Solid Waste: Assessment, Monitoring and Remediation
Twardowska, Allen, Kettrup and Lacy (Editors)

IV.5

Principles of vadose and saturated zones monitoring in solid waste sites exemplified in mining waste dumps

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IV.5.1. Introduction

IV.5.1.1. Approach to vadose zone monitoring

Waste disposal sites (landfills) can be classified as potential non-point small-area sources of aquatic environmental contamination, though the area occupied by such facilities can range from several tens of square meters to several hundred hectares. They may be formed either as waste dumps on the land surface, or onto the land as impoundments. In both cases waste is exposed to atmospheric conditions and partakes in the natural (climatic) circulation of water. Atmospheric precipitation that infiltrates through the waste layer washes the soluble compounds out of the landfill and carries them through the vadose zone to the groundwater (Fig. IV.5.1). This may affect groundwater quality adversely.

The amount of atmospheric precipitation that percolates through the vadose zone depends upon the infiltration rate I. In the average conditions in Poland, infiltration rate accounts for $I \approx 100\ \mathrm{mm/year} = 0.1\ \mathrm{m/year}$, thus through 1 ha of land surface percolates annually about $1000\ \mathrm{m}^3$ of water. Average infiltration rate in the vadose zone formed from anthropogenic materials (e.g. coal mining waste in Upper Silesia coal basin in Poland) accounts for $I \approx 400\ \mathrm{mm/year} = 0.4\ \mathrm{m/year}$, hence about $4000\ \mathrm{m}^3$ of water percolates annually through 1 ha of waste dump.

These data show that at solid waste disposal facilities, particular attention should be paid to the conditions of contaminant migration in the anthropogenic vadose zone of a landfill and in the natural vadose zone beneath the landfill base. The need for vadose zone monitoring is a logical consequence of a failure for preventing contamination by means of saturated zone monitoring. It is well known that the alert provided by groundwater monitoring in the saturated zone is often too late to prevent significant degradation of recoverable groundwater resources, as the contaminant should occur in groundwater in the detectable levels to be noticed.

Vadose zone research was initiated more than two decades ago in the USA and since then has been recognized as an essential element of groundwater protection. The concept

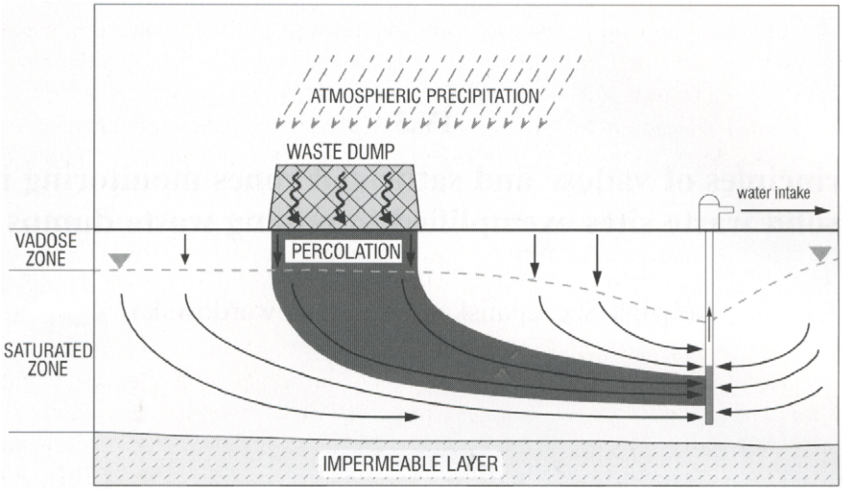

Figure IV.5.1. Scheme of water and contaminant flow in the porous aquifer with free water table (after Griffin, 1991).

behind developing vadose zone monitoring was to provide an early means to detect, and subsequently, intercept or remediate contaminants release from waste disposal facilities before they infiltrate into the saturated zone and degrade recoverable groundwater resources. By providing an early warning for taking instant remedial actions, the potential costs, and the potential for loss of recoverable groundwater resources can be greatly reduced.

U.S. EPA actions concerning vadose zone monitoring in the Resource Conservation and Recovery Act (RCRA) program started in 1978, when EPA proposed to require vadose zone monitoring for all RCRA hazardous waste landfills, surface impoundments, and land treatment facilities. The proposal was based on the research conducted at the EPA Environmental Systems Monitoring Laboratory in Las Vegas (EMSL-LV), in co-operation with the American Society for Testing and Materials (ASTM). After a decade of developing vadose zone monitoring techniques and significant advancements of this technology, the EPA proposed in 1988 to require vadose zone monitoring on a site-specific basis at RCRA hazardous waste landfills, surface impoundments, and waste piles, where the perennial water table is relatively deep and such system can provide effective early alert (U.S. EPA, 1988). In particular, it has been concluded that vadose zone monitoring equipment can be installed effectively at new facilities prior to the construction, or at the periphery of the unit to detect the contaminants migrating horizontally in the subsurface, or at the unlined solid waste management units (SWMUs), or even for many facilities equipped with liners and leachate collection systems that are prone to failure (Durant and Myers, 1995). In general, the EPA considers vadose zone monitoring to be used in conjunction with saturated zone monitoring. In sites where vadose zone monitoring system sufficiently meets the goal of early detection of contaminant release to the subsurface, it was found appropriate to eventually reduce the scope of saturated zone monitoring.

To facilitate implementing vadose zone monitoring at RCRA and other SWMUs facilities on a case-by-case basis, the EPA has developed a two-volume guidance manual on subsurface characterization and monitoring techniques (Boulding, 1993), which contains summary descriptions of more than 280 specific field methods that explain sampling procedures, frequency and sample analysis for adequate techniques. The comprehensive EPA policy directives and guidance manuals provide detailed criteria for selecting, designing and implementing site-specific vadose zone monitoring programs and response remedial actions. The methods and criteria are closely related to the standard methods developed in 1980–1994 by the ASTM, mostly by Committee D18 on Soil and Rock, and published in the recent Volumes 4.08 and 4.09 of ASTM's annual books of standards continuously updated (since 1994–2003).

The most comprehensive guidelines to vadose zone characterization and monitoring representing the synthesis of basic concepts, principles, and limitations of available routine monitoring techniques, QA/QC, as well as remediation and control of contaminants from hazardous sites, with a guide to major references including U.S. EPA and ASTM guidelines are addressed in excellent handbooks by Boulding (1995), Wilson (1995), Wilson et al. (1995) and Looney and Falta (2000).

Both EPA manuals and the aforementioned publications consider monitoring of the natural vadose zone directly beneath or at the periphery of the unit. In hydrogeologic environments where the water table of the saturated zone is too near the land surface and hence the vadose zone is too thin to ensure early detection of contaminant release to the subsurface, vadose zone monitoring may be considered inappropriate. At the same time, unlined SW landfills are in general the most problematic, as a rule being sited in areas with both a thin and permeable vadose zone, such as abandoned sand and gravel pits, or old strip mines. This means that the groundwater in these sites is the most vulnerable to contamination, which is though not adequately considered by the monitoring requirements and regulations.

The basic difference in the approach represented by the EPA and reflected in guidelines (Boulding, 1993, 1995; Wilson et al., 1995, Boulding and Ginn, 2003) and our approach is that according to our experience, the vadose zone screening/monitoring in the SW landfills, in particular in unlined ones, should comprise not just the natural vadose zone beneath the landfill, but also the anthropogenic vadose zone, i.e. the waste layer and pore solutions in the landfill. This would definitely provide an early alert in the case when the chemical composition of pore solution percolating downward in the waste profile shows unfavorable transformations, which indicates an excessive contaminant load-approaching groundwater. Typical examples of vertical redistribution of contaminant loads in non-hazardous waste dumps or landfills are presented in Chapters III.6 (coal mining waste dumps, USCB, Poland: Figs. III.6.20–III.6.22) and III.7 (power plant fly ash pond after closure, USCB, Poland: Figs. III.7.7–III.7.8 a–e). In general, the contaminant loads in pore solutions of non-hazardous waste comprise macro-components, e.g. sulfates, chlorides, nitrates, alkalis, etc. or inorganics (heavy metals), though toxic organic constituents may also occur in considerable amounts in the deeper parts of the vertical profile of a dump due to the vertical redistribution, as was found for steel and iron furnace slag and foundry waste (Twardowska et al., 2000). In any event, the contamination of groundwater by these constituents may make it unfit for any use (e.g. Twardowska et al., 1999; Twardowska and Szczepańska, 2002).

