

## V.4

# Evaluating the susceptibility of aquifers to pollution

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### V.4.1. Introduction

The largest reservoirs on earth are formed by groundwater, the oceans, and the atmosphere; for the most part, the groundwater and the atmosphere are transient depots and the oceans are final depots for pollutants.

On many continents, groundwater is the only resource available for irrigation, for manufacturing industrial products, for producing energy, and also for drinking water. This is especially true for semi-arid and arid regions and all regions with extremely high permeable rocks (e.g. soluble rocks and Quaternary gravels).

A review of the average distribution of water on the continents is given in Table V.4.1. Comparing the groundwater quantities with the annual discharge from the continents (Table V.4.1) and assuming that all of the groundwater participates evenly in the water cycle, the minimum turnover time would be 180 years. In other words, the average groundwater outflows on the continents would not yet show significant effects of the pollutants released since the start of the industrial age. Although this holds true for certain regions of the continents, in general this is not the case. Groundwater responds to instantaneous, short-term and to long-term contamination impacts as well.

The causes for this are diverse and lie in the sensitivity with which the groundwater in the aquifer systems reacts to any substance input in undesired concentrations (pollution). These reaction mechanisms have their roots

- in the intensity, with which the groundwater is incorporated in the water cycle and used by people,
- in the transport potential of the different discharge components,
- in the hydrodynamic properties of aquifers and aquifer systems, and
- in microbiological activities in the aquifers.

### V.4.2. Importance of groundwater

The importance of groundwater becomes obvious, if one looks at its usage in the different continents (Table V.4.2); surface water usage is also included in this account. Table V.4.2 shows that in the developing countries the demand for irrigation water is the highest. In the industrialized nations, however, the demand for water for energy and industrial production

*Table V.4.1.* Review of the average distribution of waters on the continents. Ice distribution is not considered.

Groundwater	8,000,000 km <sup>3</sup>
Lake water	226,000 km <sup>3</sup>
Soil moisture	62,000 km <sup>3</sup>
Air humidity	15,000 km <sup>3</sup>
River water	1000 km <sup>3</sup>
Discharge from the continents	45,000 km <sup>3</sup>

is the highest. Worldwide, the water use for household purposes, including drinking water, is the lowest for all countries. The average total water demand of about 3000 km<sup>3</sup>/year (1990) amounts to only 6.6%, the drinking water demand to only 0.5% of the average discharge from the continents. Of course, these statistics do not incorporate the uneven distribution of renewable water resources in the different climatic and geological zones on earth. It indicates, however, that the water quantities as an average will actually not be a predominant problem.

Demands on groundwater quality are the highest for drinking water as well as for waters for food processing and much lower for all the other water usages. Water quality is primarily dependent upon geogenic factors and nowadays is also strongly influenced by anthropogenic factors. Worldwide, this has led to a decrease of water quality on the continents, especially in the urban areas and in the coastal regions. As a result, actual availability of groundwater mostly depends less from renewable water, but from water quality.

In many regions of the world, the high demand for water for non-household uses has drawn away attention from maintaining groundwater quality and that way led to an unequilibrated competition between maintaining the natural ecological elements and functions for self-purification processes and water usage for production purposes.

By nature, groundwater is poor in nutrients that promote self-purification processes in surface waters. Due to anthropogenic impacts, groundwater can suffer considerable losses

*Table V.4.2.* Water demand statistics for individual regions in km<sup>3</sup> (1990). Till 1995 the world water demand has increased to about 5000 km<sup>3</sup>.

Regions	Irrigation	Energy production	Industrial production	Household use	Total
Asia	1400	68	31	98	1597
Africa	61	11	4	12	88
Australia	13	8	6	2	29
South America	35	6	4	11	56
North America	205	232	77	38	552
Europe	116	176	184	40	516
Total	1830	501	306	201	2838

in quality. Generally, these appear only slowly underground and once begun, these processes require much more time to fade away than they needed to build up.

Today's increased technical possibilities for using groundwater, the prevailing production-oriented usage of groundwater, and the lack of sewage water treatment in many countries have led to a reduction in the amount of water in some developing countries available for household use and food processing. Therefore, it has become crucial to evaluate the susceptibility of the groundwater reservoirs with regards to its vulnerability to contamination, when developing, exploiting, and managing the aquifers.

### **V.4.3. Dynamics of groundwater within the water cycle**

Only a small portion of the existing groundwater (Table V.4.1) is directly recharged by precipitation (infiltration) or undergoes indirect recharge (bank filtration, artificial groundwater recharge). The majority of the groundwater acts as a long-term reservoir. Only the recharged portion of groundwater is available for management purposes and contributes simultaneously to important ecological functions of the surface water. Any management of the long-term resources that started recently in developing and industrialized countries presents on a long run irretrievable groundwater consumption.

Groundwater recharge occurs in all regions of the continents; the desert regions receive very small amounts ( $<5$  mm/a) very irregularly, the tropical regions receive an annual amount of less than 150 mm and the humid regions an annual amount of less than 1000 mm. Recent investigations have shown that groundwater recharge occurs even in permafrost regions, albeit only little.

However, this groundwater recharge, which can also transport pollutants into the underground, does not flow through the entire thickness of the aquifer. Instead, it mainly flows through the aquifer system near the surface (Seiler and Lindner, 1995).

The groundwater flows in aquifers and each aquifer has its individual hydraulic properties. In unconsolidated aquifers, the hydraulic conductivity and porosity generally decrease with increasing depth and statistically, the hydraulic conductivity of the fissures decreases too with increasing depth in consolidated, fissured aquifers; only in areas with deep reaching tectonic faults do the hydraulic conductivity of the fissures often reach a depth of several thousand meters.

The bedding of aquifers and the general and discontinuous decrease of the hydraulic conductivities with depth are the major reasons that the groundwater recharge is not distributed equally among all the aquifers.

To present the quantitative turnover of groundwater recharge in the individual sections of the aquifer systems (Fig. V.4.1), the groundwater movement in a section of an aquifer system between the undergroundwater divide and the receiving stream with layers of different hydraulic conductivity has been simulated numerically in two dimensions; the groundwater surface receives a recharge of 150 mm/a, the groundwater flows through all the layers and finally reaches the receiving stream as a surface discharge. The numerical simulation of scenarios with generally known hydraulic conductivity/depth distributions (Fig. V.4.2) and the calculated amounts of groundwater turnover in the individual layers (in percent of groundwater recharge) leads to the conclusion that generally more than 85%

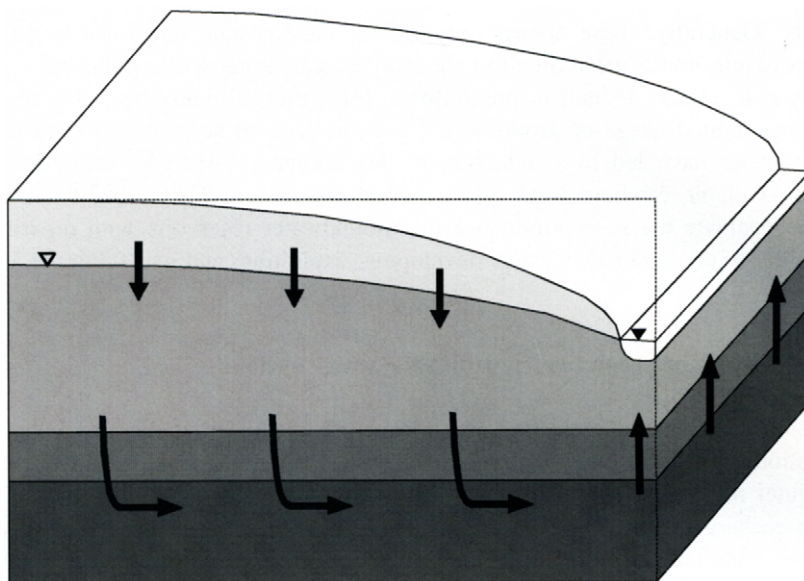


Figure V.4.1. The two-dimensional modeling plane presenting the influence of conductivity distributions in sediments upon the distribution of the groundwater recharge in the individual layers. A groundwater recharge of 150 mm/a was assumed with no overland discharge or interflow and no groundwater underflows the receiving stream.

of the groundwater recharge occurs in near surface layers and that less than 15% of the groundwater recharge reaches also deep lying aquifers.

