

**Studies of colloids and their
importance for repository
performance assessment**

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This report concerns a study which was conducted for SKB. The conclusions and viewpoints presented in the report are those of the author(s) and do not necessarily coincide with those of the client.

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ABSTRACT

English

The processes, parameters and data used to evaluate the potential of nuclide transport by a colloid facilitated mechanism are reviewed and discussed in this report. Both steady-state (present situation) and possible future non-steady-state (changing situation) hydrogeochemistry in the geosphere are covered.

In the steady-state scenario, the colloid (clay, silica, iron(III) hydroxide) concentration is around $20-45 \mu\text{g}\cdot\text{l}^{-1}$ which is considered to be a low value. The low colloid concentration is justified by the large attachment factor to the rock which reduces the stability of the colloids in the aquifer. Both reversible and irreversible sorption processes are reviewed.

In the non-steady-state scenario, changes of hydrogeochemical properties may induce larger colloid concentrations. This increase of concentration is however limited and relaxation is always observed after any change. Emphasis is placed on the glaciation-deglaciation scenario.

Swedish

Kända processer, parametrar och data har använts för att utvärdera kolloidtransport både under nu rådande och framtida föränderliga grundvattenförhållanden.

Under rådande förhållanden är kolloidhalten av storleksordningen $20-45 \mu\text{g}\cdot\text{l}^{-1}$. Kolloiderna består av ler-, kisel- och järnhydroxidpartiklar. Den låga kolloidhalten förklaras av att kolloiderna fastnar på bergytter. Både reversibel och icke reversibel sorption har undersökts.

Om grundvattenförhållandena förändras kan stora mängder kolloider bildas. Denna ökning är dock tidsbegränsad och en relaxtionsfas inträder relativt snabbt. Dylika förändringar kan uppträda t.ex. i samband med isavsmältning efter en glaciation.

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SUMMARY

This report evaluates colloid data and processes which can be used in radionuclide transport modelling in the Swedish performance assessment of a high level waste repository. Two different scenarios in the geosphere are discussed: a steady-state situation and a non-steady-state situation in the aquifer including gradient and temporal changes of hydrogeochemical conditions.

In the steady-state scenario, the Swedish granitic saline and non-saline groundwaters have a median colloid content of 20-45 $\mu\text{g}\cdot\text{l}^{-1}$ for 50-500nm colloid sizes respectively. This result which has been corrected for disturbance (artefact) of precipitation of calcite and sulphur compounds is comparable to the Swiss colloid concentration in deep granitic groundwaters. The Swedish colloids consist of clay, silica and iron(III) hydroxide colloids. Their stability and transport properties, which are closely linked, are discussed in this report. In relatively saline water, the expected attachment factor α of the colloids to the host rock is relatively high, justifying their low concentration. The sorption properties of safety relevant nuclide on the colloid phase are considered in this report. Reversible sorption is limited, with distribution coefficient values K_p of around $10^6 \text{ g}\cdot\text{ml}^{-1}$ for 20nm colloids. However, irreversible sorption may take place eg, on clay colloids. This new data (eg, the new colloid concentration of 20 $\mu\text{g}\cdot\text{l}^{-1}$ which is one order of magnitude smaller than that estimated earlier) may be used for the safety assessment study in a steady-state scenario.

In the non-steady-state scenario, the impact of physical and/or chemical changes and gradients in the aquifer on the colloid concentration is discussed. Examples of single processes are presented. Coupled processes are also studied with emphasis on the glaciation-deglaciation scenario. In this context, the effect of water flow rate changes (physical) coupled with salinity changes (chemical) are considered. Changes of colloid concentrations as a result of a potential meltwater inflow cannot be modelled. In the field and laboratory where large changes of hydrochemical conditions may be tested, their effect on the colloid concentration increase is limited to one or two orders of magnitude. In all cases, after such a colloid concentration increase, a relaxation is always observed which brings the colloid concentration back to the same level of concentration that was observed before the disturbance. The new concept of a colloid concentration peak (the same colloid concentration as used in an early study ie, 400 $\mu\text{g}\cdot\text{l}^{-1}$) as a result of the inflow of meltwater in the geosphere may be utilised for the safety assessment calculation during the meltwater infiltration period only.

With the influx of glacial water, it is expected that the dilution of the deep groundwater is not affected by more than 50% around the repository. In these conditions, the calcium concentration of the mixed water at the near-field/far-field interface is not expected to decrease below 10^{-4} molar. It will always

remain large enough and the attachment factor α of the bentonite particles will also remain large (around 1). The influx of glacial water will not be conducive to the re-suspension of the bentonite in the far-field. During deglaciation, a maximum increase of one order of magnitude of the original colloid concentration may be expected if the gradient and changes are sharp and large enough to destabilise the attached colloids in the aquifer. Even if such a smooth peak in colloid concentration did take place it would be limited in time and space. The original colloid concentration ($20 \mu\text{g}\cdot\text{l}^{-1}$) will always be restored after such an event.

1. INTRODUCTION

A general scenario in all safety assessment studies is that radionuclides released from the waste matrix may contaminate the groundwater at trace level (eg, 10^{-10} molar). Trace element behaviour in water may be affected by their association with the colloid phase. The mobility and apparent solubility of these nuclides could then be enhanced by their speciation changes and their association with colloids.

Since **colloids are present** in all waters, only groundwater colloids were taken into account in the initial performance assessment studies. Colloid facilitated transport mechanisms were evaluated considering the geosphere as a source of colloids with constant concentration in the groundwater. Their presence is a consequence of solid - water interaction, and they are generated when rocks interact with the groundwater. An attempt to correlate their concentration and their stability is currently in progress (Degueldre, 1995). These studies are difficult because colloid sampling is not always properly carried out and because the uncontaminated colloid concentration obtained may also be a function of the history of the aquifer prior to sampling (McCarthy and Degueldre, 1993). Numerous colloid data are systematically affected by interfering phases such as precipitates which are generated during sample preparation or by additional components induced by perturbation in the aquifer during water collection. These perturbing phases are often erroneously measured as colloids and thereby increase the concentration recorded. They are called **artefacts**.

In order to assess the colloid concentration results, **colloid stability** was studied to justify their concentration value in groundwaters (Liklema, 1978; Degueldre, 1995). It was shown that the more stable the colloids, the more they occur. Colloid stability may be quantified experimentally by studying the effect of pH or ion concentration on model colloid populations. It may also be estimated by using the DLVO (Dejarguin-Landau-Verwey-Overbeek) theory (Fusch, 1934; Ruckenstein and Prieve, 1973) or by applying other advanced theories such as Secondary-Minimum-Monte-Carlo-Simulation (SMMCS) (pers. com. O'Melia, 1994). However, the use of these theories is restricted and it remains difficult to explain real systems such as colloids in the hydrogeochemical environment (Grauer, 1993).

Colloid mobility is required to allow a colloid facilitated transport of radionuclides which is relevant to safety assessment. This is a function of the pathways and porosity of the system (Toran and Palumbo, 1992) as well as of both colloid and groundwater compositions.

The groundwater colloids themselves are not important for a performance assessment. It is the transport in their contaminated form which is important. The contamination of the colloids is a result of the sorption of relevant radionuclides onto them. The **sorption processes** onto the colloids must therefore be understood. In mineralised groundwater, surface complexations

are the strongest sorption mechanisms because ion exchanges of radionuclides compete with that of inactive ions. The reversibility of the uptake process is of prime importance because the contaminated colloids when they reach the clean aquifer can decontaminate and then be transported further in an uncontaminated form.

The literature and **preliminary model** studies for safety assessment available today mainly deal with a constant colloid concentration in the aquifer (van der Lee et al., 1994; Vuorinen, 1987; Smith and Degueldre, 1991; Grindrod and Worth, 1990).

A global conservative migration scenario describing water in contact with the waste and its migration through the far-field was presented by SKB (Allard et al., 1991). It takes into account:

- the irreversible sorption of nuclides on groundwater colloids present at a constant concentration of $100 \mu\text{g}\cdot\text{l}^{-1}$ and a maximum concentration of $400 \mu\text{g}\cdot\text{l}^{-1}$,
- a colloid retention of zero (no colloid attachment on the host rock),
- K_p values for the sorption of nuclides onto these colloids are however estimated on the basis of K_d values for the sorption of these nuclides onto larger particles,
- the water flowing straight to a well where the water is directly used for human consumption.

For this scenario, a dose was calculated for each nuclide leached from a typical SKB waste package. It was shown that the individual dose for humans drinking this well water is below the legal concentration limit.

In laboratory and field tests it was observed that the colloid concentration may be enhanced temporally when the hydrogeochemical **conditions change**. For example, when the groundwater pumping rate changes or when a salinity gradient is imposed through a porous rock sample, particles are freed in the collected water. This has been neglected in earlier safety studies. A novel approach in this performance assessment study is to consider transient phenomena, bearing in mind that the mechanisms of colloid generation include coupled processes induced by chemical and physical gradients or changes. Each interface eg, matrix-canister, canister-bentonite, bentonite near- and/or far-field is also affected by gradients of temperature, pressure, and chemical components (pH, pe). In this study the colloid generation is actually considered between the near-field/far-field interface, and, in the far-field (rock/water interface). The bentonite barrier is assumed to act as a colloid filter for the corrosion generated colloids (from the repository waste and the canister).

Since the deep aquifer can be affected by water mixing processes as a consequence of **glaciation-deglaciation**, the input of fresh water in the geosphere should change the ionic strength of the water. At the near-field/far-field interface this change may therefore generate a suspension of the

bentonite particles from the near-field. Furthermore, canister corrosion may induce gas production. Thus the recent concept of colloid transport at the gas-water interface must also be considered. These approaches must be discussed in a realistic way and the limitation of each process must be carefully evaluated.

The first part of this report reviews the current Swedish colloid results using statistical analysis to track artefacts and to derive in situ colloid concentrations. These results are subsequently compared to those reported in international colloid studies. This work is relevant for performance assessment because it considers accurate colloid data and discusses processes which stabilise colloids or enhance their generation and transport properties during steady-state conditions. The impact of non-steady-state situations in the aquifer on the colloid population is evaluated and discussed in the second part of this report.

2. COLLOIDS IN THE GEOSPHERE, STEADY STATE APPROACH

Constant groundwater flow and no variations in the concentration of groundwater components are assumed in this Section. Colloids from the near-field are not discussed here.

2.1 SWEDISH COLLOID RESULTS

2.1.1 Methodology and data production

Colloid sampling and characterisation was recently reviewed by McCarthy and Degueldre (1993). It was pointed out that sampling must be optimised in all cases prior to characterisation. Today various techniques (including single particle counting to colloid bulk analysis) are available involving on-site sampling and characterisation on-line or off-line. For example, samples can be taken on site and characterised directly at the site by light scattering. Alternatively samples can be separated by filtration and filters can be sent for characterisation to external laboratories. In all cases artefact tracking is required (see also Degueldre et al., 1990) for example, the volume of the groundwater sample and how this affects the colloid concentration. The analytical procedure of the characterisation must be sensitive enough to detect elements at trace level (eg, ICP or XRF).

Deep granitic groundwaters in Sweden have been analysed for a period of ten years for their content of natural colloids using an on-line filtering system isolated from the influence of the atmosphere.

The colloid concentration of deep groundwaters is usually around the $\mu\text{g}\cdot\text{l}^{-1}$ level and thus the contamination risks are obvious during water sampling and colloid sample preparation. "Unnatural" particles can be generated, even proceeding to on-line sample preparation. The sampling errors are due to: excessively high or low pump rates, contamination from borehole activities, complex hydrological situations, contamination from tubes varying the composition of the groundwater, air contamination (O_2 uptake), losses or uptake of CO_2 , aggregation-coagulation effects, long storage times prior to analyses, analytical errors etc. Some errors in the sampling and analysis of colloids are easy to avoid, others are difficult or impossible to evaluate. These problems have been addressed among others by Laaksoharju et al. (1994).

The method generally used in the Swedish program has been **on-line filtering**. The same volume of water was passed through each filter in the order of 400, 200 and 50nm. Filters, particularly the finer ones, often became clogged and this created problems during sampling and adversely affected the size distribution. The filters used were made of polycarbonate. The Al, Si, S, Ca, Mn and Fe contents were determined with energy dispersive XRF (X-Ray Fluorescence). The quantities of analysed elements on the filters were recalculated in $\mu\text{g}\cdot\text{l}^{-1}$ (ppb) considering the water flow through the filters. The

final element concentration was corrected for any enhancement of a given element, obtained by running reference blank filters. Improved methods including N₂-atmosphere, smaller filtered volumes, single filters, particle counting etc. has been used recently in the Swedish programme (Laaksoharju et al. in Banwart ed., 1995). The results from colloid analyses evaluated here are mostly from earlier SKB campaigns. All information has been extracted from GEOTAB, the SKB database.