IV.5.1.2. Vadose and saturated zones monitoring technologies

The widely practiced vadose zone monitoring methods comprise direct soil-core and soil-pore liquid techniques as well as soil gas methods. For reconnaissance of vadose and saturated zones contamination from hazardous and non-hazardous disposal waste facilities, very useful methods might be fast developing, non-intrusive geophysical methods (e.g. electrical resistivity, conductivity, electromagnetic induction (EMI), active microwave, thermal infrared, electro-optical sensors, dielectric sensors, gamma–gamma, computer assisted tomography (CAT) scan, induced polarization (IP), time domain reflectometry (TDR, also called TDEM—time domain electromagnetics), ground-penetrating radar (GPR), very low frequency (VLF) electromagnetic resistivity measuring the ratio of electric to magnetic fields, neutron moderation, etc.). These methods can provide indirect evidence of contamination in the anthropogenic and natural vadose and saturated zones of a disposal site, and should be used in conjunction with the direct techniques. Their comprehensive overview is presented in the handbooks on soil, vadose zone and groundwater characterization and monitoring (Wilson et al., 1995; Looney and Falta, 2000; Boulding and Ginn, 2003). Geophysical and remote sensing techniques have for a long time been successfully used for screening and monitoring mining sites, and are particularly helpful in evaluation of the vulnerability of waste rock to acidification, and contaminant migration in the vadose and saturated zones. Reported applications of geophysics for characterizing mine waste include GPR and geoelectrical methods such as direct current (DC) resistivity, EMI, IP, magnetometry (MAG) (Conyers and Goodman, 1997; Campbell et al., 1999; Campbell, 2000; Campbell and Fitterman, 2000), or using imaging spectroscopy for mapping acidic mine waste (Swayze et al., 2000). In general, geophysical methods in mine waste contamination studies can be applied in several fields such as: (i) for characterizing natural stratigraphic conditions (GPR, EM, DC, and seismic methods); (ii) characterizing direction of flow of a contaminant plume, e.g. acid rock drainage (ARD), (EM, DC); (iii) detecting subsurface anthropogenic materials and preferential flow (MAG, EMI, GPR, VLF, metal detection); and (iv) locating buried waste (EMI, MAG, metal detection).

Examples of using EM, VLF and magnetics to locate preferential flow, trace edges of coal mine backfill, and find buried refuse that was a source of ARD (Schuek, 2000) and integration of airborne magnetic, electromagnetic and radiometric methods to survey abandoned mine lands (Smith et al., 2000; McDougal et al., 2000; Painter et al., 2000) illustrate particular usefulness of geophysics to trace ARD plumes and investigate active contaminant leach at mine waste piles.

A number of recent innovative emerging techniques for monitoring and measuring the chemical and physical characteristics of the vadose zone are particularly promising for vadose zone monitoring with respect to time, cost and accuracy, and amenability to generate data and information in near-real time. These techniques comprise chemical sensors, miniaturized or field-portable laboratory instrumentation, non-invasive characterization techniques, and minimally invasive techniques, as well as data and information management tools. Chemical sensors (mass, fiber optical, electrochemical, radiochemical, and thermal) are emerging techniques that can be used for field screening/monitoring mainly for organics, but also for inorganics, metals and radionuclides. Of these sensors, remote fiber-optic monitors appear to be the fastest developing and the most promising technique for detection of hazardous waste contaminants in environmentally viable and safe way. In this book, the fiber-optic monitoring techniques developed in Oak Ridge National Laboratory for this purpose, have been presented in Chapter IV.3.

A new emerging class of field kits is based on immunoassay techniques, which use antibodies that have a high degree of affinity to target organic analytes. The recognition by U.S. EPA of the utility of immunoassay kits for field screening and field analytical applications resulted in proposal to include BTEX, PCBs and pentachlorophenol immunoassay-based screening method into the RCRA manual for field screening and analytical methods SW-846 (U.S. EPA, 1986–2003), which is periodically updated. Use of biomonitors based on immunological principles and bioassays for environmental applications that can be particularly useful for monitoring of both anthropogenic and natural vadose zone in the hazardous and solid waste sites are summarized in Chapters IV.4.1 and IV.4.2.

Other innovative directions in the vadose zone monitoring and evaluation of hazardous waste site cleanup efficiency is miniaturization of laboratory instrumentation, in particular, development of field portable gas chromatography (GC), gas chromatography/mass spectrometry (GC/MS) and ion mobility spectrometry (IMS) for different organic contaminants (e.g. PAH, PCB, VOCs, etc.) (Meuzelaar, 1993, 2001). Both immunoassay kits and wearable instruments require samples to be brought to the surface, though their great advantage is amenability for providing qualitative and quantitative data in a near-real time.

Particular difficulties in characterization of chemical contamination in the subsurface by volatile and semi-volatile organic contaminants (VOCs and SVOCs) and soil gas due to their extreme instability directed the efforts toward developing sampling technologies that prevent or minimize the loss of volatiles. These technologies, in particular different enhanced sorbent devices, are discussed by Koglin et al. (1995) in the review of the emerging technologies for monitoring of the vadose zone.

One of the most developed new technologies for disposal sites characterization and analysis is the cone penetrometer integrated with real-time, downhole sensing devices (e.g. site characterization and analysis penetrometer system (SCAPS)) (CMST, 2000; Knowlton et al., 1995; Koglin et al., 1995; U.S. Army Corps of Engineers, 1998; U.S. DOE, 1998). The cone penetrometer consists of: (i) a steel cone that is hydraulically pushed into the ground while *in situ* measurements are continuously collected and transported to the surface for data interpretation and visualization; (ii) a 20–40 t truck equipped with vertical hydraulic rams that are used to force a sensor probe 25.4–50.8 mm into the ground with a penetration rate typically 12–15 m/h (Fig. IV.5.2).

Standard cone penetrometers collect stratigraphic information and are equipped with strain gages used for determination of soil type; for this purpose acoustic cones are also used. Other sensors available include measurements of temperature, pH, γ-radioactivity, pressure (P) and shear (S) waves. Time domain reflectometry (TDR) sensors use an electromagnetic pulse to measure the dielectric constant of the soil and to calculate the volumetric soil moisture content. Fiber-optic RH sensors measure relative humidity, which can be used to calculate capillary pore pressure in unsaturated soils. The SCAPS penetrometers are equipped with several other types of sensors; inorganic contaminants are assessed with use of an electrical resistivity sensor, electrochemical sensors have been developed to detect explosives such as TNT in soils. Active and passive radiation probes have been developed to detect radionuclides (U.S. DOE, 1998). Other measurement

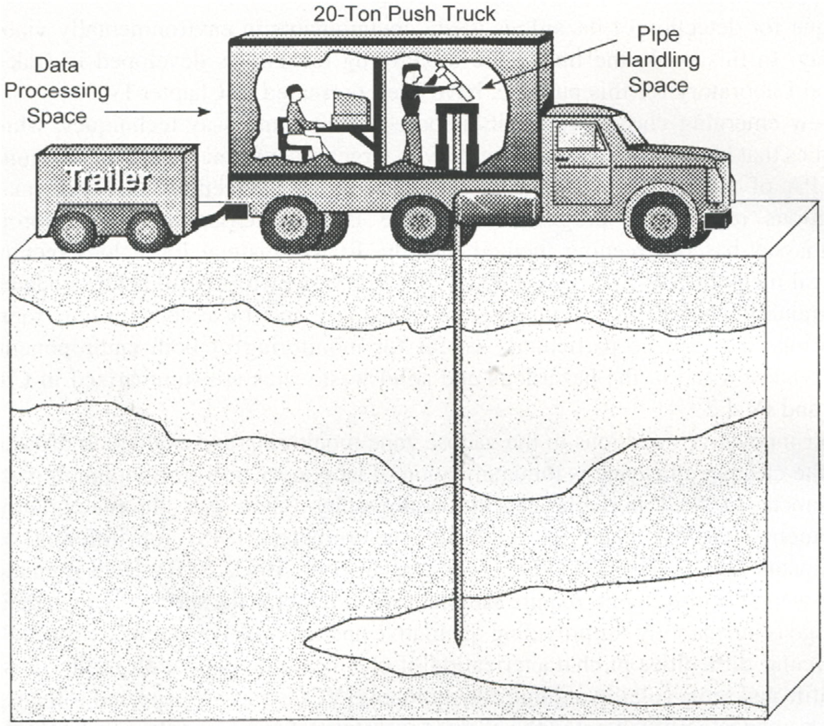

Figure IV.5.2. Site characterization and analysis penetrometer system (SCAPS) technology (after U.S. DOE, 1998).