Thus, the groundwater recharge is divided in an active, near surface zone with young groundwater (< 50 years old) and a passive, deep groundwater recharge zone (Fig. V.4.3) with water ages exceeding 50 years and reaching many thousand years (see below). Both of these groundwater recharge zones occur worldwide.

- In semi-arid to arid regions, the active groundwater recharge zone has a thickness of a few decimeters or meters, increases in the tropics to decameters and is in humid areas are less than 100 m thick; its thickness depends on the recharge and storage properties (hydraulic conductivity and effective porosity) of the system.
- The passive groundwater recharge zone can achieve a thickness of several hundred meters and is underlain by the connate groundwater (Engelhardt, 1960), i.e. from groundwater that did not return to the biosphere for millions of years (Fig. V.4.3).

The interface between the active and passive recharge zone can be identified in depth profiles by sudden changes in the age of the water and in part also by changes in the water quality. If isochrones are incorporated in the numerical simulation of the scenarios of the groundwater recharge (Fig. V.4.4), it can be seen that the age of the water at the base of the active groundwater recharge zone in effect increases very rapidly to several hundred to thousand of years. If the natural depth distribution of  $^3\text{H}$ ,  $^{14}\text{C}$ , and e.g. ion exchange waters (Fig. V.4.5) are considered, it is obvious that the tritium content does not gradually decrease with depth, but instead suddenly. The  $^{14}\text{C}$ -concentrations also show a similar

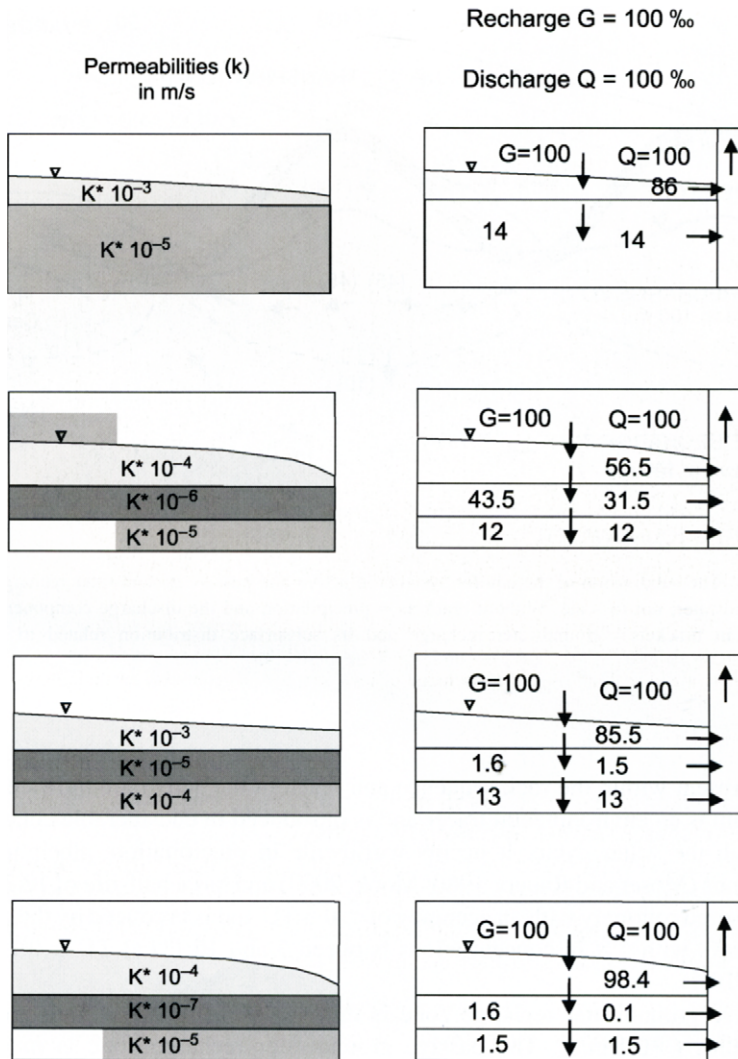


Figure V.4.2. Hydraulic conductivity series, as seen frequently in nature (left columns) and the percentage of the groundwater recharge in the individual layers (right columns).  $G$  = recharge (100%),  $Q$  = discharge (100%).

sudden decrease and also the ion exchange waters occur clearly more frequently below this depth. It has been proven practicable to define the boundary between the active and passive groundwater recharge areas with the radioactive environmental isotope  $^3\text{H}$ . To do this, the tritium naught line (TNL) is defined, an interface below which the tritium concentrations have fallen to values under the detection limit in routine measurements (Seiler and Lindner, 1995). Frequently, this boundary is also defined with the salt water/fresh water interface under the continents (Richter and Lillich, 1975). This requires, however, that salt