2.1.2 Statistical modelling of the existing inorganic colloid data

GEOTAB contains geological, hydrological and geochemical data from the last 10 years of investigations. The groundwater data from the following investigation sites have been compiled: Taavinunnen, Kamlunge, Gideå, Svartboberget, Finnsjön, Fjällveden, Kråkemåla, and the Äspö area. From this material only Finnsjön and the Äspö area contained complete colloid data and stable isotopes in the groundwaters. The material used contained data from 10 different boreholes, 5 different places (Laxemar, Ävrö, Äspö, Finnsjön and the Redox zone), 28 different observations and a total number of 84 analyses consisting of: Al, Ca, Mn, S, Si and Fe of the colloid phase isolated on the filters. These samples were generally produced during the last 5 years and were of a higher quality than the earlier sampling. The analytical material used is depicted as matrix scatter plots in Appendix 1.

In order to sort the colloid material, various groundwaters have to be classified. Principal Component Analysis (PCA) which is a standard multivariate technique was used. The strength of this approach is that several or all variables in a data matrix can be examined simultaneously. Greater resolution is possible and the character of the data in a general data matrix is therefore more easily identified than using univariate analysis, where only one variable is compared at a time (Wold, 1987). This technique has been used earlier at Äspö to classify and compare groundwater data (Laaksoharju and Nilsson, 1989; Laaksoharju, 1990; Smellie and Laaksoharju, 1992). The major element concentrations: Cl, Ca, Na, Mg, K, SO₄, HCO₃, and the isotopes ²H, ¹⁸O and ³H have been used in the Principal Component Analysis of the data material. These components are known from other studies to describe most of the variability in the data. The result of the PCA-analysis and the identified end-members is shown in Figure 2-1.

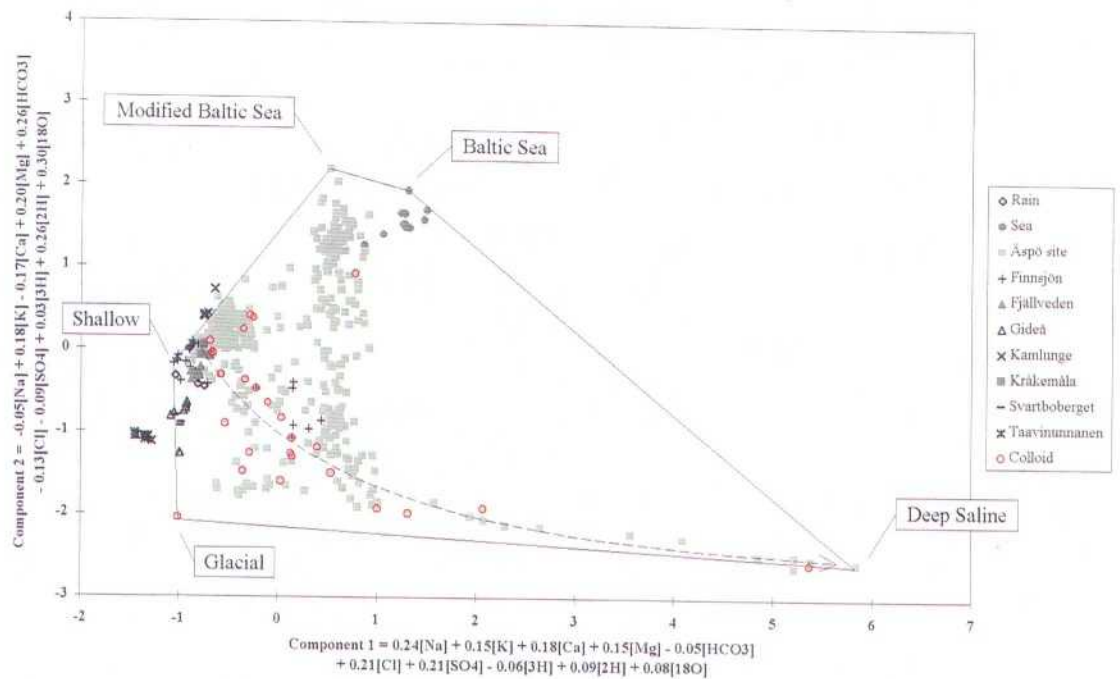


Figure 2-1. Principal Component plot based on the major element concentrations and isotopes (^2H , ^3H , ^{18}O) from GEOTAB. The internal relationship of the main groundwater types found in Sweden is shown. The arrow indicates the evolution path that contains most of the colloid sampling (along the shallow-deep saline groundwater). The weights for the different terms are shown in the equations for the first and second Principal Components respectively. These components together account for 70% of the variability, or the information in the data.

The selected end-members in Figure 2-1 represent extreme waters found in Swedish granitic formation. The **Glacial** end-member has been determined as an old glacial water based on the stable isotope values which indicate cold climate recharge ($^{18}\text{O} = -15.8$ SMOW and $^2\text{H} = -124.8$ SMOW) in combination with an apparent ^{14}C age of 31,365 years (Smellie and Laaksoharju, 1992). This type of water is found in Äspö KAS03:129-134m. **Deep Saline** water represents the brine type of water found in KLX02:1631-1681m (Laaksoharju et al., 1995), **Baltic sea** water represents modern Baltic sea water (SEA01). **Modified Baltic Sea** water represents an older than modern Baltic sea water obtained in the HRL tunnel below the modern Baltic sea. In this water sulphate reduction may occur (Laaksoharju ed., 1995). The **Shallow** waters are represented by the water found in HBH02:7.5m. The selected end-members are believed to best represent waters that interact in different degrees and portions in groundwaters sampled in Sweden. The arrow in Figure 2-1 indicates the groundwater evolution path that contains most of the colloid sampling information along the shallow-deep saline groundwater flowpath. This flowpath follows the first Principal Component which has a high loading for Cl. This simply means that the colloid observations can be best separated by using the salinity in the groundwater. The sum of the elements associated with the collo

content in the groundwater (Figure 2-2). This confirms that the colloid material is correlated with the salinity of the groundwater. There seems to be an increase in the colloid content in many of the observations when the salinity of the groundwater is higher than 2000 mg·l⁻¹. The material has therefore been subdivided into non-saline (Cl <2000 mg·l⁻¹) groundwaters and saline (Cl >2000 mg·l⁻¹) groundwaters. The median, minimum, maximum, lower quartile and upper quartile has been calculated for all observations of the non saline and the saline waters in Appendix 2.

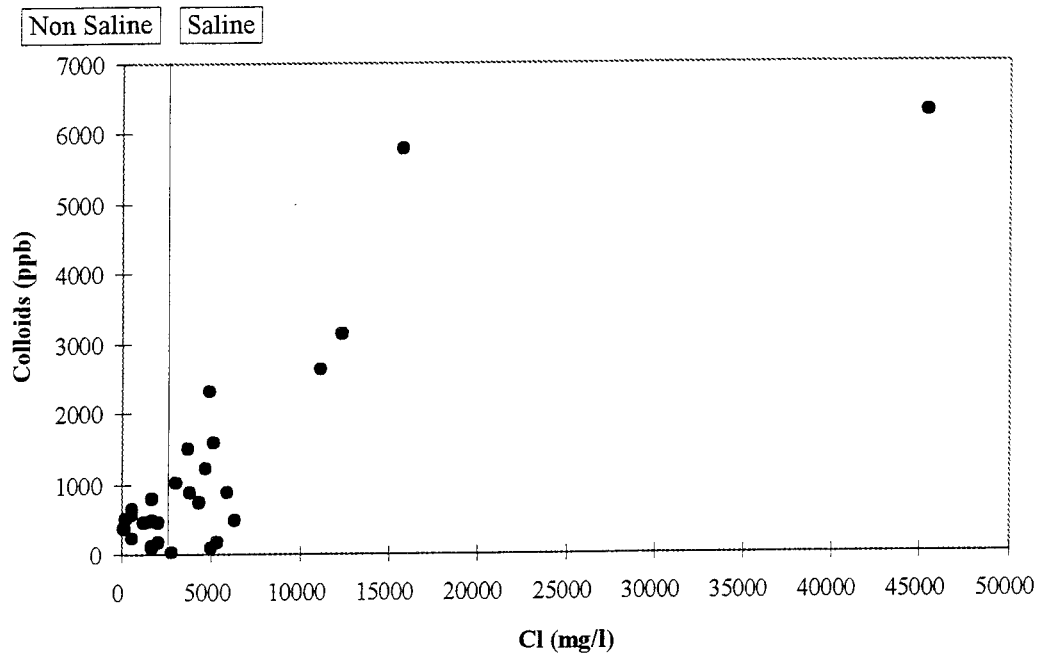


Figure 2-2: The sum of the elements associated with the colloid phase is plotted against the Cl content in the groundwater.

The matrix scatter plots in Appendix 1 are an excellent tool to analyse the data with. As an example, some important variables for this study have been extracted from Appendix 1 and are shown in Figure 2-3. This information can be used to demonstrate that the increase of colloid concentration with the salinity of the groundwater is due to calcite precipitation. It is done as follows. First it can be noted that the Ca on the filters (Ca^P where ^P = particle bound) correlates with the Cl (see Cl/Ca^P). The Ca^P correlates with depth, and waters containing high Ca^P are generally sampled at large depths (>500m) (see Depth/Ca^P in Figure 2-3). Exposing a groundwater from a borehole to atmospheric pressure can result in significant gassing/degassing (eg, CO₂) with subsequent changes in the pH and associated changes of the chemical speciation of cations capable of forming carbonate complexes. The partial pressure of carbon dioxide is low in high saline groundwaters and contamination with atmospheric gases can subsequently change the pH which then may cause calcite precipitation (see CO₂_G/Ca^P in Figure 2-3). The calculated saturation index for calcite based on downhole pH measurements has a low correlation with the Ca measured on the filters (see Calcite/Ca^P in

Figure 2-3). The calcium in the groundwater and on the filter shows a higher degree of correlation (see Ca/CaP in Figure 2-3). This indicates that Ca in the water is the main source for calcium on the filters. The calcium content on the filters is therefore not natural and should be regarded as a sampling artefact.

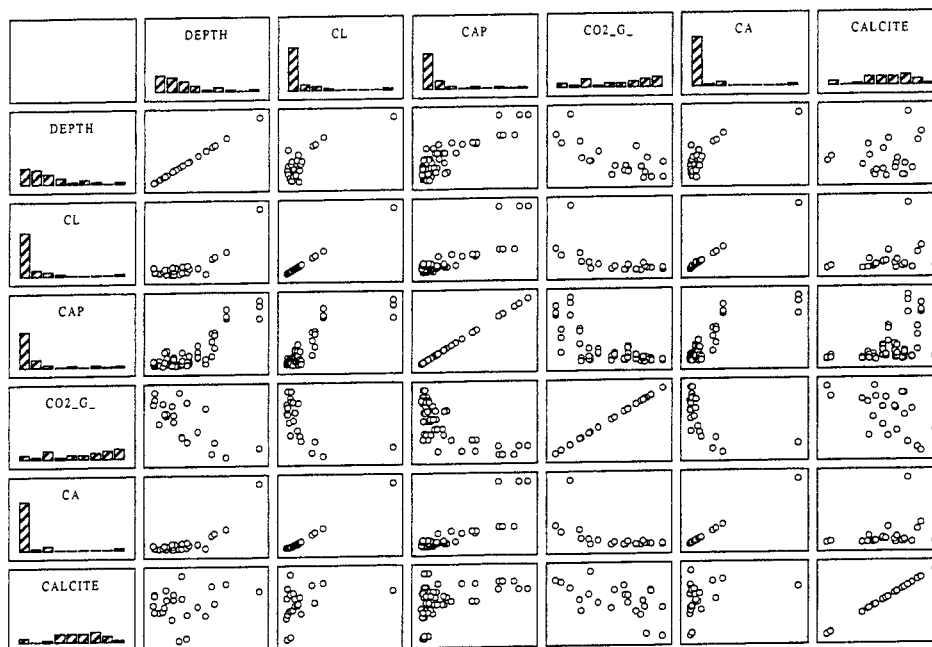


Figure 2-3: Correlation between different variables shown as matrix plots extracted from Appendix 1. Calcium concentration associated with the particles (CaP), depth (DEPTH), partial pressure of carbon dioxide (CO₂G), calcium concentration in the water (Ca) and the calculated saturation index for calcite (CALCITE) is shown.