capabilities include IR fiber-optic chemical sensors (Ewing et al., 1995; Nau et al., 1995) or fiber-optic laser-induced breakdown spectroscopy (FOLIBS) system for detection of heavy metal contamination (Thierault and Lieberman, 1995) and other fiber-optic heavy metals detection systems (e.g. SEA, Inc.), and integration of a fast gas chromatograph into the cone penetrometer system. A fiber-optic-based laser-induced fluorescence (LIF) system is used for detection of heavy fuel fractions. The research efforts in the last decade have been focused on the development and improvement of LIF sensors for cone penetrometers of different types, e.g. a laser-induced fluorescence excitation–emission matrix probe (LIF-EEM) (Lin et al., 1995), other fiber-optic LIF sensors (Knowles and Lieberman, 1995; Nielsen et al., 1995), or the single-wavelength fluorescence and multi-channel sensor (Haas and Forney, 1995) for evaluation of organic contaminants. The rapid optical screening tool system (ROST) can be used to screen the subsurface for petroleum hydrocarbons. An LIF method has also been demonstrated to characterize PAH contamination (Stepan, 1999). One of the last developments comprise the dense non-aqueous phase liquid (DNAPL) toolbox that includes the following technologies: standard sensors for lithologic delineation, LIF probes, ribbon non-aqueous phase liquid (NAPL) sampler, field Raman spectrograph, GeoVis™ Soil video imaging system, Cone Permeameter™, Geoprobe™ membrane interface probe, and various sediment and groundwater samplers (CMST, 2000). An example of a SCAPS-LIF probe is given in Figure IV.5.3.

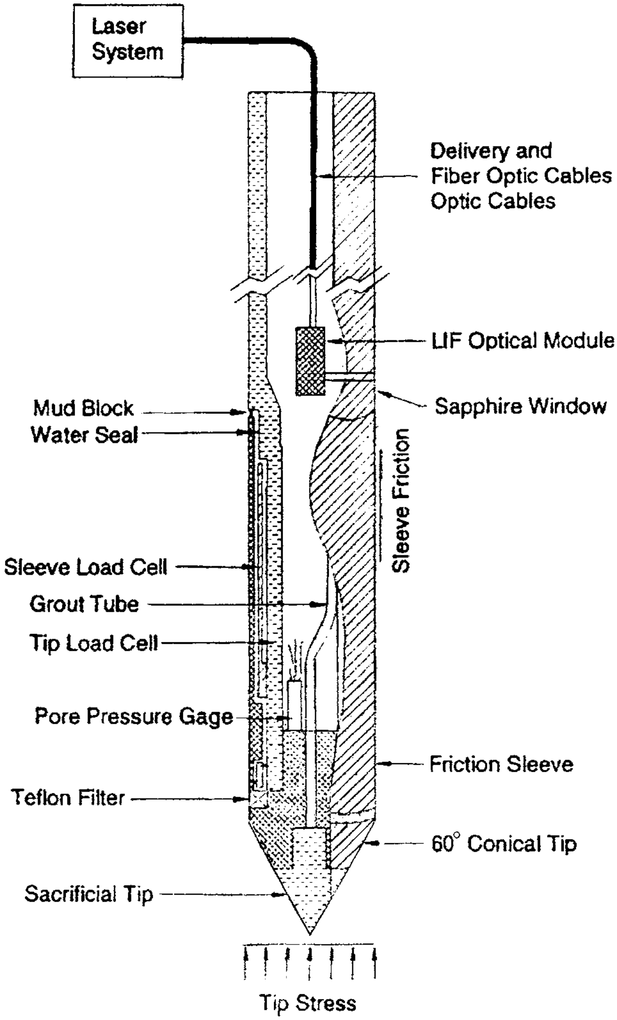

Figure IV.5.3. SCAPS-LIF probe (NaRaD, S. Lieberman, 619-553-2778), after U.S. DOE (1998).

Cone penetrometer technology can be adapted for other new sensors to measure various types of contaminants and other chemical characteristics of the subsurface; it can be used to install piezometers for soil vapor and groundwater measurements and to collect soil and water samples and is claimed to be less expensive when compared to drilling and sampling. Some limitations of use at all lithologic formations have been recently overcome by further advances in technology, e.g. by developing a vibratory assist device to allow penetration through hard rock zones. The most advanced currently available LIF sensors are though still too expensive to be applied commonly.

Therefore, conventional vadose zone monitoring methods such as direct soil-core and soil-pore liquid techniques as well as soil gas methods are still widely in use. Below, alternative invasive techniques for generating qualitative and quantitative information about contaminants in the vadose zone and some basic principles of vadose and saturated zone monitoring in the waste disposal facilities are presented.

IV.5.2. Basic principles of vadose and saturated zone monitoring in the SWMU sites

IV.5.2.1. Basic concepts

Migration of contaminants in the SWMU site occurs in two zones (Fig. IV.5.1):

- Vertical migration in the vadose zone (non-vertical movement in the layers of different permeability can also occur);
- Horizontal migration in the aquifer (in the saturated zone).

The vadose zone therefore plays a role of a retardation barrier against contaminant migration to the aquifer (saturated zone). The basic parameter to be considered in risk assessment to groundwater from waste disposal facilities is the mean time of water migration from the land surface to the aquifer (t_a):

$$t_a = \frac{x}{U_a} \qquad (a - \text{years}) \tag{IV.5.1}$$

where x is the thickness of the vadose zone (m) and U_a, the actual velocity of vertical migration of water in the vadose zone (m/year).

Classification of a risk for water quality in the saturated zone as a function of a vertical migration time (t_a) is presented in Table IV.5.1.

Mean time of horizontal flow in the aquifer (t) is evaluated from the equation:

$$t = \frac{L}{U} \qquad (\text{years}) \tag{IV.5.2}$$

where L is the migration distance (m) and U, the actual velocity of water flow (m/year).

Actual velocity of water flow U in the aquifer:

$$U = \frac{V}{n_0} \qquad (\text{m/year}) \tag{IV.5.3}$$

Table IV.5.1. Classification of a risk for aquifer water quality based on the time of water migration (t_a) through vadose zone (after Kleczkowski, 1991a).

Class of risk	Extent of risk	Mean time of water migration from the land surface to aquifer t_a (years)
A	High	<5
B	Moderate	5–25
C	Low	25–100
D	Practically no risk	>100

where V is the mean filtration velocity (m/year) and n_0, the active porosity (dimensionless).

As

$$V = kJ \qquad (\mathrm{m/year}) \tag{IV.5.4}$$

where k is the hydraulic conductivity (m/year) and J, the hydraulic gradient (dimensionless), the actual velocity of water flow is:

$$U = \frac{kJ}{n_0} \qquad (\mathrm{m/year}) \tag{IV.5.5}$$

Velocity of a horizontal water flow can be classified in accordance with the principles applied in mapping major aquifers (Kleczkowski, 1991a,b) (Table IV.5.2).

Data presented above show that velocity of water flow in aquifers is generally low, scarcely tens or hundreds of meters per year.

Time of migration of conservative contaminants (that do not react with the ambient water–soil environment, $R = 1$) is equal to the time of water migration t_a and t, estimated from Equations (IV.5.1) and (IV.5.2) on the basis of actual water flow velocity in the vadose zone (U_a) and in the saturated zone (U).

For the contaminants amenable to sorption ($R > 1$) the migration time (t_a^*) is R times longer than that of conservative contaminants (t, t_a):

$$t_a^* = Rt \tag{IV.5.6}$$

where R is the retardation coefficient evaluated from the sorption isotherms.