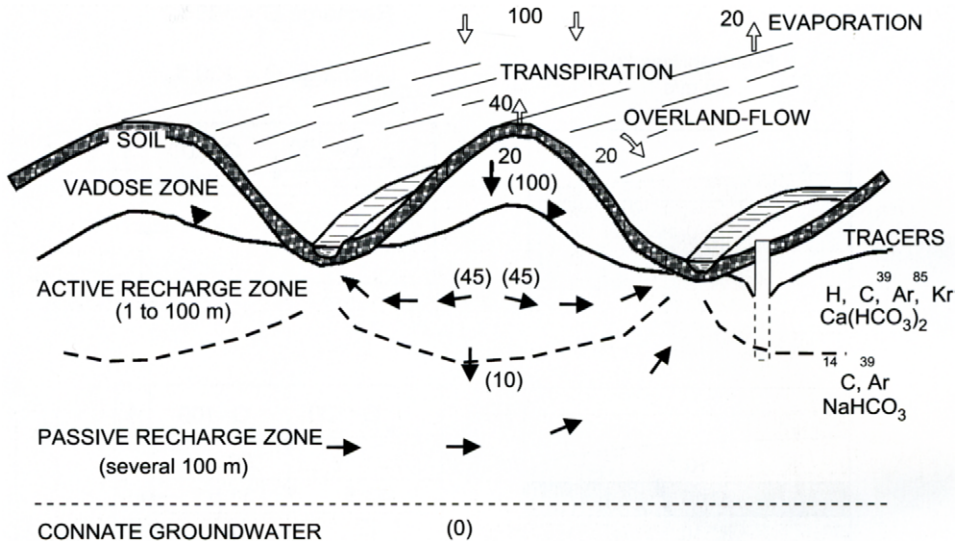
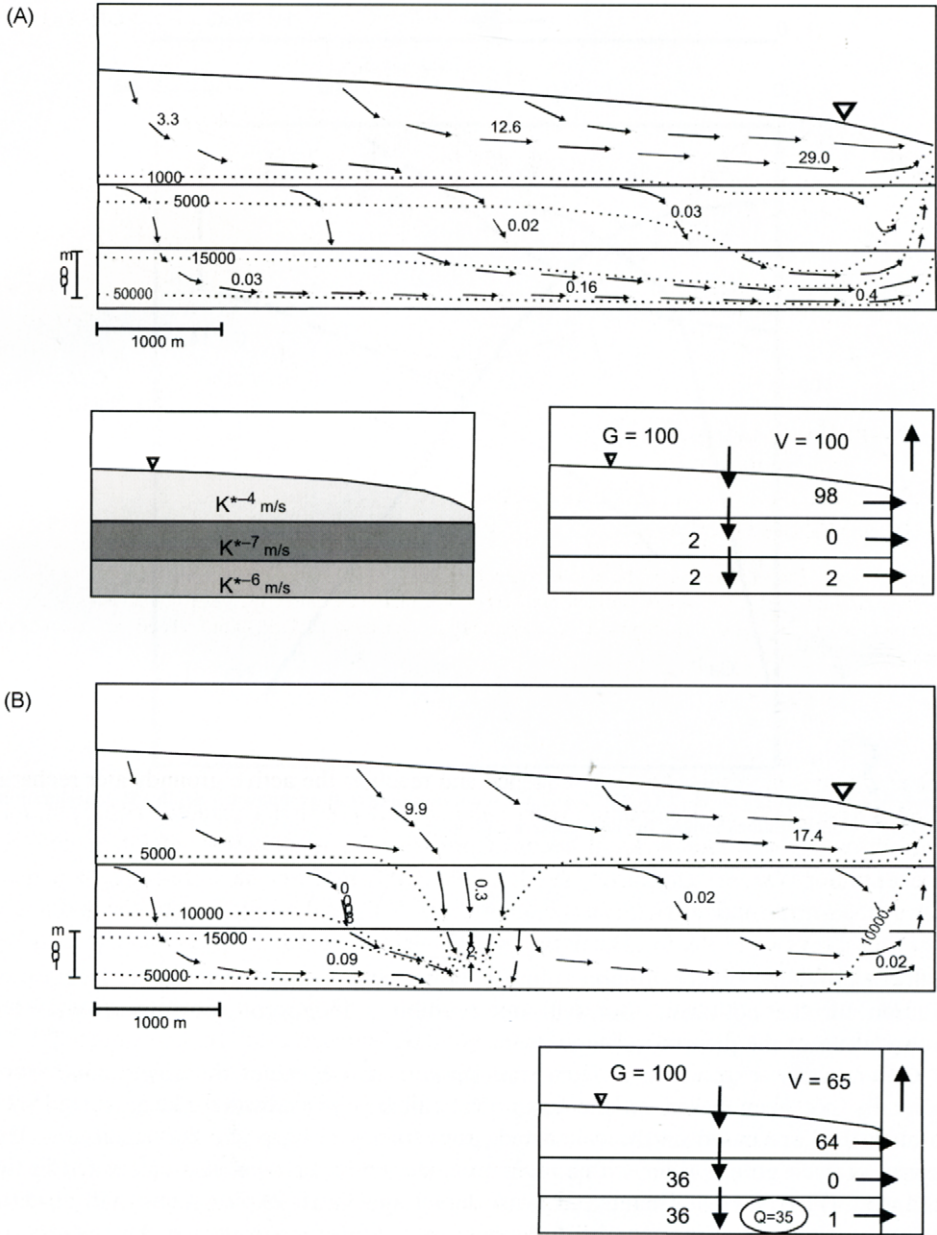


Figure V.4.3. The subdivision of the aquifer systems in active and passive groundwater recharge zones and connate groundwater; not to scale. Without brackets = precipitation and the discharge components related to precipitation; in brackets = groundwater recharge and its subsurface distribution related to groundwater recharge.

rocks are present within the rock sequence and reach to the active groundwater recharge zone. Tritium is an environmental tracer and is introduced in considerable concentrations only through the water cycle. It occurs worldwide in precipitation, albeit in different concentrations (Moser and Rauert, 1980; Mook, 2000) and has a half-life of 12.34 years. It can be measured routinely with an accuracy of  $\pm 0.5$  TU and is produced by the interaction of cosmic rays with  $^{14}\text{N}$  in concentrations between 5 and 15 TU (1 TU = 0.113 Bq/l or  $^3\text{H}/^1\text{H} = 1/10^{18}$ ).

The active groundwater recharge zone is thin ( $< 100$  m) and has high groundwater flow velocities ( $> 0.1$  m/d). The passive groundwater recharge zone is much thicker ( $> 300$  m) and has groundwater flow velocities under 0.01 m/d. Due to the high dilution volume, which results from the low groundwater flow velocities and the large groundwater thickness, the passive groundwater recharge zone reacts much slower to pollutant inputs than the active groundwater recharge zone. Only the connate water stays free of pollutants – in principle. Connate groundwaters (Engelhardt, 1960) are usually not used as drinking or industrial water due to their chemical composition; however, they can be and are used in balneology.

Following the development of water supply in the last 100 years, it turns out that in many countries at the beginning river waters, later waters of springs and shallow wells, and finally of deep wells that penetrate the passive groundwater recharge zone, have been used for water supply. The reason for this development was, among others, impairment of the groundwater quality for drinking water purposes. Mostly the sources of contamination have been disregarded and the water quality problems have been solved by digging deeper



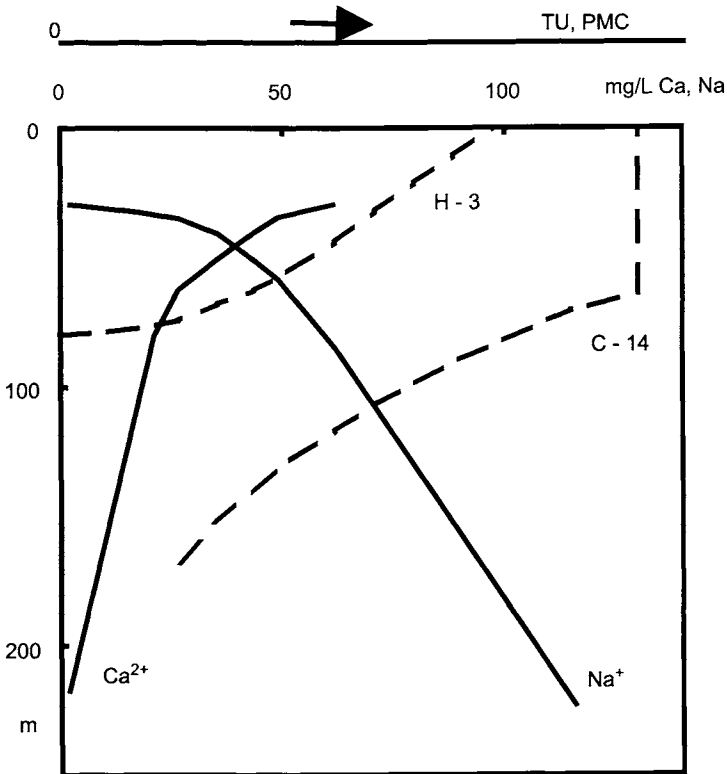


Figure V.4.5. Changes in  $^3\text{H}$ -,  $^{14}\text{C}$ -,  $\text{Ca}^{2+}$ - and  $\text{Na}^+$ -concentrations with groundwater depth;  $^3\text{H}$ -concentrations in TU,  $^{14}\text{C}$ -concentrations in pmc (percent modern carbon).

wells or providing bigger dilution volumes. However, this only produces an increase of dilution for the pollutants, but will not prohibit a long-term contamination of the groundwater of the passive recharge zone.