When tracking artefacts using both statistical and PHREEQE (Parkhurst et al., 1987) approaches, it appears that calcite precipitation may be due to sampling artefacts rather than due to mixing with water which is also known to result in calcite precipitation. Mixing should be of less importance since most of the Ca is found in deep stagnant groundwaters and not in shallow or intermediate waters. Chemical changes due to oxidation (eg, HS⁻ yielding SO₄²⁻ and/or colloidal sulphur), can induce sulphate and sulphur precipitation as well as sulphide dissolution. Fe(OH)₃/FeOOH precipitation can also be due to redox changes which could be an effect of sampling (eg, air intrusion, oxygen retention in the equipment).

2.1.3 Results and discussion

The following scoping calculations were made in order to establish a colloid concentration in the non-saline and saline waters found in Sweden. It was assumed that the analysed elements were associated with the following phases: Ca with calcite (calcium carbonate), Fe with goethite (iron oxy-hydroxide), S with pyrite (iron sulphide), Si with quartz (silica oxide), Mn with pyrolusite (manganese oxide) and Al with Al-K-Mg-illite (clay mineral). These phases are expected to constitute many of the colloid phases found in

granitic groundwater in Sweden. However, SEM/EDS measurements have indicated that the colloids are often conglomerates of phases containing different minor elements (Al, Mg, Fe and S) in association with Ca and Si. A value of the in-situ colloid concentration may still be calculated using the above phases, but the effect from known artefacts must first be compensated for. The major artefact calcite precipitation, may cause co-precipitation of other elements which causes the measured concentration of colloids to be overestimated. One way to make a reasonable and still conservative estimation of in-situ colloid concentration is to omit the load from calcite precipitation.

The second highest colloid load is from S which is also believed to be an artefact. One reason may be due to the overestimation of *SP*. High sulphur content in the colloid phase is measured in the deep groundwaters and in the shallow groundwaters (Figure 2-4). In the deep groundwaters sulphide present in water oxidises rapidly on the membrane to elemental S and sulphate, producing a covering of sulphur polymer on the filter. In the shallow groundwater which is still anoxic, similar processes may occur although natural oxidation in the upper part of the aquifer cannot totally be ruled out. In the Baltic sea iron sulphide $\text{FeS}_{0.86}$ formation has been discussed by Bågander and Carman (1994). An influx of Baltic Sea water has been found at many locations at Äspö (Laaksoharju and Skårman, 1995). This influx could cause a higher sulphur content in the colloid phase. However, one colloid observation in the near vicinity of the Baltic Sea (Figure 2-4) did not support this. This may support the idea that the possible iron sulphide colloids seem to stick to the rock matrix rather than stay in solution. Finally, if S was present as iron sulphide (troilite or pyrite) a correlation between *FeP* and *SP* (*P* = particle bound) would have been found in Appendix 1. This is not the case however. In addition *SP* values decrease with volume, which indicates an artefact. It must also be noted that traces of non-colloidal sulphate obtained during drying the membrane may also contaminate the membrane prior to XRF analysis.

The load from S on the colloid phase has therefore been diminished in the sum of final particulate material content. Further studies are needed to fully clarify the role of the sulphur-containing colloids.

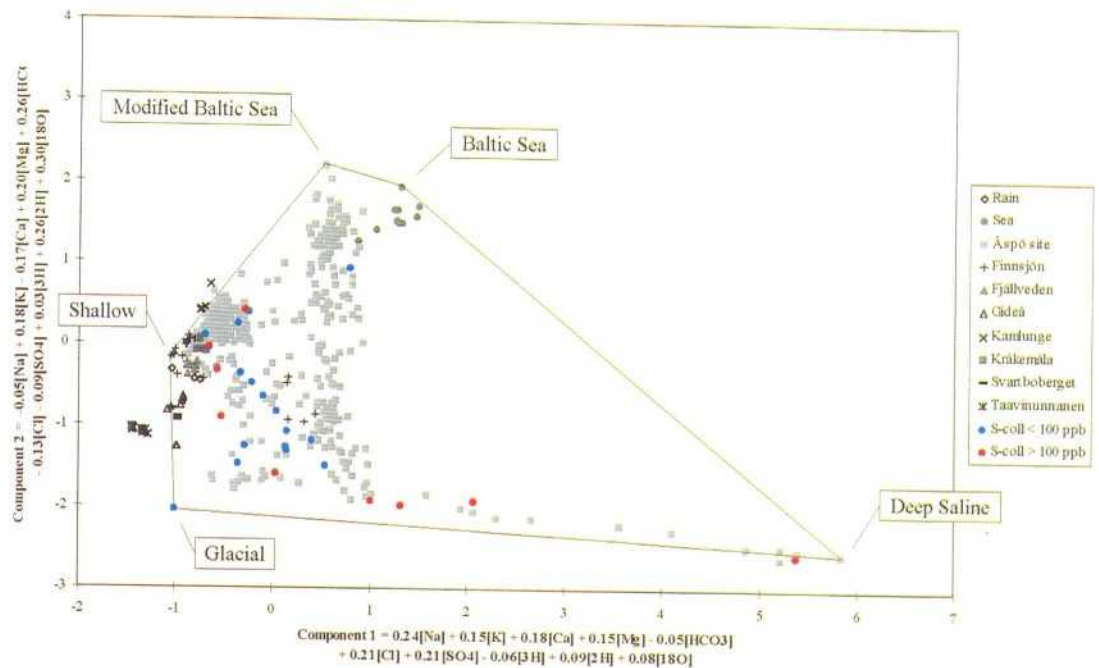


Figure 2-4: The Principal Component plot has been used to depict groundwaters with high sulphur content ($S\text{-coll} > 100\text{ppb}$) in the colloid phase and groundwaters with low sulphur content ($S\text{-coll} < 100\text{ppb}$). High values are obtained in deep and shallow groundwaters, low values in observations affected by the influx of Baltic Sea water.

The calculated colloid concentration, for the non-saline and the saline groundwaters including and then omitting the effect from calcite precipitation and the pyrite component, is presented in Table 2-1. It must be noted that this is coherent with the coagulation theory which foresees a larger colloid concentration in the less saline water.

Table 2-1: Results of element analyses (Al, Ca, Fe, Mn, Si and S) when sampling non-saline and saline groundwaters in Sweden and the calculated colloid concentrations assuming the colloids form distinct phases. The final colloid concentration is calculated by omitting the load from calcite, identified as a major artefact and S (ie, pyrite) as an oxidation artefact.

	[Coll] Non Saline Groundwaters			[Coll] Saline Groundwaters		
	min	max	median	min	max	median
General Conditions						
Cl in the groundwater (mg/L)	73	2010	1455	2800	45500	5035
Depth (m)	70	801	274	50	1563	496
Element analysis (ppb)						
Filter pore size (nm)	50-400	50-400	50-400	50-400	50-400	50-400
Al	0.06	13.00	1.35	0.10	89.00	0.60
Ca	27.50	251.58	78.12	2.10	2330.23	302.50
Fe	0.03	60.00	6.02	0.10	848.00	4.35
Mn	0.03	1.50	0.68	0.08	6.50	0.60
Si	0.60	49.76	5.03	0.10	47.00	1.30
S	0.48	243.47	29.30	0.10	108.15	23.00
<i>Sum</i>	29	619	121	3	3429	332
Calculated colloid phases (ppb)						
Ca as Calcite CaCO_3	68.7	628.2	195.1	5.2	5818	755
Fe as Fe(OH)_3	0.0	114.8	11.5	0.2	1623	8
S as Pyrite FeS_2	1.8	911.0	109.6	0.4	405	86
Si as SiO_2	1.3	106.5	10.8	0.2	101	3
Mn as Mn(OH)_2	0.0	2.4	1.1	0.1	11	0
Al as K-Mg-Illite clay: $\text{K}_{0.6}\text{Mg}_{0.25}\text{Al}_{2.3}\text{Si}_{3.5}\text{O}_{10}(\text{OH})_2$	0.9	185.0	19.2	1.4	1266	9
<i>Sum (ppb)</i>	73	1948	347	8	9223	862
<i>Sum (ppb) omitting the Calcite load</i>	4	1320	152	2	3405	107
<i>Sum (ppb) omitting the Calcite and Pyrite load</i>	2	409	43	2	3000	21

Calculations in Table 2-1 show that the major elements forming colloids are, in descending order, calcite (artefact), sulphur, iron and silica. The conclusion is that the inorganic colloid contents in both saline and non-saline water systems have a median colloid content of around 20-45 $\mu\text{g}\cdot\text{l}^{-1}$ (min. = 2 $\mu\text{g}\cdot\text{l}^{-1}$, max. = 3000 $\mu\text{g}\cdot\text{l}^{-1}$) which is in accordance with other studies.

The colloids are composed of iron hydroxides, clay minerals (eg, illite, kaolinite and smectite) and silica/quartz. Humic substances can also appear as colloids or as coatings on inorganic particles. It is important to point out once again that it is difficult to correctly sample and analyse those colloids present in the groundwater. There is a tendency to generate particles by the sampling procedure and concentrations reported by us and others are generally overestimates. Colloidal species which may finally have a role in the migration of radionuclides on a local scale (1km) of the geosphere during thousands of years are the ones that are mainly considered. The colloids of interest for performance safety must therefore be stable and not formed during or after sampling. The ultimate aim is not to describe the colloid concentration in the sample, but rather how the sample reflects the undisturbed in-situ conditions. Therefore the load has been reduced from the artefacts in the final calculations above.

The results reported earlier as colloid concentration values (when the load from artefacts was not completely removed) and which were used in the performance assessment of spent fuel disposal in the SKB-91 study (SKB 91, 1992; Allard et al., 1991) were as follows: a central value of 0.1 $\text{mg}\cdot\text{l}^{-1}$ and a maximum value of 0.4 $\text{mg}\cdot\text{l}^{-1}$.

These values were used for the calculation of radionuclide transport by colloids. For the calculations it was also necessary to obtain values for the uptake of radionuclides on the natural groundwater colloids. The natural colloid composition was therefore used as a reference in laboratory experiments on radionuclide sorption on colloids.

2.1.4 Comparison with results from Canada and Switzerland

The Swedish colloid results may be compared with those obtained in granitic groundwater from other countries (see Table 2-2). Results from Canada and Switzerland were selected because these were artefact-free and the data was obtained after careful sampling. The median Swedish colloid concentration is comparable to that reported in the Swiss study for deep granitic groundwater and a concentration of 20-45 $\mu\text{g}\cdot\text{l}^{-1}$ (median, artefact-free) should be used in the Swedish safety assessment study as background concentration.

Table 2-2: Comparison of colloid concentration and composition from selected granitic aquifers - references: Sweden - this study; Switzerland - Degueldre, 1994; Canada - Vilks et al., 1991.

Sampling site	[coll]/ $\mu\text{g}\cdot\text{l}^{-1}$	size/nm	composition	pH	depth/m
Sweden					
Non saline	43 \pm 43	50-500	clay, silica, Fe(OH) ₃	8.0	275
Saline	20 \pm 20	50-500	clay, silica, Fe(OH) ₃	7.9	500
Switzerland					
Leuggern	30 \pm 10	10-1000	clay, silica	7.8	1650
Zurzach	10 \pm 5	10-1000	clay, silica	8.0	500
Canada	300 \pm 300	5-1000	clay, org., calcite	8.?	500

2.2 STABILITY AND MOBILITY

2.2.1 Colloid stability The presence or concentration of colloids in the groundwater is restricted by their stability. Colloids may aggregate or attach themselves onto the host rock (sticking). Colloid attachment may be quantified using the attachment factor α defined as:

$$\alpha = \text{frequency of attachment} / \text{frequency of collision}$$

The attachment factor varies from 1 when the attachment occurs for each collision, to 0 when the attachment is impossible. α may be estimated using the DLVO theory or the SMMCS approach. However, both estimates may be unrealistic (Grauer, 1992). The only important effect is the effect on Na and Ca concentration. This concentration may be similar on different scales and tends to increase α when the salt concentration increases (see Figure 2-5). Recently, it was reported by O'Melia (pers. com. O'Melia, 1995) that the attachment of latex colloids on quartz particles is reversible in Na solutions and irreversible (laboratory experiment) in Ca solutions. In addition it was also reported that TOC may have a strong effect on a colloid population, re-stabilising them in groundwater when TOC concentration increases (O'Melia and Tiller, 1993).

In reality, α increases when salinity increases, when the temperature increases, when TOC decreases, and when the pH decreases. Aggregation may also have a direct impact on the sorption properties (Degueldre and Wernli, 1993).

