Water Quality Assessments - A Guide to Use of Biota, Sediments and Water in Environmental Monitoring - Second Edition
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Chapter 9* - Groundwater

**This chapter was prepared by J. Chilton*

9.1. Introduction

Water from beneath the ground has been exploited for domestic use, livestock and irrigation since the earliest times. Although the precise nature of its occurrence was not necessarily understood, successful methods of bringing the water to the surface have been developed and groundwater use has grown consistently ever since. It is, however, common for the dominant role of groundwater in the freshwater part of the hydrological cycle to be overlooked. Groundwater is easily the most important component and constitutes about two thirds of the freshwater resources of the world and, if the polar ice caps and glaciers are not considered, groundwater accounts for nearly all usable freshwater (see Table 1.1). Even if consideration is further limited to only the most active and accessible groundwater bodies (estimated by Lvovitch (1972) at 4×10^6 km^3) then they constitute 95 per cent of total freshwater. Lakes, swamps, reservoirs and rivers account for 3.5 per cent and soil moisture accounts for only 1.5 per cent (Freeze and Cherry, 1979). The dominant role of groundwater resources is clear and their use and protection is, therefore, of fundamental importance to human life and economic activity.

It is easy for the importance of groundwater in water supplies to be underestimated. It is customary to think of groundwater as being more important in arid or semi-arid areas and surface water as more important in humid areas. However, inventories of groundwater and surface water use reveal the worldwide importance of groundwater. The reasons for this include its convenient availability close to where water is required, its excellent natural quality (which is generally adequate for potable supplies with little or no treatment) and the relatively low capital cost of development. Development in stages, to keep pace with rising demand, is usually more easily achieved for ground-water than for surface water.

In the USA where groundwater is important in all climatic regions, it accounts for about 50 per cent of livestock and irrigation water use, and just under 40 per cent of public water supplies. In rural areas of the USA, 96 per cent of domestic water is supplied from groundwater (Todd, 1980). Some very large cities are totally dependent on groundwater. In Latin America, many of the continent's largest cities, Mexico City, Lima, Buenos Aires and Santiago, obtain a significant proportion of their municipal water supply from groundwater. In the valley of Mexico City, over 1,000 deep wells supply $3,200 \times 10^6$ m^3

day^{-1}, which is about 95 per cent of the total supply to a population of nearly 20 million people (Foster *et al.,* 1987). In Europe also, groundwater has always played a major part in water supplies. The proportion of groundwater in drinking water supplies in some European countries in 1988 was (UNEP, 1989):

Denmark	98%	Netherlands	67%
Portugal	94%	Luxembourg	66%
Germany Fed. Rep.	89%	Sweden	49%
Italy	88%	United Kingdom	35%
Switzerland	75%	Spain	20 %
Belgium	67%	Norway	15%

Many of the major cities of Europe are, therefore, dependent on ground-water. In Africa and Asia, most of the largest cities use surface water, but many millions of people in the rural areas are dependent on groundwater. For many millions more, particularly in sub-Saharan Africa, who do not as yet have any form of improved supply, untreated groundwater supplies from protected wells with handpumps are likely to be their best solution for many years to come.

Water is drawn from the ground for a variety of uses, principally community water supply, farming (both livestock and irrigated cultivation) and industrial processes. Unlike surface water, groundwater is rarely used *in situ* for non-consumptive purposes such as recreation and fisheries, except occasionally where it comes to the surface as springs. Consequently, ground-water quality assessment is invariably directed towards factors which may lessen the suitability of pumped groundwater with respect to its potability and use in agriculture and industry.

The overall goal of a groundwater quality assessment programme, as for surface water programmes, is to obtain a comprehensive picture of the spatial distribution of groundwater quality and of the changes in time that occur, either naturally, or under the influence of man (Wilkinson and Edworthy, 1981). The benefits of well designed and executed programmes are that timely water quality management, and/or pollution control measures, can be taken which are based on comprehensive and appropriate water quality information. Each specific assessment programme is designed to meet a specific objective, or several objectives, which are related in each case to relevant water quality issues and water uses (see Chapter 2).

Two principal features of groundwater bodies distinguish them from surface water bodies. Firstly, the relatively slow movement of water through the ground means that residence times in groundwaters are generally orders of magnitude longer than in surface waters (see Table 1.1). Once polluted, a groundwater body could remain so for decades, or even for hundreds of years, because the natural processes of through-flushing are so slow. Secondly, there is a considerable degree of physico-chemical and chemical interdependence between the water and the containing material. The word groundwater, without further qualification, is generally understood to mean all the water underground, occupying the voids within geological formations. It follows, therefore, that in dealing with groundwater, the properties of both the ground and the water are important, and there is considerable scope for water quality to be modified by interaction between the two, as described in section 9.3. The scope for such modification is in turn enhanced by the long residence times, which depend on the size and type of the groundwater body (Figure 1.2). To appreciate the particular difficulties of monitoring ground-water bodies, it is necessary first to identify and to discuss briefly those properties of ground and water that are relevant to the occurrence and movement of groundwater. This is done in the following sections. Only a brief summary is possible, but further information is available in Price (1985) which gives a general introduction to the subject. Comprehensive descriptions of hydrogeology are given by Freeze and Cherry (1979), Todd (1980) and Driscoll(1986).

9.2. Characteristics of groundwater bodies

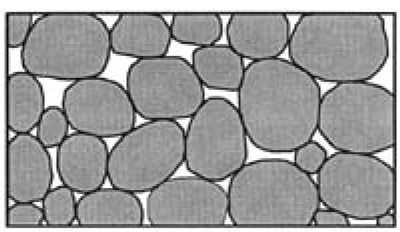
9.2.1 Occurrence of groundwater

Groundwater occurs in many different geological formations. Nearly all rocks in the upper part of the Earth's crust, whatever their type, origin or age, possess openings called pores or voids. In unconsolidated, granular materials the voids are the spaces between the grains (Figure 9.1a), which may become reduced by compaction and cementation (Figure 9.1d). In consolidated rocks, the only voids may be the fractures or fissures, which are generally restricted but may be enlarged by solution (Figure 9.1 e, f). The volume of water contained in the rock depends on the percentage of these openings or pores in a given volume of the rock, which is termed the porosity of the rock. More pore spaces result in higher porosity and more stored water. Typical porosity ranges for common geological materials are shown in Table 9.1.

Only a part of the water contained in the fully-saturated pores can be abstracted and used. Under the influence of gravity when, for example, the water level falls, part of the water drains from the pores and part remains held by surface tension and molecular effects. The ratio of the volume of water that will drain under gravity from an initially saturated rock mass to the total volume of that rock (including the enclosed water) is defined as the specific yield of the material, and is usually expressed as a percentage. Typical values are shown in Table 9.1.

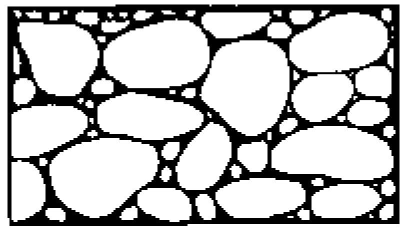
Figure 9.1 Rock texture and porosity of typical aquifer materials (Based on Todd, 1980)

a) Well sorted sedimentary deposit with high porosity



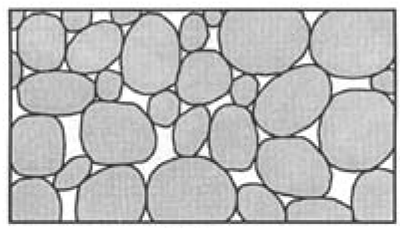
(a)

b) Poorly sorted sedimentary deposit with low porosity



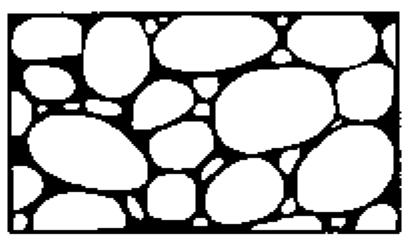
(b)

c) Well sorted sedimentary deposit of porous pebbles, resulting in high overall porosity



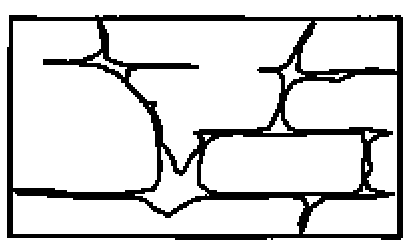
(c)

d) Well sorted sedimentary deposit in which porosity has been reduced by cement deposited between grains



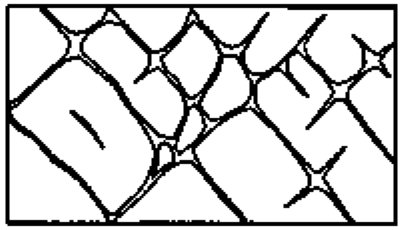
(d)

e) Consolidated rock rendered porous by solution



(e)

f) Consolidated rock rendered porous by fracturing



(f)

Groundwater is not usually static but flows through the rock. The ease with which water can flow through a rock mass depends on a combination of the size of the pores and the degree to which they are inter-connected. This is defined as the permeability of the rock. Materials which permit water to pass through them easily are said to be permeable and those which permit water to pass only with difficulty, or not at all, are described as impermeable. A layer of rock that is sufficiently porous to store water and permeable enough to transmit water in quantities that can be economically exploited is called an aquifer. Groundwater flow may take place through the spaces between the grains or through fissures (Figure 9.1), or by a combination of the two in, for example, a jointed sandstone or limestone. For any aquifer, distinguishing whether inter-granular or fissure flow predominates is fundamental to understanding the hydrogeology and to designing monitoring systems, particularly for point source pollution incidents.

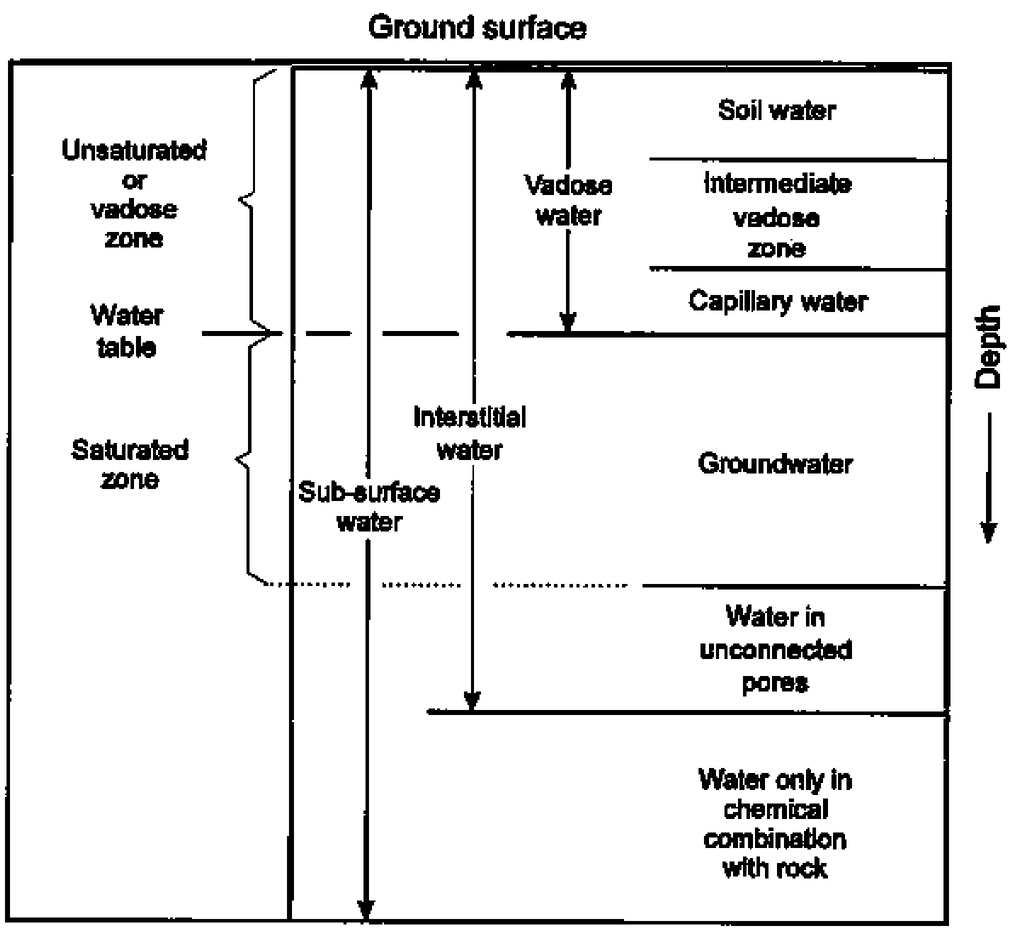
Table 9.1 Porosity and specific yield of geological materials

Material	**Porosity (%)**	**Specific yield (%)**
Unconsolidated sediments		
Gravel	25-35	15-30
Sand	25-45	10-30
Silt	35-50	5-10
Clay	45-55	1-5
Sand and gravel	20-30	10-20
Glacial till	20-30	5-15
Consolidated rocks		
Sandstone	5-30	3-15
Limestone and dolomite	1-20	0.5-10
Karst limestone	5-30	2-15
Shale	1-10	0.5-5
Vesicular basalt	10-40	5-15
Fractured basalt	5-30	2-10
Tuff	10-60	5-20
Fresh granite and gneiss	0.01-2	< 0.1
Weathered granite and gneiss	1-15	0.5-5

Sources: Freeze and Cherry, 1979; Todd, 1980; Driscoll, 1986

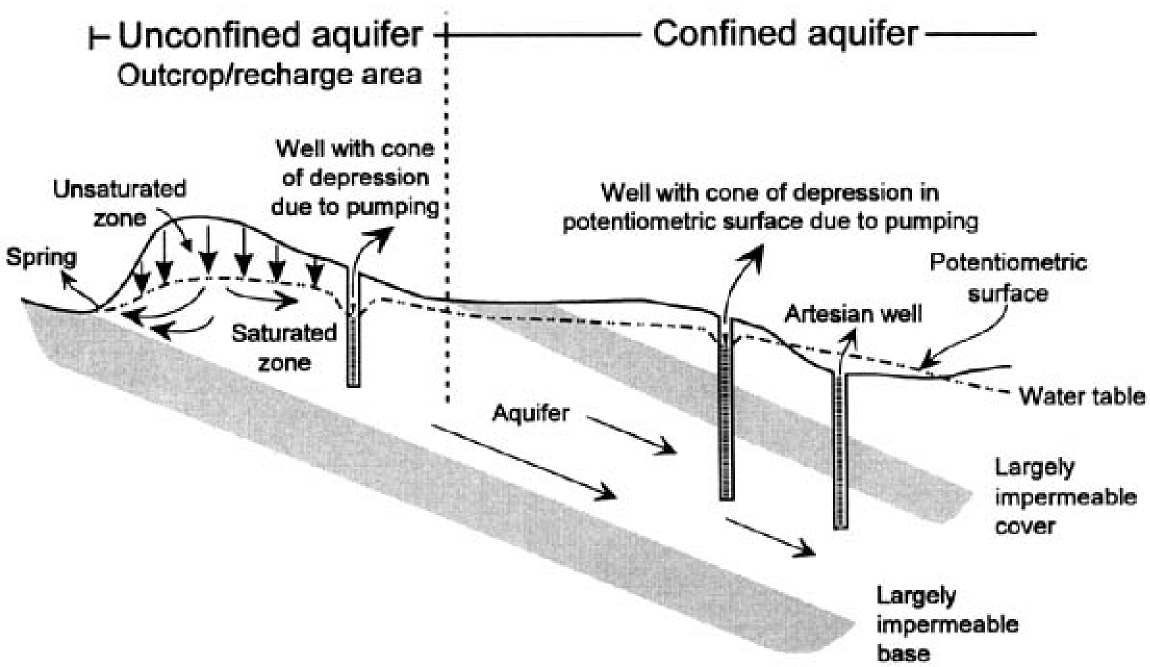
When rain falls, some infiltrates into the soil. Part of this moisture is taken up by the roots of plants and some moves deeper under the influence of gravity. In the rock nearest to the ground surface, the empty spaces are partly filled with water and partly with air. This is known as the unsaturated or vadose zone (Figure 9.2), and can vary in depth from nothing to tens of metres. In the unsaturated zone, soil, air and water are in contact and may react with each other. Downward water movement in the unsaturated zone is slow, less than 10 m a^{-1} and often less than 1 m a^{-1}. Residence times in the unsaturated zone depend on its thickness and can vary from almost nothing to tens of years. At greater depths, all the empty spaces are completely filled with water and this is called the saturated zone. If a hole is dug or drilled down into the saturated zone, water will flow from the ground into the hole and settle at the depth below which all the pore spaces are filled with water. This level is the water table and forms the upper surface of the saturated zone, at which the fluid pressure in the pores is exactly atmospheric. Strictly speaking, the term groundwater refers only to the saturated zone below the water table. All water that occurs naturally beneath the Earth's surface, including saturated and unsaturated zones, is called sub-surface water (Figure 9.2).

Figure 9.2 Classification of sub-surface water (Modified from Driscoll, 1986)



A further important way of characterising aquifers is useful in any discussion of the development and protection of groundwater resources. An unconfined aquifer is one in which the upper limit of the zone of saturation, i.e. the water table, is at atmospheric pressure. At any depth below the water table the pressure is greater than atmospheric and at any point above the water table the pressure is less than atmospheric. In a confined aquifer the effective aquifer thickness extends between two impermeable layers. At any point in the aquifer, the water pressure is greater than atmospheric. If a well is drilled through the confining layer into the aquifer, water will rise up into the well until the column of water in the well balances the pressure in the aquifer. An imaginary surface joining the water level in many wells in a confined aquifer is called the potentiometric surface. For a phreatic aquifer, that is the first unconfined aquifer to be formed from the land surface, the potentiometric surface corresponds to the water table found in wells or boreholes. The flow directions of groundwater are perpendicular to the isolines of the potentiometric surface. An example is given later in Figure 9.26.

Figure 9.3 Schematic cross-section illustrating confined and unconfined aquifers



If the pressure in a confined aquifer is such that the potentiometric surface comes above ground level, then the well will overflow and is said to be artesian. The two types of aquifer are shown in Figure 9.3. Clearly, a confined aquifer with thick overlying impermeable clays is likely to be much less vulnerable to pollution than an unconfined aquifer.

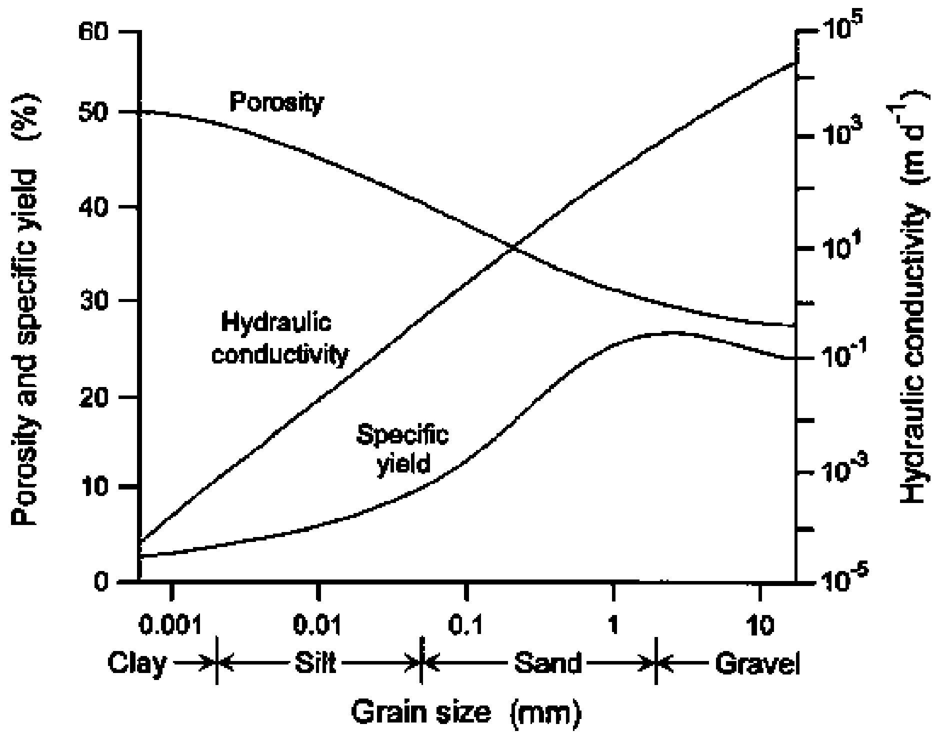
9.2.2 Groundwater flow

The flow of water through an aquifer is governed by Darcy's Law, which states that the rate of flow is directly proportional to the hydraulic gradient:

$$Q = - K\, i\, A$$

where Q is the rate of flow through unit area A under hydraulic gradient i. The hydraulic gradient dh/dl is the difference between the levels of the potentiometric surface at any two points divided by the horizontal distance between them. The parameter *K* is known as the hydraulic conductivity, and is a measure of the permeability of the material through which the water is flowing. The similarity between Darcy's Law and the other important laws of physics governing the flow of both electricity and heat should be noted. For clean, granular materials, hydraulic conductivity increases with grain size (Figure 9.4). Typical ranges of hydraulic conductivity for the main geological materials are shown in Figure 9.5.

Figure 9.4 Porosity, specific yield and hydraulic conductivity of granular materials (Modified from Davis and De Wiest, 1966)

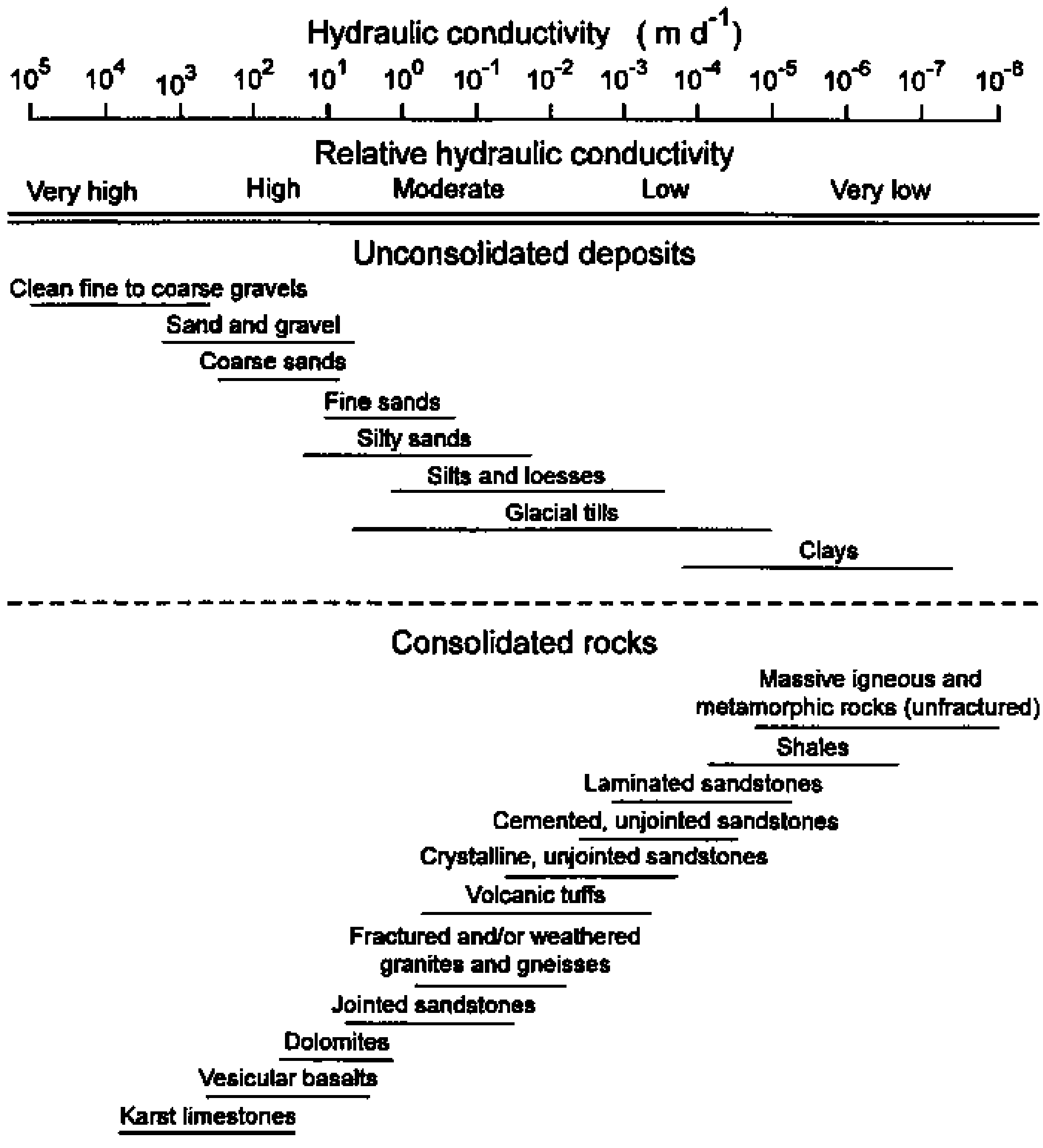


Darcy's Law can be written in several forms. By substituting *V* for the term *Q/A* it can take the form:

$$V = -K\frac{dh}{dl}$$

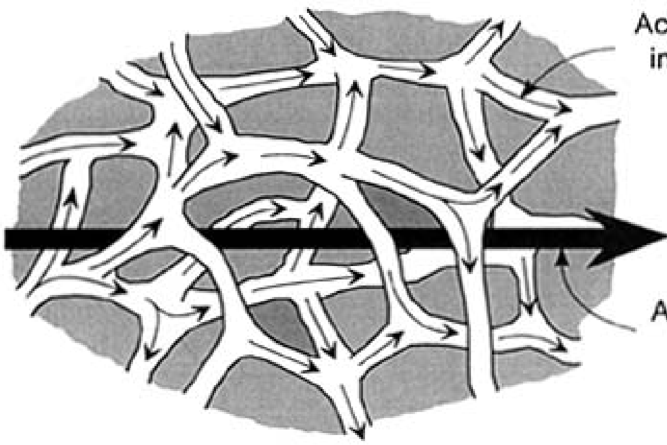
in which the term *V* is commonly referred to as the specific discharge. The specific discharge does not represent the actual velocity of groundwater flow through the aquifer material. To determine specific discharge, the volumetric flux *Q* is divided by the full cross-sectional area, which includes both solids and voids. Clearly, flow can only take place through the pore spaces, and the actual velocity of groundwater flow can be calculated if the porosity is known. Most aquifer materials in which inter-granular flow predominates have porosities of 20 to 40 per cent, so the actual groundwater flow velocity is three to five times the specific discharge. However, this average velocity in the direction of groundwater flow does not represent the true velocity of water particles travelling through the pore spaces. These microscopic velocities are generally greater, as the inter-granular flow pathways are irregular and longer than the average, linear, macroscopic pathway (Figure 9.6). The true microscopic velocities are seldom of interest, and are in any case almost impossible to determine.

Figure 9.5 Range of hydraulic conductivity values for geological materials (Based on Driscoll, 1986 and Todd, 1980)



Darcy's Law provides a valid description of the flow of groundwater in most naturally occurring hydrogeological conditions. Thus, in general, it holds true for fractured rock aquifers as well as granular materials, although there are particular problems associated with the analysis of flow in fractured media which are outside the scope of the present discussion. Fissure flow conditions are characterised by low porosity and high permeability, and very high flow velocities may result, especially where a small number of fissures are enlarged by solution (Figure 9.1e). Extensive development of solution in limestone areas can result in karst terrain, characterised by solution channels, closed depressions, sink holes and caves, into which all traces of surface drainage may disappear. Extreme groundwater velocities of several kilometres per day have been observed in tracer experiments in such areas. These conditions can be very favourable for groundwater abstraction, but aquifers of this type are often highly vulnerable to all types of pollution and difficult to monitor.