Groundwater exploitations from the passive groundwater recharge zone have generally not been based on the low, yet available groundwater recharge (<15%). Instead, they are based on the calculated groundwater recharge for the landscape. The impact of such groundwater withdrawals from deeper layers was also calculated in the scenarios (Fig. V.4.4). Thereby, it was shown that such exploitations will lead to hydraulic short-cuts between the different zones, if the groundwater exploitation is higher than the natural, respective to the aquifer related yield. The resulting groundwater deficit must then be compensated (DVWK, 1983, 1987). This compensation process can last for several years to decades and keeps the hydrodynamic system for a long run in transient conditions. Thus, it appears that a quantitatively and qualitatively secure water supply from the passive zone was available if applied in accordance to the aquifer specific recharge else leads to a long-term contamination input into a groundwater zone that would have been naturally protected in the long run.

#### V.4.4. Transport potential of discharge components

Discharge comprises three major components (Fig. V.4.6):

- overland discharge,
- interflow, and
- base flow from groundwater recharge.

All of these flow components can transport pollutants, either by erosion or dissolved or particle favored and achieve quite different impacts on ground- and surface water. Solid wastes either disposed in the landfills, or spread on the land surface as dry and wet particulate deposition or in purpose as excessive agrochemicals or common fill used in civil engineering, are ultimate sources of contaminants leached and transported to groundwater by infiltrating precipitation waters. The main pools of contaminants for these discharge components are the land surface and the effective root zone (Luckner, 1994).

Overland discharge and interflow only originate in sediments and rocks of hilly terrains with limited infiltration capacity. Normally, they do not occur in plains. Both of these discharges develop flow velocities of several kilometers and meters per day, respectively. Contrary groundwater recharge in non-fissured sediments has flow velocities of less than 0.02 m/d.

Interflow, which develops in hilly terrains out of bypass flow (Seiler, 1997), can wash a considerable amount of pollutants out of the effective root zone and thus contributes to the groundwater protection (Hellmeier, 2001). On the other hand, such a discharge of pollutants during precipitation (Fig. V.4.7) leads to main shock impacts

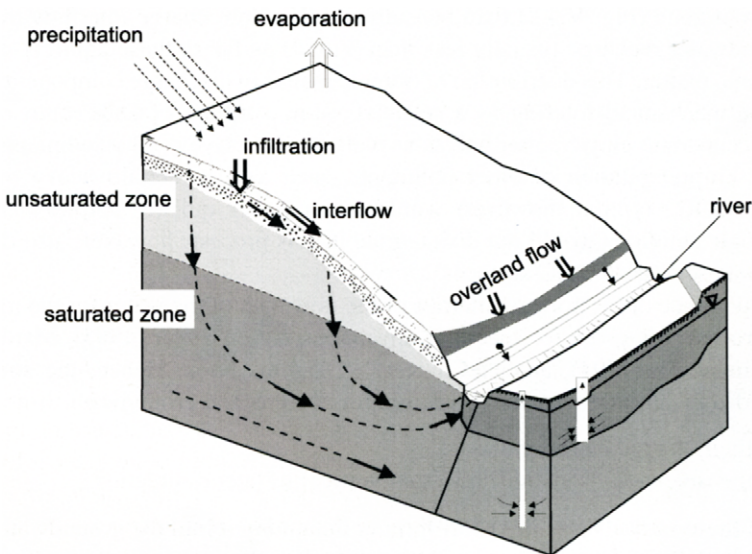


Figure V.4.6. Block diagram of a landscape with the three most important discharge components.

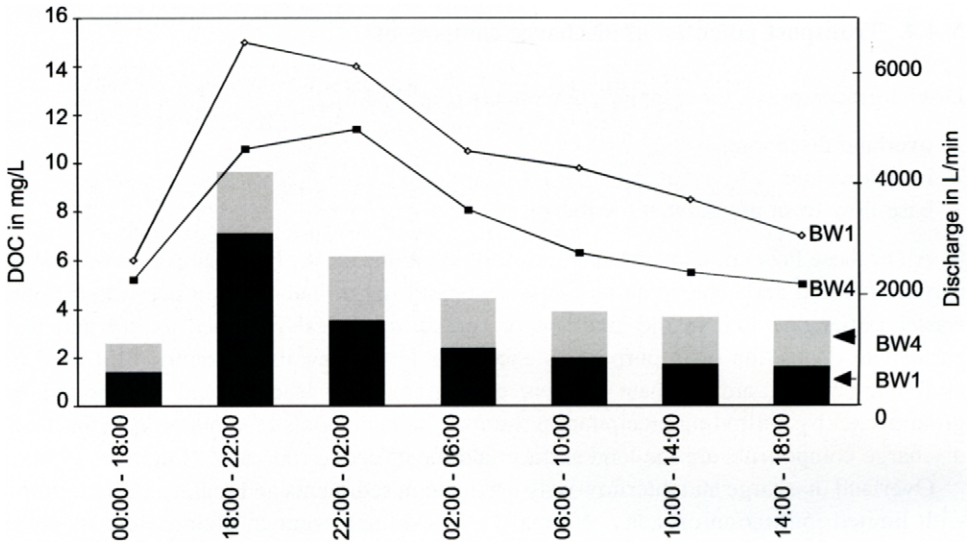


Figure V.4.7. DOC discharge after precipitation. BW1 = area with 100% firs, BW4 = forested area, about 20% used agriculturally. The DOC-concentrations in the discharge during precipitation are higher in the surface runoff than in the unsaturated zone under the effective root zone.

- to rivers, ponds, and lakes in hilly terrains and
- to groundwater in plains.

Mechanical filtration occurs in both saturated and unsaturated flow and impeded particle favored transport much more in the unsaturated zone. This is certainly a major reason why, e.g. DOC occurs in much larger concentrations in the seepage waters of the effective root zone (Fig. V.4.8) than beneath and therefore charge interflow much more than groundwater recharge (usually less than 3 mg/l) as far as these aquifers are free of fossil organic matter. This distribution of organic matter in discharge components is due as well to the mechanical filtering as a selective, slow sorption of DOC onto rocks; this sorption occurs onto clay-free sediments as well as onto clayey-silty sediments and also favors the co-precipitation of other pollutants, such as, e.g. certain heavy metals and pesticides. DOC sorption, however, would reach rather quickly sorption capacity if microbiologic activities would not disintegrate it; this process, however, also deliberates contaminants sorbed on DOC surfaces.

The importance of the pore geometries of the flow path of the water for the mechanical filtering processes is given for specified substances, e.g. in Seiler (1985), Matthess et al. (1991), Kim et al. (1994), and in Chapter V.2 of this book. Hence, the soil and the unsaturated zone provide a certain protection for aquifers and groundwater from pollution

- due to their storage capacity and
- due to the diversion of matter flow from vertical to lateral directions.

Both of these impacts prohibit short-term pollutant input into the groundwater, favor a momentary dilution of the pollutants, but can lead to long-term unwanted background levels of pollutants that may reach groundwater.