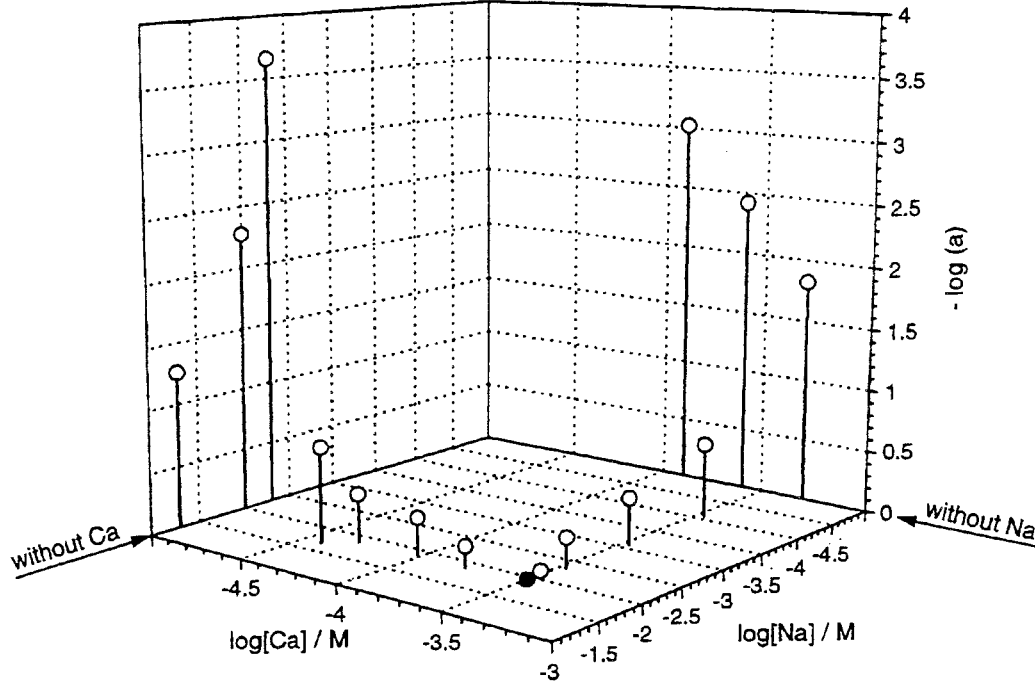


Figure 2-5: Effect of Na and Ca concentration on the attachment factor α of montmorillonite colloids. Conditions: $pH = 8$, colloid size $\geq 100nm$, $TOC < 10^{-6}M$, initial colloid concentration $10 mg \cdot l^{-1}$ (Degueldre, 1994).

2.2.2 Colloid mobility

In transport models, colloids are often assumed to move at the same speed as the water. Their retardation is then a consequence of their attachment on to the host rock of the aquifer which acts as a colloid filter. The kinetics of filtration in porous media was described by Yao et al. (1971) in Equation 2-1:

$$\frac{d[coll]}{dL} = -\frac{3}{2} \cdot \alpha \cdot \beta \cdot \frac{(1-\varepsilon)}{\varnothing_s} \cdot [coll] \quad 2-1$$

Where $[coll]$ is the colloid concentration for a specific size (eg, $pt \cdot l^{-1}$),

L the travel distance for the colloids,

\varnothing_s the size of the grain of the solid aquifer whose porosity is ε ,

and α and β describe the kinetics of the deposition. α is the attachment factor (it is a pure chemical parameter), and β is the probability of a collision (physical parameter).

Using the values of α , an evaluation of the propagation of the contaminated colloid front may be calculated. However, for safety reasons, the worst case ($\alpha=0$) is usually considered (no colloid retention in the aquifer).

Mobility tests were performed in the laboratory, for example, using a porous column (de Cayeux et al., 1990) or in the field (Harvey et al., 1989). Using Equation 2-1 fully quantified the transport of the colloids and determined the attachment factors.

A recent study suggested using a coupled processes model incorporating the colloidal phase (Ivanovich et al., 1993) to estimate nuclide transport by colloid transport mechanisms. The experimental study is, however, restricted to thorium pseudocolloids only.

2.3 MECHANISMS FOR UPTAKE OF RADIONUCLIDES ON COLLOIDS

2.3.1 Reversible sorption

In most of the Scandinavian granitic groundwaters the ionic strength is rather large ($>10^{-2}\text{M}$) and the uptake on colloids is mainly driven by surface complexation (Degueldre et al., 1994; Charlet et al., 1993; Bradbury and Baeyens, 1993) because the ion exchange (Vilks and Degueldre, 1991) is restricted by the large excess of cations in the groundwater. Recent work has suggested a pragmatic way to predict the sorption coefficients onto colloid surfaces coated with specific groups. In a natural system it is recognised that the dominant groups are $>\text{FeOH}$ and that the sorption properties may be modelled using these groups. Recently this approach was successfully used and K_p values were evaluated (Figure 2-6) on the basis of the thermodynamic properties of the species. A maximum K_p value of $10^5 \text{ ml}\cdot\text{g}^{-1}$ was calculated for colloids of 200nm in size; and for colloids of 20nm the maximum K_p value was around $10^6 \text{ ml}\cdot\text{g}^{-1}$. The latter value is recommended for further calculations. However, it is premature to assume that sorption is only reversible.

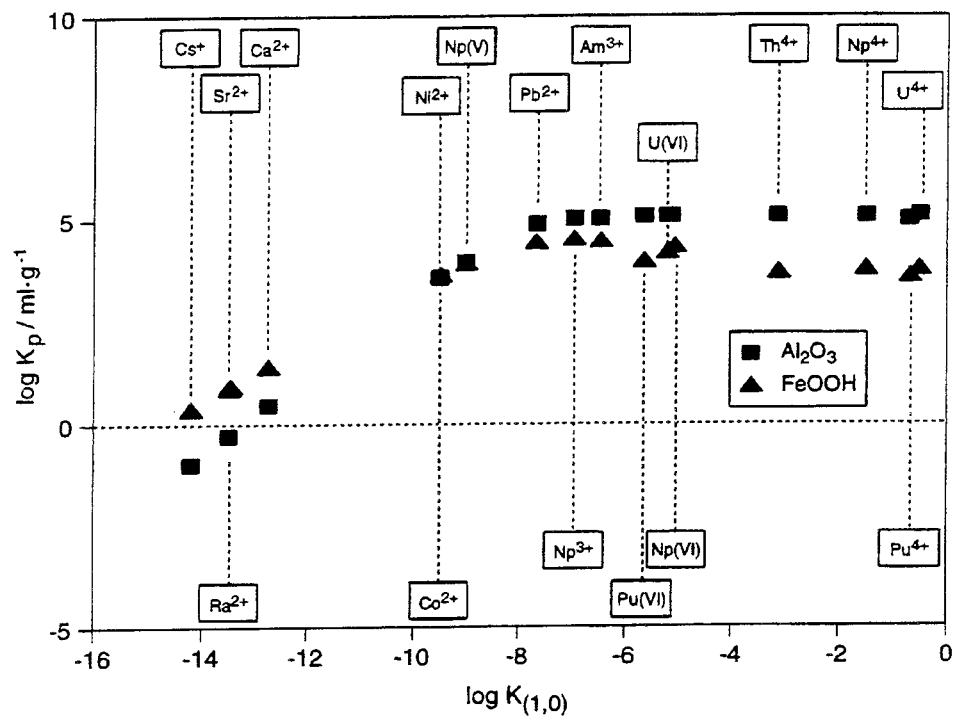


Figure 2-6: Calculated K_p values for nuclide species sorption on model colloids at pH 8 as a function of the first hydrolysis constant of these species. Conditions: colloid size 200nm, site density 3nm^{-3} , density $2\text{ g}\cdot\text{cm}^{-3}$ (Degueldre et al., 1994).

2.3.2 Irreversibility

Reversible uptake includes a desorption as a result of the sorption. If the kinetics of desorption are considered, the decontamination half-life has to be compared to the colloid (or more conservatively to the water) transport time. If irreversible sorption takes place, the mechanism responsible for this process has to be discussed. This could be due to:

- Sorption followed by colloid aggregation
- Sorption followed by chemical masking (eg, by coating)
- Sorption followed by incrustation
- Sorption into internal site followed by internal structural change (eg, in clay inter layer)

In porous media, when the colloid concentration is very low, the aggregation of colloids is unlikely compared to the probability of colloid sticking onto the host rock. This is true when the colloid surface is very small compared to the host rock surface. Coating and secondary phase formation on the colloid surface may be reversible. However, for the travel time scale, it is difficult to

quantify the process. Encrustation may be considered unlikely at low temperature (except when the nuclides are formed in the frame of radionuclide decay including recoil). The last process remains the more likely for irreversible sorption. However, other processes are also possible.

Sorption on calcite colloids may be irreversible because it is generally coupled with recrystallisation (fast). The sorption-recrystallisation coupling allows the nuclide to be built into the lattice of the carbonate where it cannot be easily desorbed. This situation is similar to sorption coupled to phase generation, it turns to coprecipitation if the saturation index of calcite increases. In this case the contaminated calcite colloids may be further stabilised by coatings (eg, iron oxide-hydroxide, organics) that yields mixed contaminated colloids (agglomerates). The situation is different on clay. For example transport of radioactive caesium by colloidal particles was found in the environment (Oak Ridge) as described by Sobocinski et al. (1990) and Cerling et al. (1990). Caesium may be irreversibly associated with the colloids. However, the transition time of the particles is around 10 years, which is too short compared to the geological time scale considered in this study. Finally, selective uptake may be noted for elements such as Se, Sn, I. This is due to synergetic effect, specific groups, and bioorganic processes. However, these mechanisms are unlikely to be found in a groundwater where the bacterial concentration is not expected to exceed $50 \text{ ng}\cdot\text{ml}^{-1}$ (SKB 91, 1992). In the case of irreversible uptake, it is not recommended to use the K_d concept for the sorption on colloids; only an uptake capacity, defined by the cation exchange capacity of the surface complexation capacity of the colloids (Degueldre et al., 1990), should be considered.

3. OTHER COLLOID GENERATION AND TRANSPORT MECHANISMS, NON STEADY STATE APPROACH.

In this Section variations in the groundwater flow rate and composition are anticipated. The possibility that colloids are generated in the near-field of a nuclear repository is also considered.

During a transient situation, colloid stability depends on the interaction changes between single particles with the aquifer solids (McDowell-Boyer et al., 1986; Lyklema, 1978). Destabilisation can occur due to compression of the double layer by increasing ion concentration, by pH-induced changes in surface charges, or by the presence of strongly binding ions that decrease the net charge of the colloids. Generally, any changes of pH, pe, ionic strength, T, or P induce phase changes and generation of these colloids. The gradients in an actinide contaminated far-field may also induce the formation of actinide polymers which may attach to other colloids and form new combined entities. Stabilisation may then take place when the chemical situation is normalised. When comparing the energy required for colloid attachment, it must also be noted that the kinetic energy of the colloids (associated with the water velocity) may have a direct impact on their sticking behaviour.

Consequently transport of colloids through porous or fractured media is enhanced by conditions promoting the stability of colloids or the generation of combined colloids. The presence of gradients (eg, temperature or concentration) at the canister/near-field interface, near-field/far-field interface, between two geological formations in the geosphere, may also be the source of colloids. In the Swedish scenario only one geological formation is considered and the bentonite clay buffer in the near-field is a total filter for colloids which may be generated from either canister or waste matrix (SKB 91, 1992).

The material in this Section was used to evaluate the effect on potential nuclide transport by colloid-facilitated mechanisms of the glaciation/deglaciation process expected to take place over the next 100,000 years.

3.1 PHYSICAL CHANGE

3.1.1 Change of flow rate

The water flow rate is a parameter that one can easily change at a borehole head. In the Cigar Lake colloid study (Vilks et al., 1993), the colloid concentration was observed to be constant in stationary conditions with a pumping rate ranging from 4 to 25 l·h⁻¹. Occasionally, colloid concentration increases were also noted during transient conditions. Independently, in a study at Wellenberg, the flow rate at the head of an artesian well was changed and observations in steady state situations indicated that the colloid concentration was not dependent on the water flow rate (50, 100 and 200 ml·min⁻¹). However, in a transient situation, corresponding to a flow rate from zero to 300 ml·min⁻¹, a colloid concentration peak was recorded. After a period corresponding to a flush-out of the water from the bore hole tubing and packer space, colloids generated by the change of flow rate in the aquifer were detected. It must be noted that the change induces an increase of colloid concentration of 1 to 2 orders of magnitude and that after the increase of colloid concentration, a decrease was observed corresponding to a relaxation down to the original concentration recorded earlier (Figure 3-1).