Figure 9.6 Average linear and microscopic flow paths (Based on Freeze and Cherry, 1979)



Although most of the flow in a fractured rock aquifer is through the fissures, very slow groundwater movement may occur through the interconnected voids of the matrix. An aquifer of this type is characterised by high porosity and low permeability in the matrix with low porosity and high permeability in the fractures. The volume of groundwater stored in the matrix may be 20 to 100 times greater than the volume in the fractures. Many limestones, some sandstones and certain types of volcanic rocks exhibit these characteristics, and such aquifers can be said to have dual porosity and permeability. One of the most intensively used and well studied aquifers of this type is the chalk of north west Europe.

9.2.3 Groundwater flow systems

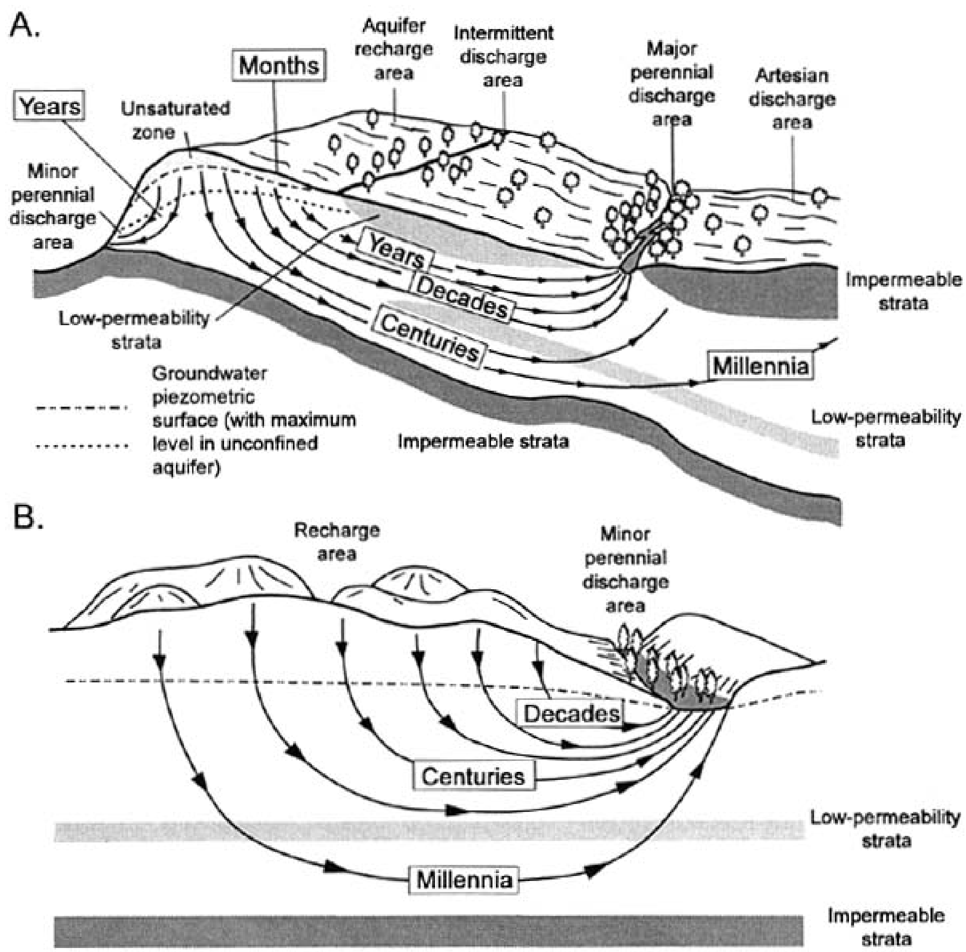
The origin of fresh groundwater is normally atmospheric precipitation of some kind, either by direct infiltration of rainfall or indirectly from rivers, lakes or canals. Groundwater is, in turn, the origin of much stream-flow and an important flow component to lakes and oceans and is, therefore, an integral part of the hydrological cycle. Water bodies such as marshes are often transitional between groundwater and surface water (see Figure 1.1). At all places where surface and groundwater meet, the inter-connection between the two must be appreciated in relation to the design of water quality assessment programmes.

Within the context of the overall cycle, the groundwater flow system is a useful concept in describing both the physical occurrence and geochemical evolution of groundwater. A groundwater flow system is a three dimensional, closed system containing the flow paths from the point at which water enters an aquifer to the topographically lower point where it leaves. Infiltration of rainfall on high ground occurs in a recharge area in which the hydraulic head decreases with depth and net saturated flow is downwards away from the water table. After moving slowly through the aquifer down the hydraulic gradient, groundwater leaves the aquifer by springs, swamps and baseflow to streams or to the ocean in a discharge area. In a discharge area, the hydraulic head increases with depth and the net saturated flow is upwards towards the water table. In a recharge area, the water table often lies at considerable depth beneath a thick unsaturated zone. In a discharge area, the water table is usually at, or very near, the ground surface. Rivers, canals, lakes and reservoirs may either discharge to, or recharge from, groundwater and the relationship may change seasonally or over a longer time span.

Examples of groundwater flow regimes under humid and semi-arid climatic conditions are shown in Figure 9.7. In large, deep aquifers, ground-water may be moving slowly, at rates of a few metres per year, from recharge to discharge areas for hundreds or thousands of years. In small, shallow aquifers, recharge and discharge areas may be much closer or even adjacent to each other, and residence times can be restricted to a few months or years (Figure 9.7). In arid and semi-arid areas, groundwater discharge areas are often characterised by poor quality groundwater, particularly with high salinity. In these areas, groundwater discharge occurs from seepages or salt marshes with distinctive vegetation, known as salinas or playas, in which evapotranspiration at high rates, over a long time, has led to a build up of salinity.

Water movement in the unsaturated zone cannot be determined by the same methods as for the saturated zone because water will not freely enter void spaces unless the soil-water pressure is greater than atmospheric. Therefore, special methods must be used to measure water movement in the unsaturated zone, and the most common are tensiometers, psychrometers and neutron moisture logging. Groundwater flow in the unsaturated zone is dominantly vertical, and is related to the rate of infiltration, the storage capacity of the vadose zone and the depth to the water table. Downward movement in granular materials is generally a few metres per year or less, and residence times of tens of years are thus possible where the unsaturated zone is thick. In fissured or dual porosity aquifers exposed at the surface, rapid infiltration through the fissures can by-pass the unsaturated zone completely, dramatically reducing residence times and making the aquifer highly vulnerable to pollution.

Figure 9.7 Groundwater flow systems A. Humid regions B. Arid regions. The residence periods indicated are typical order-of-magnitude values from time of recharge to point of discharge. (After Foster and Hirata, 1988)



9.2.4 Matrix of aquifer types

It is usual to classify all rock types as igneous, sedimentary or metamorphic, depending on their origins. Igneous rocks are formed by the cooling and solidifying of molten rock or magma, after it has been either forced (intruded) into other rocks below ground or extruded at the ground surface as a volcanic eruption. Sedimentary rocks are formed as a result of the deposition of particles (often resulting from the weathering and erosion of other rocks) usually under water on the sea bed, or in rivers, lakes and reservoirs. Metamorphic rocks are formed by the alteration of other rocks under the action of heat and pressure.

Most of the world's important regional aquifers are of sedimentary origin. Igneous and metamorphic rocks are generally far less important as sources of groundwater. As a result of their method of formation, igneous and metamorphic rocks often have such low initial porosity and permeability (Table 9.1, Figure 9.5) that they do not form important aquifers. In time, and particularly with elevation to the ground surface and prolonged exposure to weathering, joints may be formed and opened up so that sufficient porosity and permeability is created to allow the igneous and metamorphic rocks to be called aquifers. Locally these weathered rocks can be vital sources of water. In contrast, sediments usually start their geological lives with a high porosity and a permeability which is closely related to grain size (Figure 9.4). With time, they may be buried and compacted, and cement may form between the grains (Figure 9.1 d); such consolidated or indurated sedimentary rocks may have low porosity and permeability and only become aquifers by virtue of their joints and fractures.

Table 9.2 represents an attempt to construct a matrix of major aquifer types. The aquifers in the upper part of the table are largely unconsolidated and relatively young in geological terms. They are, therefore, invariably found close to the surface and form aquifers at shallow depths, usually less than 100 to 200 m. Much older sediments which started life as unconsolidated materials are found at greater depths. There are, for example, ancient glacial deposits many millions of years old but these have been deeply buried and compacted and have lost so much of their original porosity that they can no longer be considered as aquifers. Unconsolidated materials in which inter-granular flow dominates are, therefore, restricted to relatively shallow depths (Table 9.2). Unconsolidated materials are widely distributed and very important as aquifers. They are characterised by generally good natural water quality because of the short to moderate residence times and regular recharge, at least in humid areas, but are often highly vulnerable to anthropogenic influences.

Table 9.2 Major aquifer types

	Principal aquifer features					**Examples of groundwater bodies**		
Geological environment	**Formations**	**Lithologies**	**Class**	**Dominant porosity**	**Groundwater flow regime**	**Small shallow**	**Large shallow**	**Large deep**
Glacial deposits	Eskers, kames, terraces, fans, moraines, buried valleys	Sands, gravels mixed sands and gravels, boulders, clay lenses	SU	P	I	Canada, Northern USA, Denmark		
Fluvial deposits	Terraces, fans, buried valleys	Sands, silts, day lenses	SU	P	I	World-wide		
	Alluvium	Sand, silts	SU	P	I	World-wide	Netherlands N Germany Indo-Gangetic Plain	
Deltaic deposits	Alluvium	Fine sands, silts	SU	P	I		Lower Indus Banglade	

							sh Mekong	
Aeolian deposits	Dune sands	Sands	SU	P	I	Netherlands		
	Loess	Silts	SU	P	I	?	China	
Marine deposits	Limestones, dolomites	Oolites, marls	SC	P,S S	F,I	Caribbean Islands	Florida	N Europe chalk Ogallala (USA)
	Karst limestones		SC	S	F	Caribbean Islands	Yucatan, Jaffna Yugoslavia	
	Sandstones	Cemented sand grains	SC	S	F,I			Nubian, Karoo Great Artesian Basin (Australia)
Volcanics	Ashes	Disaggregated fragments	IU	P	I			
	Lavas	Fine-grained crystalline	IC	S	F	Hawaii	Deccan basalts Columbia River Plateau (USA)	Karoo basalts
	Tuffs	Cemented grains	IC		I,F	Central America		
Igneous and metamorphic	Granites, schists, gneisses (fresh)	Crystalline	IMC	S	F			
	Granites, gneisses, schists (weathered)	Disaggregated crystalline	IMC	S	F,I		Africa, India, Sri Lanka, Brazil	

Rock class and degree of consolidation: S Sedimentary; I Igneous; M Metamorphic; U Unconsolidated; C Consolidated

Dominant porosity: P Primary; S Secondary

Groundwater flow regime: I Intergranular; F Fissure

Consolidated formations produce shallow and deep aquifers (Table 9.2). Sedimentary formations originally formed at shallow depths and having high inter-granular porosity may have been deeply buried, compacted, cemented and subsequently brought back to the ground surface by major earth movements. If secondary porosity has developed (Table 9.1), they then form shallow aquifers in which both fissure flow and inter-granular flow may be important (Table 9.2). Such aquifers may be highly productive and have generally good natural groundwater quality if there is regular recharge, but may be especially vulnerable to pollution because of the possibility of very rapid flow in fissures (Figure 9.5).

Similarly, igneous and metamorphic rocks having low original porosity when they were formed at great depths within the earth's crust, can form important (but usually relatively low yielding) shallow aquifers when brought to the earth's surface and exposed to weathering processes. At greater depths, without weathering most of the fissures are closed and insufficient secondary porosity or permeability develops for such rocks to be considered aquifers (Table 9.2).

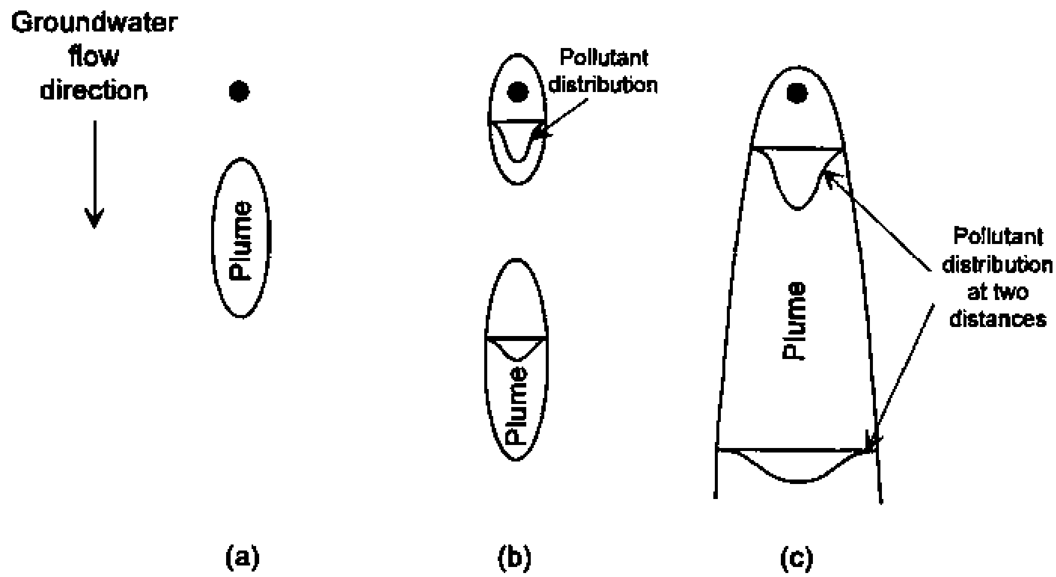
Igneous volcanic rocks come to the ground surface as molten lava or as ejected fragmentary material. These fragments may be welded by the great heat and gasses associated with volcanic activity and form tuffs. Fine grained crystalline lavas have almost no primary porosity. Their usefulness as aquifers depends on features such as cooling joints, the blocky nature of some lava types, and the weathering of the top of an individual flow before the next one is extruded over it. Shallow volcanic aquifers have generally good natural water quality for the same reasons as other shallow aquifers, except that in areas of current or recent volcanic or hydrothermal activity, specific constituents, particularly fluoride may present problems.

Deep, consolidated sedimentary formations are characterised by slow groundwater movement, long residence times, ample opportunity for dissolution of minerals and, therefore, often poor natural water quality. As a result of their great depth, such formations are often confined beneath thick sequences of low permeability clays or shales, and are generally less vulnerable to anthropogenic influences.

9.2.5 Contaminant transport

In water quality investigations, the groundwater flow system is considered in terms of its ability to transport dissolved substances or solutes, which may be natural chemical constituents or contaminants. Solutes are transported by the bulk movement of the flowing groundwater, a process termed advection. However, when a small volume of solute, either a contaminant or an artificial tracer, is released into an aquifer it will spread out from the expected advective flow path. Instead it forms a plume of diluted solute, which broadens both along, and perpendicular to, the flow direction (Figure 9.8). Two processes can contribute to this phenomenon (Freeze and Cherry, 1979). The first is molecular diffusion in the direction of the concentration gradient due to the thermal-kinetic energy of the solute particles. This process is only important at low velocities. Much more important is the mechanical dispersion which arises from the tortuosity of the pore channels in a granular aquifer and of the fissures in a fractured aquifer, and from the different speeds of groundwater flow in channels or fissures of different widths.

Figure 9.8 Plume of contaminant generated from a) a slug source or spill, b) an intermittent source, and c) a continuous source (Modified from Barcelona *et al*., 1985)



In dual porosity aquifers, groundwater solute concentrations may be significantly different in the fissures than in the matrix. In either diffuse or point source pollution, the water moving in the fissures may have a concentration higher than the water in the matrix, known as the pore-water; hence diffusion from fissures to matrix will occur. In time, if the source of pollutant is removed, the solute concentration in the fissures may decline below that of the pore-water, and diffusion in the opposite direction may result. Ground-water sampling at this stage by conventional methods would sample the fissure water and perhaps miss an important reservoir of pollutant in the rock matrix.

If pollutants find their way into aquifers in the immiscible phase, transport will be governed by completely different factors from those which determine groundwater flow, notably the density and viscosity of the immiscible fluid (Lawrence and Foster, 1987). The aromatic hydrocarbons are less dense, and generally more viscous, than water. In the immiscible phase they tend to “float” at the water table (Figure 9.9), and subsequent lateral migration depends on the hydraulic gradient. In this position they can rise with rising water levels, and during subsequent recession, the hydrocarbon may be held by surface tension effects in the pore spaces of the unsaturated zone. In contrast, the chlorinated solvents in the immiscible phase have a considerably higher density and lower viscosity than water, a combination of properties that can result in rapid and deep penetration into aquifers (Schwille, 1981, 1988). These solvents may reach the base of an aquifer, where they could accumulate in depressions or migrate down slope, irrespective of the direction of groundwater flow.

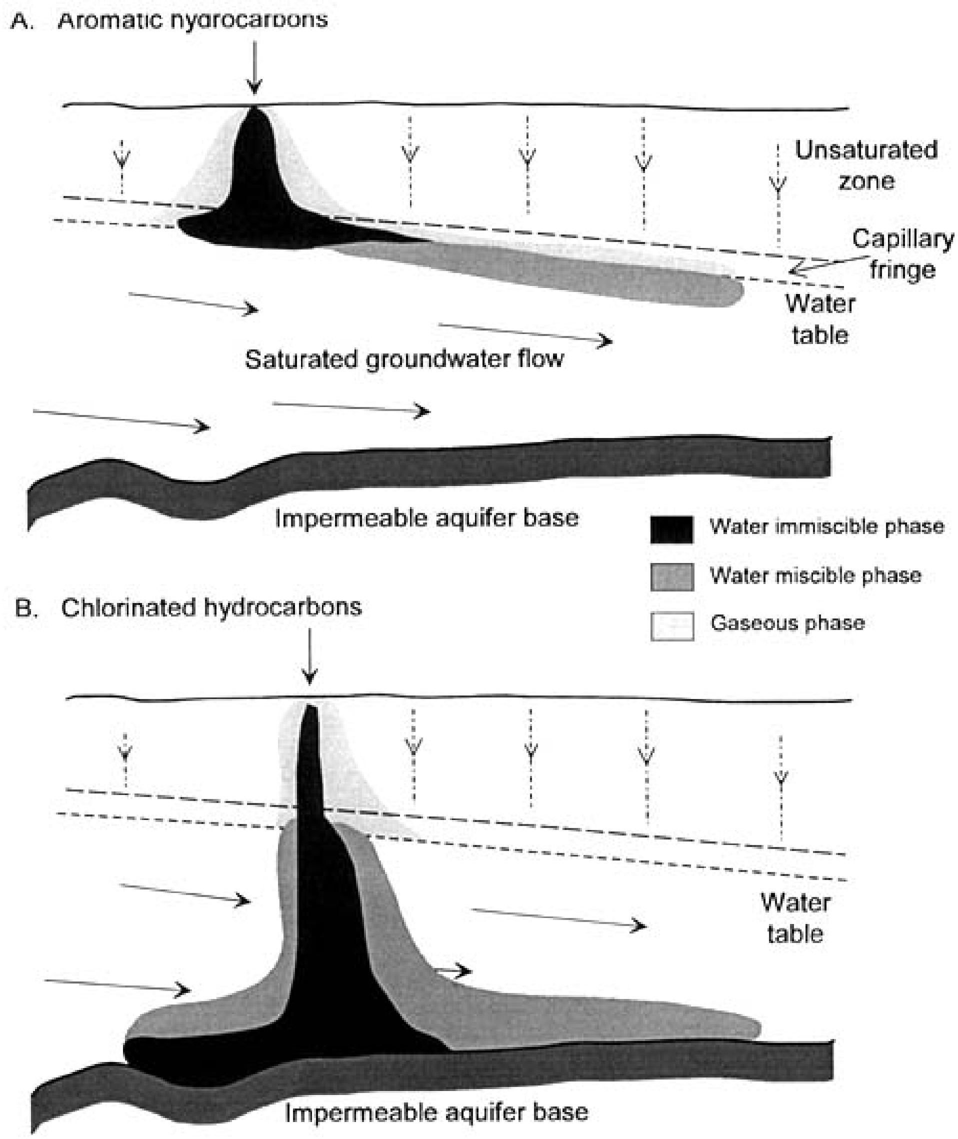
An important factor controlling the persistence of solvents in groundwater is the extent to which the immiscible phase can displace water from a fine grained porous matrix of, for example, a fissured limestone aquifer. This depends on the surface tension properties of the immiscible fluid relative to water and to the minerals of the aquifer matrix (Lawrence and Foster, 1987). It seems likely that considerable excess pressure would be required to enable the immiscible phase of most organic fluids to penetrate into the matrix. Once in the aquifer, an immiscible body of dense solvent could act as a buried pollution source, perhaps for many years or decades. The rate of dissolution will depend on the solubility of the particular compound in water, the rate of groundwater flow and the degree of mixing that is permitted by the distribution of the solvent in relation to the local hydraulic structure and flow pathways. The complexity of behaviour by an immiscible contaminant presents considerable problems for the design of assessment programmes and for the interpretation of the monitoring data.

9.2.6 Chemical characteristics of groundwater

Since groundwater often occurs in association with geological materials containing soluble minerals, higher concentrations of dissolved salts are normally expected in groundwater relative to surface water. The type and concentration of salts depends on the geological environment and the source and movement of the water. A brief description of the principal natural chemical changes that occur in groundwater systems is given in section 9.3. The following paragraphs outline the main ways in which the natural characteristics of groundwater affect water quality for different uses.

A simple hydrochemical classification divides groundwaters into meteoric, connate and juvenile. Meteoric groundwater, easily the most important, is derived from rainfall and infiltration within the normal hydrological cycle (Figure 9.7) and is subjected to the type of hydrochemical evolution described in section 9.3. Groundwater originating as sea water which has been entrapped in the pores of marine sediments since their time of deposition is generally referred to as connate water. The term has usually been applied to saline water encountered at great depths in old sedimentary formations. It is now accepted that meteoric groundwater can eventually become equally saline, and that entrapped sea water can become modified and moved from its original place of entrapment. It is doubtful whether groundwater exists that meets the original definition of connate water, and the non-generic term formation water is preferred by many authors. Connate water is, perhaps, useful to describe groundwater that has been removed from atmospheric circulation for a significant period of geological time. Formation waters are not usually developed for water supplies because of their high salinity. However, they may become involved in the assessment of saline intrusions caused by the overpumping of overlying aquifers.

Figure 9.9 Generalised distribution of hydrocarbon phases down a groundwater gradient following a surface spillage (After Lawrence and Foster, 1987 in Meybeck *et al.,* 1989)



Juvenile groundwater describes the relatively small amounts of water which have not previously been involved in the circulating system of the hydrological cycle, but are derived from igneous processes within the earth. However, juvenile groundwater often cannot be distinguished geochemically from meteoric groundwater that has circulated to great depths and become involved in igneous processes. True juvenile waters unmixed with meteoric water are rare and of very localised extent, and are not normally associated with the development and assessment of fresh groundwater resources.

The natural chemical quality of groundwater is generally good, but elevated concentrations of a number of constituents can cause problems for water use, as described in Chapter 3. The relative abundance of the constituents dissolved in groundwater is given in Table 9.3, and a summary of the natural sources and range of concentrations of the principal constituents of groundwater is given in Table 9.4.

High iron levels in groundwater are widely reported from developing countries, where they are often an important water quality issue. Consumers may reject untreated groundwater from handpump supplies if it has a high iron concentration in favour of unprotected surface water sources with low iron levels but which may have gross bacteriological pollution. The situation is made worse in many areas by the corrosion of ferrous well linings and pump components. Obtaining representative samples for iron in groundwater presents particular difficulties because of the transformations brought about by the change in oxidation/reduction status which occurs on lifting the water from the aquifer to ground level.

9.2.7 Biological characteristics of groundwater

Groundwater quality can be influenced directly and indirectly by microbiological processes, which can transform both inorganic and organic constituents of groundwater. These biological transformations usually hasten geochemical processes (Chapelle, 1993). Single and multi-celled organisms have become adapted to using the dissolved material and suspended solids in the water and solid matter in the aquifer in their metabolism, and then releasing the metabolic products back into the water (Matthess, 1982). There is practically no geological environment at, or near, the earth's surface where the pH and Eh conditions will not support some form of organic life (Chilton and West, 1992). In addition to groups tolerating extremes of pH and Eh, there are groups of microbes which prefer low temperatures (psychrophiles), others which prefer high temperature (thermophiles) (Ehrlich, 1990), and yet others which are tolerant of high pressures. However, the most biologically favourable environments generally occur in warm, humid conditions.

Table 9.3 Relative abundance of dissolved constituents in groundwater

Major constituents (1.0 to 1,000 mg l^{-1})	**Secondary constituents (0.01 to 10.0 mg l^{-1})**	**Minor constituents (0.0001 to 0.1 mg l^{-1})**
Sodium Calcium Magnesium Bicarbonate Sulphate Chloride Silica	Iron Aluminium Potassium Carbonate Nitrate Fluoride Boron Selenium	Arsenic Barium Bromide Cadmium Chromium Cobalt Copper Iodide Lead Lithium Manganese Nickel Phosphate Strontium Uranium Zinc

Source: After Todd, 1980

Micro-organisms do not affect the direction of reactions governed by the thermodynamic constraints of the system, but they do affect their rate. Sulphides, for example, can be oxidised without microbial help, but microbial processes can greatly speed up oxidation to the extent that, under optimum moisture and temperature conditions, they become dominant over physical and chemical factors.

All organic compounds can act as potential sources of energy for organisms. Most organisms require oxygen for respiration (aerobic respiration) and the breakdown of organic matter, but when oxygen concentrations are depleted some bacteria can use alternatives, such as nitrate, sulphate and carbon dioxide (anaerobic respiration). Organisms which can live in the presence of oxygen (or without it) are known as facultative anaerobes. In contrast, obligate anaerobes are organisms which do not like oxygen. The presence or absence of oxygen is, therefore, one of the most important factors affecting microbial activity, but not the only one. For an organism to grow and multiply, nutrients must be supplied in an appropriate mix, which satisfies carbon, energy, nitrogen and mineral requirements (Ehrlich, 1990).