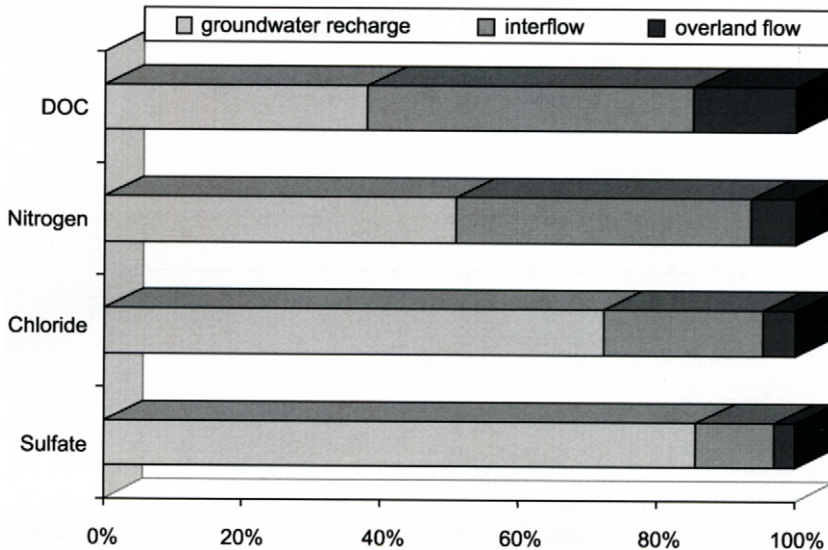


Figure V.4.8. DOC distributions as compared to nitrogen, chloride, and sulfate concentrations in groundwater recharge, interflow and overland flow (from left to right) in tertiary sediments of a hilly area (Scheyern, Germany). Chlorides and sulfates are transported as dissolved matter and the distributions in the three discharge components are in the same relations as the three discharge components itself. DOC transport differs significantly from this pattern because it is mechanically filtered within the effective root zone and does not reach groundwater as much as directly surface water through interflow; nitrogen shows a transient behavior as it is transported partially by DOC, partially as dissolved matter (Hellmeier, 2001).

#### V.4.5. Rock properties and the susceptibility of aquifers to contaminants

Water bearing rocks are called aquifers and normally they are subdivided into unconsolidated and consolidated aquifers. Unconsolidated aquifers are generally porous and do not have a secondary porosity such as fissures. Consolidated aquifers, on the other hand, were physically or chemically solidified after sedimentation (sedimentary rocks) or crystallized (crystalline rocks) by metamorphic processes; fissures provide a secondary porosity. In easily soluble rocks, the fissures can be widened into solution cavities. Some of the consolidated aquifers can also have a primary porosity, such as, e.g. many sandstones, carbonate reefs, or Cretaceous chinks. Such rocks with fissures and matrix pores belong to the heterogeneous-porous media, because their flow velocities are not continuous, but have usually a bimodal or polymodal frequency distribution.

The importance of rock porosities (pores, fissures, solution cavities) with regard to pollutant behavior and thus the susceptibility of the aquifers to pollution is closely connected with the groundwater flow velocities and the hydrodynamic dispersion:

- The movement of the groundwater is positively correlated with the size of the rock pores. In general, the hydraulic conductivity of the rock changes proportionally to the square of the porosity. The proportionality constant is closely coupled with the sediment genesis and the diagenesis of the rock. Figure V.4.9 shows a broad review of rock hydraulic conductivities.

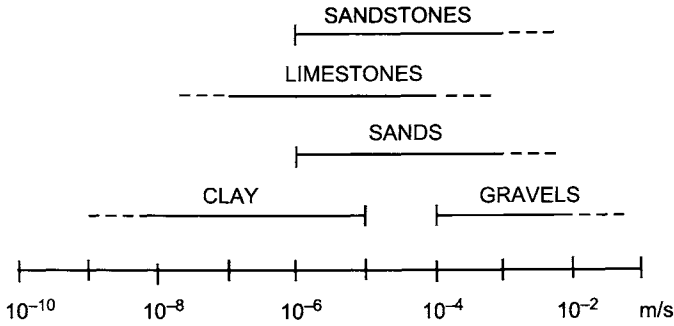


Figure V.4.9. Hydraulic conductivities of sediments and rocks.

- The groundwater flow velocity depends on the rock hydraulic conductivity and the groundwater recharge as well.
- The larger the rock pores are, the lower are the lateral and transverse hydrodynamic dispersion, i.e. the dilution, and most groundwater flow velocities will increase. Examples for the hydrodynamic dispersion of non-reactive tracers are shown in Figure V.4.10 (Lallemand-Barres and Peaudecerf, 1978; Freeze and Cherry, 1979; Seiler, 1985; Seiler et al., 1989; Glaser, 1997).

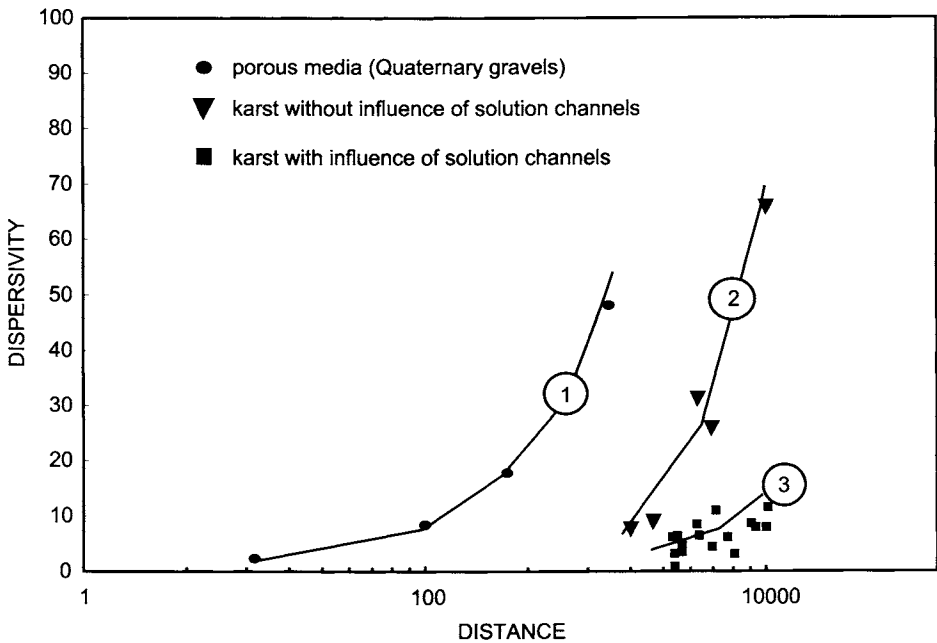


Figure V.4.10. Dispersion of non-reactive tracers in solution cavities (curve 3), in gravels (curve 1), and in fissured bedded rocks with low matrix porosity (curve 2).

In biporous media, in addition to the hydrodynamic dispersion, there is also a preferential lateral dilution component (Sudicky and Frind, 1981) due to molecular diffusion (Fig. V.4.10, top curve). This increases the dilution process,

- as long as a concentration gradient exists between the draining (e.g. fissures) and storing section (e.g. porous matrix) in heterogeneous-porous media,
- dead end fissures favor the transition of fissure flow into matrix flow, and
- the larger the spacing of fissures, respectively, the longer the flow distances within the matrix are getting.

These geological boundary conditions in fissured rocks with matrix porosity can lead to a creeping pollutant charge in the rock matrix with low hydraulic conductivity, which mostly cannot be discovered in time with conventional monitoring methods for ground- and drinking water.