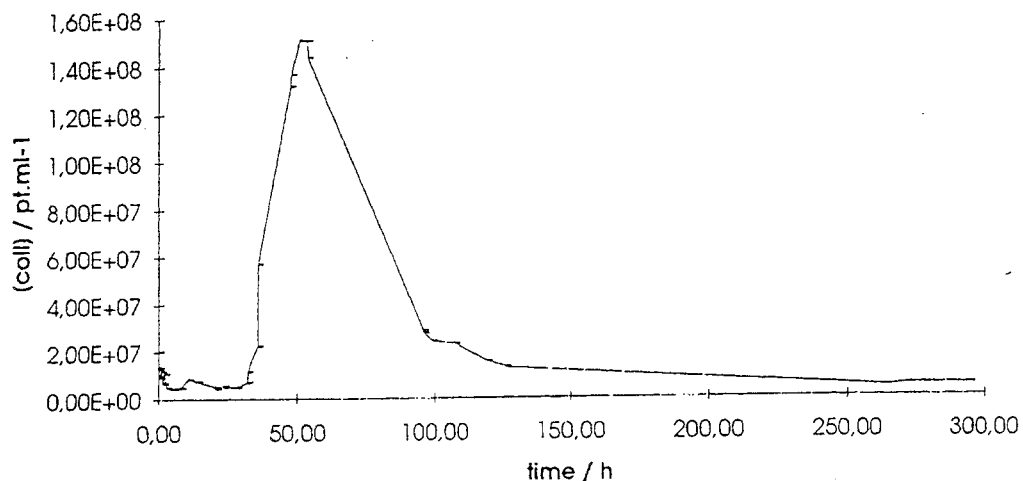


Figure 3-1: Changes of colloid concentration recorded at the well head of an artesian bore hole. Conditions: starting at time 0 with a flow rate of about 300 ml·min⁻¹. After 13 days of relaxation (no flow), cumulative colloid concentration for sizes >100nm, in well SB6 Nagra, in Wolfenschieden, Switzerland are shown. Note: the colloid concentration increased after 2 days corresponding to the colloid front due to erosion/re-suspension in the aquifer after the remaining water volumes from the tubing and interpacker space had been flushed out.

3.1.2 Change of spatial dimension

Pressure on the rock may result in fracture size variations and consequently in flow rate variations. These variations are closely connected because the fractures are large while the porosity of the water bearing zone is variable.

This must however be considered as part of a glaciation-deglaciation scenario, because the weight of the ice cover (1-2km) on the formation may cause pressure increases and subsequent fissure thickness reduction, reducing the water accessibility to the deep aquifer. This situation changes during deglaciation when the melting ice cover disappears, creating larger water flows (SKB 91, 1992) which could increase the generation of colloids in the groundwater.

Tidal effects are periodical and it is known that they induce pressure variation in crystalline shear zones (Frick et al., 1992). This may induce colloid generation by mechanical friction of the rock or by changes in the water flow rate in the water-bearing fissure. However, this effect is negligible in Sweden and it is not expected to intensify with time because the distance between the earth and the moon is increasing. Finally, tectonic activities are weak and are not a significant source of colloids in the Swedish groundwaters considered in this study.

3.1.3 Change of temperature

Since the ionic double layer thickness is a function of temperature, temperature changes also produce changes in the attachment factor (see Section 2.2.1). The attachment factor decreases when the temperature increases (Grauer, 1993). This is primarily due to the decrease of the double layer and the increase of the Brownian energy of the colloids ($k \cdot T$).

Results from field studies (Degueldre, 1994) indicate that the greater the difference between the geothermal temperature (Chalcedony geothermometer) and the temperature of the water at the sampling point, the larger the colloid concentration found. This is valid when the chemistry of two waters are comparable. This indicated that colloid generation also increased in these systems.

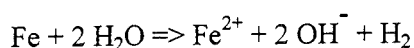
Temperature increases may be due to the presence of the radioactive waste package, hydrothermal activity or local magmatic intrusion. However, gradients due to these processes or activities are negligible or unlikely in Sweden. It is therefore reasonable to assume a constant temperature of about 15°C in the system from the near-field to the biosphere (SKB 91, 1992).

3.2 CHEMICAL CHANGES

3.2.1 Change of pH

A change of pH may induce colloids by generation or destabilisation. Sharp gradients in groundwater pH due to infiltration of acidic mining waste were associated with higher levels of colloids in the unconsolidated alluvium in Arizona (Puls et al., 1992).

The pH change may take place between the near-field (eg, the pH of bentonite pore water ranges from 8.6 to 9.2 according to Curti (1993), or from 6.8 to 9.3 according to MBT95 (1995)) and the crystalline formation (groundwater of the far-field is around 7.5). Since the pH changes from a higher to a lower pH value or is roughly invariant at the near-field/far-field interface, stabilisation is expected (Stumm and Morgan, 1981; Grauer, 1992). Therefore, this pH gradient is assumed not to contribute to the resuspension of clay particles from the compacted bentonite. Furthermore, if canister corrosion takes place, it is likely that the pH of the pore water increases according to the following reaction :



However, if the pH may then locally rise to 10 at the canister/near-field interface, it is unlikely that this pH will reach the near-field/far-field interface due to the pH-buffering effect of the bentonite. This pH change could enhance resuspension of the bentonite colloids in the groundwater. Even the geosphere waters are buffered and their pH is expected to remain around 8 even if diluted with an influx of fresh water.

3.2.2 Change of pe

A change of pe may induce colloid generation by decementation (Gschwend et al., 1990). This is mainly the case with particles originally cemented by iron(III) oxy-hydroxide which are liberated when the pe drops because the iron(III) cement is dissolved as iron(II). Even if iron(III) colloids were found in the Swedish groundwaters, this scenario would be unlikely because the aquifer studied by Gschwend was contaminated with organics and the reduction by hydrogen gas production in the Swedish geosphere is expected to be very slow (at 15°C).

3.2.3 Change of ionic strength

A change of salinity can drastically modify the colloid stability and induces a dispersion of these particles called "peptisation" in soil science (Khilar and Fogler, 1984). For example in column experiments, clay particles are eluted (see Figure 3-2) when the concentration becomes less than a marginal value concentration eg, $C = 0.16 \text{ M NaCl}$, see Fauré et al. (1994). The amount of particles leaving the matrix is only dependent on the initial and final

concentration of NaCl in the feed water. The particle concentration in the effluent depends on the volume used to vary the salt concentration. In systems fed with NaHCO_3 or Na_2CO_3 solutions, the effect on the marginal value is only slightly different. However, the release of particles due to these salinity changes induces dramatic changes in the hydrodynamic properties of the porous medium. Preferential pathways are observed as well as local clogging of the column. The concentration of colloids during these changes cannot be quantified analytically; only an empirical estimate depending on the porous medium (length, proportion of clay, pH, etc.) is suggested in the Faure et al. (1994) study.

A salinity decrease, for example, may be caused by the influx of glacial meltwater or by the influx of groundwaters of different origin and compositions. Such an event could in principle create a local and temporary increase of the colloid concentration of one or two orders of magnitudes. It must be noted that the tests were carried out with Na solution and not Ca solution which cements the colloids.

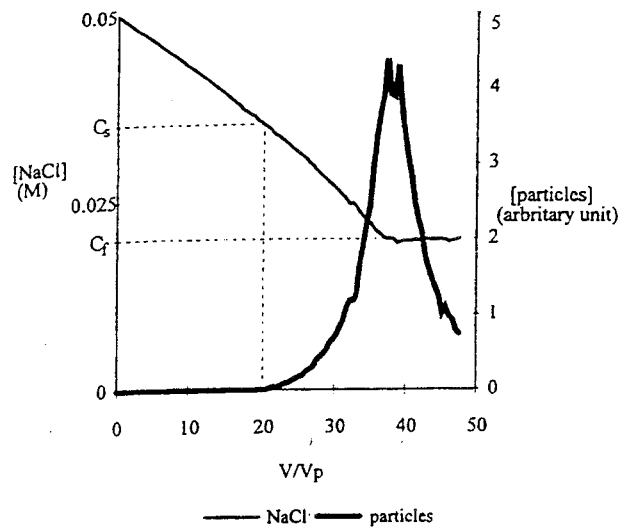


Figure 3-2: Influence of salinity concentration on colloid concentration. Conditions: elution of clay colloids through column with Na solution gradient (Faure et al., 1994).

3.3 EFFECT OF COUPLED PROCESSES

3.3.1 Coupling of physical processes

Physical processes are occasionally coupled together and their complementary action contributes to colloid generation. An example of coupled physical processes with a possible effect on colloid generation is the variation of ice sheet thickness affecting the pressure on fracture zone thicknesses and, consequently, on flow rate variation. These phenomena are coupled together and are discussed in a separate section.

3.3.2 Coupling of chemical processes

Dilution may affect the pH and salt concentrations and thereby influence colloid stability. These processes complement each other. In addition intrusion of oxic fresh water can increase the Eh and generate Fe(III) hydroxo colloids. Introduction of fresh water may also modify the rock water interaction processes and reequilibration may induce colloid generation. However, these processes are balanced by the dilution of fresh colloid-free water.

Coupling chemical and physical processes

Since the ice melting in the glaciation-deglaciation scenario includes major water input and flow rate modifications, it must also include the redox, ionic strength and pH changes associated with these physical variations. In addition, a more complex coupled process or mechanism may also take place independently of the deglaciation scenario.

Another example is the effect of colloid transport or remediation by gas bubbles which was recently studied by Wan and Wilson (1992, 1994). These authors observed the attachment of clay colloids at the gas-water interface. A gas source may be due to the corrosion of the canister; degassing is also possible by pressure release (upward flow, large pressure at depth vs. near the surface) or may also be due to an increase in temperature (eg, higher temperature due to the source around the near-field). In the Swedish scenario the latter effect is likely to be negligible (SKB 91, 1992).

A conservative scenario has been evaluated (Neretnieks and Ernstson, 1994) where all the gas produced by canister corrosion forms small bubbles and their surface becomes covered by a monolayer of colloids emanating from the bentonite buffer saturated with the nuclides; the colloids attached irreversibly to the gas bubbles migrate with the bubbles in the groundwater to the ground surface. A basic assumption in this scenario is the mobilisation of contaminated colloids by detachment followed by flotation with the bubbles. The gas water interface always adsorbs negatively and positively charged particles: (eg, clay particles) at high and low ionic strengths. The degree of this adsorption increases when the ionic strength increases and the adsorption of particles on gas bubbles can be irreversible. The interaction studied by Wan and Wilson (1992) is consistent with the classic DLVO theory.

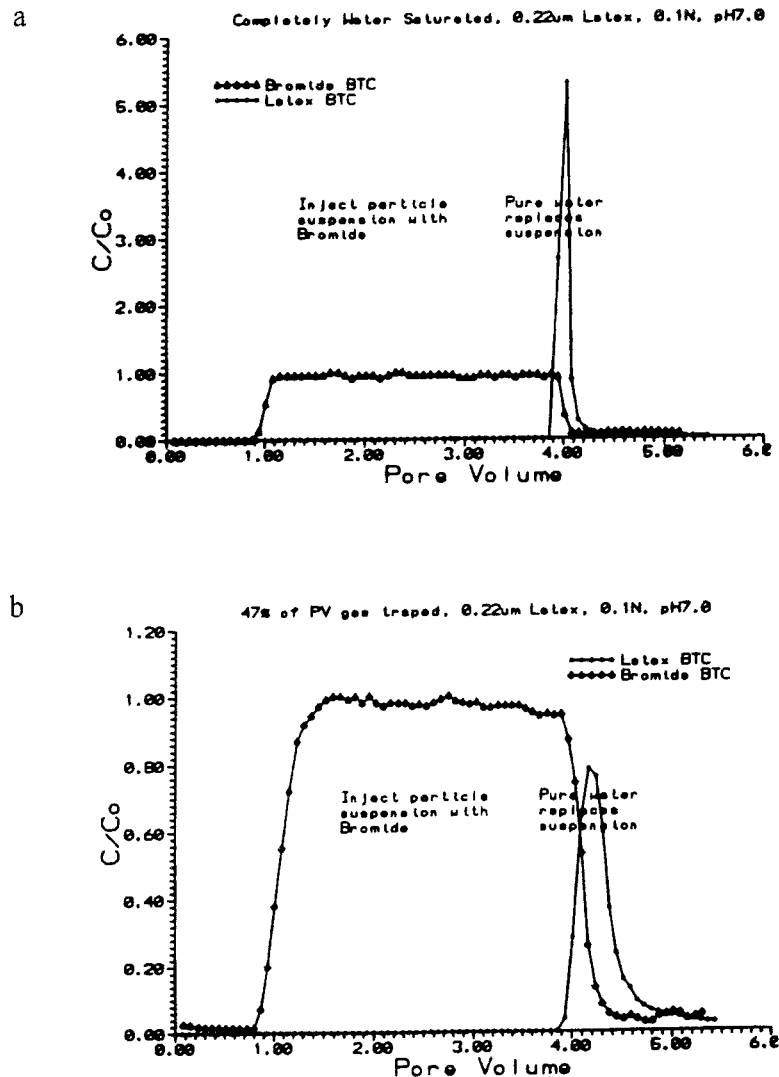
However, this very interesting scenario requires additional consideration to be adapted in a quantitative and realistic way. It is based on the gas source and is

limited by the bubble production which in turn is limited by the gas production rate and the gas solubility. Changes in gas saturation are corrosion rate sensitive and over-saturation in such a system is like the corrosion rate - a function of time. The gas phase generated during corrosion of the canister may isolate the canister/near-field interface reducing the corrosion rate by isolating the interface. Even if corrosion may continue because of water vapour in the gas phase, the corrosion rate must decrease when passing from first contact with saline water to that with the moist gas. Under these conditions, is the gas production still large enough to yield over-saturation at the near-field/far-field interface?