Table 9.4 Sources and concentrations of natural groundwater components

Component	Natural sources	Concentration in natural water
Dissolved solids	Mineral constituents dissolved in water	Usually < 5,000 mg l^{-1}, but some brines contain as much as 300,000 mg l^{-1}
Nitrate	Atmosphere, legumes, plant debris, animal excrement	Usually < 10 mg l^{-1}
Sodium	Feldspars (albite), clay minerals, evaporites such as halite, NaCl, industrial wastes	Generally < 200 mg l^{-1}; about 10,000 mg l^{-1} in sea water; ~ 25,000 mg l^{-1} in brines
Potassium	Feldspars (orthoclase, microcline), feldspathoids, some micas, clay minerals	Usually < 10 mg l^{-1}, but up to 100 mg l^{-1} in hot springs and 25,000 mg l^{-1} in brines
Calcium	Amphiboles, feldspars, gypsum, pyroxenes, dolomite, aragonite, calcite, day minerals	Usually < 100 mg l^{-1}, but brines may contain up to 75,000 mg l^{-1}
Magnesium	Amphiboles, olivine, pyroxenes, dolomite, magnesite, clay minerals	Usually < 50 mg l^{-1}; about 1,000 mg l^{-1} in ocean water; brines may have 57,000 mg l^{-1}
Carbonate	Limestone, dolomite	Usually < 10 mg l^{-1}, but can exceed 50 mg l^{-1} in water highly charged with sodium
Bicarbonate	Limestone, dolomite	Usually < 500 mg l^{-1}, but can exceed 1,000 mg l^{-1} in water highly charged with CO_2
Chloride	Sedimentary rock (evaporites), a little from igneous rocks	Usually < 10 mg l^{-1} in humid areas; up to 1,000 mg l^{-1} in more arid regions; approximately 19,300 mg l^{-1} in sea water and up to 200,000 mg l^{-1} in brines
Sulphate	Oxidation of sulphide ores, gypsum, anhydrite	Usually < 300 mg l^{-1}, except in wells influenced by acid mine drainage; up to 200,000 mg l^{-1} in some brines

Silica	Feldspars, ferromagnesian and clay minerals, amorphous silica, chert and opal	Ranges from 1-30 mg l^{-1} but as much as 100 mg l^{-1} can occur and concentrations may reach 4,000 mg l^{-1} in brines
Fluoride	Amphiboles (hornblende), apatite, fluorite, mica	Usually < 10 mg l^{-1}, but up to 1,600 mg l^{-1} in brines
Iron	Igneous rocks: amphiboles, ferromagnesian micas, FeS, FeS_2 and magnetite, Fe_3O_4. Sandstone rocks: oxides, carbonates, sulphides or iron clay minerals	Usually < 0.5 mg l^{-1} in fully aerated water; groundwater with pH < 8 can contain 10 mg l^{-1}; infrequently, 50 mg l^{-1} may be present
Manganese	Arises from soils and sediments. Metamorphic and sedimentary rocks and mica biotite and amphibole hornblende minerals contain large quantities of Mn	Usually < 0.2 mg l^{-1}; groundwater contains > 10 mg l^{-1}

Source: Modified from Todd, 1980

Most micro-organisms grow on solid surfaces and, therefore, coat the grains of the soil or aquifer. They attach themselves with extra-cellular polysaccharides, forming a protective biofilm which can be very difficult to remove. Up to 95 per cent of the bacterial population may be attached in this way rather than being in the groundwater itself. However, transport in the flowing groundwater is also possible. The population density of micro-organisms depends on the supply of nutrients and removal of harmful metabolic products (Matthess, 1982). Thus, in general terms, higher rates of groundwater flow supply more nutrients and remove the metabolic products more readily. Microbe populations are largest in the nutrient-rich humic upper parts of the soil, and decline with decreasing nutrient supply and oxygen availability at greater depths. Many sub-surface microbes, however, prefer lower nutrient conditions. In the presence of energy sources, such as organic material, anaerobic microbial activity can take place far below the soil and has been observed at depths of hundreds and even thousands of metres. The depth to which such activity is possible is determined by the nutrient supply and, in addition, pH, Eh, salt content, groundwater temperature and the permeability of the aquifer.

Microbiological activity primarily affects compounds of nitrogen and sulphur, and some of the metals, principally iron and manganese. Sulphate reduction by obligate aerobes is one of the most important biological processes in groundwater. Nitrogen compounds are affected by both nitrifying and denitrifying bacteria. Reduction of nitrate by denitrifying bacteria occurs in the presence of organic material in anaerobic conditions, leading to the production of nitrite which is then broken down further to elemental nitrogen. The possibility of enhancing natural denitrification is currently receiving attention in relation to the problem of nitrate in groundwater. Under aerobic conditions, ammonia (which may be produced during the decomposition of organic matter) is oxidised to nitrite and nitrate. Likewise iron can be subjected to either reduction or oxidation, depending on the Eh and pH conditions of the groundwater. In favourable microbiological environments, massive growth of iron bacteria can cause clogging of well screens and loss of permeability of aquifer material close to wells, and may require special monitoring and remedial action.

Micro-organisms can break down complex organic materials dissolved in groundwater. Under anaerobic conditions, microbial breakdown proceeds either as a methane fermentation or by reduction of sulphate and nitrate (Matthess, 1982). Microbial decomposition has been demonstrated for a whole range of organic compounds, including fuel hydrocarbons, chlorinated solvents and pesticides. Under ideal conditions, all organic materials would eventually be converted to the simplest inorganic compounds. In practice, complete breakdown is never reached, and intermediate products of equal or even greater toxicity and persistence may be produced. Nevertheless, the possibility of using indigenous or introduced microbial populations and enhancing the nutritional status of their environment is currently an important area of research in the quest for effective techniques for remediation of contaminated aquifers (NRC, 1993).

A principal microbiological concern in groundwater is the health hazard posed by faecal contamination. Of the four types of pathogens contained in human excreta, only bacteria and viruses are likely to be small enough to be transmitted through the soil and aquifer matrix to groundwater bodies (Lewis *et al.,* 1982). The soil has long been recognised as a most effective defence against groundwater contamination by faecal organisms, and a number of processes (see section 9.3) combine to remove pathogens from infiltrating water on its way to the water table. Not all soils are equally effective in this respect. In addition, many human activities which can cause groundwater pollution involve the removal of the soil altogether (section 9.4). Bacteriological contamination of groundwater remains a major concern, especially where many dispersed, shallow dug wells or boreholes provide protected but untreated domestic water supplies.

In summary, microbiological processes may influence groundwater quality both positively and negatively. The former include reducing nitrate and sulphate contents of groundwater, and removal of organic pollutants. The latter include the production of hydrogen sulphide and soluble metals, production of gas and biofilm fouling of well screens and distribution pipes.

9.3. Water-soil-rock interactions

9.3.1 Natural hydrochemical evolution

In treating groundwater, the importance of the physical properties of both ground and water have been summarised and used in section 9.2 to assist in defining a matrix of groundwater bodies. Similarly, it follows that the chemical properties of both ground and water are important in determining the quality of groundwater. The natural quality of groundwater (as described briefly in section 9.2.6) is, therefore, controlled by the geochemistry of the lithosphere, the solid portion of the earth, and the hydrochemistry of the hydrosphere, the aqueous portion of the earth.

Table 9.5 Average composition of igneous and some sedimentary rocks

Element	Igneous rocks (ppm)	Sedimentary rocks		
		Sandstone (ppm)	Shale (ppm)	Carbonates (ppm)
Si	285,000	359,000	260,000	34
Al	79,500	32,100	80,100	8,970
Fe	42,200	18,600	38,800	8,190
Ca	36,200	22,400	22,500	272,000
Na	28,100	3,870	4,850	393
K	25,700	13,200	24,900	2,390
Mg	17,600	8,100	16,400	45,300
Ti	4,830	1,950	4,440	377
P	1,100	539	733	281
Mn	937	392	575	842
F	715	220	560	112
Ba	595	193	250	30
S	410	945	1,850	4,550
Sr	368	28	290	617
C	320	13,800	15,300	113,500
Cl	305	15	170	305
Cr	198	120	423	7.1
Cu	97	15	45	4.4
Ni	94	2.6	29	13
Zn	80	16	130	16
Co	23	0.33	8.1	0.12
Pb	16	14	80	16
Hg	0.33	0.057	0.27	0.046
Se	0.05	0.52	0.60	0.32

Source: Hem, 1989

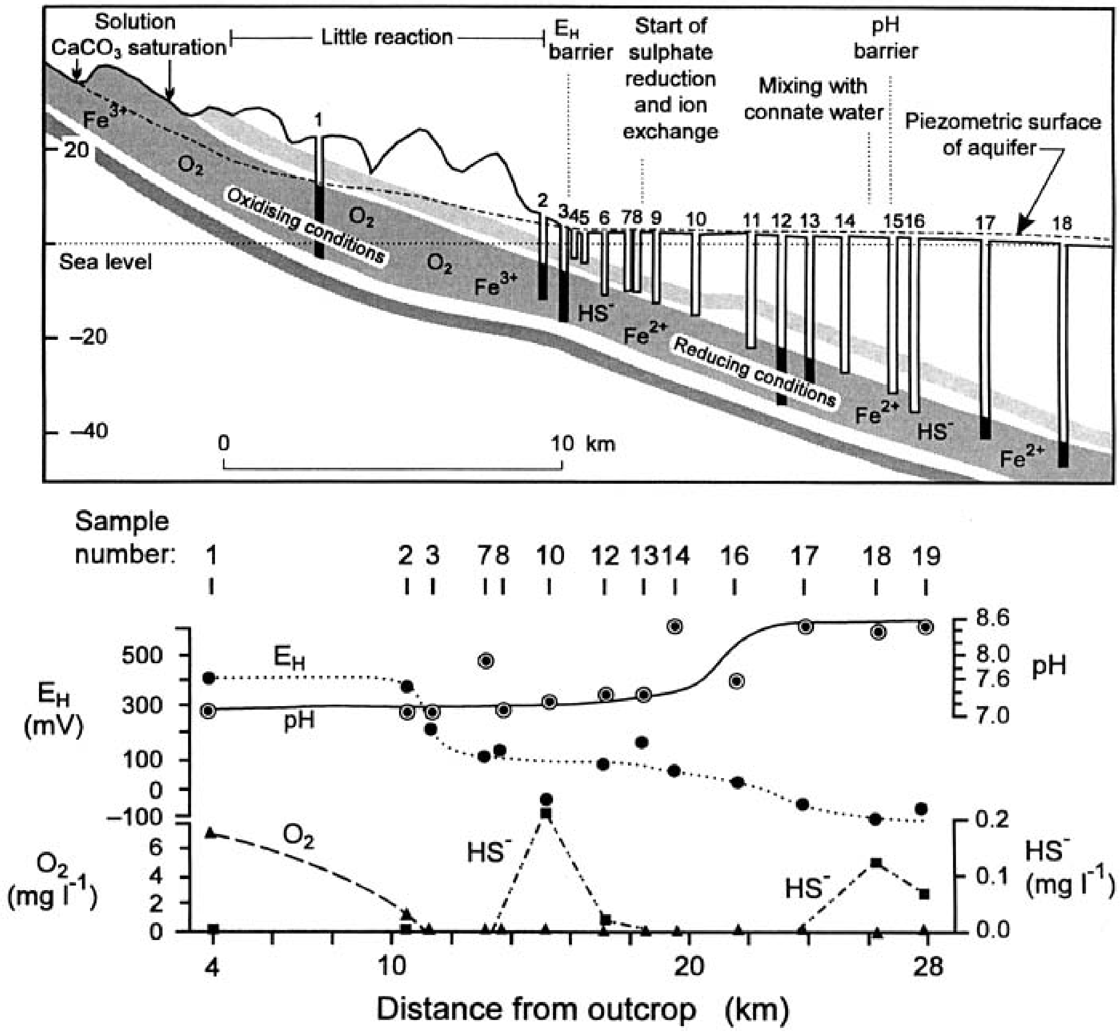
The average composition of igneous and sedimentary rocks is shown in Table 9.5. About 95 per cent of the earth's crust to a depth of 16 km is composed of igneous rocks (Hem, 1989). Ninety-eight per cent of the crust consists of the common alumino-silicates, made up of various combinations of the uppermost seven elements in the first column of Table 9.5, plus oxygen. Most usable groundwater occurs at depths of < 1 km, where sedimentary rocks predominate. Moreover, it is clear from section 9.2 that igneous rocks generally have low porosity and permeability and are much less important as aquifers on a global scale than sedimentary rocks.

Atmospheric precipitation infiltrating through the soil dissolves CO_2 produced by biological activity. The resulting solution of weak carbonic acid dissolves soluble minerals from the underlying rocks. A second process operating during passage through the soil is the consumption by soil organisms of some of the oxygen which was dissolved in the rainfall. These reactions occur in the soil and the top few metres of the underlying rock. In temperate and humid climates with significant recharge, groundwater moves continuously and relatively rapidly through the outcrop area of an aquifer (Figure 9.7); hence contact time with the rock matrix is relatively short. Readily soluble minerals will be removed, but insufficient contact time exists for less soluble minerals to be taken up. Groundwater in the outcrop areas of aquifers is likely to be low in overall mineralisation, with the natural constituents depending on the materials of which the rocks are made.

In igneous rocks, the restricted opportunity for reactions to take place is accentuated by the fact that groundwater storage and flow is predominantly in fissures, giving short residence times and low contact surface area. Groundwater in igneous rocks is, therefore, often exceptionally lightly mineralised, although characterised by high silica contents (Hem, 1989). Pure siliceous sands or sandstones without a soluble cement also contain ground-water with very low total dissolved solids (Matthess, 1982). In such aquifers, the dissolved constituents which are present come mainly from other sources, such as rainfall and dry deposition, especially sodium, chloride and sulphate which, in coastal regions, may exceed calcium, magnesium and bicarbonate. Sulphate may also be produced by the oxidation of metallic sulphides which are present in small amounts in many rock types. The presence of soluble cement may produce increased concentrations of the major ions. Ground-waters in carbonate rocks have pH values above 7, and mineral contents usually dominated by bicarbonate and calcium.

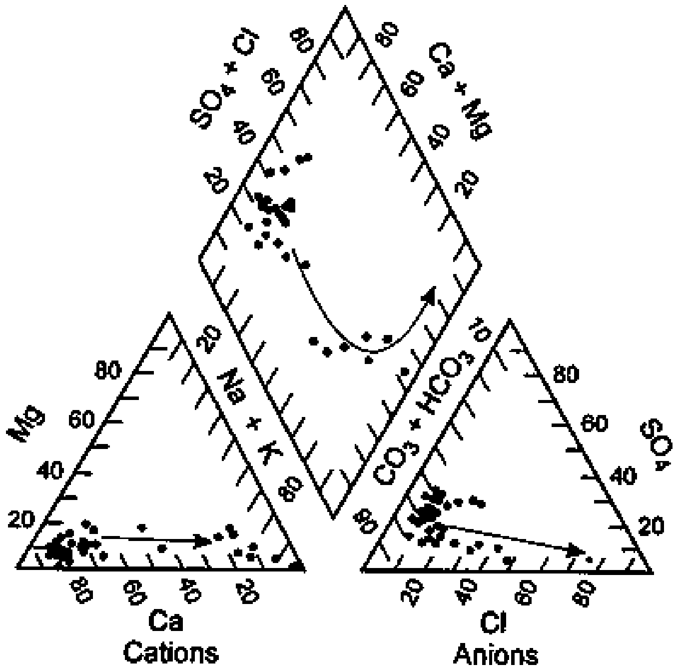
In many small and/or shallow aquifers the hydrochemistry does not evolve further. If, however, an aquifer dips below a confining layer, a sequence of hydrochemical processes occurs with progressive distance down gradient from the outcrop. These processes are clearly observed in the Lincolnshire Limestone of eastern England (Edmunds, 1973). This Middle Jurassic limestone outcrops in eastern England, and dips eastwards at less than one degree beneath confining strata consisting of clays, shales and marls (Figure 9.10). Dissolution of calcium and bicarbonate is active in the recharge area, such that Ca^{2+} is soon saturated with respect to calcite (Edmunds, 1973). As the water moves down dip, further modifications are at first limited. By observing the Eh of pumped samples, a sharp redox barrier was defined some 10-12 km east of the onset of confining conditions (Figure 9.10). An Eh of + 440 mV on the oxidising side, buffered by the presence of dissolved oxygen fell sharply to + 150 mV, corresponding with the complete exhaustion of oxygen. The water then became steadily more reducing down dip, as demonstrated by the presence of sulphide species (Figure 9.10), detectable by smell at many of the pumping boreholes. Reducing conditions also led to an increase in solubility of iron and denitrification of nitrate to N_2.

Figure 9.10 Section through the Lincolnshire Limestone, UK to show down dip oxidation and reduction processes (After Edmunds, 1973)



Beyond the redox barrier, there was a distance of several kilometres before any further chemical changes could be observed. Calcium was buffered until about 14 km from the outcrop. Eastwards of this point, Na^+ in the ground-water gradually increased at the expense of Ca^{2+} (Figure 9.11). This process of ion exchange of sodium on the aquifer particles for calcium in the water tends towards equilibrium, and produces a natural softening of the water. Calcium concentrations were almost insignificant and the reaction was practically complete by about 20 km from the outcrop. Bicarbonate rose gradually and the pH increased until buffering occurred at 8.3 (Figure 9.10). Sulphate concentration was constant in the oxidising water, probably reflecting the lack of additional solution of sulphate after initial infiltration. There was a sudden decrease in sulphate concentration 3 km beyond the redox barrier, resulting from sulphate reduction. Twenty-three kilometres away from the confining layer, the increase in Na^+ was greater than the equivalent concentration of Ca^{2+}. This was accompanied by an increase in chloride, and marks the point at which meteoric water moving down dip mixes with much older formation water, as shown in the triangular diagram used to illustrate groundwater analyses (Figure 9.11).

Figure 9.11 Down dip changes in major ions in the Lincolnshire Limestone, UK (After Downing and Williams, 1969)



The observed hydrochemical changes down dip can thus be interpreted in terms of oxidation/reduction, ion exchange and mixing processes. Similar sequences have been described from the Atlantic Coastal Plain of the USA (Back, 1966) and the UK Chalk (Edmunds *et al.,* 1987), and probably occur widely in humid, temperate regions. In arid and semi-arid areas, evapotranspiration rates are much higher, and exceed rainfall for much of the year. Recharge is limited, and often occurs in very restricted areas. In the example in Figure 9.7, recharge occurs in a mountainous region of somewhat heavier rainfall and travels slowly through the aquifer, dissolving soluble salts as it goes. The long residence times and incomplete flushing of soluble minerals produce groundwaters which are generally of the sodium chloride type. In the discharge area, evapotranspiration is often too great to allow a perennial stream to develop. Salt is concentrated in soil and water by direct evaporation and a salt marsh or sabkha may be formed.

9.3.2 Reactions related to anthropogenic effects

Physico-chemical reactions between soil or rock and water are of considerable importance when evaluating or predicting the nature of anthropogenic impacts on groundwater quality. In this respect the unsaturated zone, and particularly the soil, deserves special attention since it represents the first and most important natural defence against groundwater pollution (Lewis *et al.,* 1982; Foster, 1985; Matthess *et al.,* 1985). This is as a result of its position between the land surface and the water table and because a number of processes of pollutant attenuation are more favoured by the environments of the soil and the unsaturated zone (Figure 9.12).

Water movement in the unsaturated zone is largely vertical and normally slow. The chemical condition (see section 9.3.1) is usually aerobic and frequently alkaline. Thus, as suggested by Foster and Hirata (1988), there is considerable potential for:

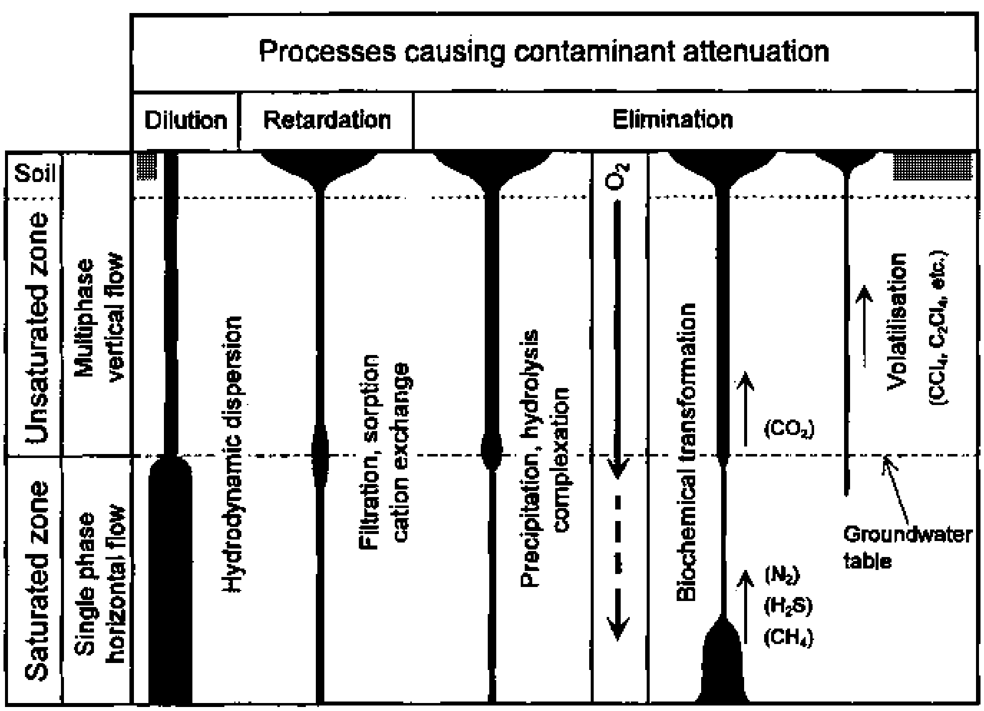
- interception, sorption and elimination of pathogenic bacteria and viruses,

- attenuation of trace elements and other inorganic compounds by precipitation, sorption or cation exchange, and

- sorption and biodegradation of many hydrocarbons and synthetic organic compounds.

The widths of the respective lines in Figure 9.12 indicate that most of these processes occur at their highest rates in the more biologically active soil zone, because of its higher content of clay minerals and organic matter and the much larger bacterial population. However, as noted in section 9.2.7, many activities which can cause groundwater pollution involve the complete removal, or by-passing, of the soil zone. The characteristics of the soil also influence the scope for nutrient and pesticide leaching from a given agricultural activity and whether acid aerial deposition is neutralised. The continuation of these processes at greater depths, albeit to a lesser degree, is more likely in sedimentary materials (in which inter-granular flow predominates) than in consolidated rocks (in which flow is largely restricted to fissures). Some of these processes are applicable to a range of possible constituents of percolating groundwater, while others are more restricted in their effects (Table 9.6).

Figure 9.12 Processes causing contaminant attenuation in groundwater systems. The thickness of the corresponding line indicates the relative importance of the process in the soil, above, at and below the groundwater table. (After Foster and Hirata, 1988 and Gowler, 1983)



9.4. Groundwater quality issues

Groundwater quality is the sum of natural and anthropogenic influences. Multi-purpose monitoring may be directed towards a whole range of ground-water quality issues and embrace many variables. Other categories of monitoring may concentrate on a single quality issue such as agricultural pesticides, an industrial spill, saline intrusion, etc. Table 9.7 lists and categorises the human activities that may potentially pollute groundwater and identifies the main pollutants in each case. These have been linked to the principal uses to which groundwater is put and to three levels of industrial development, so that the most important current and future groundwater quality issues can be identified. A major subdivision into urban, industrial and agricultural is made, although it is clear that there is considerable overlap between the first two. Some of the activities generating serious pollution risks are common to highly industrialised, newly industrialising and low development countries (mainly agricultural based economies), but those presenting the most serious threats differ significantly (Table 9.7). The most important groundwater quality issues world-wide are described briefly in the following sections.

Table 9.6 Processes which may affect constituents of groundwater

Constituent	Physical		Geochemical						Biochemical	
	Dispersion	Filtration	Complexation	Ionic strength	Acid-base	Oxidation-reduction	Precipitation-solution	Adsorption-desorption	Decay, respiration	Cell synthesis
Cl^-, Br^-	xx									
NO^{3-}	xx					xx			xx	xx
SO_4^{2-}	xx		x	x	x	xx		x	x	
HCO_3^-	xx		x	x	xx		xx		xx	
PO_4^{3-}	xx		xx	xx	xx		xx	xx	xx	xx
Na^+	xx			x				xx		
K^+	xx			x				xx		
NH_4^+	xx		xx	x	xx	xx		xx	xx	xx
Ca^{2+}	xx		x	xx			x	xx		
Mg^{2+}	xx		x	xx			x	xx		
Fe^{2+}	xx		xx	xx	xx	xx	xx	xx		
Mn^{2+}	xx		xx	xx	xx	xx	xx	xx		
Fe^{3+} and Mn^{4+} oxyhydroxides	xx	xx			xx	xx	xx			
Trace elements	xx		xx	xx		xx	xx	xx		
Organic solutes	xx		xx	x	xx	xx	x	x	xx	xx
Micro-organisms	xx	xx				xx			xx	xx

xx Major control

x Minor control

Source: UNESCO/WHO, 1978

Table 9.7 Principal activities potentially causing groundwater pollution

Activity	Principal characteristics of pollution					Stage of development[1]			Impact of water use		
	Distribution	Category	Main types of pollutant	Relative hydraulic surcharge	Soil zone by-passed	I	II	III	Drinking	Agricultural	Industrial
Urhanisation											
Unsewered sanitation	ur	P-D	pno	x	✓	xxxx	xx	x	xxxx		x
Land discharge of sewage	ur	P-D	nsop	x		x	x	x	xx	x	x
Stream discharge of sewage	ur	P-L	nop	xx	✓	x	x		xx	x	x
Sewage oxidation lagoons	u	P	opn	xx	✓	x	xx	x	xx		x
Sewer leakage	u	P-L	opn	x	✓			xx	x		x
Landfill, solid waste disposal	ur	P	osnh		✓	x	xx	xxx	x		x
Highway drainage soak-aways	ur	P-L	so	xx	✓	x	xx	xx	xx	x	x
Wellhead contamination	ur	P	pn		✓	xxx	x		xxx		
Industrial development											
Process water/effluent lagoons	u	P	ohs	xx	✓	x	xx	xx	xx		x
Tank and pipeline leakage	u	P	oh		✓	x	xx	xxx	xx		xx
Accidental spillages	ur	P	oh	xx		x	xx	xxx	xxx		xx
Land discharge of effluent	u	P-D	ohs	x		x	xx	xx	x	x	x
Stream discharge of effluent	u	P-L	ohs	xx	✓	x	x	x	x	x	x
Landfill disposal		P	ohs		✓	x	xxx	xxx	xx		x

residues and waste ur											
Well disposal of effluent	u	P	ohs	xx	✓		x	x	xx		x
Aerial fallout	ur	D	a					xx	x	x	x
Agricultural development											
Cultivation with:											
Agrochemicals	r	D	no			x	xx	xxx	xxx	x	x
Irrigation	r	D	sno	x		xx	xx	x	xxx	xxxx	x
Sludge and slurry	r	D	nos			x	x	xx	xx	x	x
Wastewater irrigation	r	D	nosp	x			xx	x	xx	xx	
Livestock rearing/crop processing:											
Unlined effluent lagoons	r	P	pno	x	✓	x	x	xx	x	x	
Land discharge of effluent	r	P-D	nsop	x	✓	x	x	xx	x	x	
Stream discharge of effluent r	r	P-L	onp	x	✓	x	x	xx	x	x	
Mining development											
Mine drainage discharge	ru	P-L	sha	xx	✓	x	xx	xx	xx	x	x
Process water/sludge lagoons	ru	P	hsa	xx	✓	x	xx	xx	xx	x	x
Solid mine tailings	ru	P	hsa		✓	x	xx	xx	xx	x	x
Oilfield brine disposal	r	P	s	x	✓		x	x	xx	x	x
Hydraulic disturbance	ru	D	s		na		x	x	xx	x	x
Groundwater resource management											
Saline intrusion	ur	D-L	s		na	x	x	xx	xxx	xxx	xx
Recovering water levels	u	D	so		na			x	x		x

Distribution: u Urban; r Rural

Category: P Point; D Diffuse; L Line

Types of pollutant: p Faecal pathogens; n Nutrients; o Organic micropollutants; h Heavy metals; s Salinity; a Acidification

x to xxxx Increasing importance or impact
na Not applicable

[1] Stages of development: I Low development; II Newly industrialising; III Highly industrialised

The differentiation between point and diffuse sources has been discussed in Chapter 1. Although there is overlap between the two (Table 9.7), as many small point sources may become a diffuse source, the distinction is of fundamental importance, especially in relation to prevention and control of groundwater pollution, and hence in the design of monitoring undertaken to evaluate the effectiveness of pollution control measures. It is apparent, for example, from Table 9.7 and from the brief descriptions below that ground-water quality issues may be global, national or regional, affect the whole of one or more aquifers, or be restricted to the immediate vicinity of a single contaminant source. Therefore, the scale and type of monitoring operations required depend on the issue, or combination of issues, and the size of groundwater body affected. These are described further in section 9.5 and illustrated by examples in section 9.6.

9.4.1 Unsewered domestic sanitation

The impetus of the International Water Supply and Sanitation Decade (from 1981 to 1990) produced considerable efforts in many countries to improve health by investment in water supply and sanitation programmes. These often comprise the provision of largely untreated rural and small-scale urban water supplies from groundwater and the construction of unsewered, on-site sanitation facilities using various types of latrines. Under certain hydro-geological conditions, unsewered sanitation can cause severe groundwater contamination by pathogenic micro-organisms and nitrate, which may largely negate the expected health benefits of such programmes. In some circumstances, therefore, these two low-cost technologies may be incompatible (Lewis *et al.,* 1982; Foster *et al.,* 1987).