Typical examples for heterogeneous-porous media are the reef rocks from the Upper Jurassic of the Franconian Alb. They have fissure porosities of less than 2 vol.% with hydraulic conductivities around  $10^{-3}$ – $10^{-4}$  m/s (Seiler et al., 1991) and matrix porosities between 5 and 10 vol.% with hydraulic conductivity under  $10^{-7}$  m/s (Weiss, 1987). In the same area there are also bedded rocks without matrix porosity and with the same fissure porosity and hydraulic conductivity as the reef rocks. Here it was shown by tracer experiments that

- 5 kg of non-reactive tracers, injected instantaneously into the groundwater, were diluted in the reef rocks after flow distances of 2 km to concentrations under their detection limit (2–20 ng/l). In contrast, in the neighboring bedded facies with less than 4 vol.% of matrix porosity, the tracer was still detected in high concentrations after a flow distance of more than 10 km and 50–100% of the injected tracer was recovered.
- In cases of heterogeneous-porous media, the tracer can also reach springs that discharge old water. Here, young and old waters get mixed along the fissure pathway.

In the investigated case, the pollutant input into both aquifer types was the same as the amount of the infiltrating water. Due to the mentioned differences in the rock properties, the groundwater from the bedded facies has clearly higher pollutant concentrations than the groundwater from the reef facies (Fig. V.4.11). The aquifer without matrix porosity shows the current pollution situation, whereas the aquifer with matrix porosity is not yet fully charged with pollutants, i.e. is creepingly charged, as long as microbiological processes in the rock matrix do not decay the pollutants (Seiler et al., 1996a,b).

In heterogeneous-porous media, through a suitable combination of classical hydrogeological, geochemical, and isotope investigation methods, the process of charge of the matrix with pollutants can be determined and process-orientated numerically modeled (Seiler et al., 1991; Seiler, 1997). In the case mentioned above, the mean residence time of the groundwater was determined in the rock matrix by sampling under dry-weather flow conditions for the environmental tracer  $^3\text{H}$ ; as a result residence times of several decades to a few centuries (Fig. V.4.12) were determined (Seiler et al., 1996a,b); the determination of the groundwater flow velocities in the fissures of the same aquifer (0.5–1 km/d) was done with fluorescent tracers and was coupled with a tracer balance (less than 1% recovery). Tracer balance and flow velocity provide quantitative information about the extent of the tracer exchange between the fissures and the porous rock matrix.

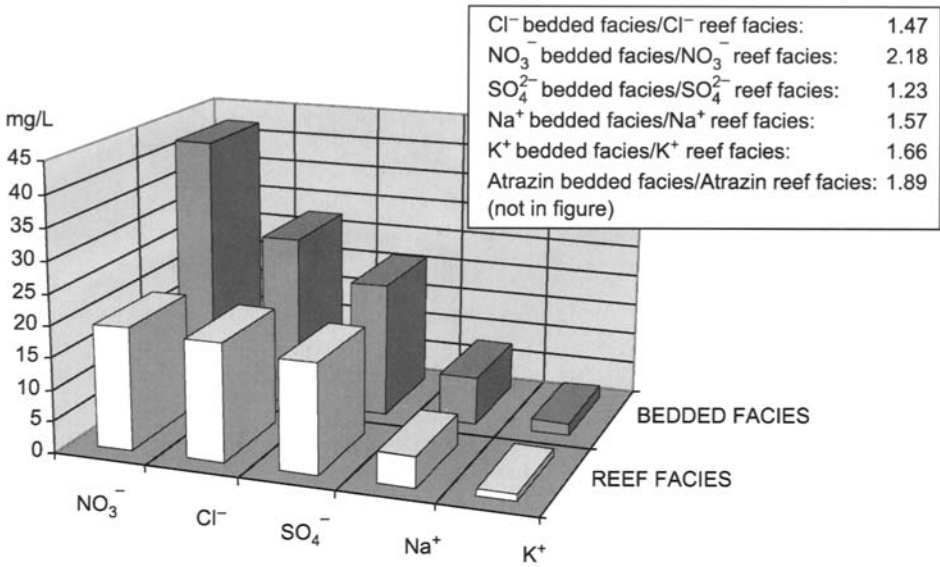


Figure V.4.11. Average concentrations of agrochemicals in groundwater from aquifers without (bedded facies) and with matrix porosity (reef rocks) in the Franconian Alb, Germany (after Glaser, 1997).

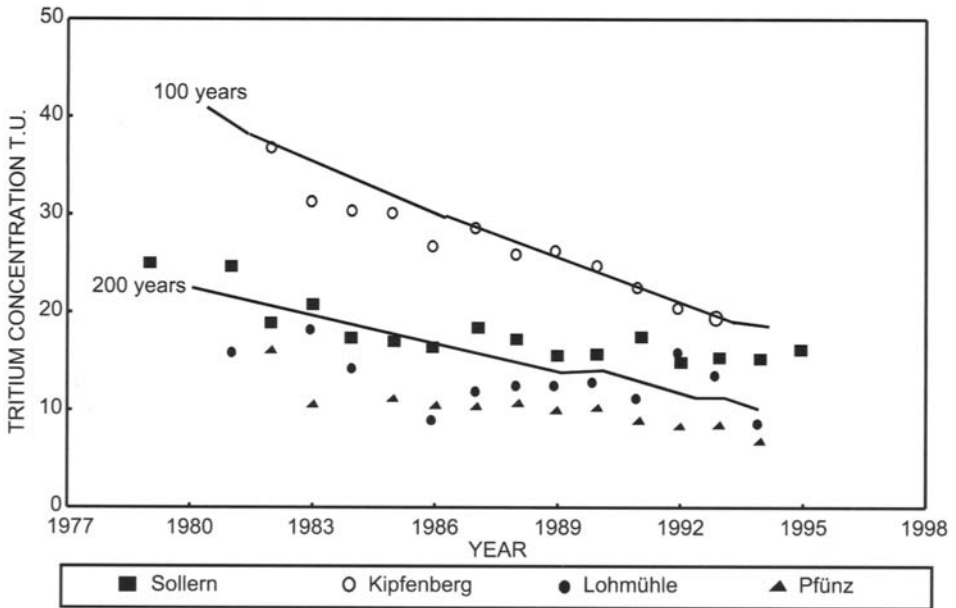


Figure V.4.12. Tritium concentrations and mean residence times of some groundwaters from the Franconian Alb.

Together with physical investigations of the rocks on determination of matrix porosities, pore sizes and matrix hydraulic conductivity, these data provide the basis for a numerical simulation of the pollution charge of the rock matrix, which can be compared with the measured values of the pollution charge in the groundwater from this heterogeneous-porous media.

#### **V.4.6. Microbiological activities in aquifers**

Common scientific teaching states that aquifers are poor in microorganisms, which catalytically influence chemical reactions. In contrast, the soil zone with abundant organic matter has a very high microbiological disintegration potential.

It is known from past investigations that microbiological activity occurs in confined aquifers with slow groundwater movement (Rietti-Shati et al., 1996) and in ore mines; however, in open aquifers, such reactions were rarely recognized up to now.

Recent microbiological–hydrochemical investigations have shown, however, that microbiological reactions in unconfined and even in heterogeneous-porous aquifers take place more often than previously assumed (Seiler et al., 1996a,b; Seiler, 1997).

Most aquifers appear to have a natural small microbiological population density, which can increase as a whole or selectively, as soon as nutrients (carbon, phosphate, nitrogen) and sufficient energy sources are available. This can happen, e.g. through the influx of organic pollutants that are easy to decay and will start after a certain incubation time needed to increase the microbiologic population. An example of this process can be taken from the fissured-porous rocks of the Upper Jurassic Limestones in the Franconian Alb. Here it was demonstrated (Seiler et al., 1996a,b) that under agriculturally used land, higher counts of colony forming microorganisms appear than under forest areas (Fig. V.4.13). The groundwater under the agricultural areas has a high (Fig. V.4.14), and the biofilms in these rocks have an even higher denitrification capacity. In comparison, the groundwater and biofilms under forest areas have a much lower denitrification potential. In the unfrosted areas of these regions, household sewage water and agrochemicals infiltrate into the soil. Under the forests, only atmospheric pollutants infiltrate the soil via interception.