Particle attachment on the bubbles requires that the particles considered are initially free in the water phase, or form a cohesionless phase. This condition is not fulfilled at the near-field/far-field interface because the water bearing feature through which the gas has to move had earlier contained or still does contain groundwater. The ionic strength is large enough to induce reaction with the sodium bentonite. Actually, when the gas bubble diffuses through the near-field, the bentonite is not in a sodium-form any more but a calcium bentonite one. It must be recognised that this material is not cohesionless, the attachment coefficient (between two montmorillonites particles) is large (Figure 2-5) and these particles are then not free for attachment onto the potential bubbles.

Particle transport with gas bubbles can only take place in a local environment because trapped bubbles may also stop the contaminated colloid transport (see Figure 3-3). In addition, a dilution factor relative to distance is generally accepted in a safety assessment study. Again, the gas which may be initially over-saturated may become under-saturated with time as a function of the distance from the repository. During transport, colloids on which relevant nuclides are sorbed are able to desorb these nuclides. This has been recently observed when the sorption takes place by ion exchange or surface complexation. This may not happen if colloidal particles aggregate together after sorption. Finally, if moving gas-water interfaces can strip and carry away particles from a source, re-deposition can also take place in another part of the aquifer which has different channelling and porosity properties. Considering the impact of fresh water intrusion, gas should dissolve because the infiltration water is not saturated with the gas produced in situ.

This interesting scenario should be developed further. This is however beyond the scope of this study. To our knowledge, there is no natural analogue system which supports this scenario. It is therefore not considered in this report.



Figures 3-3a and b: Effect of the presence of gas-water interface on colloid migration. Conditions - a) Infiltration in column without gas, b) infiltration in column with gas. Note: colloids are trapped by gas bubbles (Wan and Wilson, 1992).

3.4 RELAXATION PROCESS

Following any hydrogeochemical change in the aquifer, the system tends to re-equilibrate as observed in field tests (see Figure 3-1) or in the laboratory (see Figure 3-2). Physical or chemical gradients decrease therefore in intensity. The aquifer may return to a comparable situation with the original one and the colloid concentration also returns to what it was before the transient situation.

Any change of T, P, ionic strength, pH or pe characterising the non-steady-state situation may induce a release of colloids according to preferential processes. After system modification (physical and/or chemical), two trends are possible:

- a) either the colloid generation decreases because production is limited by the colloid quantity able to be released and the colloid concentration in the groundwater subsequently decreases, or,
- b) because the system re-equilibrates (after flow rate change or ionic gradient), the colloids re-attach onto the host rock thus reducing the colloid concentration too.

Both processes yield the same result: a colloid concentration relaxation and a return to that observed for the steady state situation (see Figures 3-1 and 3-2).

In the hydrogeological system considered in this report, because of the scale of the system, reduced parameter fluctuations are expected because of the buffer capacity of the aquifer eg, pH, ionic strength, redox, instead of sharp changes like those observed in the laboratory or during field tests. Thus large increases (2-3 orders of magnitude) of the colloid concentration are not expected.

4. EFFECT OF DEGLACIATION

Probably the most dramatic change in the repository conditions will be the predicted new glacial period over Fennoscandia. This glacial period, known as the second glaciation 20000AP. (After Present), is expected to cover the northern part of Sweden. A third glaciation 60000AP has also been predicted that will cover the whole of Sweden (Ahlbom et al., 1991). When this ice sheet finally melts the glacial meltwater may be injected into the basement and affect the groundwater and hence increase the colloid concentration. The conceptual model for the injection of glacial meltwater at Äspö when the continental ice was melting and retreating is shown in Figure 4-1 (Laaksoharju and Skärman, 1995).

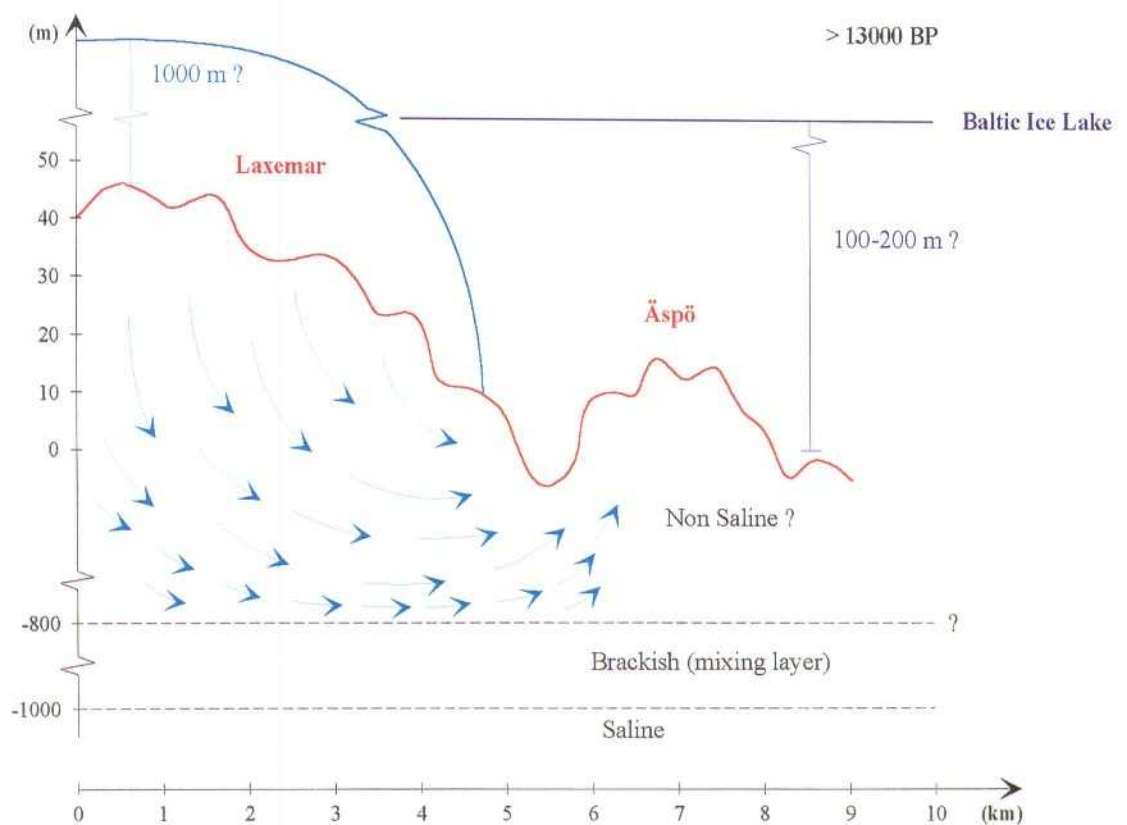


Figure 4-1: Injection of Glacial meltwater into the basement shown as a profile of Äspö and the mainland modified after I. Rhén, (pers. com. 1995).

In order to estimate the effect of a future glacial melting process, traces of the last glaciation in present water had to be traced at the Äspö site. This was done by Smellie and Laaksoharju (1992); Laaksoharju and Skärman (1995). The Principal Component analysis plot (Figure 4-2) was used as a phase diagram to calculate the mixing ratios. Calculations concerning the influx of glacial meltwater from the last glaciation in the present groundwater is shown. From the results of the calculations it can be seen that most of the observations at Äspö were affected by the influx from the last glaciation. Observations at 500m depth seem to contain up to 40-60% of glacial meltwater according to these calculations.

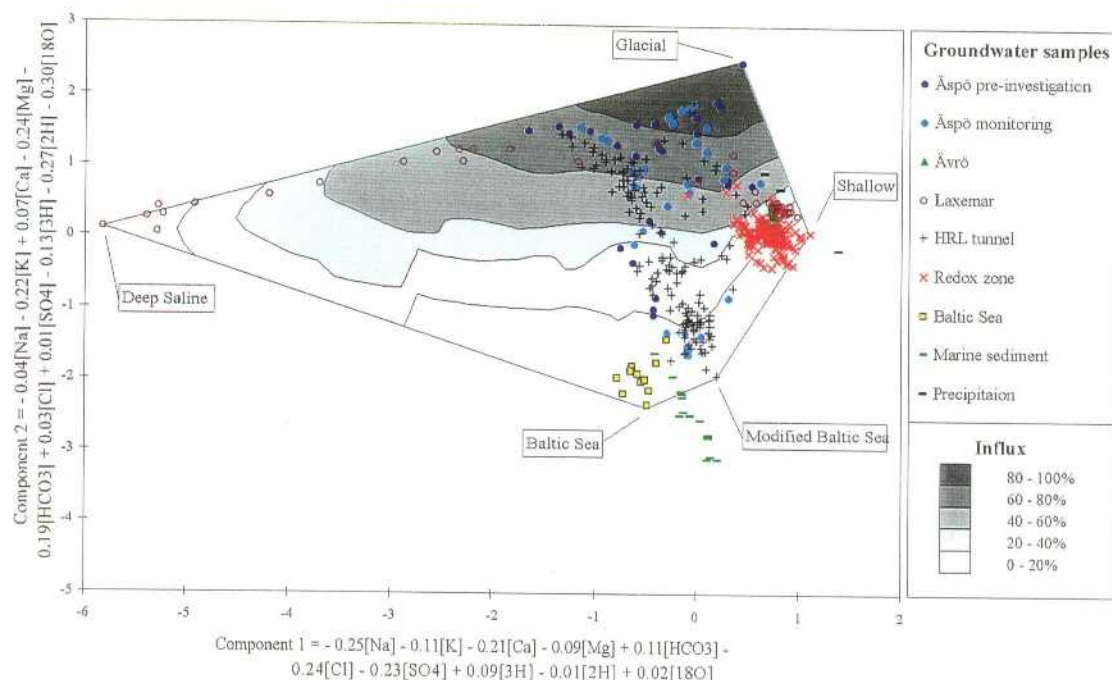


Figure 4-2: Principal Component plot used to calculate the influx of 0-20%, 20-40%, 40-60% and 80-100% of glacial meltwater from the last glaciation into the groundwaters observed at Äspö. Equations are shown for the first and second Principal Component respectively (Laaksoharju and Skärman, 1995).

Glacial meltwater is assumed to have an oxic rainwater composition when being injected into the basement. Based on the experience from the Large Scale Redox Experiment (Banwart et al., 1994) oxygen was rapidly consumed in the basement by dissolved organic matter and possibly iron(II) containing minerals. It is therefore assumed that the original oxic glacial meltwater will become reducing with time. When glacial meltwater is injected the colloid concentration is expected to rapidly increase from what is generally found in the Äspö groundwaters ($20\text{--}45\text{ }\mu\text{g}\cdot\text{l}^{-1}$), with a factor of 100 for example, due to the opening of new flow paths, introduction of mixing processes, increased fracture erosion etc. After this relatively short initial period a relaxation phase will occur when the colloid concentration will rapidly decrease to about $20\text{ }\mu\text{g}\cdot\text{l}^{-1}$ (assuming an influx of 50% glacial water at 500m repository depth). The dilution of the groundwater will deplete but also stabilise the existing colloid population (Khilar and Folger, 1984; Grauer, 1992; Fauré et al., 1994). Infiltration of glacial meltwater is therefore believed to reduce the pre-existing colloid concentration. It must however be noted that infiltration of fresh water will decrease the ionic strength of the groundwater and increase the colloid generation (see Figure 3-2). Salinity gradients resulting from the infiltration of glacial meltwater in the geosphere will be smaller than the gradient imposed in column experiments in the laboratory, generating a colloid concentration peak of 1 or 2 orders of magnitude larger than the background colloid concentration. Therefore the colloid concentration peak resulting from the infiltration of meltwater in the geosphere is not expected to be larger than 10 times the present background concentration. Currently the colloid concentration in the saline waters is $20\text{ }\mu\text{g}\cdot\text{l}^{-1}$ while in the non-saline

waters (some of which are mixed waters) it is $45 \mu\text{g}\cdot\text{l}^{-1}$. The ratio of colloid concentration for the two types of water is only 2.