For prediction of contaminant flow and transport in the vadose zone in space and time, numerous 2D and 3D computer models have been developed, and new ones suitable for a variety of vadose zone applications continuously appear in the market (e.g. GMS 4.0 or WHI UnSat Suite Plus packages). Nevertheless, the present state of the art still does not provide reliable computer-modeled simulations that may be used alone, but are considered to be the most useful in iterative monitoring/modeling combination with expert opinion for predicting and quantifying environmental risk from contaminated sites (Fogg et al., 1995; Cramer and Cullen, 1995).

Due to long-term impact of mining activity on the ground and surfacewater quality, lasting for decades, geochemical modeling of mine drainage formation in time and space

Table IV.5.2. Classification of water flow velocity (U) in the major aquifers (after Kleczkowski, 1991a,b).

Mean actual velocity of water flow U (m/year)	Character of flow
<10	Very slow
10–30	Slow
30–100	Moderately fast
100–300	Fast
>300	Very fast

with consideration of kinetics of sulfide oxidation and buffering capacity of a material is an integral part of predictive models that are still under development (Foos, 1997; Szczepańska and Kmiecik, 2001), along with commercially available geochemical computer programs, e.g. AquaChem having a direct interface to the popular PHREEQC model (SSG, 2003; Waterloo Hydrogeologic, 2003).

IV.5.2.2. ***Factors affecting quality of hydrogeochemical data***

High economic value and commonly occurring risk to the shallow unprotected aquifers constrains the necessity of continuous water quality observations in the specially established monitoring network in the SWMUs and hazardous waste sites under RCRA. Monitoring of old sites should be preceded by site screening and analysis of site history and construction details if available. Water monitoring can be defined as repeated (with a defined frequency) analysis of water quality in permanent points, data processing and prognosis of trends to support actions focused on interception and remediation of adverse anthropogenic impact on the aquatic environment.

Sampling frequency of water quality monitoring in the vadose zone (pore solutions) and in the saturated zone (groundwater) is designed site-specifically, depending upon the velocity of water flow in these zones.

Sampling procedure is the first particularly important stage in the screening/monitoring of anthropogenic (waste layer) and natural vadose zone (pore solutions), as well as saturated zone (groundwater). To assure satisfactory analytical quality of data reported in the monitoring protocol (required level of precision and accuracy), proper procedures in a whole monitoring process should be used, starting from site selection, sampling technique, sample collection, processing and preparation for analysis, analytical methods, and data handling. Errors occurring in these stages should be summarized and will affect the final data given in the report (hydrogeochemical data – water physico-chemical characteristics). According to estimates, about 30% of the total error originates from sample collection and transport, 60% is due to sample processing and preparation for analysis, and barely 10% are analytical errors (Nielsen, 1991). QA/QC should not be limited just to the testing stage, but should comprise all the stages where errors can occur, with particular respect to the first stage, i.e. sampling (Fig. IV.5.4).

One of the major objectives of monitoring is evaluation of water quality changes in time or space. In the first case, the trends in water quality changes in time are assessed in the selected points of the monitoring network. In the second case, the extension of the contaminated zone from the contamination source is evaluated on the basis of measurements conducted in the local monitoring network. Thus, water sampling should be adequately time- and spatially representative. The time representativity depends on the sampling frequency that should enable assessment of the changes of water chemical composition resulting from the impact of the different sources of contamination. The frequency of sampling is assessed on the basis of hydrogeological premises depending upon the sampling depth and actual water flow velocity.

The spatial representativity is connected with sampling in the monitoring network (from permanent points), from the defined aquifer, constant depth and in the defined time intervals. Below, the general principles of representative sampling of the vadose and

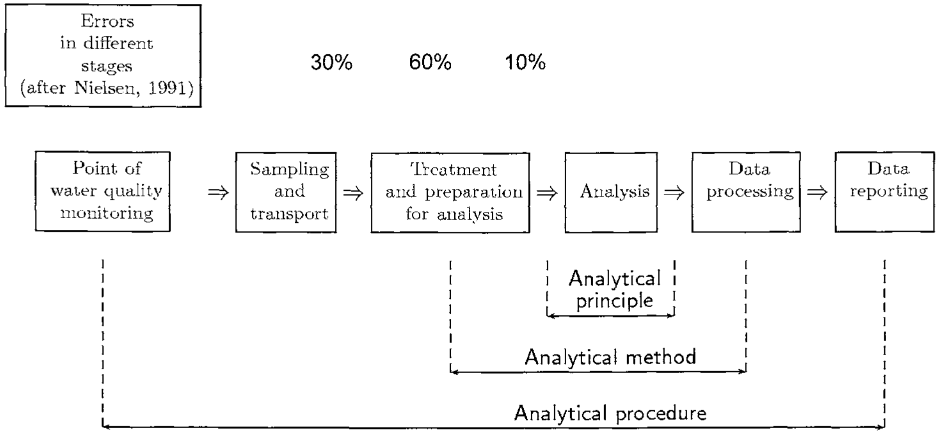

Figure IV.5.4. Monitoring of water quality. Phases of water quality monitoring procedures and relative error rates occurring in the subsequent phases.

saturated zones by invasive methods and QA/QC programs for assessment of errors occurring in the different phases of water quality monitoring (Fig. IV.5.4) will be discussed.

IV.5.2.3. Vadose and saturated zones sampling

Measurements of geochemical profiles of mine and other waste dumps have become a routine part of geoenvironmental investigations (Robertson et al., 1998; Twardowska et al., 1999, 2000; Helgen et al., 2000; Twardowska and Szczepańska, 2002).

Assessment of contaminant migration in the natural or anthropogenic vadose zone by invasive methods comprises the following procedure:

- Drilling of monitoring boreholes.
- Sampling of natural and anthropogenic soils from the defined depth intervals.
- Extraction of pore solutions from the soil samples (pressure methods, water extracts, solvent extracts).
- Analysis of constituent concentrations characteristic for the given source of contamination (indicator analysis) with use of adequate analytical methods.

Changes of chemical composition of pore solutions in the vadose zone in conjunction with the vertical redistribution of contaminant loads and transformations of chemical composition of pore solutions are observed in adequate depth and time intervals defined on the basis of the mean vertical infiltration velocity of precipitation waters U_a and conservative constituents, which migrate with the same velocity $R = 1$. This parameter can be assessed approximately from the mean infiltration rate and volumetric moisture content of soils in the vadose zone:

$$U_a = \frac{I}{W_0} \qquad \text{(m/year)} \tag{IV.5.7}$$

where I is the mean infiltration rate of precipitation waters in the vadose zone (m/year) and W_0, the volumetric moisture content of soils in the vadose zone (dimensionless).

In the natural conditions of Poland mean infiltration rate I accounts for about 17% of atmospheric precipitation H (Pazdro and Kozerski, 1990), i.e. about 100 mm/year (0.1 m/year).

In Table IV.5.3 are given mean migration velocities assessed from Equation (IV.5.7) for typical natural soils of the vadose zone (loess) of several major groundwater basins (MGWB) in Poland. The mean volumetric moisture content of loess $W_0 = 0.30$ (Fig. IV.5.5A), hence the mean vertical migration velocity $U_a = 0.33$ m/year.

In the anthropogenic soils (e.g. coal mining waste of USCB, Poland) the infiltration rate is much higher compared to the mean values for natural soils. Infiltration rates for coal mining waste assessed from long-term lyzimetric studies in the natural hydrologic cycle (Fig. IV.5.6) account for $I = 0.68H$, i.e. about 400 mm/year (0.4 m/year). For volumetric moisture content of coal mining waste $W_0 = 0.12$ (Fig. IV.5.7), the mean vertical velocity of water migration U_a in the vadose zone of waste dump accounts for 3.33 m/year.

The vertical migration velocities U_a estimated from Equation (IV.5.7) on the basis of infiltration rate and volumetric moisture content of soils in the vadose zone appeared to be consistent with the data obtained from the field studies on conservative tracer (Cl^- ion) migration. Migration velocity evaluated from the Cl^- ion breakthrough curves (Fig. IV.5.5B) with use of the CXTFIT program (Parker and Van Genuchten, 1984) accounted for 0.31 m/year for loess, and 4.45–2.04 m/year for coal mining waste depending on the extent of weathering disintegration of waste at the dumping site (Table IV.5.4).