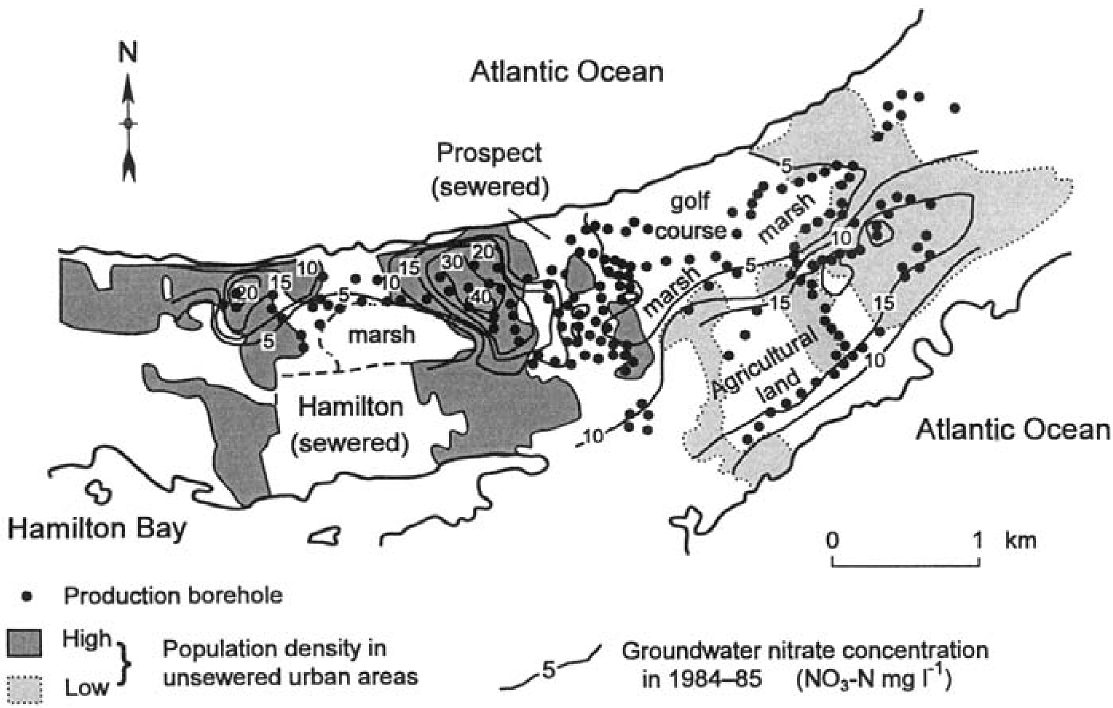
Unsewered sanitation consists of the installation of either septic tanks or pit latrines of the ventilated, dry or pour-flush types. There are important differences between the two in relation to the risk of groundwater contamination (Foster *et al.,* 1987). Septic tank soak-aways discharge at higher levels in the soil profile than pit latrines and conditions may be more favourable for pathogen elimination. Pit latrines are often deep excavations (to allow a long useful life) and the soil may be entirely removed. The hydraulic loading from septic tank soak-aways is likely to be less than for some of the pit latrine types. Septic tanks are lined and their solid effluent of high nitrogen content is periodically removed, whereas most pit latrines are unlined and the solid material remains in the ground. For these reasons, septic tanks are likely to pose a less serious threat to groundwater than pit latrines. If domestic waste-water is also discharged to unsewered sanitation, there is an added risk of groundwater contamination by the increasing range of organic compounds used in household products such as detergents and disinfectants.

The impact of unsewered sanitation is felt particularly in relation to drinking water (Table 9.7). Contamination of groundwater supplies by unsewered sanitation has been the proven vector of pathogens in numerous disease outbreaks. In unconsolidated deposits, filtration, adsorption and inactivation during migration through one metre or less of unsaturated, fine-grained strata normally reduces pathogen numbers to acceptable levels (Lewis *et al.,* 1982). Problems usually arise only where the water table is so shallow that on-site sanitation systems discharge directly into the saturated zone. The risk to groundwater may be enhanced by persistent organisms, and considerable uncertainty remains about the persistence in aquifers of some pathogens, especially viruses.

Whilst bacteriological contamination of shallow wells and boreholes in all types of geological formations is believed to be widespread, migration of pathogens through unconsolidated strata to deep water supply wells is unlikely. In these situations, bacteriological contamination is most probably direct and localised, reflecting poor well design and/or construction and sanitary completion rather than aquifer pollution. Thus, in many large cities, the impact of unsewered sanitation may not be significant because municipal water supplies are drawn from surface water and then treated, or drawn from relatively distant and deep, well-protected aquifers. Often the most serious problems arise in medium to smaller sized towns and in densely populated peri-urban and rural areas where local, shallower, and often untreated, groundwater sources are used. In these circumstances, direct pollution of the source at the wellhead by the users, by livestock and by wastewater may be a serious problem.

The nitrogen compounds in excreta do not represent such an immediate hazard to groundwater as pathogens, but can cause more widespread and persistent problems. Unsewered sanitation has been shown to cause increased nitrate concentrations in the underlying groundwater at many localities in, for example, South America, Africa and India. An example is shown in Figure 9.13. The impact is often demonstrated by rising nitrate concentrations in public supply wells, as in Greater Buenos Aires (Foster *et al.,* 1987) and Bermuda (Figure 9.14). It is possible to make semi-quantitative estimates of the concentration of persistent and mobile contaminants, such as nitrate and chloride, in groundwater recharge by using the following formula (Foster and Hirata, 1988):

Figure 9.13 Correlation of high nitrate concentrations in groundwater with densely populated areas in Bermuda with unsewered sanitation (Redrawn from Thomson and Foster, 1986)



$$C = \frac{1{,}000 \times a \times A \times F}{0.365 \times A \times U + 10I}$$

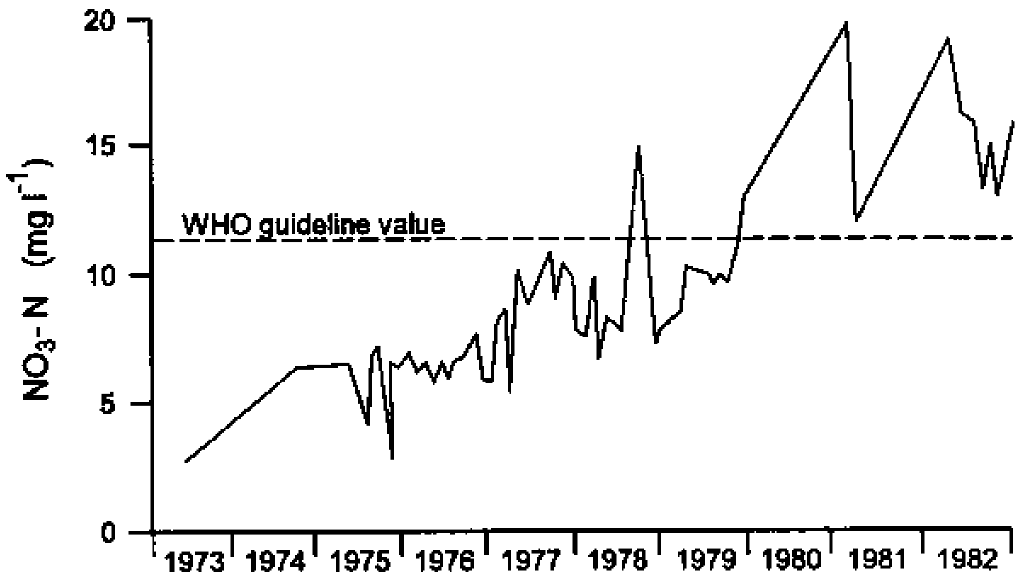
where:

C = the concentration (mg l^{-1}) of the contaminant in recharge
a = the unit weight of nitrogen or chloride in excreta (kg capita^{-1} a^{-1})
A = population density (persons ha^{-1})
F = proportion of excreted nitrogen oxidised to nitrate and leached
U = non-consumptive portion of total water use (1 capita^{-1} day^{-1})
I = natural rate of rainfall infiltration (mm a^{-1})

Greatest uncertainty surrounds the proportion of the total nitrogen load that will be oxidised and leached into the groundwater recharge.

Groundwater pollution by unsewered sanitation is most likely to occur where soils are thin or absent, where fissures allow rapid movement and where the water table is shallow. A further example of groundwater pollution of this type is described by Lewis *et al.* (1980) in weathered metamorphic rocks in southern Africa. Groundwater quality assessment activities related to unsewered sanitation can generally be considered as surveillance (as defined in Chapter 2).

Figure 9.14 Composite trend of average nitrate concentrations in public supplies derived from the Bermuda central groundwater lens (After Thomson and Foster, 1986)



9.4.2 Disposal of liquid urban and industrial waste

An example of groundwater pollution from disposal of liquid waste is described by Everett (1980). Liquid wastes containing chromium and cadmium from metal plating processes were disposed of, virtually untreated, into lagoons recharging directly to a glacial sand aquifer in Long Island, New York. No records were kept of the types and quantities of the wastes, and disposal continued in this way for many years. Eventually chromium was detected in a water supply well, and extensive investigations were undertaken to define the lateral and vertical extent of the plume of polluted groundwater (Figures 9.15 and 9.16). The difference of behaviour between cadmium and chromium should be noted since their maximum concentrations do not coincide. Further examples of groundwater pollution of this type are described by Williams *et al.* (1984) and Foster *et al.* (1987).

Typical methods of wastewater disposal include infiltration ponds, spreading or spraying onto the ground surface and discharge to streams or dry stream beds which may provide a rapid pollution pathway to underlying, shallow aquifers. Current practices are reviewed by Foster *et al.* (1994). In some areas, deep soak-aways or abandoned wells are used for the disposal of liquid domestic, industrial or farming waste into aquifers. There are many thousands of such wells in the USA. Even if the intention is to dispose of the waste at depth, improper sealing or corrosion of well linings often produces leaks and subsequent pollution of the shallow groundwater which is used for water supplies. In recent years, attention has been given to the possibility of injecting treated municipal sewage into aquifers to enhance recharge or establish hydraulic barriers against saline intrusion. Well established examples of disposal by this method are described for a humid temperate environment by Beard and Giles (1990), and for more arid climates by Bouwer (1991) and Idelovitch and Michail (1984).

Figure 9.15 Monitoring wells, water table contours and the contaminant plume associated with disposal of liquid industrial waste, Long Island, New York (Modified from Everett, 1980)

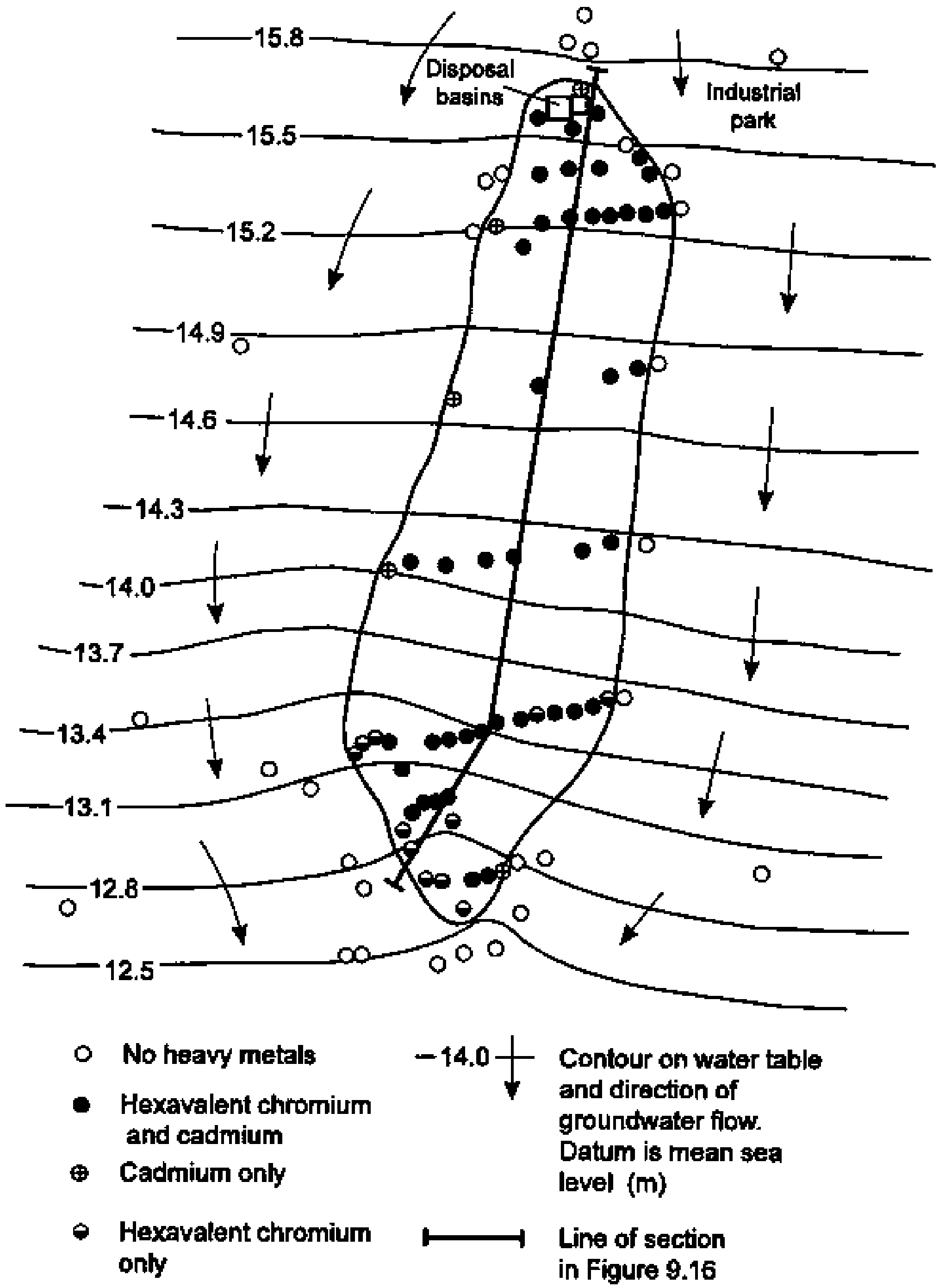
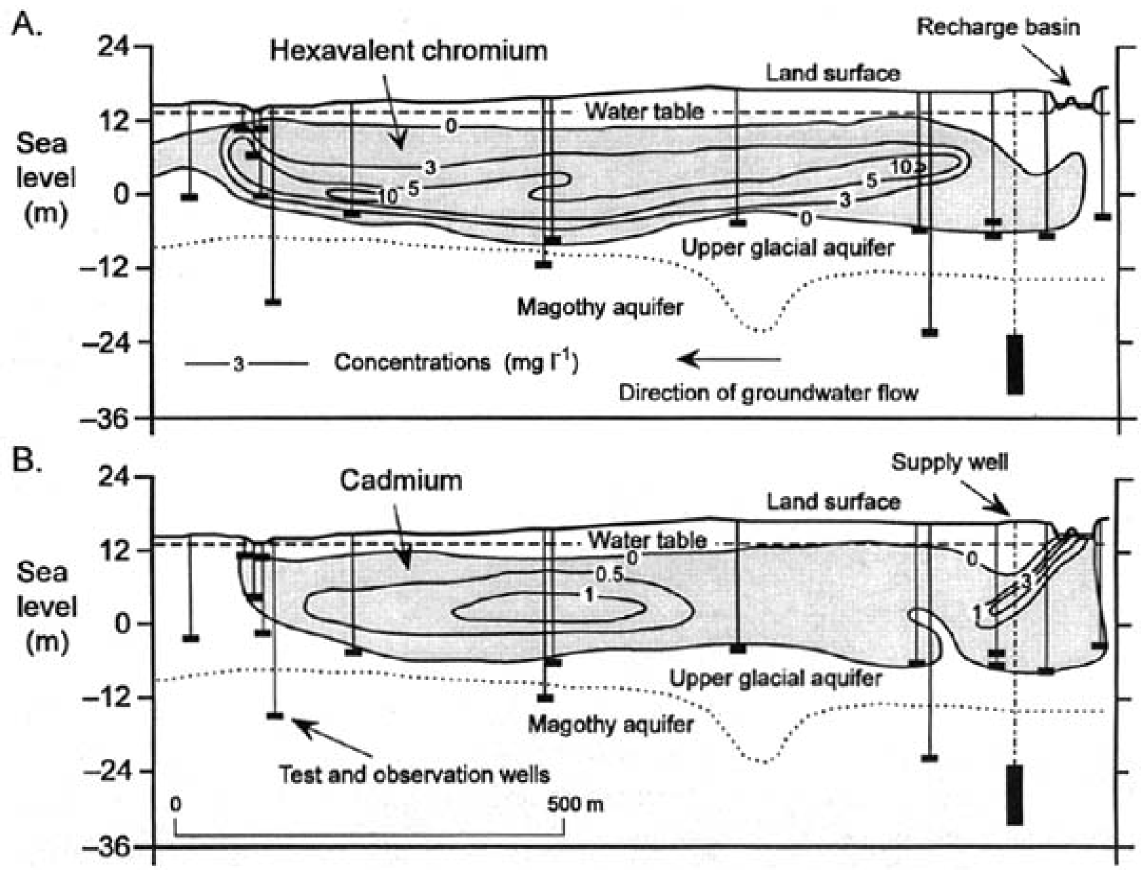


Figure 9.16 Vertical distribution of hexavalent chromium and cadmium along the centre line of the plume resulting from disposal of liquid industrial waste, Long Island, New York (Modified from Perlmutter and Lieber, 1970 in Everett, 1980)



In urban areas which do have mains sewerage systems, an economical method of partial treatment of sewage is wastewater stabilisation by retention in shallow oxidation lagoons before subsequent discharge into rivers or onto land or subsequent re-use for irrigation (see section 9.4.8). These lagoons are often unlined and, over suitably coarse-textured soils, may have high rates of seepage loss, especially immediately after construction and each time the lagoon is cleaned. This can have a considerable local impact on groundwater quality, particularly in relation to nitrogen and trace organic compounds and where groundwater is used for drinking water supplies (Geake *et al.,* 1986).

Sanitary sewers are intended to be watertight; if they were completely so, they would present no threat to groundwater. In practice, leakage is a common problem, especially from old sewers, and may be caused by defective pipes, poor workmanship, breakage by tree roots, settlement and rupturing from soil slippage or seismic activity. The problem could be greater but for the fact that the suspended solids in raw sewage can clog the cracks and the soil around the pipes and be, to some extent, self-sealing.

9.4.3 Disposal of solid domestic and industrial waste

The most common method of disposal of solid municipal waste is by deposition in landfills. A landfill can be any area of land used for the deposit of mainly solid wastes and they constitute important potential sources of groundwater pollution (Everett, 1980; Foster *et al.,* 1987). Sanitary landfills are usually planned, located, designed and constructed according to engineering specifications to minimise the impact on the environment, including groundwater quality. The engineering methods adopted include lining and capping, compaction of fill and control of surface water inflow. Of the 100,000 landfills in the USA, only about ten per cent could be described as sanitary (Everett, 1980), and the remainder are open dumps.

The principal threat to groundwater comes from the leachate generated from the fill material. The design of a sanitary landfill aims to minimise leachate development by sealing the fill from rainfall, run-off and adjacent groundwater. For significant leachate to be produced, a flow of water through the fill is required. Possible sources include precipitation, moisture in the refuse, and surface water or groundwater flowing into the landfill. Therefore, the volume of leachate generated can be estimated by a simple water balance (Foster and Hirata, 1988), and is greater in humid regions with high rainfall, plenty of surface water and shallow water tables, and is much less in arid regions.

The chemical composition depends on the nature and age of the landfill and the leaching rate. Most municipal solid wastes contain little hazardous material, but solid wastes of industrial origin may contain a much higher proportion of toxic constituents, such as metals and organic pollutants. The most serious threats to groundwater occur where there is uncontrolled tipping rather than controlled, sanitary landfill. There may be no record of the nature and quantity of materials disposed of, which could include hazardous industrial wastes, such as drums of toxic liquid effluent. In Europe and North America this situation existed until the mid-1970s, when uncontrolled tipping was replaced by managed, sanitary landfill facilities. Abandoned landfills can, therefore, represent a potential hazard to groundwater for years or decades.

9.4.4 Accidents and leaks

Groundwater pollution incidents from major industrial complexes are becoming more common and are often the subject of major, expensive investigations and clean-up activities. The causes include accidents during transportation, spillages due to operational failures and leaks due to corrosion or structural failure of pipes or tanks. The result is a point source of pollution which may be short and intense in the case of an accidental spillage or small but continuous in the case of an undiscovered leak. In either case, dangerous chemicals may be discharged into the environment and serious groundwater pollution can be caused. Petroleum and petroleum products are the most important because they are widely used. Fuel stations with buried tanks are universal and pollution of this type is not restricted to industrial, or even urban, areas. Moreover, accidents during road or rail transport may result in spillages of, for example, industrial solvents in rural areas. Very few steel underground tanks are protected from corrosion and it has been estimated that up to 25 per cent of fuel storage tanks in the USA leak (Canter *et al.,* 1987).

Figure 9.17 The range of possibilities for groundwater contamination by industrial organic compounds (After Lawrence and Foster, 1991)

Duration of discharge	Probability of immiscible phase		
	Low	**Moderate**	**High**
Continuous	Leaking industrial sewers and lagoons	Infiltration from landfill waste disposal	Leaking underground storage tanks
Intermittent	Courtyard drainage	Effluent soak-aways	
Single pulse			Major spillages

A major control over how a pollution plume develops and migrates in groundwater, and hence on how it should be monitored, is the mechanism by which the contaminant enters the sub-surface. An overnight leakage of several thousands of litres of solvent will produce a different plume to that caused by contaminated drainage regularly infiltrating from an industrial site. In the former situation, significant leaks of fuel oils, oil derivatives or chlorinated solvents are likely to result in the immiscible phase penetrating into the aquifer, whilst in the latter situation shallow contamination of the aqueous phase will result. An indication of the probability of the immiscible phase liquid being present in the groundwater is shown in Figure 9.17. Two examples of contrasting pollution incidents of this type are summarised by Lawrence and Foster (1991). Monitoring of the development of the plume has to take account of the behaviour of the light and dense immiscible phase contaminants described in section 9.2.5.

The cost of aquifer restoration measures and/or provision of alternative water supplies after major incidents of this type may run into many millions of US dollars. Much of this expenditure may depend on the information obtained from the groundwater quality assessment programme, the design and operation of which becomes especially critical if a legal case is involved.

9.4.5 Acid deposition

Of the total groundwater resources, that part which is in active circulation and potentially available for abstraction for whatever purpose is derived mainly from rainwater infiltration through the soil to underlying aquifers. The acidity of this groundwater depends on the inputs of acidity from internal (biological, soil and rock) and external (atmospheric and anthropogenic) sources, and on the ability of the soil to attenuate them (Kinniburgh and Edmunds, 1986). Some groundwater is, therefore, vulnerable to acid deposition, a collective term for a wide variety of processes by which acidic substances are transferred from the atmosphere to vegetation, land, or water surfaces. For convenience, acid deposition can be divided into wet and dry deposition.

The nature, origins and effects of acid deposition have been most studied in the highly industrialised areas of Europe and North America, where the problem of acid deposition is felt most acutely. In the UK, it is thought that about two thirds of the present acidity of the rain can be attributed to sulphur dioxide emissions from the burning of fossil fuels and one third to nitrous emissions from internal combustion engines. The average pH of rainfall over much of the UK is 4.2 - 4.5. Dry deposition of both emissions is also an important source of acidity.

The attenuation of acidity in the infiltrating rainwater depends on the time available for reactions with the soil and the rock. The residence time of water in the soil and the aquifer is, therefore, an important factor in determining the acidity of groundwater, together with the mineral composition of the aquifer material. Thus limestone aquifers with their high calcium carbonate content are well buffered and will not be adversely affected by acid deposition. Other non-carbonate aquifers may be protected by the neutralising of acid infiltration as it passes through overlying strata. The most vulnerable aquifers are shallow sands, sandstones and shales with relatively short residence times. The most serious consequences of acidification of groundwater are the increased mobilisation of trace elements, especially aluminium, in soils and aquifers, and the increased solubility of some metals in water distribution systems, both resulting from the lowering of the pH.

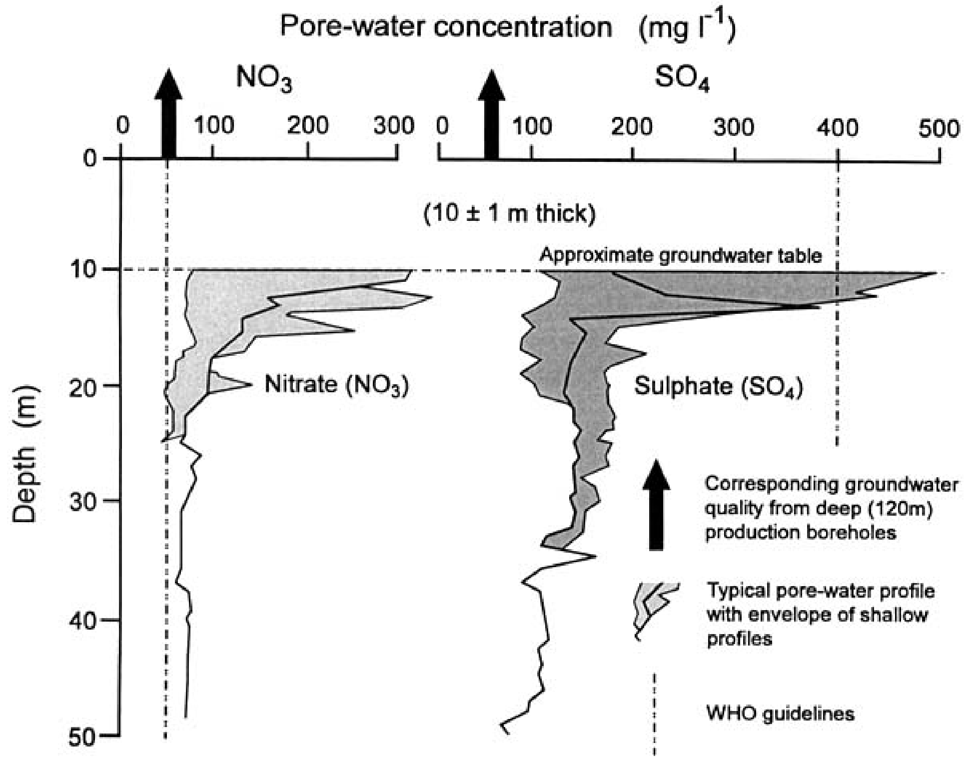
9.4.6 Cultivation with agrochemicals

Agricultural land-use and cultivation practices have been shown to exert major influences on groundwater quality. Under certain circumstances serious groundwater pollution can be caused by agricultural activities, the influence of which may be very important because of the large areas of aquifer affected. Of particular concern is the leaching of fertilisers and pesticides from regular, intensive cultivation, with or without irrigation, of cereal and horticultural crops. The changes in groundwater quality brought about by the clearing of natural vegetation and ploughing up of virgin land for new cultivation are also important. The impact of cultivation practices on groundwater quality is greatest, as are most anthropogenic effects, where relatively shallow, unconfined aquifers are used for potable supply in areas where there is no alternative.

The impact of modern agricultural practices on groundwater quality became fully apparent in some regions of industrialised countries during the latter part of the 1970s. Detailed scientific investigations demonstrated that high rates of leaching to groundwater of nitrate and other mobile ions occurred from many soil types under continuous cultivation sustained by large applications of inorganic fertilisers. In the USA, for example, fertiliser use doubled between 1950 and 1970 from 20 to 40 million tons, and the percentage of nitrogen in all fertilisers increased from 6 to 20 per cent. A similar pattern occurred in Europe (OECD, 1986) and is now also occurring in rapidly industrialising countries such as India, in response to growing population and food demands. In the developing world, annual consumption of nitrogen fertiliser has more than tripled since 1975 (Conway and Pretty, 1991).