The microbiologically effective colonization of the limestones of the Upper Jurassic is efficient solely in the fissured-porous media. Their matrix porosity is 5–10 vol.% and the pore openings are large enough for microbiological colonization ( $>5\ \mu\text{m}$ ). The importance of the differences in the microbiological colonization of the rocks with (reef facies) and without matrix porosity (bedded facies) is shown by a statistical comparison of some pollutants in the reef facies and the bedded facies (Fig. V.4.11). The nitrate, chloride, sodium, potassium, and sulfate concentrations (Seiler, 1997) correspond in the bedded facies to that calculated from the releases of agrochemicals and sewage waters in landscapes as compared to the groundwater recharge. In the reef facies, however, these values are generally low, because the rock matrix with porosities between 5 and 10 vol.% produces a higher dilution of the pollutants than the facies without the matrix porosity, but simultaneously increases a creeping storage of pollutants. This charging process began about 40–50 years ago with the increased use of agrochemicals. For sodium, potassium, chloride, and sulfate, the average concentration ratio in the groundwater from the reef and bedded facies is 1:1.5 and reflects the slow storage of pollutants in the matrix; yet for

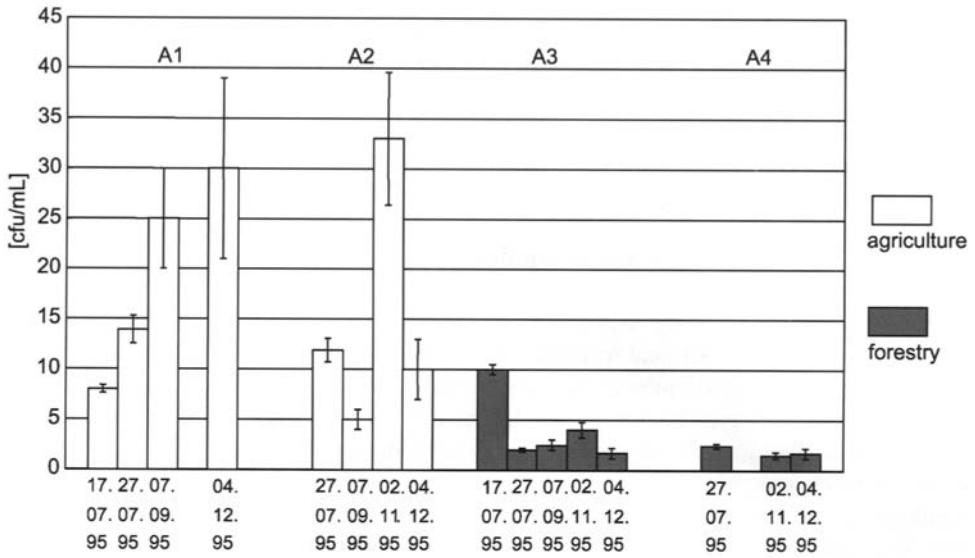


Figure V.4.13. Colony forming units (cfu) in groundwaters under agriculture (left two column groups) and forest areas (next two columns) in the reef facies and in a mixed reef/bedded facies (right columns).

nitrate it is 1:2. This is undoubtedly due to the decay process in the facies with the matrix potential, which is – as shown above – microbiologically controlled and which works against the pollutant storage in the rock matrix. Such a microbiological decay also appears to be present for atrazine.

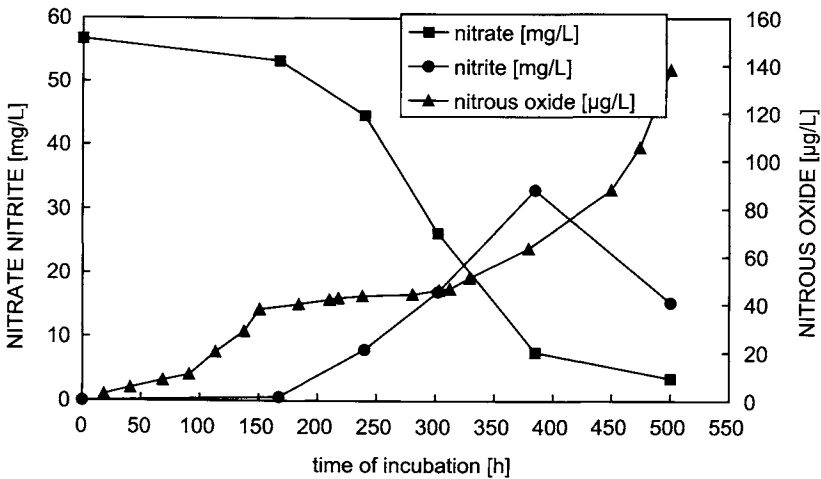


Figure V.4.14. Denitrification in a karst groundwater, which was dosed with 50 mg/l nitrate. NO<sub>2</sub> and N<sub>2</sub>O gas form subsequently. The N<sub>2</sub>-production could not be measured and was not prohibited.

An example that shows how fast microbiological populations in aquifers can increase as a result of pollutant input can be seen in the Caracas Aquifer under the city area of Caracas, Venezuela. Up to about 50 years ago, the city only covered the western part of the confluence of the Guaire and Valle rivers (Fig. V.4.15) but has since expanded astronomically over to the eastern part. The city receives its drinking water from nearby surface water reservoirs. The influx amounts to  $17 \text{ m}^3/\text{s}$ , and  $0.9 \text{ m}^3/\text{s}$  (Seiler, 2000) are lost into the groundwater due to leaks in the water supply systems under the city area. These losses are a significant source of the groundwater recharge under the city (Seiler, 1997).

The sewage water from Caracas is collected in a sewage system, which also leaks. Therefore, under the old and new city areas (Seiler, 1997) the chloride content in the groundwater is higher than in the imported drinking water, which percolates into the ground, and in the natural groundwater recharge. However, the nitrate content in the groundwater under the old parts of the city amounts to only a few milligrams per liter and the water contains some nitrite and ammonium; however, the groundwater under the new city has higher nitrate contents than in the drinking water. This proves qualitatively that denitrification processes effectively occur in the groundwater under the old city and that it has not yet started underground the new city, in spite of comparable pollutant inputs. Obviously, here an incubation time of a few decades is needed to increase the microbiological populations to a degree that considerable denitrification may occur. In both parts of the city area, the mean residence times of the groundwater are consistently around 4–5 years.

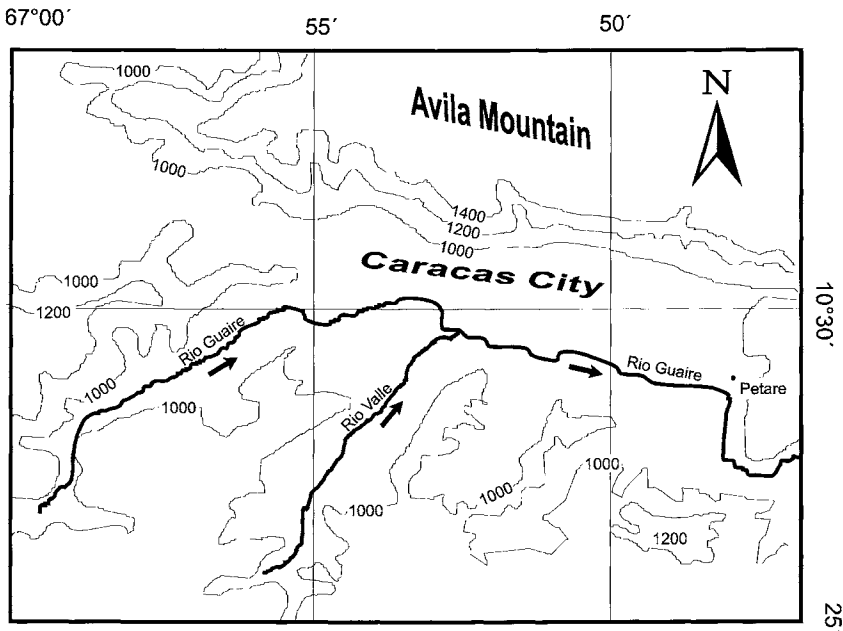


Figure V.4.15. Map of the city of Caracas in Venezuela. The old city lies in the west and the new city in the east.