Therefore, for assessment of the depth and frequency of the vadose zone sampling, the actual vertical migration velocity U_a estimated from mean infiltration rate I and volumetric moisture content W_0 may be used. Volumetric moisture content W_0 may be assessed from the natural moisture content W_n and volumetric density of soil ρ_d:

$$W_0 = \frac{W_n \rho_d}{100} \tag{IV.5.8}$$

where W_n is the natural moisture content (wt%); ρ_d, the volumetric density of soil (g/cm^3); $\rho_d = \rho/(100 + W_n)100$, and ρ, the volumetric density of soil (g/cm^3).

Table IV.5.3. Mean vertical migration velocities U_a in the vadose zone formed from natural and anthropogenic soils.

Soil type		Atmospheric precipitation H (m/year)	Infiltration I (m/year)	Volumetric moisture content W_0 (m/year)	$U_a = \frac{I}{W_0}$ (m/year)
Natural	Loess	0.60	$0.60 \times 0.17 = 0.10$	0.30	0.33
Anthropogenic	Coal mining waste, USCB	0.60	$0.60 \times 0.68 = 0.40$	0.12	3.33

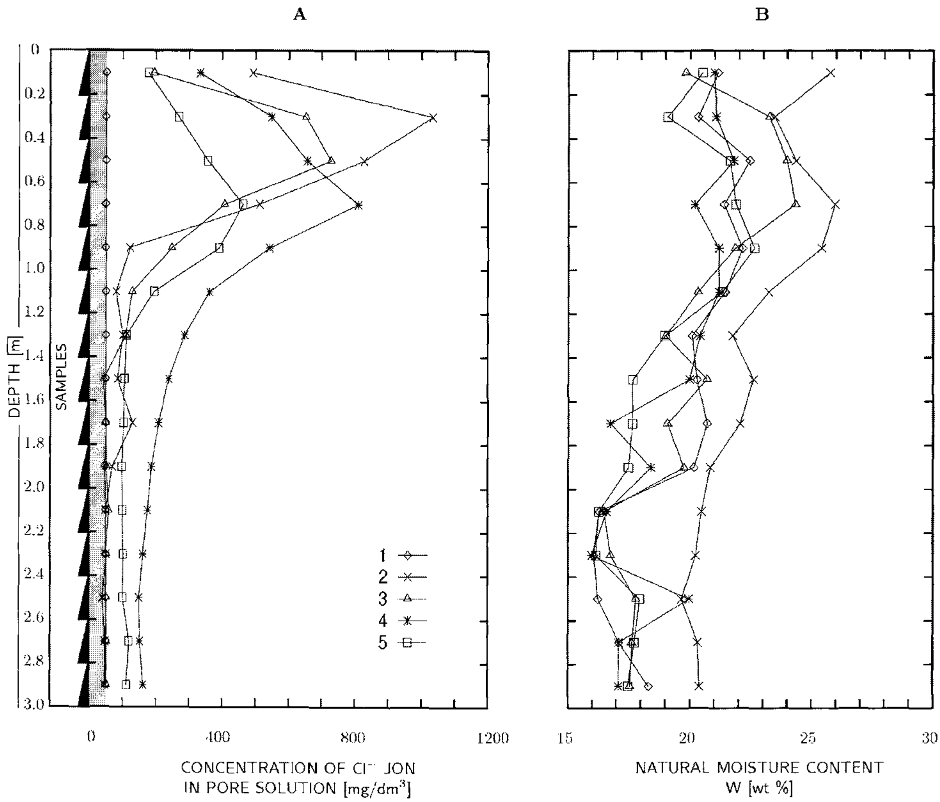

Figure IV.5.5. Conditions of contaminant migration in the vadose zone formed from natural soils. Loess insulating Major Groundwater Basin MGWB 459 in the Krakow area, Poland (after Bury (1994)). A – Scheme of soil-core sampling for observation of Cl^- ion migration. 1 – natural hydrogeochemical background of Cl^- ion preceding tracer injection ($t = 0$), 2–5 – Cl^- breakthrough curves for different time periods from the moment of tracer injection: 2 – after 7 months ($t = 7$), 3 – after 12 months ($t = 12$), 4 – after 24 months ($t = 24$), 5 – after 60 months ($t = 60$). B – Scheme of pore water sampling for evaluation of loess moisture content in the vadose zone.

Location of samples in the vertical profile of the vadose zone and frequency of sampling should be assessed on the basis of the actual migration velocity U_a. For loess, samples have been taken every 0.3 m along the vertical profile (Fig. IV.5.5A,B), while coal mining waste were sampled in 1.0 m intervals (Fig. IV.5.7) with sampling frequency ≥ 1 year.

Sampling frequency should be estimated individually, depending upon the time of water percolation through analyzed vadose zone (t_a). As can be concluded from Equation (IV.5.1), time t_a depends on the vadose zone thickness (x) and actual velocity of vertical migration (U_a).

Water percolation t_a through an analyzed loess layer 3 m thick will last approximately 10 years, hence the frequency of sampling should be no higher than once a year, except the initial period of monitoring, when the sampling should be more frequent due to the need of

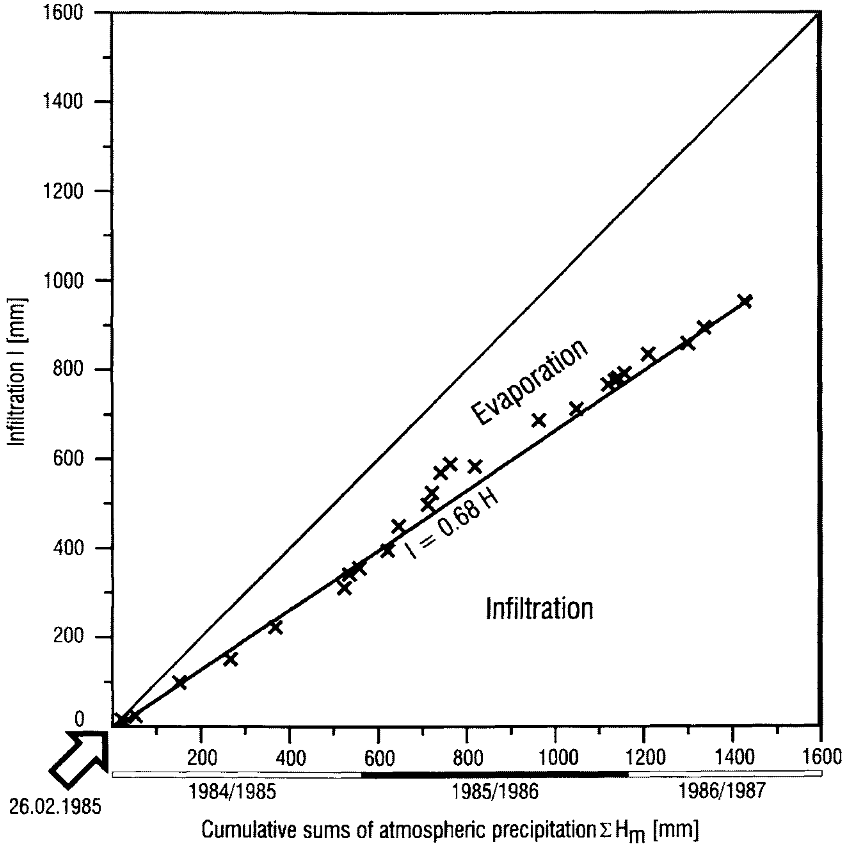

Figure IV.5.6. Infiltration rate *I* vs. precipitation *H* in coal mining waste of USCB, Poland. Lyzimetric studies in the natural hydrologic cycle.

estimation of possible changes in the uppermost layer resulting from the variability of infiltration rate in dry and wet periods.

IV.5.2.4. Monitoring of groundwater quality in the vicinity of waste disposal site

For evaluation of the natural hydrogeochemical background and observation of disposal site (SWMU) impact on groundwater quality (both in operational and post-closure periods) monitoring of groundwater is being conducted. It is classified as a local monitoring (LMGW). Its design, realization and operation in Poland follow the guidelines of the State Inspectorate of Environmental Protection (Staniewicz-Dubois, 1995) and European Union (EU, 1996; EEA, The European Environment Agency, 1999).