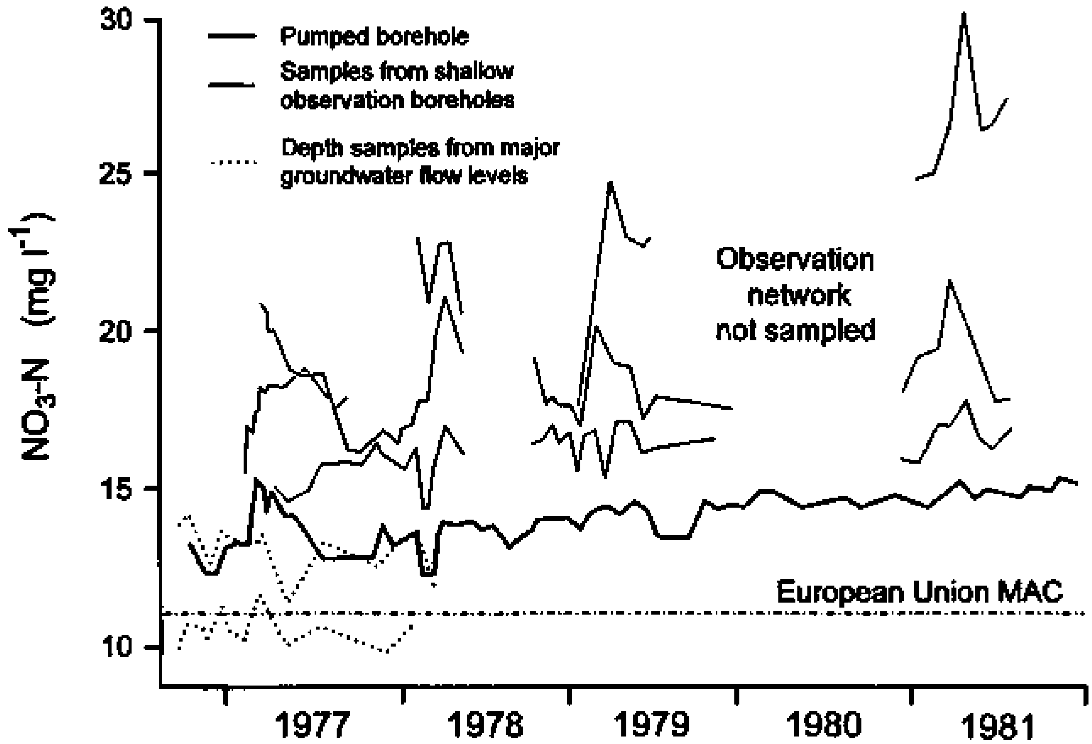
Monitoring of groundwater for compliance with legally imposed allowable nitrate concentrations, and to observe trends, is based on discharge samples from production boreholes, i.e. the water that goes to the consumer. Where there is a thick unsaturated zone in the aquifer recharge area (Figure 9.3), much higher nitrate concentrations from the most recent fertiliser applications may be observed (Foster *et al.,* 1986). The slow movement in the unsaturated zone means that there is a time lag (sometimes 10 to 20 years in the chalk aquifers of Western Europe), dependent on the thickness of the unsaturated zone, before the full impact of agricultural intensification is felt in public supply sources. Similarly, changes in land use intended to reduce nitrate leaching will take time to have an effect on nitrate concentrations in groundwater supplies. Much of the detailed research on nitrate leaching through the unsaturated zone was based on vertical profiling by extracting matrix water from undisturbed drilling samples by centrifugation. Repeat profiling of this type has been employed to examine the development of nitrate profiles in the unsaturated zone (Geake and Foster, 1989). This approach, although expensive, is likely to be required in monitoring to evaluate the effects of land use changes employed in a groundwater protection programme.

Figure 9.18 Saturated zone pore-water profiles from the Triassic Sandstone Aquifer in Yorkshire, UK. Vertical stratification of groundwater quality means that pump discharge samples do not fully reflect severity of pollution (After Parker and Foster, 1986)



Pore-water profiling has also been extensively employed in the saturated zone in the study of diffuse pollution, and has demonstrated considerable stratification of groundwater quality (Parker and Foster, 1986). Groundwater quality assessed on the basis of pumped samples from deep water supply boreholes may not reflect the severity of pollution already present in the upper part of the aquifer (Figure 9.18). Pumped samples from boreholes completed at different depths in the aquifer, and depth sampling in deep boreholes have also demonstrated quality stratification (Figure 9.19). By the time pollutants are recorded at significant concentrations in deep pumping boreholes, a large volume of the aquifer will already be polluted, and effective remedial action may be impossible.

Figure 9.19 Groundwater quality stratification in the Chalk of Eastern England (After Parker and Foster, 1986)

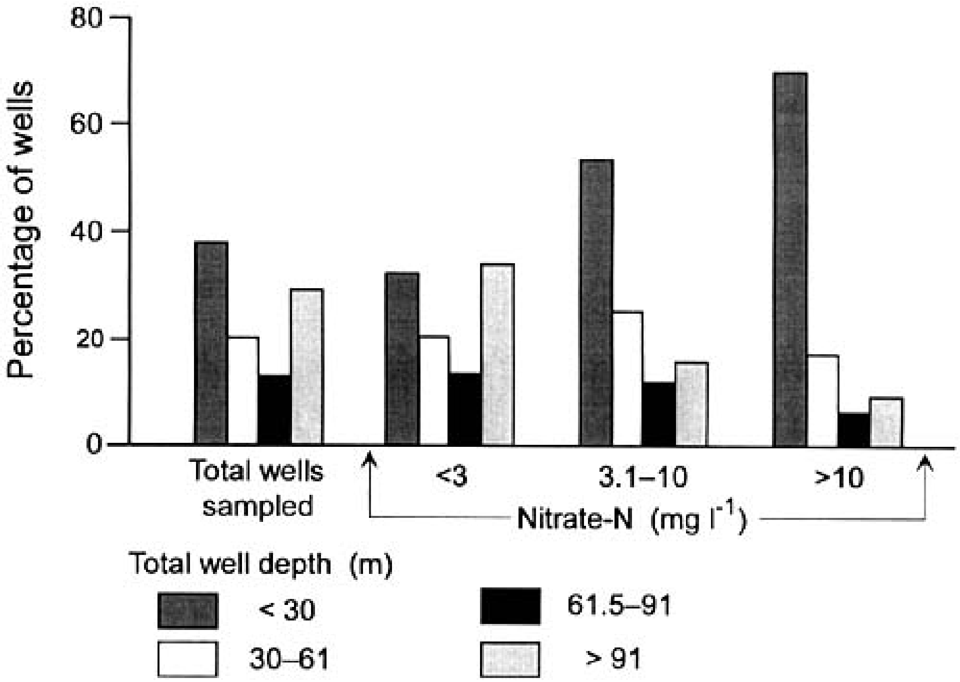


The examples in Figures 9.18 and 9.19 are from consolidated aquifers in which fissure flow is important. Such aquifers are particularly prone to marked vertical variations in groundwater quality, but thick aquifers with inter-granular flow may also exhibit such quality stratification. Thus, in the USA, a high proportion of the rural population in agricultural areas obtain their domestic water supplies from shallow, private boreholes, which suffer the impact of nitrate pollution to a much greater extent than deeper, public supply boreholes (Figure 9.20).

Evidence of the impact of tropical agriculture on groundwater quality is now becoming available. The results of a recent study (Table 9.8) show a wide variation in nitrate leaching losses, resulting from differences in soil and crop types, fertiliser application rates and irrigation practices. Nitrogen leaching losses beneath paddy cultivation are likely to be low because the flooded, largely anaerobic soil conditions during the growing season restrict oxidation of nitrogen to nitrate. In tropical regions, groundwater is most vulnerable to leaching of nitrate where (Chilton *et al.,* 1995):

- the soil and unsaturated zone are thin and permeable,
- several crops a year are grown,
- fertiliser inputs are high, and
- excess irrigation leads to rapid leaching of nutrients beyond the root zone.

Figure 9.20 Nitrate concentrations in wells of various depths in the USA (After Madison and Brunett, 1985 in Meybeck *et al*., 1989)



Much less attention has been given to the leaching of pesticides from agricultural land to the underlying groundwater, in spite of the dramatic increase in the use of pesticide compounds over the last 20 years. The use of some of the most toxic, particularly the organochlorine insecticides, is now banned or rigorously controlled. However, new compounds are continually being introduced. All are, to a greater or lesser degree, manufactured to be toxic and persistent. With improved analytical techniques, instances of pesticide occurrence in groundwater are increasingly being reported.

Many of the pesticide compounds which are widely used are unlikely to be leached significantly to groundwater from normal agricultural use as their physico-chemical properties, and methods and rates of application are not such as to promote leaching. In addition, many of the processes described in Figure 9.12 are very active in relation to pesticide compounds. There are, however, exceptions, such as the carbamate insecticides and herbicides of the carboxyacid, triazine and phenylurea groups which have been encountered in groundwater at levels which give cause for concern, in spite of the relatively rapid biodegradation rates quoted for these compounds. The published values for sorption and degradation of common pesticide compounds are for fertile, organic, clay-type soils. Much less favourable values must be expected if pesticide compounds are leached into the unsaturated and saturated zones of aquifers. Figure 9.12 indicates that the attenuation processes are much reduced in effectiveness below the soil. Even a very small percentage (1-5 per cent) of the applied pesticide leaving the soil layer could be persistent and mobile enough in the groundwater system to present a problem in relation to the very low concentrations recommended in drinking water guidelines.

Table 9.8 Summary of the results of nitrate investigations in tropical environments

Features	Sri Lanka	Barbados	India	Mexico
Aquifer type	Coastal dune sands	Coral limestone	Thin alluvium	Thick alluvium
Depth to water (m)	1-3	20-80	10	5-15
Travel time to water table or borehole screen	10 days	Days	1 year	25-40 years
Potential for dilution within aquifer	Low-moderate	Low	Low	High
Principal crops	Onions, chillies	Sugar cane	Rice	Wheat
Crops per year	2-3	1	2	1
Length of period of intensive cultivation (years)	10-20	>30	20	>30
Fertiliser type	Urea, triple super-phosphate	24-0-18 NPK	Urea	Urea, anhydrous ammonia
Application rates (kg N ha^{-1} a^{-1})	400-500	123	300	120-220
Leaching losses (kg N ha^{-1} a^{-1})	60-120	35-70	Small	?
Current groundwater nitrate concentrations (mg NO_3-N l^{-1})	10-30	5-9	2-5	2-5

Source: After Chilton *et al*., 1995

Both scientific investigation and routine monitoring of pesticides in groundwater present significant difficulties. A strategy needs to be developed to assist water utilities and regulatory authorities to protect groundwater from contamination by pesticides. It is not practical to monitor for all of the many pesticides in regular use. General screening is prohibitively expensive where potable supplies are based on a few high-discharge sources, and quite out of the question for large numbers of small, rural supplies. To be effective, the available resources for sampling and analysis should be focused by the following approach:

• Existing usage data, if available, or new surveys are required to select those compounds which are widely used.

• The pesticides and their metabolites most likely to be leached to ground-water can be identified from their physical and chemical properties.

• Toxicity information is required for the pesticides and their metabolites.

In this way, monitoring can be concentrated on the high-risk compounds, those which are widely used, mobile, toxic and persistent (Chilton *et al.,* 1994). Within such a strategy, monitoring resources should be concentrated on the most vulnerable aquifers. The hydrogeological environments which are likely to be most vulnerable to the leaching of pesticides are those with shallow water tables and coarse textured soils low in organic matter (Chilton *et al.,* 1995). These include coastal and island limestones, sands and some alluvial deposits. In many countries, the trend of growth in pesticide use closely follows that of inorganic fertilisers. Thus, in spite of the difficulties outlined above, monitoring for pesticides in drinking water is likely to become of increasing importance.

9.4.7 Salinity from irrigation

Increasing salinity resulting from the effects of irrigated agriculture is one of the oldest and most widespread forms of groundwater pollution (Meybeck *et al.,* 1989). Many, but not all, instances of waterlogging and salinisation are related to low irrigation efficiency and lack of proper drainage measures. Over-irrigation without adequate drainage can cause rises in groundwater level which result in soil and groundwater salinisation from direct phreatic evapotranspiration. The addition of further excess irrigation water to leach salts from the soil merely transfers the problem to the underlying ground-water. Additional contributions to increased salinity may come from the dissolution of salts from soil and aquifer material by the rising groundwater, and from the greatly increased infiltration from irrigated land leaching out the salts present in desert or semi-desert soils, at the time the land is first brought under irrigation.

The total area of irrigated cultivation in the world is about 270 x 10^6 ha (WRI, 1987), about 75 per cent of which is in developing countries. Up to half of this land may be affected by salinity to some extent, and productivity is seriously affected on seven per cent of the total (Meybeck *et al.,* 1989). Salinisation of fertile crop lands is occurring at a rate of 1-1.5 x 10^6 ha a^{-1} (Kovda, 1983), and irrigated land is going out of production at a rate of 30 to 50 per cent of the rate at which new land is being brought under irrigated cultivation. The fall in crop production and the consequent economic costs of this loss of agricultural land are enormous and difficult to quantify. The increase in salinity of groundwater also has a significant impact on drinking water and industrial uses (Table 9.7).

Preventing or alleviating the problem of groundwater salinity requires more efficient irrigation combined with effective drainage. Adequate drainage requires a general lowering of the water table to below 2-3 m from the ground. This can be achieved by open ditches, the drains or pumping from wells. However, even if drainage measures are implemented there are often problems associated with the disposal of saline water, and irrigation return flows have a serious impact on surface water quality in many places (Meybeck *et al.,* 1989). Assessment of river water quality has often provided the best indication of overall trends in salinity (Williams, 1987).

9.4.8 Use of urban wastewater for irrigation

Greatly improved urban water supplies with individual domestic connections have led to increasing water consumption and an increasing problem of wastewater disposal. The projected effluent production (Pescod and Alka, 1984) for some of the fastest growing major cities of the world (e.g. Mexico City and São Paulo) indicates that in these areas, water demands are escalating rapidly, and it is imperative that effluent re-use forms an integral part of the overall water resource management strategy. Agricultural re-use is consistent with the need to maintain and increase food production for rapidly expanding urban populations (Pescod, 1992). Effluent re-use is not a new concept, but controlled wastewater irrigation has not been extensively employed in developing countries because the installation of sewered sanitation has generally lagged behind improved provision of water reticulation systems. As more large-scale, urban, water-borne sewerage systems are installed, it is inevitable that effluent re-use for irrigation will become more important.

When effluent is to be re-used for irrigation, both public health and agronomic effects must be considered when assessing the possible impact on underlying groundwater. Public health considerations centre on the presence of pathogenic organisms in the effluent (Pescod and Alka, 1984) which may be transported with the infiltrating irrigation water. However, municipal wastewater may also contain hazardous trace elements, organic compounds and nutrients (Foster *et al.,* 1994). As in the case of unsewered sanitation, the likelihood of pathogens reaching the water table depends largely on their survival times and retention or adsorption in the soil. Research in the Lima area (Geake *et al.,* 1986) has demonstrated that under the soil conditions and irrigation practices at that site, elimination of faecal bacteria was less effective beneath the cultivated land than beneath the nearby wastewater stabilisation lagoons. Agronomic concerns are associated essentially with the effects of the major inorganic constituents of sewage effluent on crop yields and soil structure, as affected by the accumulation of salts, and on the danger of accumulation of trace organic and inorganic constituents in soil, crops and the underlying groundwater.

9.4.9 Mining activities

A range of groundwater pollution problems can be associated with mining activities. The nature of the pollution depends on the materials being extracted and the post-extraction processing. Coal, salt, potash, phosphate and uranium mines are major polluters (Todd, 1980). Metalliferous mineral extraction is also important, but stone, sand and gravel quarries, although more numerous and widespread, are much less important chemically. Both surface and underground mines usually extend below the water table and often major dewatering facilities are required to allow mining to proceed. The water pumped, either directly from the mine or from specially constructed boreholes, may be highly mineralised and its usual characteristics include low pH (down to pH 3) and high levels of iron, aluminium and sulphate. Disposal of this mine drainage effluent to surface water or ground-water can cause serious impacts on water quality for all uses (Table 9.7).

Pollution of groundwater can also result from the leaching of mine tailings and from settling ponds and can, therefore, be associated with both present and past mining activity. The important features of liquid and solid mine waste are similar to those described in sections 9.4.2 and 9.4.3, except that trace elements are likely to be an important constituent (Table 9.7). Examples of groundwater pollution from mine waste are described by Morrin *et al.* (1988) and from the leaching of salt from the tailings of the potash mines in Alsace, France by Meybeck *et al.* (1989). Mining activities can also have an indirect negative impact on groundwater quality where continuous large scale groundwater abstraction lowers the water table sufficiently to permit saline intrusion.

Disposal of the highly saline water produced with the oil from production wells has long been a problem (Meybeck *et al.,* 1989). In the USA, until the 1960s, the brine was usually placed in evaporation ponds. Lined pits functioned as intended, but saline water often reached shallow aquifers from unlined pits. In established oil fields, plumes of polluted groundwater remained for a long time after the pits themselves had been abandoned. Unlined pits were banned by the USA regulatory agencies in favour of reinjection into the oil-bearing formations, and there are estimated to be more than 70,000 brine disposal wells in the USA (Everett, 1980). Economic considerations have led to the use of abandoned oil production wells for brine disposal but, because they were not designed for injection, there have been many instances of faulty wells allowing brines to leak into important freshwater aquifers above the intended disposal zone. This type of groundwater pollution is obviously restricted to oil producing regions, but in such areas of the USA, for example, brines represent one of the major causes of ground-water pollution. An example of groundwater pollution by brine disposal in Arkansas is described by Everett (1980).

9.4.10 Groundwater resource management

The most important water quality change resulting from the management of groundwater resources is saline intrusion, which occurs where saline water displaces or mixes with freshwater in an aquifer. The problem is encountered in three possible circumstances:

- where there is upward advance (upconing) of saline waters of geological origin,
- where there is lateral movement from bodies of saline surface water, and
- where there is invasion of sea water into coastal or estuarine aquifers.

Under natural conditions, fresh groundwater flows towards, and discharges into, the sea. The position of the saline water-freshwater interface in a coastal or island aquifer is governed by the hydrostatic equilibrium between the two fluids of different densities. Sea water intrusion results from the development of these aquifers. Increasing groundwater withdrawals lower water levels and reduce flow towards the sea. Lowering of groundwater levels changes the hydrostatic conditions and causes local upconing of saline water below abstraction wells. Where groundwater withdrawal from a well or group of wells is sufficient to cause regional lowering of water levels or reversal of hydraulic gradients, then lateral movement of the interface will occur. The resulting pollution of the aquifer can be difficult to reverse.

There are important coastal aquifers throughout the world, and many oceanic islands are completely dependent on groundwater for their drinking water supplies. The thin lenses of freshwater on small islands are highly vulnerable to saline intrusion (and other forms of pollution), and development of these resources is often possible only by carefully controlled pumping from shallow skimming wells, accompanied by a well-developed operational surveillance programme. Avoiding saline intrusion is essential since there may be no alternative source of water supply except for the very expensive option of desalination.

Quality problems can also be caused by declining groundwater abstraction. In many of the major cities of northern Europe, groundwater usage increased for over a century with industrial growth to a peak in the 1940s and 1950s, producing falls in groundwater levels of tens of metres. Since then, decline of traditional industries and a switch to municipal water supplies has produced a dramatic decrease in private industrial groundwater abstractions, and water levels have begun to recover. The water table beneath a city may then rise, taking into the groundwater body pollutants of all types left in the dewatered part of the aquifer during the decades of industrial activity.

9.5. Assessment strategies

Groundwater bodies are always less accessible than surface water bodies. Consequently, obtaining the essential information on groundwater quality is technically difficult and costly. Significant limitations in groundwater quality assessment usually have to be accepted and need to be recognised in the interpretation and use of the monitoring results. This is often not appreciated by those responsible for establishing water quality goals or groundwater resource management strategies. Consequently, the information expectations placed on water quality assessments may be far beyond any ability to supply the information (Sanders *et al;* 1983). It is essential, therefore, for the designer of a groundwater quality assessment programme to understand and define the information objectives, and to appreciate the several types of monitoring that can exist.

9.5.1 Types of groundwater assessment

The fundamental requirement of a groundwater assessment to define the spatial distribution of water quality, applies almost invariably, regardless of the specific objective of the assessment. Three major categories of water quality assessment i.e. monitoring, survey and surveillance are defined in Chapter 2, and the various sub-categories listed in Table 2.1. Surveillance is generally related to the acceptability of water for a given use and/or the control of associated treatment processes (Foster and Gomes, 1989). Sampling for surveillance normally comprises frequent or continuous measurements on pumped water, and there is no need for samples to be representative of conditions in the aquifer. In all other cases, the requirement is to obtain analytical results from samples which are uncontaminated by the processes of sample collection and analysis and are representative of *in situ* conditions at specific points in the groundwater system.

The types of water quality assessment activity listed in Table 2.1 are not mutually exclusive. Considerable overlap exists, and there are also other approaches to defining categories of assessment (Everett, 1980). Some of the types of water quality assessment listed in Table 2.1 have inherent implications of scale. The most common types of national or regional assessment cover large areas, with sampling points tens or hundreds of kilometres apart. In contrast, emergency or impact monitoring associated with a single point source may be limited to the immediate vicinity of the actual, or potential, contaminant plume. The sampling points, whether few or many, may be confined within distances as low as a few tens or hundreds of metres. In some very detailed monitoring activities carried out for research programmes, the intensity of sampling points may be very great with, for example, sampling through an aquifer at very close depth intervals (0.1-0.5 m) to determine vertical variations in groundwater quality and mechanisms of solute transport (Foster *et al.,* 1986; Geake and Foster, 1989).

These differences in scale of operations are important and should be appreciated in establishing a monitoring strategy. They should not, however, be viewed too rigidly. Water quality assessment activities may have to be expanded in response to the results obtained. This could require any combination of extending the area covered, increasing station density, increasing sampling frequency and increasing the number of variables. For example, a basic survey, background or trend monitoring of a large groundwater basin may identify groundwater pollution which requires further investigation, and a programme of monitoring to be developed around a pollution source in a small part of the area. In some circumstances, this might lead on to assessment for enforcement or case preparation because legal action is being taken. Research may be associated with any type of assessment activity.

9.5.2 Establishment of an assessment strategy

Between the policy decision to establish a groundwater assessment programme and effective operation are a large number of decisions and steps. Simplifying assumptions must inevitably be made to reduce the complexity of groundwater quality to a level at which practical solutions are possible, allowing for the requirements for rigour implied by the information objectives. The number and type of simplifying assumptions depends on the purpose of the programme, and also on the financial resources available. Difficult decisions, implying the allocation of funds and staff, have to be made about the selection of sampling points, frequency of sampling and choice of variables. It follows that, depending on the assumptions and choices made, there are many levels of design that could be applied. The aim of the following sections is to present a general strategy for the establishment of a groundwater quality assessment programme. The purpose of such a strategy is to provide a framework for the planning and implementation of a programme that focuses on the most important water quality issues in the area covered, allows for properly designed but cost-effective monitoring of the issues, and interprets and presents the results in a way that allows pollution control measures and/or groundwater management decisions to be made.

The general strategy described here follows that shown in Figure 2.2. The wide range, world-wide, of hydrogeological conditions, groundwater uses, water quality issues and sources and scales of groundwater pollution described in the earlier sections of this chapter, means that there is an almost infinite variety of possible assessment programmes. Nevertheless, within the framework of the general strategy described here, it should be possible to establish a programme to meet all situations. In addition, the strategy outlined here can be applied in the review of existing programmes to ensure that the objectives and design of the programme can be modified as new types of pollutants become important and as improved knowledge and understanding of the hydrogeology becomes available. This is particularly important if a responsible authority wishes to move up through the stages of assessment defined below.

Proper establishment of an assessment programme can be considered as requiring two main stages. Firstly, a period of initial assessment (preliminary surveys) which may be very short (in the case of emergency surveys) or as long as a year, including surveys of pollutant sources, investigation of hydro-geological conditions (either by desk study of existing data or by field studies) and frequent sampling of available groundwater sources for a wide range of variables. This stage identifies the principal features of the ground-water quality and defines any seasonal fluctuations which may need to be taken into account. The second part of the programme entails longer-term groundwater monitoring in which the location and number of stations, sampling frequency and number of variables is established, to minimise costs but still meet the objectives of the programme. The components of each of these stages (see Figure 2.2) as they apply to groundwater are described in the following sections.

9.5.3 Defining objectives

The vital importance of groundwater resources in use and re-use for a range of different purposes requires aquifer management strategies in relation to both quantity and quality. It is unrealistic and unnecessarily expensive to measure all possible variables continuously and throughout the area. Establishing groundwater assessment implies difficult choices in respect of these options. Many authors have emphasised the need to define clearly the objectives of an assessment programme before beginning the design (Nacht, 1983; Canter *et al.,* 1987; Foster and Gomes, 1989). A number of different general reasons for groundwater assessment can be recognised in accordance with the categories of assessment given in Table 2.1:

- To develop an understanding of regional groundwater quality as an aid to better knowledge of the groundwater regime for optimal management of groundwater resources.

- To determine long-term trends in groundwater quality and to relate observed trends to human activities as a basis for informed decision making.

- To identify and monitor the locations of major pollutant sources (e.g. landfill site, mining operation or sewage disposal facility) and the movement of the pollutant in the aquifer, in relation to the design of aquifer restoration, treatment works or legal proceedings.

- To determine compliance with regulations and standards.

- To assess the effectiveness of pollution control measures, such as ground-water protection zones.

- To determine regional groundwater quality variations and background levels for studying natural processes and as a reference point for large scale and long-term anthropogenic impacts.

- To study groundwater recharge, for example chloride and tritium profiles.

- To determine flow paths and rates of groundwater movement using injected tracers.

- To determine the quality of groundwater, particularly with respect to its possible use as a source of drinking water or other non-potable uses. In the case of a major, extensively used aquifer in a highly developed region (possibly subject to both natural and artificial recharge) this could include a wide range of possible influences on groundwater quality, some of which could be specific for particular uses.

- To determine the extent and nature of an accidental pollution event (e.g. chemical leakage).

- To determine groundwater quality in the vicinity of public supply sources, threatened by point source pollution or saline intrusion, to protect the integrity of the supply and maintain its use.

• To calibrate and validate groundwater quality models which may have been developed for pollution control or resource management, for example saline intrusion, contaminant migration, prediction of nitrate trends.

The above reasons for groundwater quality assessment are only a part of the objectives. Those which are appropriate to the situation should be incorporated in a written statement of the objectives, which should also include reference to the techniques that will be employed, the data that will be obtained, the approach to data interpretation for provision of information, and the use to which the information will be put.