With the infiltration of glacial water, it is expected that the dilution of the deep groundwater will not be affected by more than 50% around the repository (see Figure 4-1). Under these conditions, the calcium concentration of the mixed water at the near-field/far-field interface is not expected to decrease below 10^{-4} molar. It will always remain above this concentration level. In the saline groundwater diluted by glacial water, the attachment factor of the bentonite particles will remain around 1 (see Figure 2-5). Therefore it may be concluded that the impact of glacial water will not favour the re-suspension of the bentonite in the far-field.

5. CONCLUSIONS

In this study, a deeper understanding of Swedish groundwater colloid properties has been achieved.

Inorganic colloid contents in both saline and non-saline water systems have median values of around 20-45 $\mu\text{g}\cdot\text{l}^{-1}$ respectively (for 50-500nm sizes), which is in accordance with the Swiss colloid results in deep crystalline groundwaters. This result was obtained by carefully correcting the former data by eliminating all interfering components.

Colloids can be generated during transient situations such as flow rate changes or salinity gradients. In addition, the coupling of these processes can also enhance the colloid concentration in the aquifer. However, it has been demonstrated that both changes and gradients are smoothed out and that the colloid generation process is attenuated in the geosphere. The level of concentration may then be enhanced by one order of magnitude over that presently observed. However, relaxation occurs after any change which reduces the colloid concentration back to the same level of concentration as before the disturbance.

With the infiltration of glacial water, it is expected that dilution of the deep groundwater will not be affected by more than 50% around the repository. In these conditions, the calcium concentration of the mixed water at the near-field/far-field interface is not expected to decrease below 10^{-4} molar. It will always remain large enough and the attachment factor α of the bentonite particles will remain large (around 1). Therefore, the impact of glacial water will not favour the re-suspension of the bentonite in the far-field.

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APPENDIX 1: RESULTS FROM THE COLLOID FILTERS AND GROUNDWATER ANALYSES IN GEOTAB SHOWN AS MATRIX SCATTER PLOTS

Chemical analyses of colloid filter samples and the groundwaters found in GEOTAB, shown as matrix scatter plots with histograms. The saturation index calculations were performed with PHREEQE (Parkhurst et al. 1987). Abbreviations used in the figures:

DEPTH = Average depth of measurement (m)

AREACODE = Area code for sampling

YEAR = Year of sampling or measurement

PORE = Filter pore size (μm)

PSIZE = Particle size

VOLUME = Groundwater volume passed through filter (ml)

ALP = Measured Al content on the filter ($\mu\text{g/l}$)

CAP = Measured Ca content on the filter ($\mu\text{g/l}$)

FEP = Measured Fe content on the filter ($\mu\text{g/l}$)

MNP = Measured Mn content on the filter ($\mu\text{g/l}$)

SIP = Measured Si content on the filter ($\mu\text{g/l}$)

SP = Measured S content on the filter ($\mu\text{g/l}$)

FILTFE = Fe content of the filtrate ($\mu\text{g/l}$)

FILTFEII = FeII content of the filtrate ($\mu\text{g/l}$)

NA = Measured Na in groundwater (mg/l)

K = Measured K in groundwater (mg/l)

CA = Measured Ca in groundwater (mg/l)

MG = Measured Mg in groundwater (mg/l)

HCO3 = Measured HCO_3 in groundwater (mg/l)

CL = Measured Cl in groundwater (mg/l)

SO4 = Measured SO_4 in groundwater (mg/l)

TR = tritium (units)

D = deuterium (per mil)

O18 = oxygen-18 (per mil)

SO4_S = SO_4 -S in groundwater (mg/l) (Sulphur by ICP-AES)

BR = Measured Br in groundwater (mg/l)

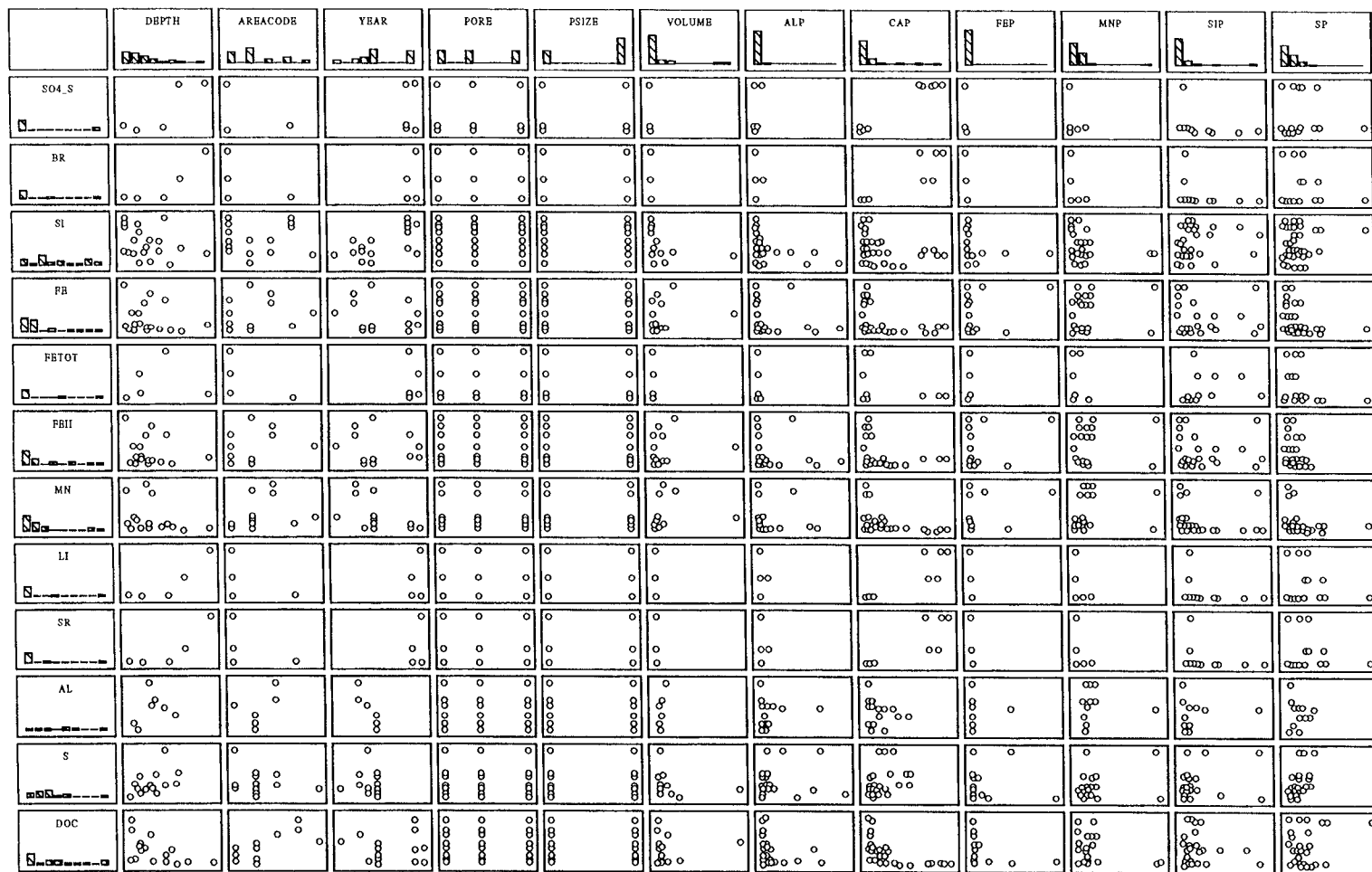
SI = Measured Si in groundwater (mg/l)

FE = Measured Fe in groundwater (mg/l)

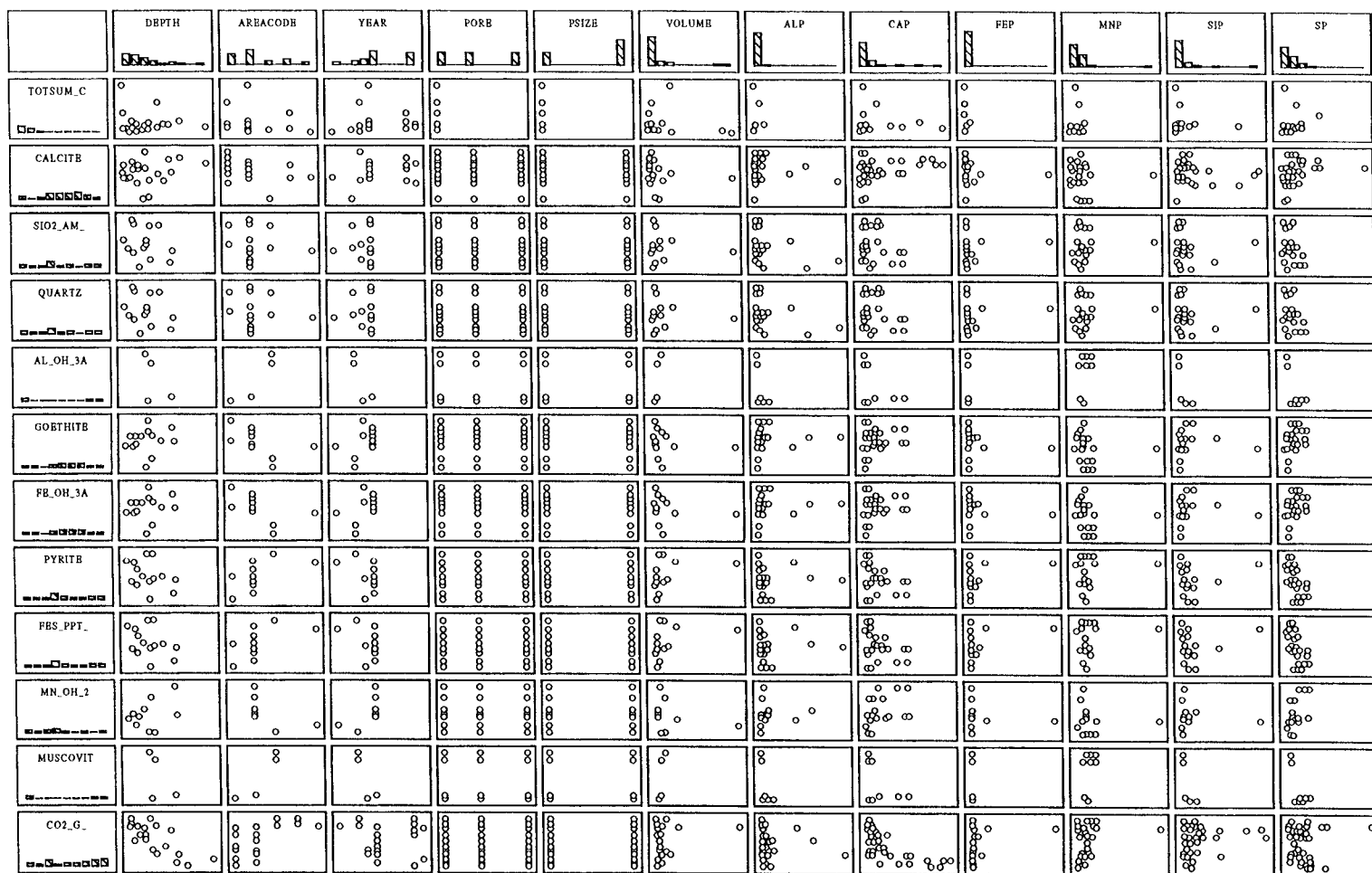
FETOT = Measured total Fe in groundwater (mg/l)
 FEII = Measured FeII in groundwater (mg/l)
 MN = Measured Mn in groundwater (mg/l)
 LI = Measured Li in groundwater (mg/l)
 SR = Measured Sr in groundwater (mg/l)
 AL = Measured Al in groundwater (mg/l)
 S = Measured S in groundwater (mg/l)
 DOC = Measured dissolved organic compounds (DOC) in groundwater (mg/l)
 TOTSUM_C = Sum of non-Ca colloids/observation
 CALCITE = Saturation index, log
 SiO2(am) = Saturation index, log
 QUARTZ = Saturation index, log
 Al(OH)3a = Saturation index, log
 GOETHITE = Saturation index, log
 Fe(OH)3a = Saturation index, log
 PYRITE = Saturation index, log
 FeS(ppt) = Saturation index, log
 Mn(OH)2 = Saturation index, log
 MUSCOVIT = Saturation index, log
 CO2(g) = Calculated partial pressure of carbon dioxide
 PHYAV = Average pH measured at surface
 PHL = Laboratory pH
 CONDL = Electrical conductivity measured at the laboratory (mS/m)
 CONDYAV = Average electrical conductivity measured at surface (mS/m)
 WFLOW = Pump rate (ml/minute)
 DRILLW = Drill water content (%)
 SELPH = Selected pH used in PHREEQE calculations
 CALCEH = Calculated Eh according to method suggested by Grenthe et al. (1992).
 USEDEH = Used Eh values in PHREEQE calculations
 SUM = The sum of ALP+CAP+FEP+MNP+SIP+SP content per filter (µg/L)
 TOTSUM = The element sum of the 0.40+0.20+0.5+0.5 µm filter (mg/l)
 SUM_CAP = Sum of non-Ca colloids/filter