The number of observation boreholes and their location depends on the SWMU size and the hydrodynamic field of ambient groundwater. The approximate density of the local monitoring network should be about 1 LMGW borehole/ha. Monitoring wells should be located in three zones (Fig. IV.5.8A,B):

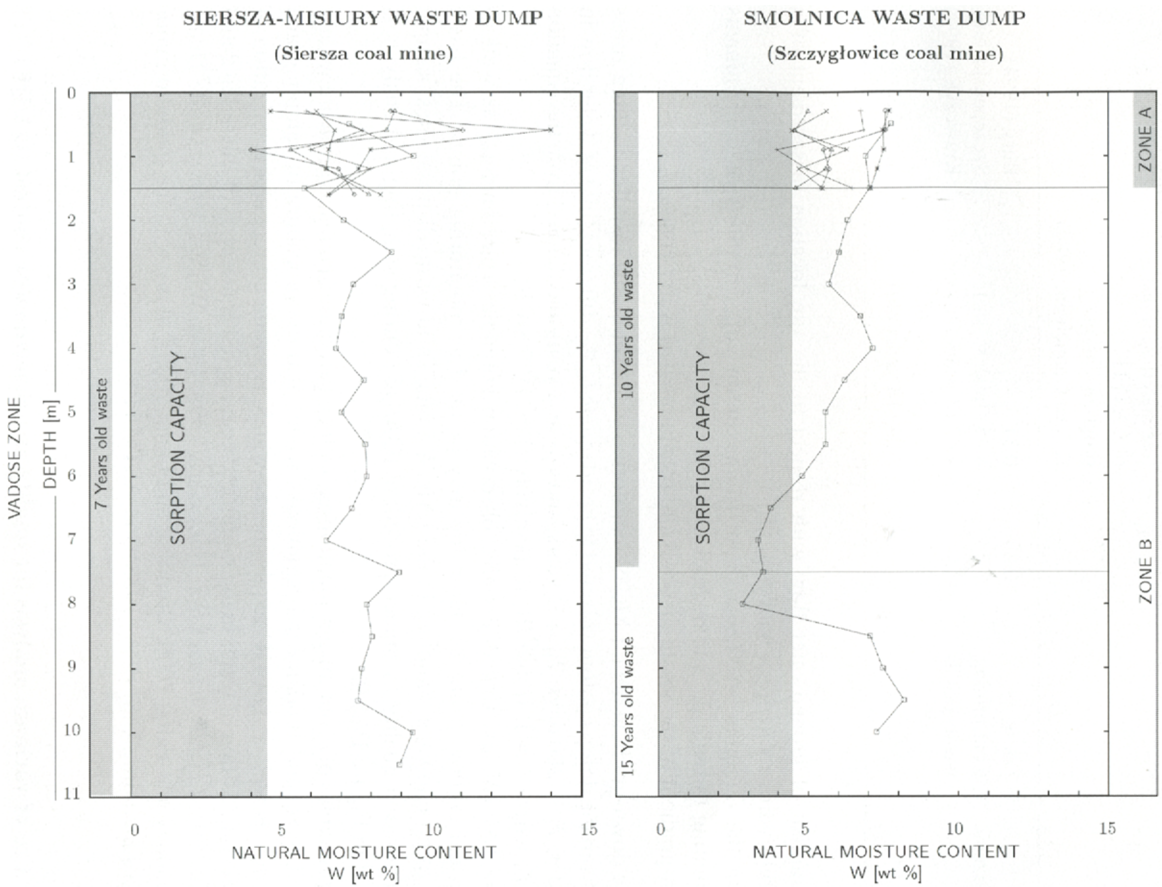

Figure IV.5.7. Conditions of contaminant migration in the anthropogenic vadose zone formed of coal mining waste dumps in the USCB, Poland. Scheme of soil-core sampling for assessment of natural moisture content in the vadose zone of the Siersza-Misiury and Smolnica dumps.

Table IV.5.4. Values of mean vertical migration velocity U_a in the natural and anthropogenic soils in the vadose zone estimated from Equation (IV.5.7) and from the program CXTFIT on the basis of breakthrough curve for conservative tracer (Cl^- ion).

Soil type		$U_a = \frac{I}{W_0}$ (m/year)	U_a (m/year) (estimated with use of CXTFIT program-Parker & Van Genuchten, 1984)	Data source
Natural	Loess	0.33	0.31	Bury (1994)
Anthropogenic	Coal mining waste USCB	3.33	4.45 (fresh waste) 4.42 (8 years old) 2.59 (12 years old) 2.04 (18 years old)	Szczepańska and Krawczyk (1994)

- up-gradient of groundwater flow with respect to the site location (assessment of a natural hydrogeochemical background);
- within the dumping site (assessment of maximum concentrations of contaminants percolating through the site bedrock);
- down-gradient of the waste disposal site (contaminated water zone).

In the down-gradient area, LMGW monitoring wells should be installed in three zones distant from the dump contour selected to be adequate to the different time of water flow in the aquifer: $T_{\mathrm{I}} < 200$ days, $T_{\mathrm{II}} = 2$ years, and $T_{\mathrm{III}} > 2$ years and defined from the equation $L_n = UT_n$ where $n =$ I, II and III, respectively (Fig. IV.5.8A,B).

In the LMGW, existing dug wells, observation boreholes (piezometers) as well as seepage and outflow from the dump toe may be used as monitoring points. The basic scheme of groundwater sampling in the LMGW in the solid waste disposal sites, sample preservation and quality assurance procedure (QA/QC) during sampling and analysis is presented in Figure IV.5.9.

The principles of groundwater monitoring, well construction and installation are widely presented in several fundamental handbooks where also the procedure of sampling and monitoring, as well as QA/QC requirements, which ascertain correct characterization of chemical composition of water in the monitored endangered aquifers, is discussed in detail (Nielsen, 1991; Lesage and Jackson, 1992; Sara, 1993, 1994; Wilson, 1995; Asante-Duah, 1996; Looney and Falta, 2000; Boulding and Ginn, 2003), along with the guide to major references including U.S. EPA (1988) and ASTM (1994–2003).

IV.5.3. Use of variance analysis for quality assurance/quality control (QA/QC) in groundwater monitoring

QA/QC in LMGW for solid waste disposal sites is of particular stringency when the monitoring data have either direct legislative liability or economic consequences in assessing fees/penalties on facility owners, or used for research purposes. QA/QC in

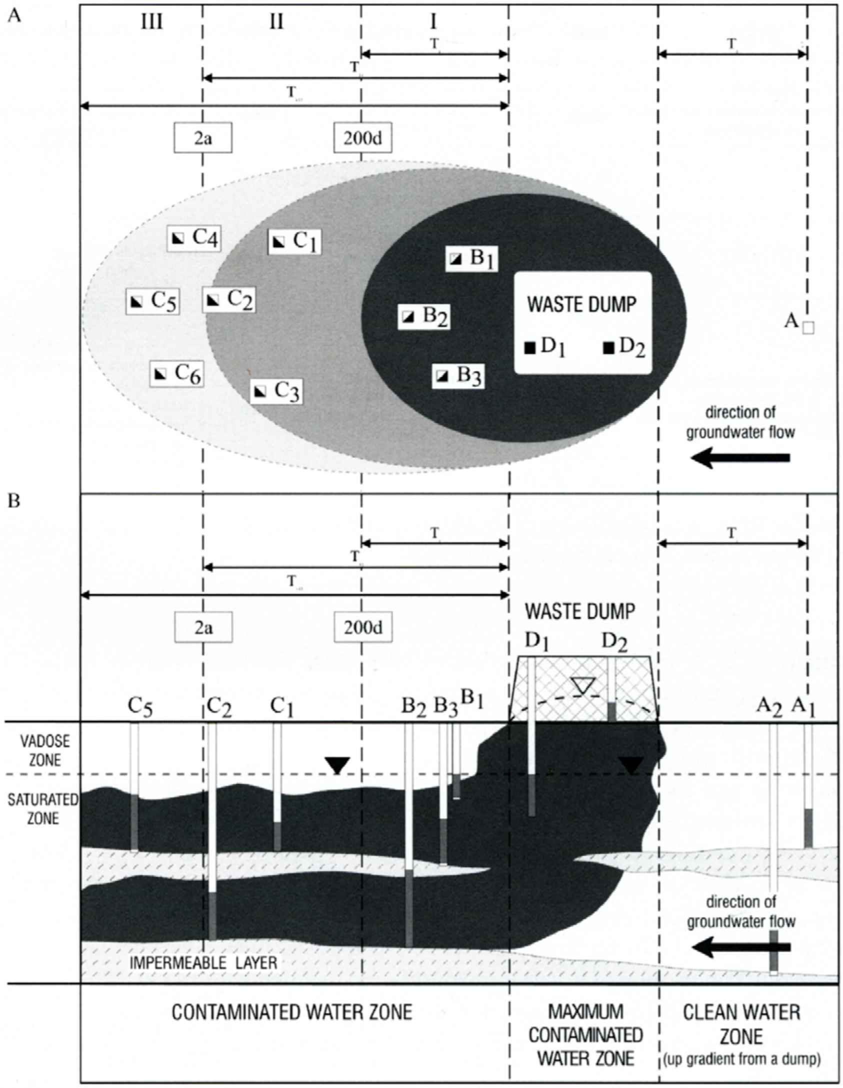

Figure IV.5.8. Local monitoring of ground water quality (LMGW) in a solid waste site. A – location of the monitoring network sites. B – Scheme of permanent monitoring well installations for observation of contaminant migration in the upper and lower level of the aquifer (after DVWK, 1992).