Both of the examples described above prove that microbiological reactions in groundwater and in aquifers may increase the resilience of the underground system in its decay of pollutants after a specific, yet still not precisely known incubation time. The extent of this resilience will probably

- reflect the intensity of the land use and
- will get limited by too high concentrations of pollutants and an unfavorable chemical environment as well (e.g. low pH).

#### V.4.7. Concluding remark

The renewable groundwater resources are particularly vulnerable to contamination from anthropogenic sources: nowadays, in parallel with fast growing demand, reduction of drinking water availability due to deterioration of groundwater quality became a serious problem of immediate concern. Solid waste disposal, non-disposal use of chemicals and waste materials on the surface of the land, and long-term wet and dry deposition from high and low emitters are the major categories of sources of groundwater contamination. Soils appear to be not capable enough of binding and holding chemicals applied to their surfaces directly or leached from waste. Where concentrations of leached substances have been encountered in groundwater, they have been orders of magnitude higher than those found in surface water. Their dilution and removal is also much slower than in surface water and may render the groundwater non-potable for the foreseeable future. To safeguard groundwater resources from deterioration, the methods of waste and man-made chemicals management and use have to be thus judicious and effective.

#### References

- DVWK, 1983. Beiträge zu tiefen Grundwässern. DVWK-Schriften, 61, 1–107 (in German).
- DVWK, 1987. Erkundung tiefer Grundwasserzirkulationssysteme. DVWK-Schriften, 81, 1–223 (in German).
- Engelhardt, V.W., 1960. Der Porenraum der Sedimente. Springer, Berlin, p. 207 (in German).
- Freeze, R.A., Cherry, J.A., 1979. Groundwater. Prentice-Hall, Englewood Cliffs, NJ, p. 604.
- Glaser, S., 1997. Der Grundwasserhaushalt in verschiedenen Faziesbereichen des Malms der Südlichen und Mittleren Frankenalb. PhD Thesis, Univ. of Munich (unpublished).
- Hellmeier, C., 2001. Stofftransport in der ungesättigten Zone der landwirtschaftlich genutzten Flächen in Scheyern/Oberbayern (Tertiärhügelland). GSF-Ber., Neuherberg, p. 183 (in German).
- Kim, J.I., Delakowitz, B., Zeh, P., Klotz, D., Lazik, D., 1994. A column experiment for the study of colloidal radionuclide migration in Gorleben aquifer systems. Radiochim. Acta, 66/67, 173–185.
- Lallemand-Barres, A., Peaudecerf, P., 1978. Recherche des relations entre les valeurs de la dispersivité macroscopique d'un milieu aquifère, ses caractéristiques et les conditions de mesures. Etude bibliographique. Hydrogéologie et Géologie de l'Ingénieur, pp. 277–284 (in French).
- Luckner, L., 1994. Zustand und Schutz des Grundwassers in den neuen Bundesländern. DVGW Schriftenreihe Wasser, 84, 135–148 (in German).
- Matthess, G., Bedbur, E., Gundermann, K.-O., Loff, M., Peters, D., 1991. Vergleichende Untersuchungen zum Filtrationsverhalten von Bakterien und organischen Partikeln in Porengrundwasserleitern. Zentralbl. Hygiene Umweltmedizin, 191, 53.
- Mook, W.G., 2000. Introduction. Theory and methods review. In: Mook, W.G. (Ed.), Environmental Isotopes in the Hydrologic Cycle. Principles and Application, Vol. I, UNESCO/IAEA public, Paris/Vienna, p. 270.

- Moser, H., Rauert, W., 1980. Tracermethoden in der Hydrologie. Schweizerbart, Stuttgart (in German).
- Richter, W., Lillich, W., 1975. Abriß der Hydrogeologie. Schweizerbart, Stuttgart, p. 281 (in German).
- Riitti-Shati, M., Ronnen, D., Mandelbaum, R., 1996. Atrazin degradation by *Pseudomonas* strain ADP entrapped in sol-gel glass. *J. Sol-Gel Sci. Technol.*, 23, 77–79.
- Schaefer, A., Usthal, P., Harms, H., Staufer, F., Dracos, T., Zehnder, A.J.B., 1998. Transport of bacteria in unsaturated porous media. *J. Contam. Hydrol.*, 33, 149–169.
- Seiler, K.-P., 1985. Results of field experiments on hydrodynamic dispersion in Quaternary gravels of southern Germany. *Sci. Base Water Res. Manag., IAHS Publ.*, 153, 351–360.
- Seiler, K.-P., 1997. Isotope study of the hydrological impact of large scale agriculture. *Int. Symp. on Isotope Tech. in the Study of Environmental Changes, IAEA, Vienna*, pp. 321–339.
- Seiler, K.-P., 2000. Man's impact on groundwater systems. In: Mook, W.G. (Ed.), *Environmental Isotopes in the Hydrologic Cycle. Principles and Application, Vol. V, UNESCO/IAEA public, Paris/Vienna*, p. 102.
- Seiler, K.-P., Lindner, W., 1995. Near surface and deep groundwater. *J. Hydrol.*, 165, 33–44.
- Seiler, K.-P., Maloszewski, P., Behrens, H., 1989. Hydrodynamic dispersion in karstified limestones and dolomites in the Upper Jurassic of the Franconian Alb. *J. Hydrol.*, 108, 235–247.
- Seiler, K.-P., Behrens, H., Hartmann, H.-W., 1991. Das Grundwasser im Malm der Südlichen Frankenalb und Aspekte seiner Gefährdung durch anthropogene Einflüsse. *Deutsche Gewässerkr. Mitteilungen*, 35, 171–179 (in German).
- Seiler, K.-P., Behrens, H., Wolf, M., 1996a. Use of artificial and environmental tracers to study storage and drainage of groundwater in the Franconian Alb, Germany, and the consequences for groundwater protection. *Proc. on Isotopes in Water Res. Management, IAEA, Vol. 2, IAEA, Vienna*, pp. 135–145.
- Seiler, K.-P., Müller, E., Hartmann, A., 1996b. Diffusive tracer exchanges and denitrification in the Karst of Southern Germany. In: Bottrell, S.H. (Ed.), *Proc. of the 4th Int. Symp. on the Geochem. of the Earth Surface of the Int. Assoc. of Geochemistry and Cosmochemistry, IAGC*, pp. 644–651.
- Sudicky, E.A., Frind, E.O., 1981. Carbon-14 dating of groundwater in confined aquifers: implication of aquitard diffusion. *Water Res.*, 17, 1060–1064.
- Weiss, E.G., 1987. Porositäten, Permeabilitäten und Verkarstungserscheinungen im Mittleren und Oberen Malm der Südlichen Frankenalb. PhD Thesis, University of Erlangen, p. 211 (in German).