
Setup Time:	0:00		
Run Time:	0:00		
Total Time:	0:00		

Bindery

Quantity	Description	Actual	Done

No Bindery operations on this job

21/08/96 6:34 pm Wed

Job Ticket

Invoice # 12375
Job# 3 of 5
Ordered on: Wed, 21/08/96 at 6:28 pm
Wanted by: Fri, 23/08/96 at 12:00 am
Proof Due:

Customer: British Geological Survey
Contact: Accounts Dept (J.D.BENNETT)
Phone: 0115 936 3255
PO Number: E82K063204
Sales Rep: PC

Proofed by: ______________________

Pre-Press

Quantity	Description	Actual	Done

No Pre-Press operations on this job

Production

Description: **FRONT COLOUR COPY**
Stock: 120g COLOURTECH
Run Size: A4
Color: WHITE
Sheets: 1 **Up:** 1
Copies: 35
Copier: Colour Copier A4
Sides: 1

Vendor: XEROX
Parent Size: A3
Finish Size: A4
Qty Ordered: 35
Parent Req'd: 18 (2 out)

	Front & Back	Actual	Done
Imp/Total:	35	______	______
Setup Time:	0:00	______	______
Run Time:	0:00	______	______
Total Time:	0:00	______	______

Bindery

Quantity	Description	Actual	Done

No Bindery operations on this job

21/08/96 6:34 pm Wed

Job Ticket

Invoice # 12375
Job# 1 of 5
Ordered on: Wed, 21/08/96 at 6:28 pm
Wanted by: Fri, 23/08/96 at 12:00 am
Proof Due:

Customer: British Geological Survey
Contact: Accounts Dept (J.D.BENNETT)
Phone: 0115 936 3255
PO Number: E82K063204
Sales Rep: PC

Proofed by: ______

Pre-Press

Quantity	Description	Actual	Done

No Pre-Press operations on this job

Production

Description: SINGLE SIDED WHITE
Stock: 80gm White Plain
Run Size: A4
Color:

Vendor: R & B
Parent Size: A4
Finish Size: A4

Sheets: 172 Up: 1
Copies: 35
Copier: A4 Docutech standard rate 3p per click
Sides: 1

Qty Ordered: 35
Parent Req'd: 6,020

	Front & Back	Actual	Done
Imp/Total:	6,020		
Setup Time:	0:00		
Run Time:	1:00		
Total Time:	1:00		

Bindery

Quantity	Description	Actual	Done

No Bindery operations on this job

21/08/96 6:34 pm Wed

Job Ticket

Invoice # 12375
Job# 2 of 5
Ordered on: Wed, 21/08/96 at 6:28 pm
Wanted by: Fri, 23/08/96 at 12:00 am
Proof Due:

Customer: British Geological Survey
Contact: Accounts Dept (J.D.BENNETT)
Phone: 0115 936 3255
PO Number: E82K063204
Sales Rep: PC

Proofed by: ____________________

Pre-Press

Quantity	Description	Actual	Done

No Pre-Press operations on this job

Production

Description: **YELLOW BLANK SHEETS**
Stock: 80gm Tint - Deep
Run Size: A4
Color:

Vendor: R & B
Parent Size: A4
Finish Size: A4

Sheets: 5 **Up:** 1
Copies: 35
Copier: Docutech A4(Account customers)
Sides: 1

Qty Ordered: 35
Parent Req'd: 175

	Front & Back	Actual	Done
Imp/Total:	175	____	____
Setup Time:	0:00	____	____
Run Time:	0:02	____	____
Total Time:	0:02	____	____

Bindery

Quantity	Description	Actual	Done

No Bindery operations on this job