9.5.4 Selecting and defining area

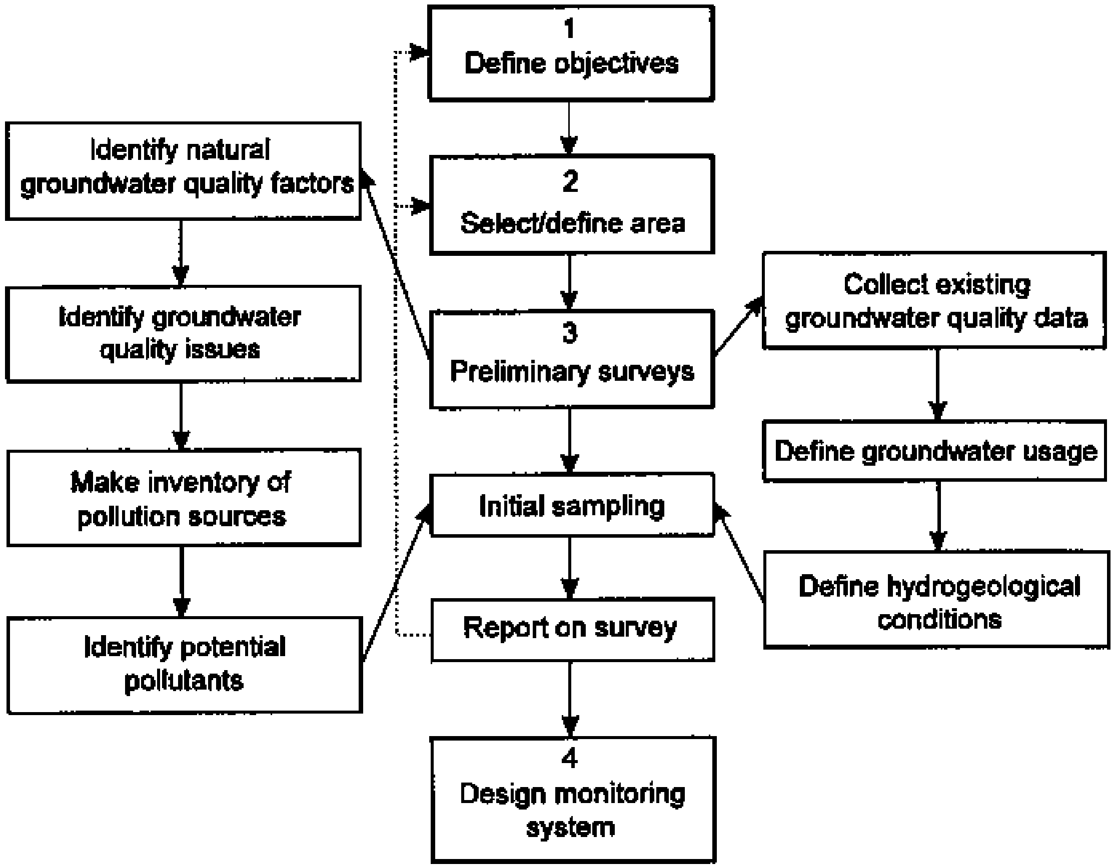
Selection of the area in which the assessment is to be established will be governed firstly by the objectives of the programme and then by a combination of administrative and hydrogeological considerations. In many cases the monitoring extends to the area of jurisdiction of the agency responsible for carrying out the monitoring, and may coincide with national, regional, state or district boundaries, or with the boundaries of major catchments. Often, however, political or administrative boundaries are very different from hydrological boundaries, and in any case groundwater divides may not coincide with surface water catchments. Pollutants from an adjoining area may enter from sources which cannot be directly studied and which are beyond the jurisdiction of any pollution control measures that may be indicated as a result of the assessment. In the case of very large catchments or major aquifers, this may even include national boundaries.

Of the assessment categories defined in Chapter 2, trend and operational monitoring are likely to be extended over the whole basin or administrative area. The extent of monitoring activities for the other categories will be related to the water quality issue or issues, and water uses, embraced by the objectives of the programme. Operational surveillance related to diffuse pollution from agriculture is likely to be extensive, concentrating on potable supplies in crop growing areas and any adjacent areas down the hydraulic gradient. Operational surveillance of saline intrusion is generally restricted to coastal areas. Operational surveillance or emergency monitoring related to point source pollution from a landfill or an industrial spill is much more restricted to the area close to the source. In many such cases it may not be possible to define the area to be studied with any degree of certainty, and preliminary surveys (see section 9.5.5) are required. In the meantime, monitoring is concentrated on potable supply sources immediately down gradient of the source, making some general estimates of the possible extent of the contaminant plume based on the aquifer type and regional hydrogeology.

9.5.5 Preliminary surveys

In all cases of the establishment of an assessment programme, an initial survey is required, the extent of which depends on the objectives of the programme, the complexity of the hydrogeology and the number and nature of the water quality issues to be addressed. For the purposes of trend monitoring, the preliminary survey may be restricted to a desk study review of existing data. In contrast, emergency surveys at a major spill will almost certainly need detailed and expensive field investigations which may have to be done very rapidly. The principal components of the preliminary survey are shown in Figure 9.21 and described in the following paragraphs.

Figure 9.21 Components of preliminary surveys for groundwater quality assessment



Collecting existing groundwater quality data

The first task is to collect all existing groundwater quality data. Some data may be held by the organisation which is proposing to carry out the assessments, especially if that is the regional or national authority responsible for water. Other sources include universities or research institutions, consultants' reports, published scientific literature, and other government departments. A certain amount of ingenuity and persistence may be required to locate fragmented and dispersed data, and judgement will often be needed to distinguish reliable from unreliable data. This information is used to identify the principal features of the natural groundwater quality in the region or around the defined area, and to provide an indication of possible anthropogenic influences which may be related to the objectives of the assessment, thus giving a first indication of possible sources of pollution.

Table 9.9 Hydrogeological data requirements for groundwater monitoring system design

Data	Possible source material
Aquifer locations, depths and aerial extent	Geological and hydrogeological maps and reports
Physical properties of aquifers, especially transmissivity	Test pumping, geological and hydrogeological maps
Areal distribution of groundwater levels	Borehole archives and observation wells
Areal distribution of depth to groundwater	Borehole archives and observation wells
Areas and magnitudes of natural groundwater recharge	Climatic and hydrological data, soils and land-use
Areas and magnitudes of artificial recharge	Irrigation records, municipal recharge schemes
Areas and magnitudes of natural groundwater discharge	Streamflow and water level data
Locations and magnitudes of groundwater abstractions	Borehole archives, municipality and irrigation department records, well owners
Directions and velocities of groundwater flow	Geological and hydrogeological maps, water level and transmissivity data

Source: Modified from Everett, 1980

Collecting hydrogeological data

Concurrently, and often from many of the same sources, existing hydro-geological information must be collected. Depending on the location and size of the defined area, this may comprise an enormous body of data, some of which may already have been interpreted and produced in report form or published works. The data requirements are summarised in Table 9.9. Again, considerable persistence may be needed to track down this information in the various national or regional organisations responsible for collecting and holding it. The objective of this is to obtain as full as possible an understanding of the hydrogeological conditions in the area designated for assessment. The principal requirement, especially where point sources of pollution are concerned, is a knowledge of the groundwater flow regime, without which assessment could be misleading and/or financially wasteful because samples are taken from the wrong locations or depths.

Defining groundwater use

Evaluation of the impact of actual or potential pollution requires knowledge of the use of the resource. In this context, it is important to define the quantities of groundwater being, or projected to be, abstracted, the locations of the major centres of pumping and the type of use to which the water is put. The impact that different pollutants have on the range of water uses is given in Table 1.3. In the case of rural water supplies, small groundwater abstractions may be spread widely over the region or area of concern. In the case of large abstractions for municipal or industrial use, major wells or well fields may draw down groundwater levels sufficiently to modify locally the ground-water flow regime; this must be accounted for in designing the programme. Information on groundwater usage may come from national or regional agencies, municipalities, public water utilities, irrigation or agriculture departments, and individual industrial or agricultural users.

Identifying quality issues and potential pollutants

A further major component of the preliminary survey is the identification of the most important water quality issues in the area. In the case of a major, extensively used aquifer, several water quality issues may be important. Groundwater quality may be affected by agricultural, industrial and urban influences. The principal activities that could potentially have an impact on groundwater quality have been described in section 9.4 and are listed in Table 9.7. The identification of waste disposal practices and the distinction between point, line and diffuse sources of pollution is particularly important at this stage of the programme.

For many assessment objectives it may be necessary to follow up a general appraisal of the main water quality issues in the area with a detailed inventory of pollution sources, including identification of potential pollutants and characterisation of the possible sub-surface contaminant load. This then becomes a groundwater pollution risk assessment, as described by Foster and Hirata (1988). Such a risk assessment combines the evaluation of likely sub-surface contaminant load (from whichever potential groundwater pollution sources have been identified) with appraisal of the vulnerability of the aquifer to pollution. This appraisal is based on existing hydrogeological knowledge, principally concerning the occurrence of groundwater, depth to water and overall lithology (Foster and Hirata, 1988). This and other empirical methods of assessment of the potential for groundwater pollution typically involve the development of numerical indices, with larger numbers denoting greater risk. Canter *et al.* (1987) review nine such methods, relating to surface impoundments, landfills, hazardous waste sites, brine disposal and pesticides. One of the most comprehensive numerical methods for assessing vulnerability of groundwater to pollution based on local hydrogeological conditions is the DRASTIC rating scheme devised by the United States Environmental Protection Agency (US EPA) (Aller *et al.,* 1987). However, the DRASTIC scheme has several major weaknesses and, therefore, must be used only by very experienced hydrogeologists. A comprehensive approach to risk assessment related to possible microbiological pollution of potable ground-water supplies is given by Lloyd and Helmer (1991).

To meet the objectives of assessing potable water quality from a heavily used aquifer in a highly developed region, the preliminary survey may need to include an extensive and detailed survey of agricultural, urban and rural waste disposal and sanitation practices and industrial activities. Information will need to be collected from a range of national and/or regional organisations, and field surveys carried out to establish such factors as fertiliser and pesticide use. Figure 9.22 is an example of a survey form used in Barbados to establish which pesticide compounds were commonly in use and their application rates. Having identified the most widely used compounds, an appraisal of their mobility and persistence from existing literature enables an initial selection to be made of the compounds most likely to be leached to groundwater and which should, therefore, be included as variables for monitoring in the programme.

Identification of potential pollutants from a landfill may be particularly difficult. A wide range of trace elements and organic pollutants may be involved and comprehensive records may be lacking for materials disposed at the site, especially for old or disused sites. When studying pollution from a spill or leak at an industrial site, only a single compound may be involved. However, a local survey to identify other sources of the same compound may be required, especially if the monitoring is related to possible legal proceedings.

Most of the discussion in the preceding paragraphs has been directed towards basic surveys. For background and trend monitoring on a regional scale a preliminary survey is necessary to locate stations not affected by immediate local pollution sources. This may present few problems in developing countries, but may be especially difficult in highly industrialised ones.

Initial sampling

The preliminary survey may require a short and intensive programme (see Table 2.2) of groundwater sampling, and this is recognised as a distinctive category of groundwater monitoring. This initial sampling programme may be essential:

• to identify the principal features of natural groundwater quality if there is no, or very little, existing groundwater quality data,

• to act as an invaluable supplement to land-use, sanitation or industrial surveys as a means of identifying potential pollutants, and

• to identify or confirm seasonal, lateral or vertical variations in ground-water quality that must be taken into account in the programme.

Figure 9.22 Example of agricultural survey form used in Barbados to establish pesticide use and the associated risk of groundwater pollution

ENVIRONMENTAL ENGINEERING DIVISION MINISTRY OF HEALTH

Name of Farm/Plantation: ____

Address: ____

Name of Manager: ____

Telephone number: ____

Rainfall area: High____________ Intermediate__________ Low__________________________

Area under vegetable production: ____

Area under sugarcane production: ____

Do you rotate between areas of cane and vegetable production? If yes, how often? ___________

Is there a water well on your property? ____

Is it used for irrigation
purposes?____
How do you dispose of old/unwanted pesticides?
____
How do you dispose of cleansing water from spraying
equipment?_____________________
Do you think it is necessary to monitor water, soil and food for pesticide
residues?__________
Any other comments?
____

(please fill in table opposite)

Insecticides used	**Active ingredient**	**Concentr. g l^{-1} (kg)**	**Amount used/year**	**Frequency**	**Years used**
1.					
2.					
3.					
4.					
5..					
6.					
7.					
8.					
Fungicides used	**Active ingredient**	**Concentr. g l^{-1} (kg)**	**Amount used/year**	**Frequency**	**Years used**
1					
2.					
3.					
4.					
5.					
6.					
7.					
8.					
Herbicides used	**Active ingredient**	**Concentr. g l^{-1} (kg)**	**Amount used/year**	**Frequency**	**Years used**
1.					
2.					
3.					
4.					
5.					
6.					
7.					
8.					

Name other pesticides that have been used in the past:
To meet these clearly defined objectives, the preliminary survey is likely to comprise frequent sampling, over a short period of time, of a large number of points for a moderate to high range of variables, the exact combination of which will depend on the circumstances and the resources available. Sampling in preliminary surveys is almost invariably restricted to existing wells, boreholes and springs (the locations of which will have been established earlier, at the time of collecting existing hydrogeological information). In most assessment programmes, this initial sampling can be completed within a year or often much less. In the case of emergency surveys of a spill, particularly if hydrogeological conditions were such that major public supply sources were threatened, the initial sampling might be completed in a matter of days. Where the main potential pollutant is faecal pathogens from unsewered sanitation, which is possibly the most common groundwater pollutant in many developing countries, the preliminary survey will always include bacteriological analysis of groundwater sources (Lloyd and Helmer, 1991).

All of the information collected in the preliminary survey must be analysed and summarised in report form. This may include, as recommendations, the design of the monitoring network, or information to be used as the basis for the design (Figure 9.21).

9.5.6 Design of groundwater quality assessment

Design of a programme to assess groundwater quality includes much more than the station location. Design consists essentially of the choice of sampling stations, sampling frequency, range of variables, methods of data production and interpretative approaches required to produce the information needed for decision making. Design, therefore, involves choices concerning all of the steps in the overall strategy outlined in Figure 2.2. The designer and operator must see the system as a whole, rather than as several separate activities. This has not always been the case in the past and, as a result, monitoring network design has probably received more attention in the literature than the overall assessment system design. Nevertheless, network design and the decisions to be made in the process remain a critically important component of the design of assessment programmes. General considerations of network design are discussed in Chapter 2, and specific problems related to groundwater bodies are further discussed below. The factors which determine network design for groundwater bodies are summarised in Table 9.10.

Number and location of sampling stations

As indicated in Table 9.10, the number and location of sampling stations is a function of the objectives and scale of assessment, the hydrogeological complexity, land-use distribution and economic considerations. The latter will inevitably constrain the comprehensiveness of the proposed network. Relating the general principles of Table 2.2 to groundwater, the possibilities range from a small number of stations dispersed over a region or aquifer to provide background and trend monitoring, or a small number of stations for emergency surveys around a spill, to a medium to high number of stations for multi-purpose monitoring or operational surveillance of potable water quality.

Table 9.10 Factors which determine sampling network design for groundwaters

Sampling point		Sampling frequency	Choice of variables
Type	Density		
Assessment objectives	Assessment objectives	Assessment objectives	Assessment objectives
Hydrogeology (complexity)	Hydrogeology (complexity)	Hydrogeology (residence time)	Water uses
	Geology (aquifer distribution)	Hydrology (seasonal influences)	Water quality issues
	Land use		Statutory requirements
	Statistical considerations	Statistical considerations	
Costs	Costs	Costs	Costs

In the Netherlands, for example, a national groundwater quality monitoring programme has been established, principally directed towards diffuse sources of pollution (van Duijvenbooden, 1993), and following the general strategy outlined here. The country is densely populated, with intensive agricultural and industrial development, and is dependent on relatively shallow and vulnerable aquifers for its water supplies. Consequently, the relatively large number of 380 monitoring points provides an overall station density of 1 per 100 km^2, with emphasis on areas of importance for drinking water supplies. The sites were chosen to give adequate coverage of the range of soil types, land use and hydrogeological conditions in the country, and avoided the influence of any localised sources of pollution (van Duijvenbooden, 1993). The national groundwater monitoring network of the former Czechoslovakia described in section 9.6.5 had a similar average station density of 1 per 250 km^2. Much wider spacing of sampling stations can be anticipated in larger and/or less developed countries.

In all cases, the designer of the sampling network must consider the importance of lateral and vertical variations in hydraulic conductivity in relation to the network scale. The required number of sampling points per unit area and unit depth of aquifer must be regarded as a function of the hydraulic heterogeneity of the groundwater system (Ward, 1979). Designing the network needs hydrogeological expertise to ensure that adequate account is taken of the complexity of the groundwater flow system.

It is highly unlikely that existing wells will be in suitable locations in relation to most types of point source monitoring, even though analyses from existing supply wells may have been the first indication of the problem. Hydrogeological conditions, especially groundwater flow rates and directions, will be the main consideration in locating observation wells in relation to identified point sources of pollution. Great care should be exercised, however, in the interpretation of limited hydrogeological data in relation to siting observation wells, especially where fissure flow predominates. Extreme heterogeneity and anisotropy of hydraulic conductivity is possible, and the plume of pollutant may take a form and direction very different to that which might be expected from regional groundwater flow based on water level contours. Monitoring wells may completely miss the plume unless this possibility is appreciated, and a wider initial network installed accordingly.

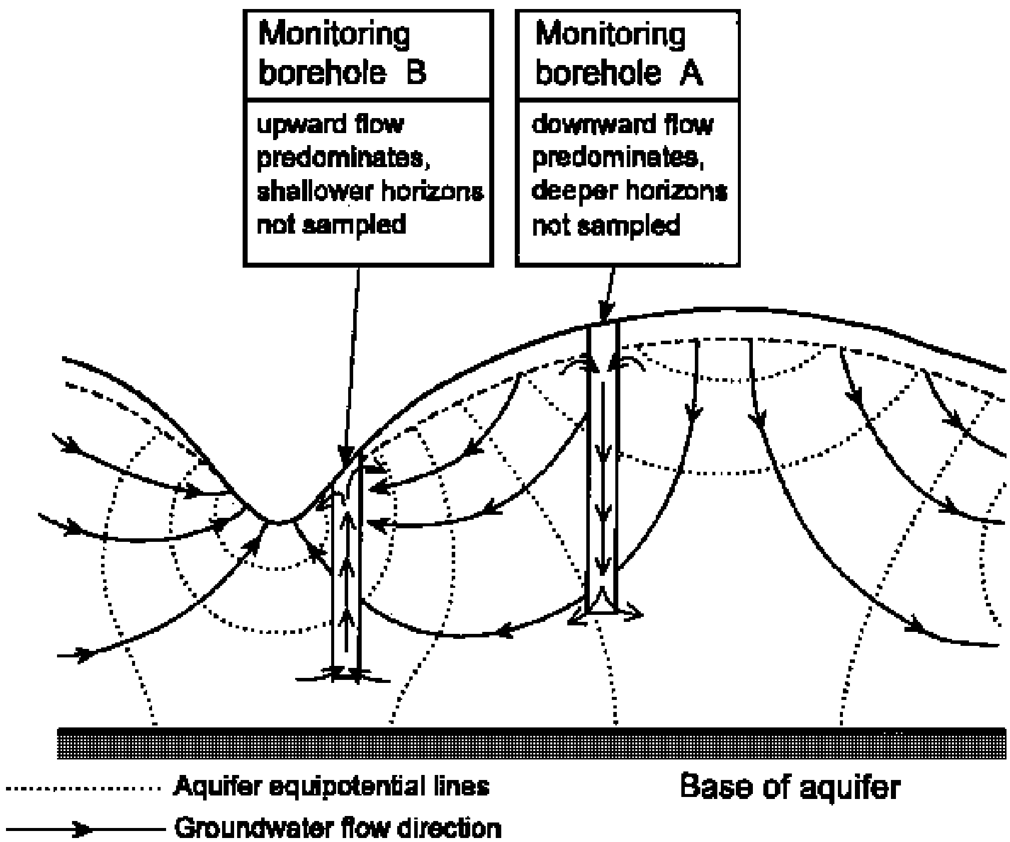
A monitoring scheme will always need to strike a balance between costs and comprehensiveness. Under the USA Resource Conservation and Recovery Act (RCRA), regulations for a single point source require a minimum of three monitoring wells down-gradient and one up-gradient. It is recommended that the three down-gradient wells should be placed in a triangular arrangement, skewed down-gradient, with two inside the existing plume and one beyond it. This will provide data on the spatial variation of groundwater levels and allow for observation of the plume migration rate. Although well-intended, such specific regulations may be of limited value (Canter *et al.,* 1987), or even dangerously misleading. It may not be possible to determine the hydraulic gradient without drilling some observation boreholes, the hydraulic gradient may be transient and reverse seasonally, or the source itself may produce a local recharge mound which masks the regional hydraulic gradient. In the case of extreme heterogeneity as described above, and in karst aquifers, these simple guidelines cannot apply. In all situations of point source monitoring, phasing of the network installation should be envisaged and budgeted for, so that the results from the initial locations can help to optimise the siting and type of additional sampling stations.

Type of groundwater sampling stations

A major consideration in network design for groundwater bodies is the type of sample station in relation to the vertical distribution of water quality. Existing groundwater quality assessment programmes, for a range of the objectives listed, depend entirely on samples taken from the pump discharge as it comes to ground level. This sample may not be representative of hydro-chemical conditions in the aquifer, because of transformations resulting from the temperature and pressure changes caused by lifting it, perhaps from great depth, to ground level. These processes include the entry of atmospheric oxygen, precipitation of pH controlled variables and loss of volatile compounds. The borehole or well may penetrate through, and draw water from, a considerable thickness of an aquifer or even more than one aquifer, and the resulting pumped sample may be a mixed one. The public supply or irrigation boreholes from which pumped samples are usually taken were designed and constructed for producing water rather than for monitoring it. This is not a problem for many assessment objectives since the mixed or aggregate pumped sample measures the quality of the water that goes to the user and is, therefore, appropriate in relation to objectives related to water use and acceptability. Samples from pumping boreholes and wells are also the basis of background and trend monitoring programmes on a regional, national or continental scale, such as the GEMS (Global Environment Monitoring System) global network (WHO, 1992). Existing boreholes selected for such programmes should be hose with the most reliable information, so that a short screen length over the most appropriate depth interval of aquifer and most comprehensive geological log, facilitate subsequent interpretation of the results.

To meet other objectives, a knowledge of the vertical distribution of groundwater quality is important to provide, for example, a three dimensional picture of the development of a contaminant plume or a saline intrusion problem. In detailed monitoring of a pollution incident, adequate definition of sample depth requires a knowledge of the design of the well and the hydraulic conditions within the well and in the surrounding aquifer. However, construction of the borehole itself may have disturbed the local groundwater flow regime. This is especially likely for boreholes open over much of their length and situated in recharge or discharge areas where there are significant vertical components of groundwater flow (Figure 9.23). The same might occur close to major abstraction boreholes. A borehole may penetrate two or more aquifers separated by impermeable strata. Different hydraulic heads may produce upward or downward components of flow, and the borehole itself may induce cross-contamination by permitting flow from one aquifer to the other. Such boreholes are generally very misleading for monitoring purposes, and representative groundwater samples are unlikely to be obtained, whatever sampling method is used. Similar problems can occur in fissured aquifers in which groundwater flow is restricted to a few major horizons which may have very different groundwater quality.

Figure 9.23 The effect of vertical flow components on fully-screened or open monitoring boreholes (After Foster and Gomes, 1989)



The installation of purpose-drilled observation boreholes to specified depths, with known screened intervals, offers the best chance of obtaining samples that are reasonably representative of conditions in the aquifer. If preferred drilling methods (Scalf *et al.,* 1981; Barcelona *et al.,* 1985; Neilsen, 1991) are employed with suitable casing materials (Barcelona *et al.,* 1983), then the sample bias will be minimised and restricted largely to the effects of sampling itself. Construction procedures for monitoring boreholes follow the same general sequence as for supply boreholes (Keely and Boateng, 1987). This comprises drilling, installation of well screen, filter pack and plain casing and the construction of a sanitary seal to prevent contamination by surface water. The drilling process may introduce fluid or water of different chemistry that contaminates the aquifer close to the borehole. Therefore, a period of cleaning and development pumping may be required after completion, to restore natural hydrochemical conditions. Observation boreholes are normally of smaller diameter (50-100 mm) and open to the aquifer over a much more limited depth interval (1-5 m). They also allow the measurement of water levels and the performance of hydraulic tests on the aquifer.

Design of monitoring boreholes must take into consideration (Foster and Gomes, 1989):

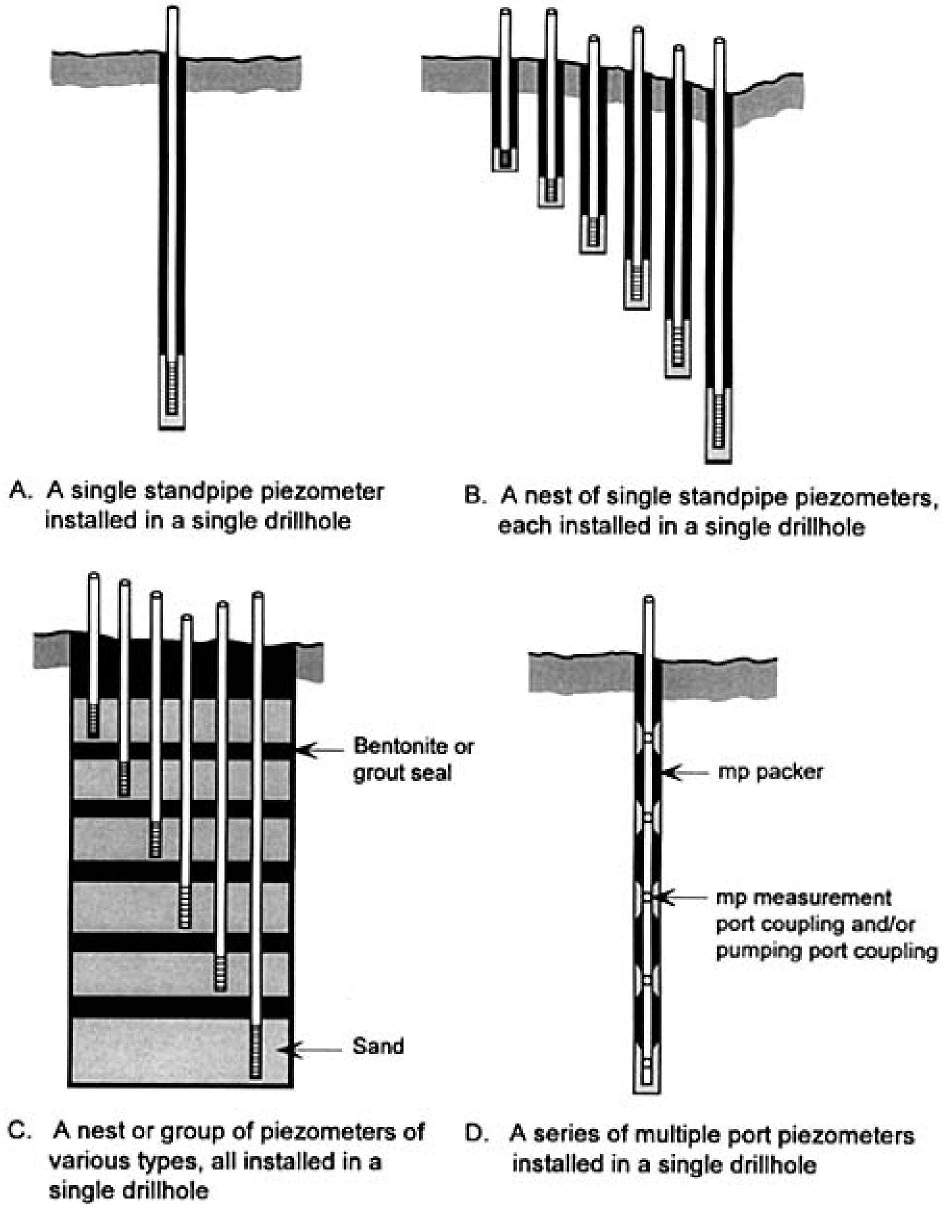
- the objectives of, and financial provision for, the programme,
- the anticipated variables of concern and their probable concentrations,
- the available, or proposed, sampling equipment, and
- the nature of the groundwater flow regime.

Single-level observation boreholes of the type described above are the simplest and most commonly used, and provide adequately representative samples to meet most monitoring objectives at modest cost. Inert materials (pvc, PTFE or stainless steel) can be used where the variables to be monitored include industrial organic compounds or pesticides.