groundwater monitoring consists of two independent field and laboratory procedures, which are aimed at identifying and eliminating errors generated during sample collection, preservation, storage and transportation to the laboratory, as well as chemical analysis in the field and in the laboratory. The basic routine QA/QC procedures comprise collection and analysis of additional samples (10–20%), which consists of blanks, field, replicate, duplicate, split, and spiked samples, as well as use of SRMs, and are widely discussed in

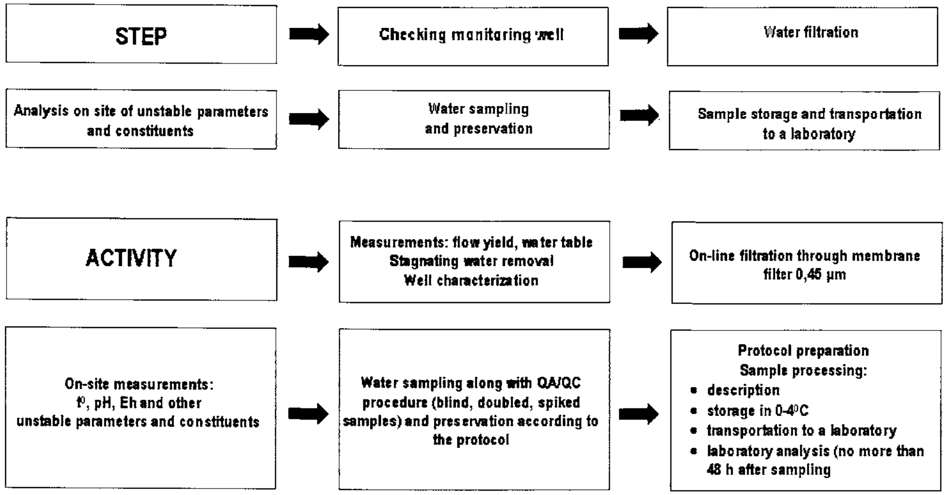

Figure IV.5.9. Basic flow diagram of water sampling, preparation and analysis procedure in monitoring of groundwater quality (after Witczak and Adamczyk, 1994.

the numerous U.S. EPA reports/guidances and other relevant sources quoted in the handbooks referred to previously. A particularly rich and systematic reference index on QA/QC in the vadose zone and groundwater monitoring is contained in the fundamental handbook prepared by Boulding (1995).

A valuable tool for analysis of precision in the groundwater quality monitoring is the analysis of variance ANOVA approach (ANOVA/MANOVA, 1984–2003) put forth by Garrett (1969) and developed in the present form by Ramsey et al. (1992). The current practice of QA/QC in groundwater monitoring shows that ANOVA is the most cost-effective method of reliably estimating random errors occurring in the sampling and analytical procedures (Szczepańska et al., 1999). Opposite to standard deviation, variance is additive and can be totaled if variance sources are independent. In the ANOVA method, natural spatial variability of physico-chemical parameters of groundwater can be assessed numerically by hydrogeochemical variance σ_g^2 for duplicate samples taken in the LMGW network.

The total observed spatial variation can be presented in the form of the total variance (σ_t^2):

$$\sigma_t^2 = \sigma_g^2 + \sigma_s^2 + \sigma_a^2 \tag{IV.5.9}$$

where σ_g^2 is the hydrogeochemical variance, σ_s^2, the sampling variance, and σ_a^2, the analytical variance.

The impact of sampling cannot be distinguished from laboratory errors without conducting the special extended quality control sample analysis (Fig. IV.5.10A), where the number of analyses is doubled with respect to the routine QC program (Fig. IV.5.10B). In the routine QC program, the technical variance σ_{tech}^2, which is a sum of the sampling and

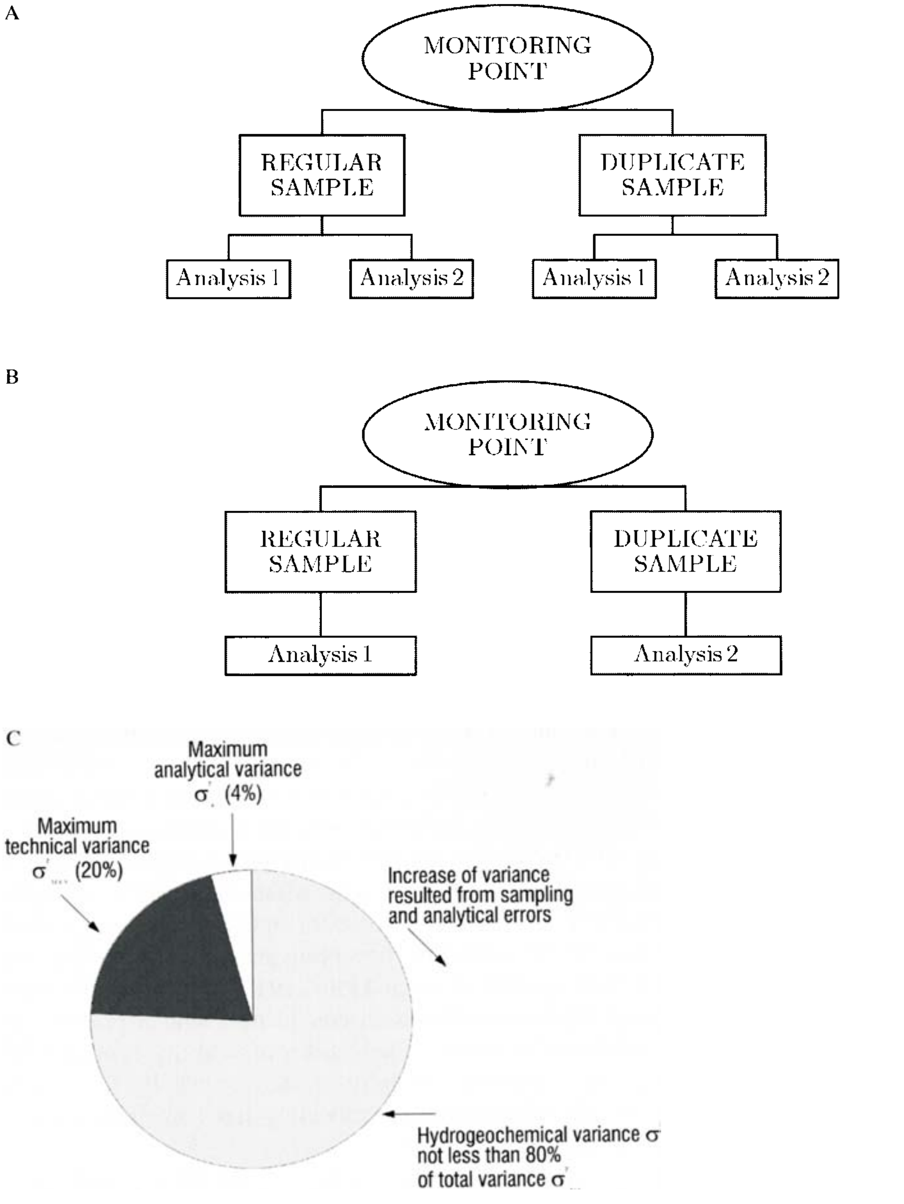

Figure IV.5.10. Sampling of groundwater quality monitoring site for ANOVA analysis of variance (after Ramsey et al., 1992). A – Diagram of procedure for evaluation of sampling variance (σ_s^2) and analytical variance (σ_a^2). B – Diagram of procedure for evaluation of technical variance ($\sigma_{tech}^2 = \sigma_s^2 + \sigma_a^2$). C – Maximum acceptable level of sampling (σ_s^2), analytical (σ_a^2) and technical variance ($\sigma_{tech}^2 = \sigma_s^2 + \sigma_a^2$).

analytical variance, is estimated:

$$\sigma_{tech}^2 = \sigma_s^2 + \sigma_a^2 \tag{IV.5.10}$$

Precision is considered satisfactory, if the technical variance (σ^2_{tech}) does not exceed 20% of the total variance, σ^2_{t}, while analytical variance, σ^2_{a}, should not exceed 4% of the total variance, σ^2_{t} (Fig. IV.5.10C) (Ramsey et al., 1992).