Where the assessment objective requires a knowledge of the vertical distribution of water quality, multi-level sampling installations are required. The simplest solution, technically, is to construct clusters of single observation boreholes completed at different depths in the aquifer (Figure 9.24) or, where necessary, in different aquifers. This is very expensive (Table 9.11), especially for moderate to deep installations. Alternatively, a nest of small-diameter observation points, each sealed above and below, can be installed in a single borehole (Figure 9.24). This approach produces economies in drilling, but placing and sealing each observation point correctly and developing them requires great care. These difficulties of execution, combined with drilling diameter limitations, normally restrict such installations to 2-5 sampling points in a vertical profile (Table 9.11). However, this type of installation has provided reliable monitoring data in many instances. The 380 groundwater sampling stations in the national network in the Netherlands (van Duijvenbooden, 1993) are purpose-built installations of this type (Figure 9.24C), with three separate, short, screened intervals at about 10, 15 and 25m depth.

More recent developments in sampling technology have seen the increasing use of multi-port installations (Figure 9.24). The borehole is completed with a single plastic (pvc or PTFE) lining tube with ports or small screens at many depth intervals. Sampling through small-diameter, permanently attached inert plastic tubes is achieved by suction at shallow water level depths (Pickens *et al.,* 1978) or syringe type positive-displacement pumps where water levels are deeper (Cherry, 1983). Further variations of multilevel sampling employ gas-driven pumps permanently attached to the small ports or screens, in which case hydraulic measurements are not possible, or installations in which the special ports are opened and sampled by a gas or electric sampler introduced into the borehole. Multi-level installations are required to achieve representative samples in the difficult groundwater systems that have vertical flow and fissure flow.

Figure 9.24 Types of groundwater monitoring installations



Groundwater sampling

A detailed discussion of groundwater sampling techniques is outside the scope of this book. Comprehensive reviews are given by Scalf *et al.* (1981), Foster and Gomes (1989) and Neilsen (1991). Driven by the increasing interest in groundwater pollution and aquifer restoration and related legal and quality assurance considerations, groundwater sampling, particularly in relation to unstable variables at low concentrations, is the focus of considerable current research. New developments and case studies are continuously reported, with new equipment always being advertised by specialist companies. Any detailed review quickly becomes outdated, but some general comments on traditional and novel methods of sampling can be made.

Table 9.11 Comparison of groundwater monitoring installations

Type of installation	Vertical sampling points	Water level measurements	Hydraulic testing	Inert materials	Drilling costs	Material costs	Overall costs
Supply borehole	Integrates over screen interval	Disturbed by pumping	Data may exist	No	None (already exists)	None	Very low
Single piezometer	1	Yes	Yes	Yes	Medium	Low	Low
Cluster of single piezometers	Several	Yes	Yes	Yes	Very high	High	High
Nest of piezometers in single borehole	2-5	Yes	Yes	Yes	High	High	Medium
Multi-port sampling systems	Many	Some types	Some types	Yes	Medium	High	High

Choice of sampling methods is an important component of monitoring system design, and is closely tied to the selection of types of sampling points. The choice is governed by the same criteria, i.e. selection of variables (which is in turn determined by the objectives of the assessment programme), the water quality issues to be addressed, the hydrogeological conditions and financial considerations. The most commonly used methods are sampling the discharge of existing production boreholes, grab sampling from non-pumping boreholes and, much less frequently, sampling during borehole drilling.

Discharge samples are normally collected in bottles from a well-head tap or directly from the pump outflow. Where this is not possible, samples are often taken from the nearest tap in the distribution system, which may be after the water has passed through a well-head storage tank. Samples collected in this way have serious limitations for water quality assessment (Table 9.12). Nevertheless, such samples are suitable for surveillance of potable water quality, provided these limitations are appreciated. Pump discharge samples may also be of value for monitoring where there are no vertical variations in water quality, or if an average sample integrated over the whole screened section of the aquifer is appropriate. This is likely to be the case in background and trend monitoring on a regional scale (dark and Baxter, 1989), provided the construction details of the borehole are well known, and the pumped sample is taken after an adequate period of pumping to allow the borehole outflow to reach hydrochemical equilibrium with inflow from the aquifer.

Table 9.12 Summary of characteristics of groundwater sampling methods in boreholes

Method	Hydrogeological representativity	Sample modification			Relative cost
		Contamination	Degassing	Atmospheric contact	
Production borehole discharge	Poor. No control over sample depth. Mixing and dilution	Moderate from well materials	Severe	Moderate to severe	Very low
			← Loss of unstable variables →		
Grab sample from non-pumping borehole	Unreliable. No control over sample depth. Vertical flows	Moderate to severe, cross-contamination. Moderate from well materials	Moderate	Moderate to high	Low
During borehole drilling	Moderate to good control over sample depth, with temporary casing	Moderate to severe from drilling fluid. Some cross-contamination	Severe	Severe	
			← Loss of unstable variables →		High

Source: Modified from Foster and Gomes, 1989

Grab sampling by bailers or depth samplers (Foster and Gomes, 1989) involves lowering of the device to a known depth in the borehole water column, closing it and raising it to the ground surface. Due to their cheapness and ease of use, such samplers have long been the mainstay of monitoring programmes but, like discharge samples, they have serious limitations (Table 9.12).

In some circumstances, groundwater samples collected during drilling may provide information on vertical variations in quality. The feasibility of doing this depends on the drilling method. The percussion and air-flush rotary methods permit the collection of water samples as drilling progresses (by bailer and air-lift pumping respectively), whereas rotary methods using mud-or water-flush do not. Samples collected during drilling are, however, prone to contamination downward from higher levels in the borehole and from the drilling fluid or compressed air (Table 9.12).

None of these traditional methods is likely to achieve the necessary precision and reliability for the assessment of groundwater pollution. Improved methods should be introduced where the need for better reliability is economically justified or where unstable variables of public health significance are involved. Table 9.13 summarises the features and suitability for groups of variables of some of the improved sampling techniques.

Sampling frequency

Sampling frequency is a function of the type and objectives of the assessment programme, the water quality issues to be addressed, the nature of the groundwater body and the logistical and financial resources for sample collection and analysis (Table 9.10). Optimum frequencies for groundwater sampling have often been set either by regulation or from statistical approaches based on analogies with surface water bodies (Nelson and Ward, 1981; Casey *et al.,* 1983). Because of the generally long residence times and relatively slow rate of evolution of groundwater quality, less frequent sampling is required than for surface water bodies. Basic surveys, background and trend monitoring could be based on annual samples for large ground-water bodies and quarterly samples for smaller aquifers. The national groundwater assessment system in the former Czechoslovakia, described in section 9.6.5, took account of aquifer size and depth in determining sampling frequency.

Table 9.13 Summary of characteristics of borehole sampling pumps

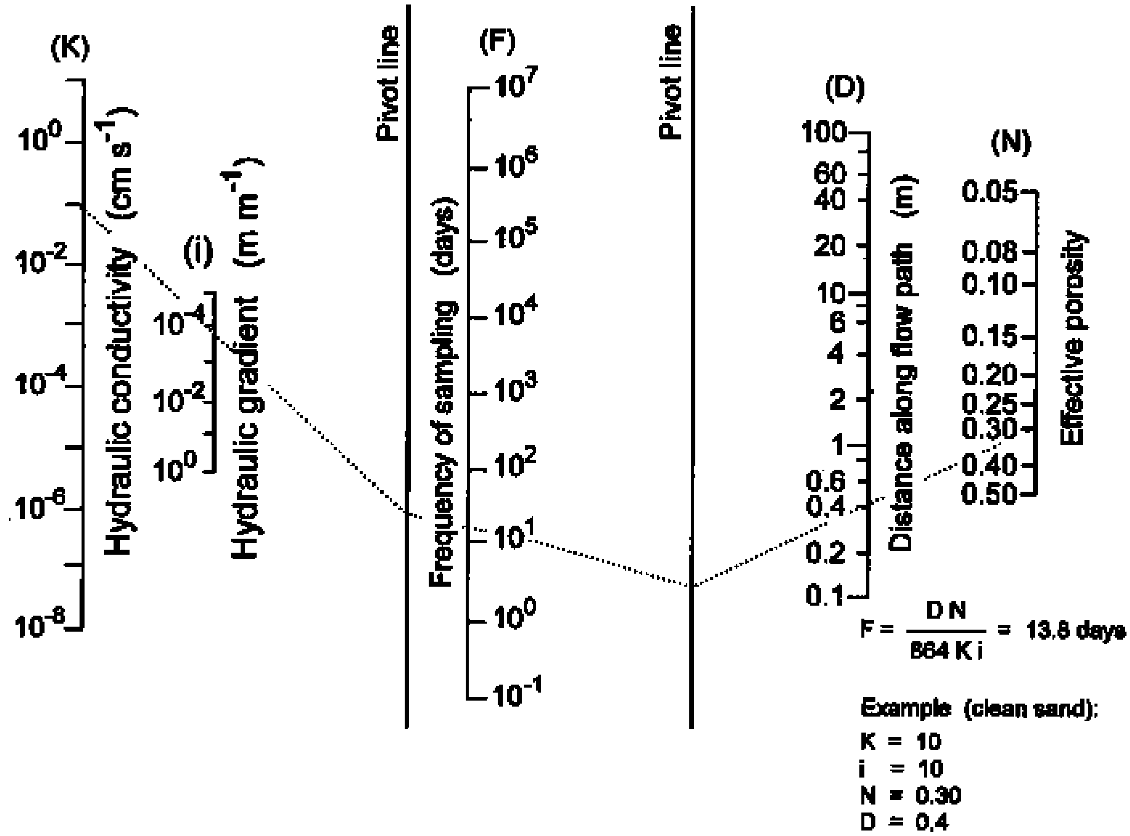
Pumping mechanism	Principle of operation	Advantages	Limitations	Relative cost
Suction lift • peristaltic • manual vacuum • centrifugal	Sample withdrawn by suction applied directly to water or via collection bottle	Highly portable Suitable for all variables given adaptation and accessories Appropriate for small diameter piezometers Gentle delivery with low pumping rates Adaptable for purging prior to sampling Pump-sample contact limited in most cases	Sampling depth limited to 8 m Degassing and aeration difficult to control	Low but increases if inert materials used
Gas-driven • double tube • continuous delivery	Positive gas pressure drives water from borehole backflow prevented by check valves	Unlimited depth Can be made of inert materials Efficient for purging prior to sampling Flow rates can be controlled Can be combined with *in-situ* sorption or depth-control packers Suitable for permanent installation	High purity inert gas needed to avoid contamination Entrance of gas in discharge line may cause degassing/volatilisation Portability reduces for deep boreholes	Low to moderate, but increases if inert materials used
Positive displacement submersible (A)	Water driven continuously from borehole by: gears or rotor assembly; gas operated plunger; gas	Unlimited depth All variables - given adaptation and use of inert materials	Moderate portability Electric centrifugals not reliable for unstable variables Piston pumps of continuous delivery	Moderate to high, especially if inert materials used

• electric centrifugal • piston pump • bladder pump	operated diaphragm	Appropriate for small diameter piezometers Can be combined with *in-situ* sorption or depth control Flow rates can be controlled Efficient for purging prior to sampling Low purity gas acceptable because no contact with sample	difficult to dean/maintain	
Positive displacement submersible (B) • manual inertial • mechanical inertial	Sample-riser tube and foot-valve assembly moved vertically, filling with water on downstroke and raising water on upstroke, with inertia maintaining downstroke	Portable, but can also be dedicated Durable Good for borehole development, purging, and hydraulic testing Flow rates can be controlled Can be made of inert materials Well suited for unstable variables	Sampling depth limited (manual to 40 m) Deeper boreholes require motor drive, reducing portability and increasing cost Time consuming Not yet well proven and documented Plastic foot valves wear with heavy use	Low for home-made manual version Increases considerably with motor drive and inert materials, which may be difficult to obtain!

Source: Foster and Gomes, 1989

More frequent sampling is required for pollution source studies, which should be related to the type of pollution source and the local hydrogeological conditions. Very frequent sampling may be required, for example, in the case of a point source of pollution close to a potable water supply in a highly vulnerable aquifer with rapid fissure flow. The nature of the source (Figure 9.8) has a direct influence on the type of pollutant plume that may be formed. The size, shape and rate of plume movement depends on the source characteristics, groundwater flow system and the chemical and microbiological attenuation of the pollutants in the plume. The size and shape of a plume from a continuous pollutant source can be estimated (Barcelona *et al.,* 1985) using a relationship developed by Todd (1980) for defining capture zones to pumping boreholes. From this, the number of sampling points in the flow path required to assess the movement of the plume can be decided. Sampling frequency can then be obtained from the hydrogeological characteristics at the site by calculating groundwater flow velocity from the Darcy equation and the effective porosity of the aquifer material. A nomogram for translating the hydraulic data into sampling frequency at various flow path lengths is given in Figure 9.25.

Figure 9.25 Sampling frequency nomogram (After Barcelona *et al*., 1985)



Choice of variables

Where the assessment objective is related primarily to groundwater pollution, the choice of variables depends on the quality constraints imposed by the principal water use, together with the likelihood of the variables thus defined, being present in troublesome concentrations from natural or anthropogenic origins. Some guidance as to the likely pollutants from a range of activities is given in Table 9.7, and a full discussion of the selection of variables for water quality assessment is given in Chapter 3.

Monitoring in the unsaturated zone

Traditional monitoring systems based on pumped samples suffer from serious inadequacies with respect to the early detection of diffuse pollution (Parker and Foster, 1986). As a result of the considerable residence times of groundwater carrying pollutants through the unsaturated zone (section 9.2), and the possibility of marked water quality stratification below the water table, a large part of an aquifer system may be seriously polluted before contaminants of diffuse origin are detected in a monitoring network based on pumped samples.

In unconfined aquifers, the unsaturated zone occupies a key position between the land surface, close to which most contaminants are discharged, and the groundwater table, from below which water supplies are drawn (Foster and Gomes, 1989). Monitoring of water quality in the unsaturated zone can give early warning of pollution and, especially if this zone is thick, could allow time for control measures to be taken to protect water supplies. Unsaturated zone monitoring is becoming more widely used to detect potential groundwater pollution from diffuse agricultural sources and from waste disposal facilities. A comprehensive discussion of unsaturated zone sampling is given by Wilson (1983).

The most common method of unsaturated zone sampling is by suction or tension. This is an *in situ,* non-destructive method in which water is drawn into a sample chamber through porous cups or plates by applying a vacuum. Once water has filled the chamber, it is drawn to the ground surface either by suction lift (limited to 8 m) or by a gas-driven pump. The suction method has the advantages of being cheap, being easy to install, being made from inert materials and allowing repeat sampling. It has, however, significant operational difficulties caused by clogging, and limitations in respect of sample representativeness due to adsorption of some variables, by-pass of infiltrating water in structured or fissured materials, and uncertainty about what volume of material is being sampled.

Much less frequently used are free-draining pan lysimeters, which consist of a collecting tray placed in the unsaturated zone from a trench or tunnel. The impermeable tray of inert material intercepts downward flow, which is drawn off into a container and sampled regularly. Representativeness is improved, especially in fissured materials by the larger area sampled, and generally by the absence of suction through porous cups. Installation, however, causes more disturbance of natural conditions and is limited to shallow depths by the access requirements.

Destructive sampling, involving the centrifuge extraction of pore water from undisturbed drill cores, is effective in the unsaturated zone. This is an established field method for the study of water quality in both unsaturated and saturated zones (Edmunds and Bath, 1976; Kinniburgh and Miles, 1983) and gives much better control over sample depth and greater confidence about sample origin than other unsaturated zone methods. Repeat coring through the unsaturated zone has been employed as a tool in research studies to observe the downward migration of, for example, nitrate and tritium peaks in the unsaturated zone (Geake and Foster, 1989) in studies of both diffuse pollution and recharge. It is, however, expensive and is not suitable for frequent, routine monitoring. The method also presents additional problems in the sampling of some classes of pollutant. In the case of organic pollutants, pore-water extraction by centrifugation is unlikely to be effective because of the loss of sample by volatilisation and because, particularly in the case of pesticides, insufficient sample volume is produced to permit laboratory analysis to the detection limits required. Methods based on solvent extraction have been used on soils and are being developed, but as yet are far from being applicable to routine monitoring.

Indirect methods of groundwater quality monitoring

In some circumstances, for specific objectives and particular variables, indirect methods of monitoring groundwater quality may be employed. The use of fluid conductivity logging in observation wells to monitor the three dimensional development of saline intrusion is an example. The use of geophysical methods is limited to situations where groundwater quality differences are sufficiently strong to cause physical contrasts. Thus, the measurement of ground resistivity by surface geophysics may be used in some hydrogeological situations to assess the lateral spread of salinity through an aquifer. For point source pollution incidents involving volatile hydrocarbons, the use of soil gas detection methods may provide a cost-effective means of studying the development of a contaminant plume. Both of these indirect methods depend, as all such methods do, on adequate control being provided from some direct sampling by investigation drilling and construction of permanent monitoring points.

9.5.7 Field operations

Although the questions of where, when and what to sample for are determined in the design of an assessment programme, and the question of how to sample is largely covered by consideration of the design of observation boreholes and sampling methods in relation to the assessment objectives, the field operations of the monitoring activities require careful planning, execution, supervision and evaluation.

Where purpose-built observation wells are required for the programme, as in the case of emergency or impact surveys of a point source, careful supervision of construction based on a clearly defined drilling contract is required. The contract must have detailed specifications for well design, drilling method, well completion materials and the methods for, and cleaning of, the well. It is particularly essential that no drilling fluid is used that will continue to affect the groundwater quality after the well has been cleaned out in preparation for sampling. Very clear and detailed records must be kept by contractor and supervisor to ensure that there are no subsequent uncertainties about the hydraulic conditions in the well, particularly the positions of open or slotted sections, that might compromise the representativeness of samples.

In both preliminary surveys and subsequent establishment of a permanent monitoring system, the construction of observation wells may provide data that are vital to the understanding of hydrogeological conditions in the area of interest. These data range from geological descriptions (logs) of the strata penetrated to detailed pumping tests to determine the hydraulic characteristics of the aquifer. Every effort should be made to collect and record these data in the most cost-effective way. In some cases of detailed local monitoring of point source pollution, especially where aquifer restoration measures are involved or contemplated, the construction of dedicated monitoring wells may be a major expense running to many thousands of US dollars. The proper preparation and supervision of drilling contracts is essential to ensure that the money is well spent and, where appropriate, to satisfy legal or case preparation requirements. Strict requirements in relation to data collection during well construction may also apply in many cases of research investigations. A comprehensive general description of well construction and hydrogeological data collection is given by Driscoll (1986), and useful discussions of well construction and data collection related specifically to monitoring are included in Everett (1980), Scalf *et al.* (1981), Barcelona *et al.* (1985) and Neilsen (1991).

Field operations also include the collection of hydrogeological data during the lifetime of the monitoring system to assist in the interpretation of the analytical results. This comprises principally the routine measurement of groundwater levels in observation wells (Table 9.14), either continuously by means of autographic recorders or more often by regular spot measurements. These data are required to provide an indication of the persistence of ground-water flow patterns. For large-scale regional monitoring, the flow system (Figure 9.7) is unlikely to change seasonally or in response to effects such as groundwater abstraction sufficiently to affect the interpretation of water quality data. A rare instance where this may not be the case is the recovery of groundwater levels resulting from declining abstraction (see section 9.4.10). On a local scale, however, changes in water table or piezometric level could be very important in the interpretation of monitoring results. This applies particularly to saline intrusion, and salinity resulting from irrigation, where the cause of the problem and its control are inextricably linked to water level changes. In emergency and impact surveys at a very local scale, modified abstraction, including the closure of polluted or threatened high-discharge public supply sources could have a significant effect on local groundwater flow patterns. Failure to determine properly these changes by water level measurements could lead to erroneous interpretation of the water quality data. In extreme cases, the results of recording changes in groundwater flow patterns could lead to a requirement for additional sampling stations.

Table 9.14 Hydrogeological data requirements during monitoring

Data	Frequency	Source	Application
Water levels	Continuous, weekly, monthly	Observation wells	Monitor changes in groundwater flow regime
Pump discharge	Rates, aggregates	Production wells (Public supply, irrigation or industrial)	Mass balance in point source pollution Monitoring saline intrusion
Rainfall, evaporation	Daily, weekly, monthly	Climate stations, meteorological department	Calculate infiltration for diffuse pollution inputs Estimate leachate volume from landfills

Within the assessment of some types of point source pollution, collection of data on the abstraction rate of polluted wells may be required (Table 9.14). Aggregate discharge data and regular pollutant concentrations may permit a simple mass balance estimation of the amount of pollutant removed by pumping.

Hydrogeological data requirements are somewhat different in relation to the assessment of diffuse pollution. Interpretation of nitrate concentrations in the unsaturated zone requires a knowledge of infiltration rates so that inputs of nitrogen by leaching from the soil can be estimated (Foster and Hirata, 1988) and related to nitrogen application rates in cultivation practices. Climatic and hydrological data are also essential for the estimation of leachate volumes from solid waste disposal sites (Everett, 1980; Foster and Hirata, 1988).

When taking samples from pump discharge, a further consideration is to ensure that the sample is representative of hydrochemical conditions in the aquifer, rather than those in the well. In the case of public supply or irrigation wells which are operating continuously (or at least for long periods) at high discharge rates, it is relatively easy to collect a sample after a sufficiently long period of pumping to be certain that the water has been drawn from the aquifer rather than from within the well itself. If, however, specialist sampling pumps with low discharge rates are used to sample non-pumping observation wells, then “purging” the well of standing water is required before sampling. Some workers have recommended "rule-of-thumb" purging guidelines based on removal of three, five or ten well-volumes. Others (Gibb *et al.,* 1981; Barcelona *et al.,* 1985) have recommended calculation of the purging requirement from the geometry and hydraulic performance of the well. In either case, the well purging operation should be verified by the inline field measurement of parameters such as Eh, pH, temperature and electrical conductivity until reasonable stability indicates that the required high proportion of aquifer water is being drawn.

Choice of field or laboratory analysis

The choice between carrying out analyses in the field or in the laboratory is part of the assessment design process. This choice is based on the stability of the selected variables, the difficulties of transporting samples, their ease of analysis and several logistical and economic considerations. Some variables (Eh, pH, dissolved oxygen) can only be properly analysed in the field, either in-line or on-site. Other variables lend themselves to easy measurement in the field as general indicators of particular aspects of water quality, such as electrical conductivity used as a guide to overall salinity. The choice between field and laboratory also involves practical considerations. The area to be studied may be remote from a suitable laboratory, in which case field analyses become essential. This applies particularly to bacteriological monitoring. The level of skill and training of the staff who will carry out the field sampling, the financial resources and transport arrangements also affect how much of the analytical work can be done in the field.

9.5.8 Data treatment

Groundwater assessment systems, for whatever objective, will generate an enormous quantity of data. Often, in the past, much less thought and research effort have gone into the vital steps in the system beyond laboratory analysis (see Figure 2.2). The result is poorly archived and inaccessible data, a major and common weakness in assessment programmes (Wilkinson and Edworthy, 1981). In the data treatment phase of groundwater quality assessment, water quality and hydrogeological data need to be converted into information that can be interpreted and used. The conversion of data into useful information (involving storage, retrieval and analysis) is described in Chapter 10.

9.5.9 Interpretation and reporting

Interpretation and reporting are parts of the process of converting data into information, and are closely linked with data treatment, often overlapping. In the case of emergency surveys and operational and early warning surveillance in relation to water used for public supplies, the analytical results must be interpreted and passed on to managers almost immediately in order to protect consumers, if necessary by shutting down the supply. In many, but not all, cases this is possible because the monitoring agency and the managing agency are the same organisation. In these circumstances, the amount of statistical treatment and interpretation of the data is minimal. In contrast, for regional background and trend monitoring and research studies, interpretation is often dependent on the accumulation of a large body of data and, therefore, occurs some time after data collection (see Table 2.2).

For most types of monitoring, the product is information presented in a form that permits an evaluation of groundwater quality with respect to the original objectives, allowing management decisions to be made (see Figure 2.2). These decisions can often be subjective, following an evaluation of the monitoring information and taking account of financial, operational and even political factors. Decisions relate both to water use management and to pollution control (see Figure 2.2). The former could include ceasing or modifying abstraction from polluted wells, blending water from more than one source before putting it into supply, installing water treatment plants, permanently abandoning wells and seeking alternative groundwater or surface water sources, or changing the use to which the abstracted water is put. The latter could include, for example, establishment of protection zones of modified land-use practice around public supply wells, closure and/or removal of point sources of pollution, improved sanitary protection measures at the well itself, and remedial measures to remove pollutants from an aquifer.

The reporting stage of assessment also includes evaluation of the programme itself in relation to the originally stated objectives. The evaluation applies to the whole programme and changes to all steps must be considered, including whether the area needs to be re-defined, whether additional hydrogeological data are required and how they should be collected, whether the monitoring system needs to be redesigned in any way, and whether improvements are required in any of the subsequent steps. In time, the objectives of the assessment need to be reviewed to ensure that they remain relevant to changing economic circumstances and development, and the shift in emphasis in water quality issues that this will bring.

9.6. Examples of groundwater assessment

The outline of the assessment strategy given above is, of necessity, generalised to encompass all types and stages of assessment. The interested reader, charged with the task of establishing an assessment programme or reviewing an existing system, may still find the possible permutations involved somewhat bewildering. The best way to overcome this is to summarise briefly a number of illustrative examples of assessment programmes, covering some of the most important types of assessment and water quality issues. Unfortunately, however, examples of good design, execution, interpretation and presentation of groundwater assessment which follow the general strategy outlined above are difficult to find. Most assessment of groundwater quality has been developed on a much more *ad hoc* basis. Nevertheless, the following examples illustrate how a general overall strategy has been applied to the establishment of groundwater assessments to meet specific objectives related to some of the principal water quality issues described in section 9.4.

9.6.1 Diffuse source pollution from agriculture

South Dade County in Florida is one of the few localities in the continental USA in which fruit and vegetables are grown during the winter (Waller and Howie, 1988). The soils of the area are thin and not naturally fertile. They are irrigated and require large inputs of fertilisers and pesticides to maintain profitable crop production. One method that is used to increase both nutrient and moisture retention capacity is the application to the soil of sludge from treated domestic wastewater. This intensively cultivated agricultural area overlies permeable limestone of the unconfined Biscayne aquifer, which is the sole source of potable and irrigation water in South Dade County. The local and State regulatory agencies became concerned about the possible impact of the agricultural activities on groundwater quality, and in 1985 a study was initiated.

The objective of the assessment was thus well defined from the outset. The area to be assessed comprised South Dade County, with particular emphasis on the 30,000 ha of cultivated land. About 85 per cent of the area is underlain by Rockdale soil composed of crushed limestone and organic debris and the remaining 15 per cent is underlain by the freshwater deposits of the Perrine marl. The hydrogeology of the area is relatively well known. Water levels are shallow and the unsaturated zone varies from < 1 m to 4.5 m thick. The highly fissured Biscayne limestone has measured hydraulic conductivities of > 3,000 m day^{-1}, and the vertical permeability greatly exceeds the horizontal permeability. The aquifer in the area under investigation is 18-37 m thick.

Table 9.15 Variables and sampling methods selected, South Dade County, Florida, USA

Variable group	Variables	Sampling method
Physical properties	Conductivity, alkalinity, pH	Tygon® tube and centrifugal pump
Nutrients	Nitrate, phosphate, dissolved organic carbon	Tygon tube and centrifugal pump, Teflon® bailer
Micronutrients	Copper, iron, magnesium, manganese, potassium, zinc	Tygon tube and centrifugal pump
Sludge-derived contaminants	Arsenic, cadmium, chloride, chromium, lead, mercury, nickel, sodium	Tygon tube and centrifugal pump
Organic constituents	Acid extractable and base-neutral compounds, volatile organic compounds, chlorophenoxy herbicides, organophosphorus insecticides, organochlorine compounds	Teflon tube, peristaltic pump and vacuum chamber Teflon bailer Teflon tube, peristaltic pump and vacuum chamber

Source: Waller and Howie, 1988

The thickness of the unsaturated zone, variations in lithology and water levels, the hydraulic gradient and the flow directions determined the locations and depths of the wells. The selected network of stations comprised two types. Baseline stations consisted of single monitoring wells to determine the groundwater quality in the upper part of the saturated zone over the whole area. Two of the baseline sites were selected to provide the regional background quality as they were hydraulically up gradient of the agricultural development area. At five test fields, representative of the two soil types and a range of crops, multiple-depth clusters of wells were installed for intensive monitoring of water quality. Details of the construction of the wells are given in Waller and Howie (1988).