IV.5.4. Use of neural networks for long-term prognosis

As has been shown, generation and leaching of contaminants from solid waste disposal units, e.g. mining waste piles may last for decades, thus posing threat to the aquatic environment. This requires conducting adequate long-term monitoring of groundwater at the disposal site. Due to complexity of weathering processes in such heterogeneous systems as ARD-generating mining waste, where contaminant release is governed with kinetically defined process of sulfide oxidation, geochemical computer modeling with use of many existing popular programs such as WATEQ4F (Ball and Nordstrom, 1991, 1994) Visual MINTEQ v. 2.01 (Allison et al., 1991) is encumbered with serious predictive uncertainties. Much greater capabilities displays PHREEQC – a computer program of USGS for speciation, batch reaction, dispersion, advective-transport and inverse geochemical calculations (Parkhurst, 1995; Scott et al., 1997; Parkhurst and Appelo, 1999; Zhu and Anderson, 2002) in particular the newest version PHREEQC Interactive 2.8.0.0 (2003), or a sophisticated software package for aqueous geochemical analysis AquaChem (SSG, 2003; Waterloo Hydrogeologic, 2003) that incorporates PHREEQC. This program allows multi-component reaction kinetics and transport modeling in complex geochemical systems and is thus the most applicable for simulation of ARD generation and transport (Appelo et al., 1998; Postma and Appelo, 1999), also with uncertain data (Parkhurst, 1997). For simulating flow and organic contaminant transport in the vadose zone of landfills, the WHI UnSatSuite graphical environment (SSG, 2003; Waterloo Hydrogeologic, 2003) provides a comprehensive tool. Nevertheless, all these programs require a developed set of input data that are often not available. In such cases, neural networks, which permit "incomplete" input data and are capable of parallel data processing, can be of use. Their beginning has been generally associated with the appearance of a historical work by McCulloch and Pitts (1943) who for the first time gave a mathematical description of the neural cell in conjunction with a problem of data analysis. The major merit of neural networks is their ability of "learning" complex pictures and trends of data change and human-like using the gained knowledge for solving new problems. Development of the neural network modeling has been supported by the progress in computer techniques and programs.

On the basis of results of long-term lysimetric studies on coal mining waste conducted by the authors in 19 years natural hydrologic cycle, it was found that for the prognosis of contaminant release from waste as a function of time and for assessment of a time span in which the concentration of leached contaminants will reach the permissible level (Directive of the Minister of Health, 2000), models of supervised neural networks of multi-layer perceptron (MLP), radial basis function (RBF) or Bayesian network (SPSS Inc., 1997, 1999) can be successfully applied.

Models of neural network were constructed with use of the Neural Connection v. 2.1 program (an example of a used simple supervised neural Bayesian network is presented in Figure IV.5.11). The details of models of supervised neural network construction and use

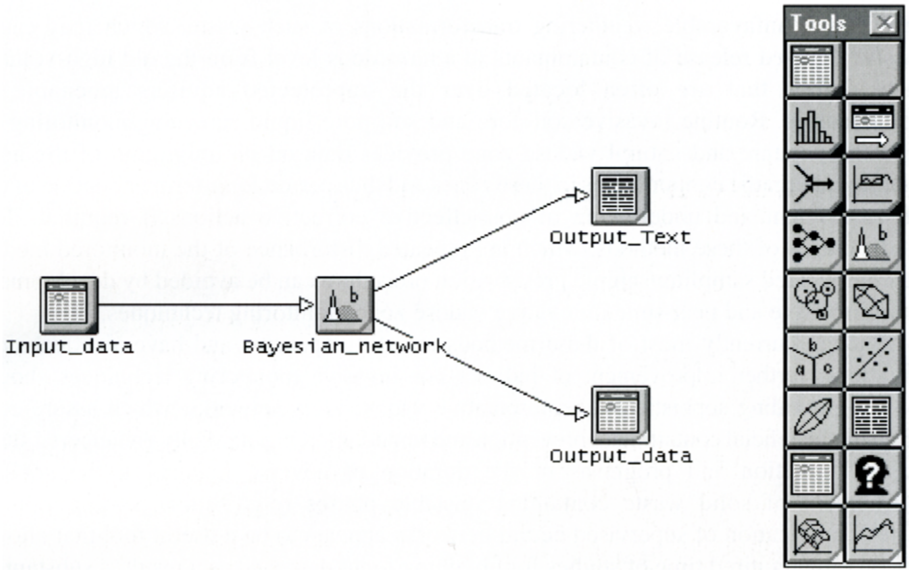

Figure IV.5.11. Scheme of a simple supervised neural Bayesian network. Screen projection from the Neural Connection program v. 2.1 (SPSS Inc., 1999).

for simulation of contaminant leaching from coal mining waste are discussed elsewhere (Kmiecik, 2000; Szczepańska and Kmiecik, 2001; Kmiecik et al., 2003).

The precision of prognosis appeared to be satisfactory for practical purposes, as the relative error does not exceed several percent. With use of neural networks, the time span in which the concentration of contaminants of interest will reach the permissible level or their release will terminate can be assessed, so that the groundwater deterioration as a function of duration of the waste disposal can be evaluated. The evaluation with use of this tool requires much lesser number of observations than in the traditional methods; thus it allows significant reduction of time and costs of a monitoring program.

IV.5.5. Concluding remarks

Monitoring of the vadose zone is gaining increased recognition in the regulatory and implementation arena as an instrument of early detection and subsequent prevention/interception/attenuation of aquifer contamination in the most cost-effective and efficient way. It should be considered complimentary to saturated zone monitoring as a warning system indicating an alarming extent of contaminant migration to the aquifer, while saturated zone monitoring provides data on the actual status of the groundwater quality in the SWMU sites.

In the USA, regulatory amendments of the U.S. EPA (1988–2003) require vadose zone monitoring at RCRA hazardous waste disposal sites on a case-by-case basis, there is though still lack of regulations in this respect in the EU and other countries. The authors' experience in the studies of non-hazardous waste disposal utilities shows that anthropogenic vadose zone (waste layer) screening/monitoring can be particularly useful in the detection of the eventual unfavorable weathering transformations of such waste, which may cause massive delayed release of contaminants in a hazardous level from the old high-volume waste dumps that are often located over the unprotected aquifers amenable to contamination. Routine invasive soil-core and soil-pore liquid screening/monitoring of the anthropogenic and natural vadose zone provides data on the migration of the non-organic and organic contaminants in these zones and thus permit long-term validation of the risk assessment and undertaking of cost-effective corrective actions if required. The disadvantages of these methods, which are repeated disturbance of the monitored media, and complicated sampling/sample preservation procedure can be avoided by development of non-invasive and near-time innovative vadose zone monitoring techniques.

Although currently most of these methods are still expensive and have also technical limitations, further improvement of indirect non-invasive monitoring techniques should result in providing sophisticated representative vadose zone networks, which assure economically beneficial contaminant prevention and remediation in potentially endangered sites.

For simulation and prognosis of the duration of adverse impact on the aquatic environment of solid waste containing unstable phases (e.g. sulfide-bearing mining wastes), application of supervised neural networks appears to be a useful tool that allows reducing the required time of studies for obtaining input data, and as a result, a substantial reduction of costs for vadose zone monitoring.

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