The choice of variables was based on historical water quality information, theoretical geochemistry, site inspections, county and state agricultural records and discussion with the local agricultural community. Appropriate sampling methods were then selected (Table 9.15), and a quality assurance programme was established. Agricultural practices are dictated by the climatic regime, therefore, a schedule of eight samples per year was chosen to cover the onset of the summer rains, rising water table, high water table and low water table, and to co-ordinate with agrochemical applications. Preliminary results indicated that the long history of agricultural activity had little effect on groundwater quality. The few instances of high concentrations of variables were related more to the storage and disposal of agricultural chemicals rather than their application to the fields.

9.6.2 Waste disposal to landfill

At Borden, approximately 80 km north west of Toronto, Canada, an abandoned landfill on an unconfined aquifer of fluvio-glacial sand has been the focus of intensive monitoring and related groundwater studies (Cherry, 1983). Landfilling operations took place from 1940 to 1976, with little record of the quantities or types of waste brought to the site. By the end of this period, the landfill covered 5.4 ha, with a thickness of 5 to 10 m. Investigation drilling in the fill material established that it mostly consisted of ash, wood and construction debris, with lesser amounts of domestic and commercial food wastes.

A range of groundwater monitoring devices has been used at the Borden site (Cherry, 1983). Hydrogeological studies commenced in 1974 with the installation of piezometers, open to the aquifer over a single, relatively short interval, and water table standpipes, open over a single but somewhat longer section. Large numbers of these were installed, and provided information on groundwater levels, flow directions and the lateral extent of the leachate plume. A broad, fan-shaped plume covered an area of about 39 ha and extended about 700 m north of the landfill in the direction of groundwater flow (Figure 9.26). The landfill had caused deterioration of groundwater quality in the shallow, fluvio-glacial sand aquifer, which was not used for water supply. The zone of contamination was separated by a clay and silt layer from a deeper aquifer which was used for water supplies. It appeared unlikely that the deeper aquifer would be affected, but the long-term impact on groundwater quality could not be predicted from this reconnaissance study, and it was decided to close the landfill. A long-term programme of assessment and research was initiated at the abandoned landfill site.

Within this extended programme, additional piezometers and standpipes were installed, supplemented by multi-level groundwater sampling points of three types (Cherry, 1983) to enable regular water samples to be taken from many depths within the aquifer. The natural groundwater quality at the site is characterised by very low overall mineralisation with, for example, chloride concentrations < 10 mg l^{-1}, and this stable, relatively non-reactive inorganic constituent was used as a convenient indicator of the extent of contamination. Figure 9.27 shows the location of sampling points and chloride distribution along a north-south section through the centre of the plume. The greatest chloride concentrations occurred in the middle of the aquifer, 50 to 150 m down gradient of the landfill, and the plume extended to the base of the shallow aquifer. Water samples taken from within the underlying clay had chloride concentrations at the background value of 10 mg l^{-1}, confirming that the clay acts as an impermeable barrier protecting the underlying deeper aquifer.

Figure 9.26 Lateral extent of contaminant plume based on chloride contours at the Borden landfill site, Canada (After Cherry, 1983)

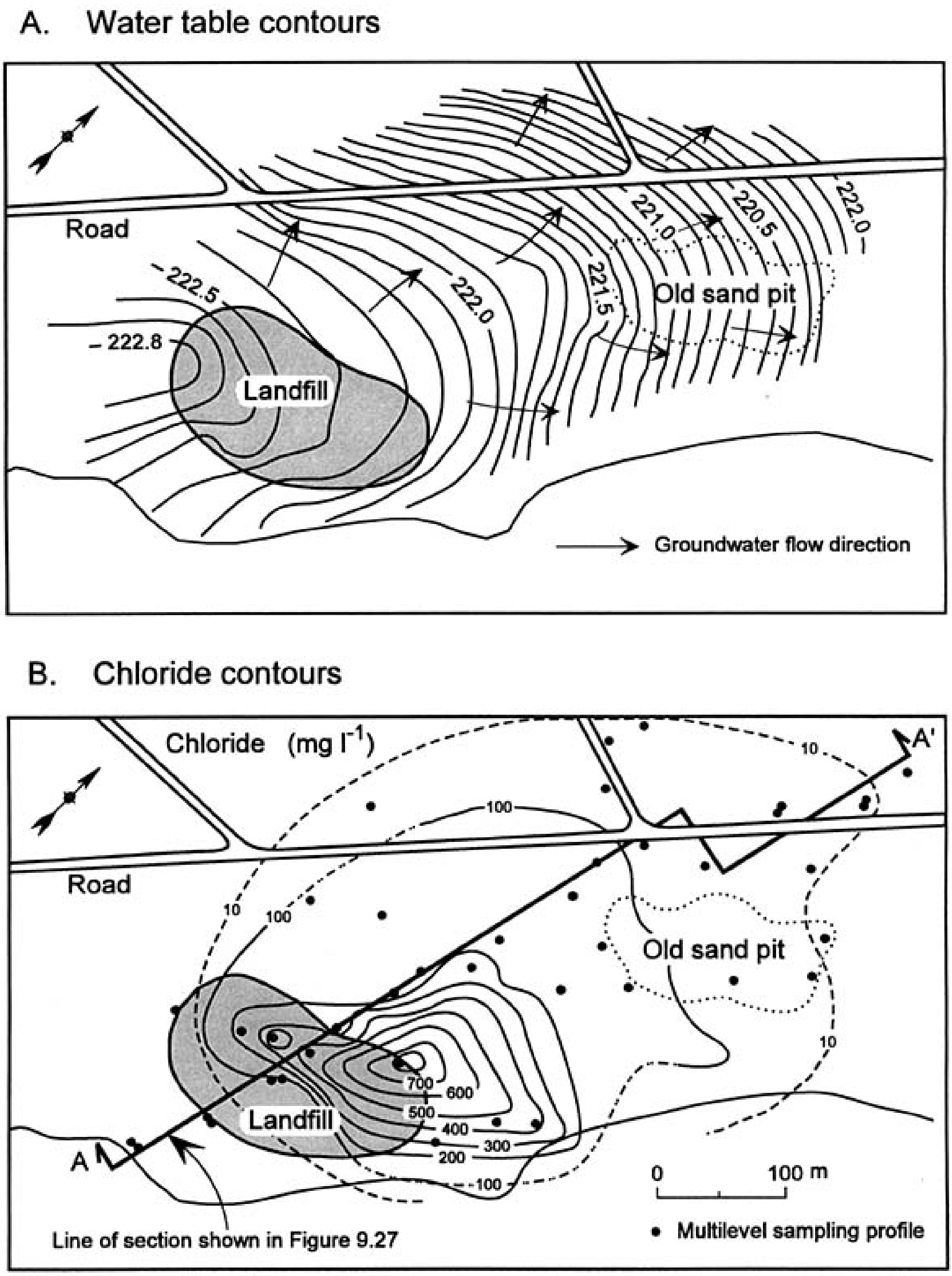
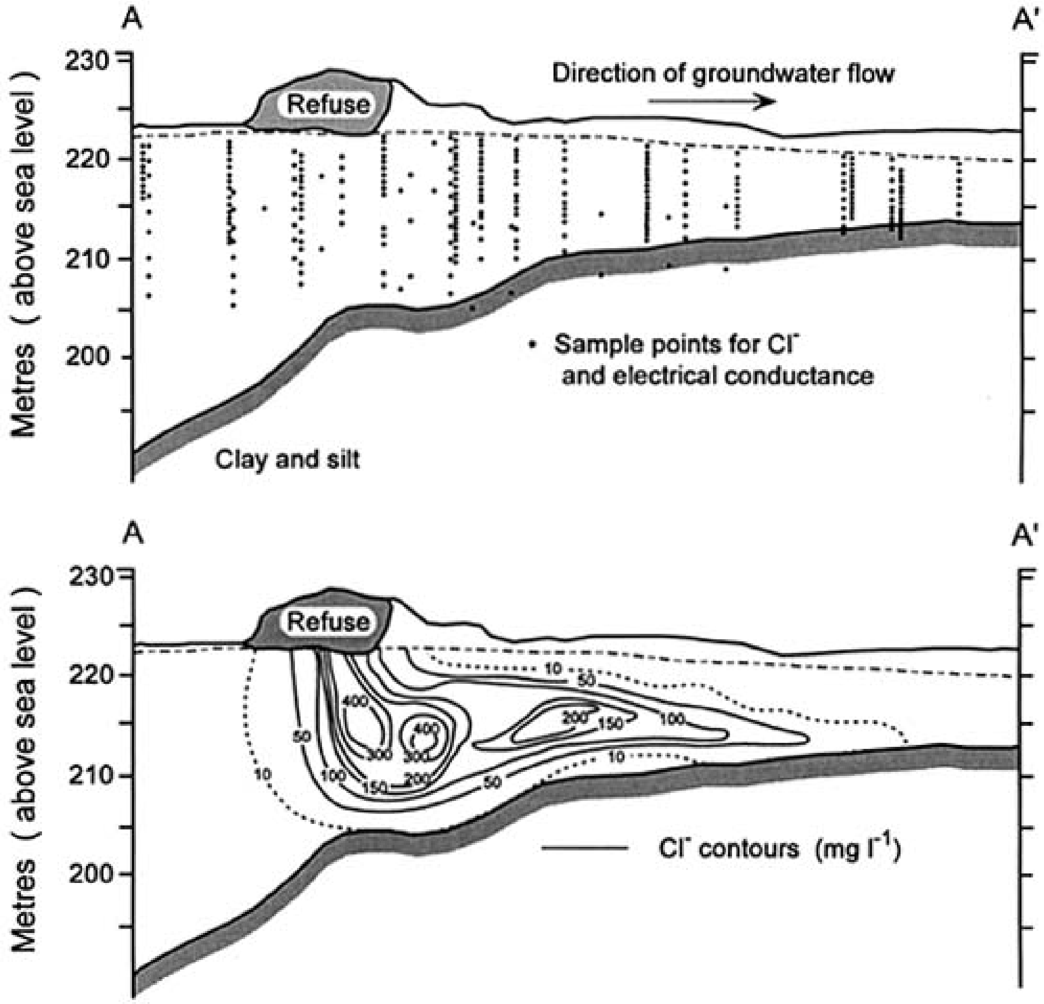


Figure 9.27 Multi-level sampling points and vertical variation of chloride in the plume at the Borden landfill site, Canada. Line of section A - A' shown in Figure 9.26. (After Cherry, 1983)



Provision of such a dense network of sampling points permitted detailed investigation of the three-dimensional distribution of permeability and hydraulic head in the aquifer, of the seasonal modifications to groundwater flow patterns, and of dispersion, together with their effects on contaminant transport and plume characteristics (Cherry, 1983). As a result of the wide range of research activities there, the Borden site is probably the most intensively monitored of all landfills, even though the impact on water supplies is minimal, and has been the model for many such monitoring exercises.

9.6.3 Bacteriological assessment of rural water supplies

The World Health Organization's (WHO) *Guidelines for Drinking-Water Quality. Volume 3* (WHO, 1985) has emphasised the importance of microbiological assessment of the risk of pollution of drinking water as an important, world-wide aspect of water quality control. Within the United Nations Water Decade much effort was directed towards the construction of new supplies, and only recently has more attention been given to the investigation and protection of the installations which supply drinking water. A risk assessment methodology has been developed by Lloyd and Helmer (1991) which aims: (i) to test and evaluate the approach described in the WHO guidelines (WHO, 1985), (ii) to provide a scientific basis for strategies of remedial action, and (iii) to develop a monitoring infrastructure that will ensure drinking water supplies are kept under continuous public health assessment. This risk assessment has been developed as a series of steps in which the causes and sources of pollution are identified and remedial strategies proposed. The principal steps are shown in Figure 9.28, and similarities to Figure 2.2 can be noted. The overall strategy has been developed from pilot studies in Peru, Indonesia and Zambia. In Indonesia, the Gunung Kidul district was chosen for a pilot project (Lloyd and Suyati, 1989). The population is 702,000, about 80 per cent of which is described as living in rural areas. An inventory of water supplies in the district (Figure 9.28: step 1) recorded over 21,000 public installations.

The next step was the planning of inspection visits, and priority was given to the facilities serving the largest populations. A general guide to the suggested frequency of such visits is given in Table 9.16. Sanitary inspection forms were designed for each type of installation listed in the inventory. The forms were based on models included in WHO (1976), and provided for:

- identification of all potential sources of contamination,
- quantification of the level of risk to each facility,
- a graphical means of explaining the risks to the users, and
- guidance to the user on remedial action required.

The check list on the form is completed by the sanitary worker, with the help of the operator or community representative, to give a sanitary inspection risk score, and identify the potential sources of pollution (Figure 9.28: step 4). Details of the forms and risk assessment are given in Lloyd and Helmer (1991). At the same time, a water sample was collected for bacteriological analysis (step 3/4) to confirm whether pollution was occurring (step 5).

The analysis serves to detect actual pollution of the groundwater and the sanitary survey identifies possible sources and establishes the risk of pollution. The two activities are thus complementary, and can be combined (step 7) in a graphical presentation (Figure 9.29) which identifies the installations that most urgently require remedial action (step 8). The graphical method also provides for supervision of field staff and evaluation of the method. Where gross faecal contamination does not correlate with a high risk determined from the sanitary inspection, then a need for urgent re-sampling is indicated (Figure 9.29). A broad spread of points from lower left to upper right of the graph for all types of water supply facilities indicates general agreement between sanitary inspections and bacteriological analyses in identifying polluted sources. This is confirmation of the robustness of the assessment system, but should not be seen as a reason to dispense with bacteriological analysis. A principal objective of the original pilot project was to establish an effective system that could be extended to other provinces over a period of years. This is now being done (steps 9 and 10), and it is anticipated that additional field experience will result in improvements to the basic methodology (Lloyd and Suyati, 1989).

Figure 9.28 Steps used in establishing a microbiological monitoring programme for drinking water supply wells (After Lloyd and Helmer, 1989)

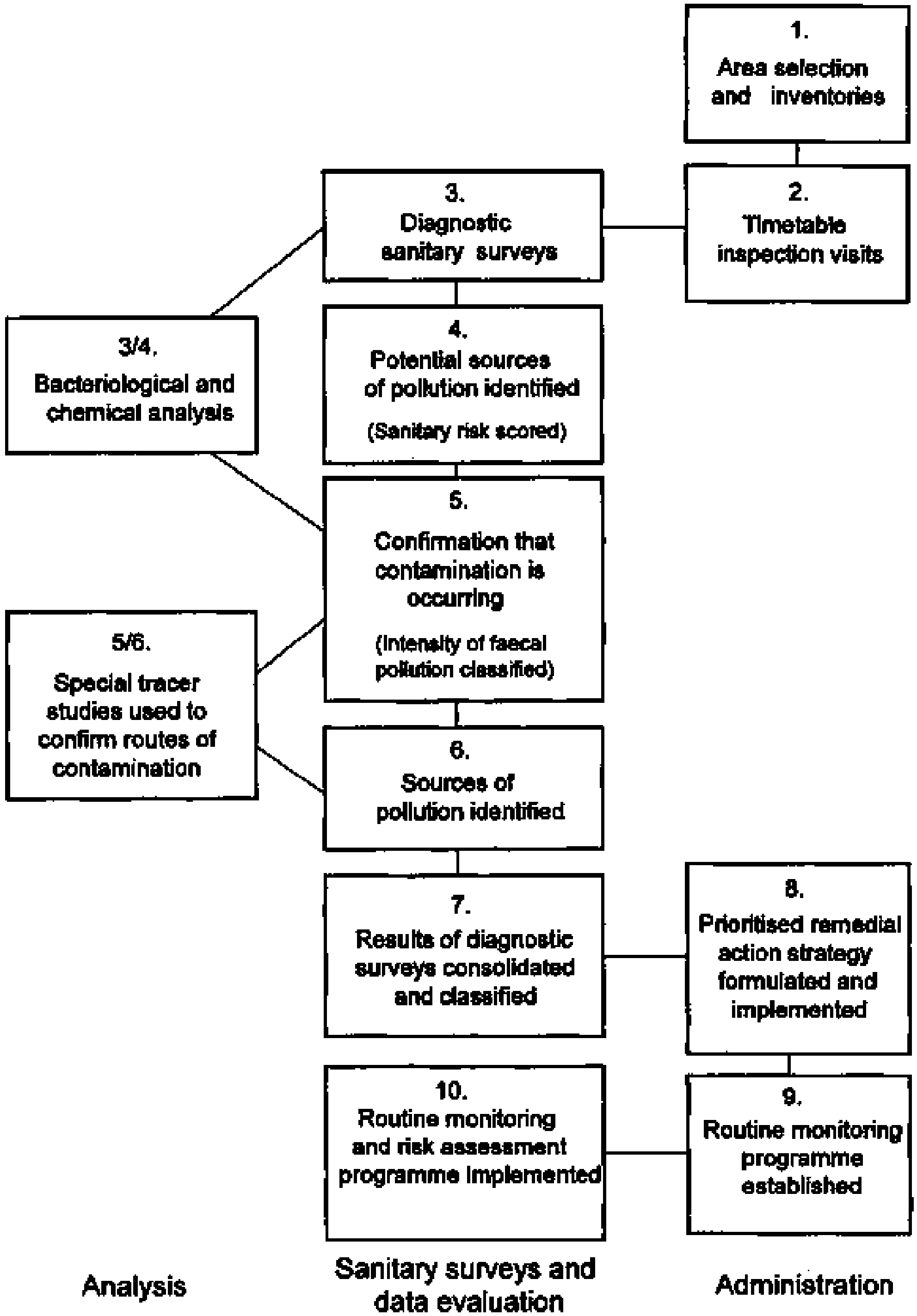


Table 9.16 Suggested frequency of groundwater source surveillance for bacteriological pollution

Population served by source	Maximum interval between sanitary inspections	Maximum interval between bacteriological samples
> 100,000	1 year	1 day
50,000 to 100,000	1 year	4 days
20,000 to 50,000	3 years	2 weeks
5,000 to 20,000	3-5 years	1 month
< 5,000 Community dug wells	Initial, then as situation demands	As situation demands
Deep and shallow tubewells	Initial, then as situation demands	As situation demands
Springs and small piped supplies from tubewells	Initial, and every 5 years, or as situation demands	As situation demands

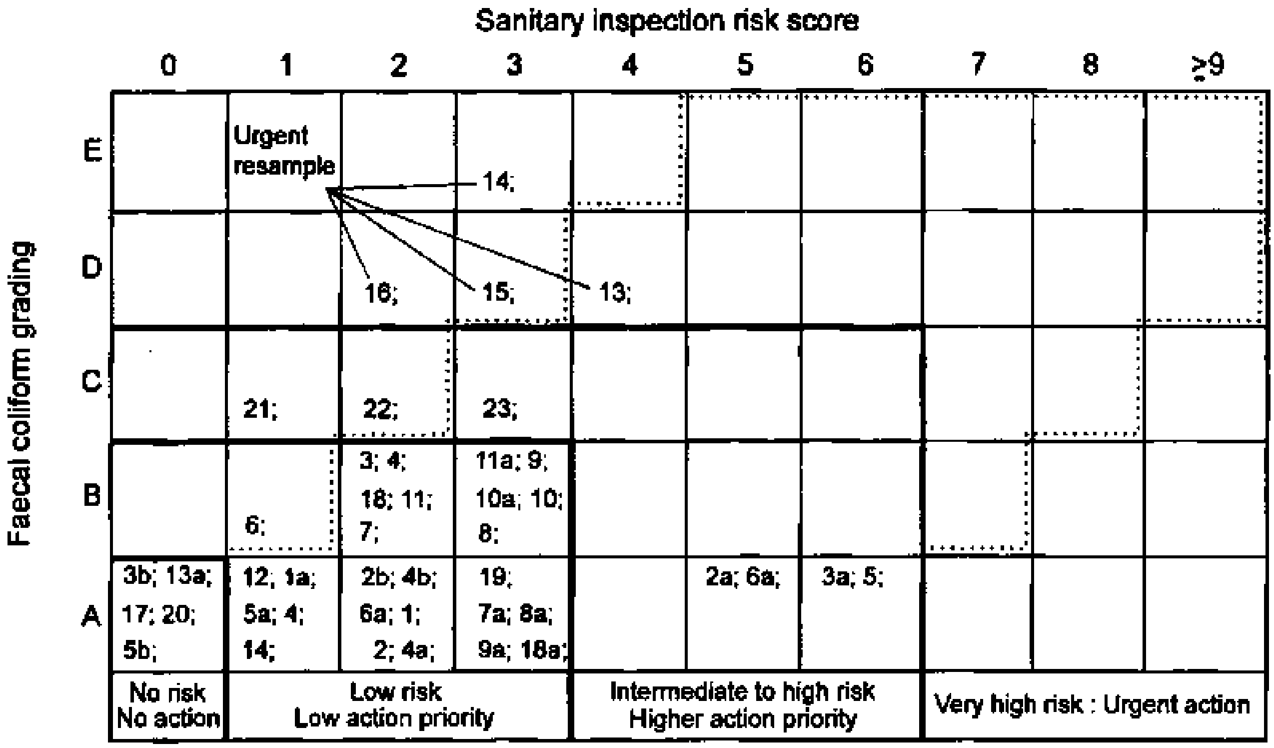
Source: Lloyd and Helmer, 1991

9.6.4 Trend monitoring

A state groundwater quality assessment programme was initially developed in California in 1974. Some 24 priority groundwater basins from a total of about 500 were designated “priority one” on the basis of population, water use, existing water quality problems, available groundwater resources and alternative sources of supply. Of the remaining basins, 180 were classed as “priority two” and 300 as “priority three” (Canter *et al.,* 1987).

Development of the assessment system began with a pilot programme in four basins. An inventory of existing wells was carried out and annual sampling was initiated by the State Department of Water Resources.

Figure 9.29 Risk analysis from sanitary inspection and bacteriological examination of a handpumped deep tubewell used for drinking water supplies (After Lloyd and Helmer, 1991)



Survey period: June - August 1988
Study area: Java, Gunung Kidul
Code:
Number = sample sequence
Letters = District, sub-districts

Concentrations of major ions were found to be comparable to historical values. Trace elements and nutrients were added to the selected variables. There was little historical data for these, and some exceeded the state drinking water standards. A structured programme for the establishment of monitoring in the remaining priority basins is currently under way. This comprises an inventory of existing water quality assessment programmes, including details of each well, and design of an improved network, using existing wells where possible, and selecting variables and monitoring frequency appropriate to each basin.

9.6.5 National groundwater quality assessment programme in former Czechoslovakia

The purpose of the former Czechoslovak national groundwater quality assessment programme was to collect background data as a baseline for evaluating the current state and forecasting the changes and trends in ground-water quality due to natural processes and human impacts. The national assessment system was in operation from 1985, and comprised 180 boreholes in shallow aquifers, 98 boreholes in deeper aquifers and 44 springs. The monitoring stations in the deeper aquifers consisted of two or three boreholes, each completed so as to monitor separately individual aquifers in complex sedimentary sequences. The network provided an average monitoring area per station of about 250 km^2. Twenty-two variables for which national drinking water standards were given were included in the basic set and, occasionally, selected organic pollutants were sampled and analysed. Sampling frequency was twice yearly for the shallow aquifers and springs, and annually for the deeper ones. As a result of political changes within former Czechoslovakia, this programme may have been modified recently.

When the programme was established, three categories of monitoring station were designated according to the GEMS classification (Meybeck, 1985). Regional monitoring programmes were established in the middle Elbe region of Bohemia and the Danube lowlands of Slovakia. The former was in operation from 1980 over an area of 3,000 km^2 of intensively cultivated farm land. The regional network in Bohemia was aimed at determining the impact of agriculture on groundwater quality. The areal and vertical distribution of nitrate and other variables were being studied. At selected stations, the sampling frequency was 6-12 times per year; at the remainder it was 2-4 times. The impact on groundwater quality from increasing use of inorganic nitrogen fertiliser, organic fertiliser, farm slurries and from septic tanks was well demonstrated by this programme (Pekney *et al.,* 1989).

Intensive monitoring at a few selected pilot stations was subsequently established to study the evolution of groundwater quality and support the regional programme. Nests of boreholes (Figure 9.24b) with short screens at different depths, were installed in shallow alluvial deposits overlying impermeable bedrock. The unsaturated zone was also being sampled.

9.7. Conclusions and recommendations

The discussion in the preceding sections provides an indication of the broadness of scope of groundwater assessment, encompassing a range of quality issues, purposes, types, scales and levels of assessment. The need to define the objectives at the very beginning has been consistently emphasised, since this will determine the variables to be analysed, the scope of the monitoring network and the financial resources required. Water quality assessment should be seen in the wider context of the management of water resources, comprising both quality and quantity aspects. Unless there is a legal and administrative framework, together with an institutional and financial commitment to appropriate follow-up action, the usefulness of the information obtained from assessment is severely limited. Wilkinson and Edworthy (1981) have identified four main reasons why groundwater assessment systems yield inadequate information:

- The objectives of the assessment were not properly defined.
- The system was installed with insufficient hydrogeological knowledge.
- There was inadequate planning of sample collection, handling, storage and analysis.
- Data were poorly archived.

Despite these inadequacies, data are often used for long-term predictions of water quality and as a basis for decisions on capital expenditure. The same is often true for surface water assessment programmes, and it is hoped that the discussion in this and the preceding chapters will assist in putting assessment on a firmer technical basis, so that sound quality management decisions can be made from relevant and reliable information.

The particular problems of monitoring groundwater bodies relate to the general complexity of flow and contaminant transport in both saturated and unsaturated zones. This is a product of the physical, chemical and biological interactions between soil, rock and water, and the generally slow movement of groundwater compared to other water bodies. The outcome is complex lateral and vertical variations in water quality which are difficult to sample properly and which present major difficulties for interpretation. The need for adequate hydrogeological expertise throughout the assessment and water quality management process cannot be over-emphasised.

To meet assessment objectives related primarily to water use, sampling of pump discharges is usually the simplest and most economical method and remains generally adequate, although there is often scope for improvement in collection and handling of samples and in the choice of variables. However, sampling pump discharges has severe limitations where the objectives are related either to the provision of early warning of a pollution threat to supply wells from diffuse or point sources, or to defining the precise distribution of pollution in an aquifer. Production well samples may be completely inadequate for this purpose because there is insufficient control over the sample depth combined with possible loss of unstable variables. Purpose-built observation wells, open over a selected, short interval and sampled by a device appropriate to the chosen variables are required.

As a result of the generally slow movement of groundwater compared to surface water, once a groundwater body has become polluted it may remain affected for a very long time. Serious groundwater pollution can result in the use of an important aquifer being discontinued, at least locally, for decades. The cost of restoration measures may run into tens of millions of US dollars. Considerable research effort is now devoted to groundwater quality problems, including assessment, and the results have been used in the preparation of this guidebook.

Subjects requiring further assessment effort include sampling for volatile organic compounds in groundwater. Sampling for pesticides occurring at very low concentrations in the unsaturated zone presents particular difficulties. However, such samples are likely to become of increasing importance in first the study, and then the monitoring, of possible diffuse sources of agricultural pesticides. Other future groundwater quality assessment requirements can be anticipated in relation to increasing wastewater re-use for irrigation, problems such as methane generation and other pollutants from reclaimed urban industrial sites, and from rising groundwater levels.

9.8. References

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