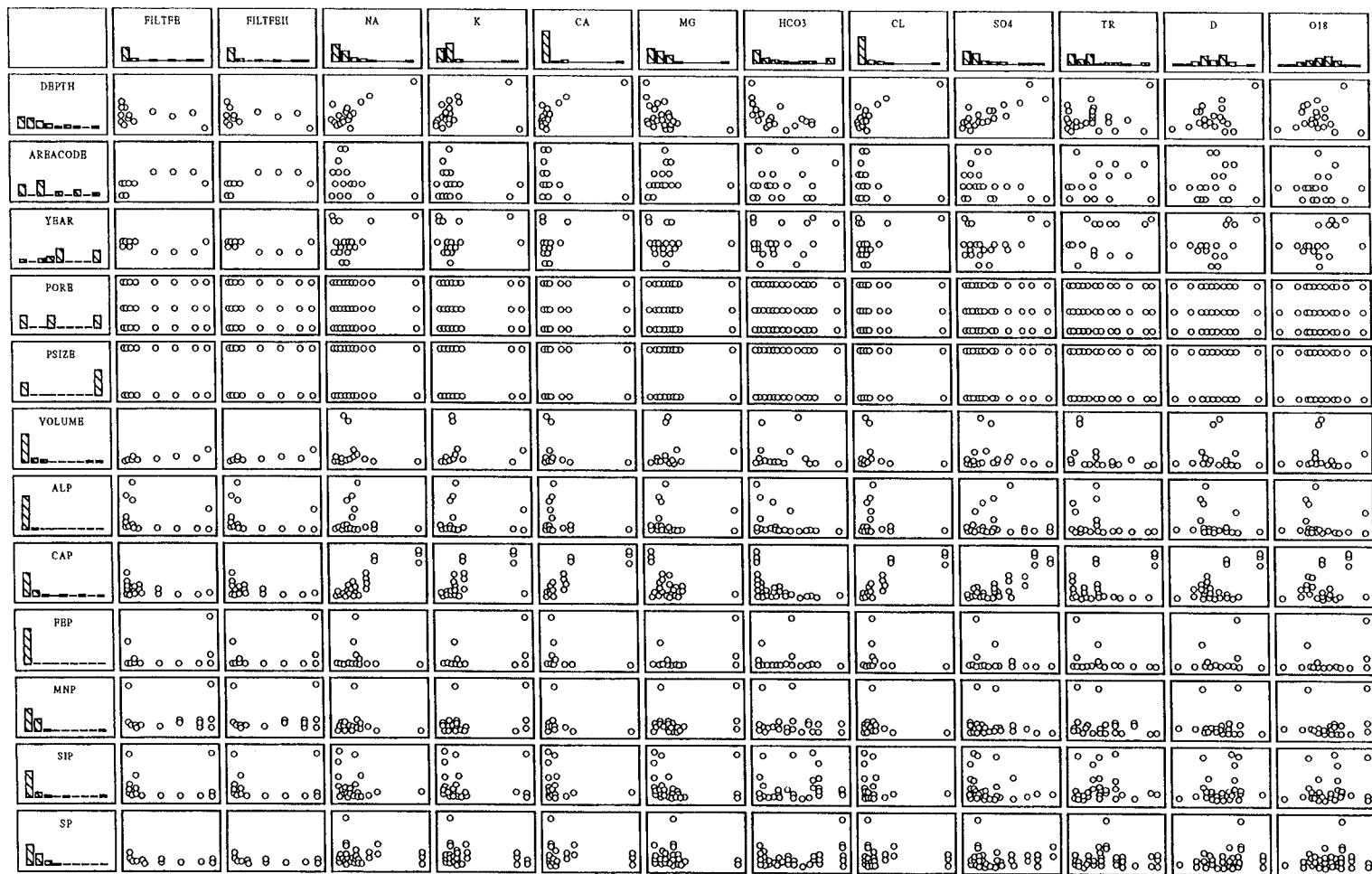
	DEPTH	AREACODE	YEAR	PORE	PSIZE	VOLUME	ALP	CAP	REP	MNP	SIP	SP
DEPTH												
AREACODE												
YEAR												
PORE												
PSIZE												
VOLUME												
ALP												
CAP												
REP												
MNP												
SIP												
SP												



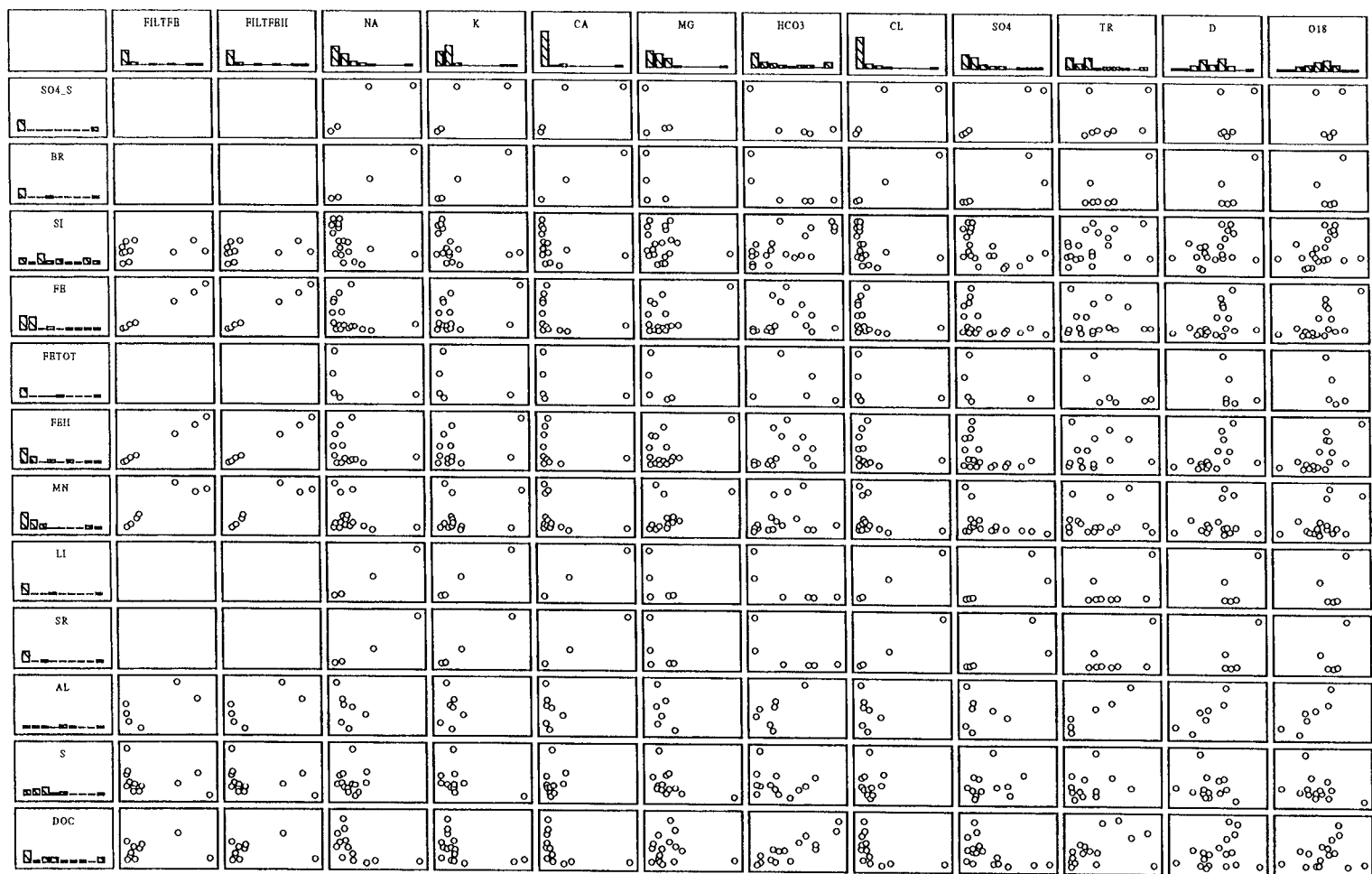


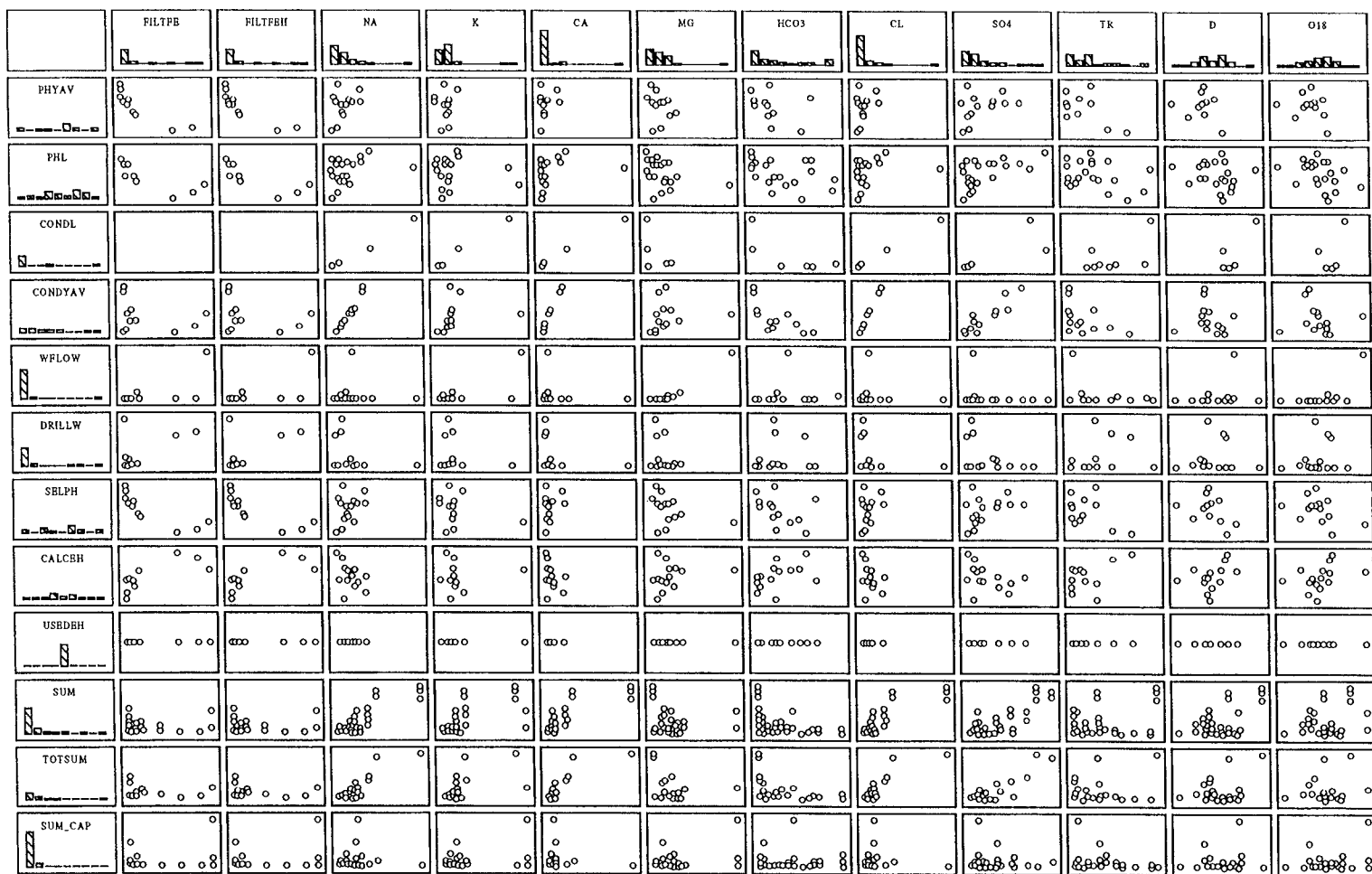


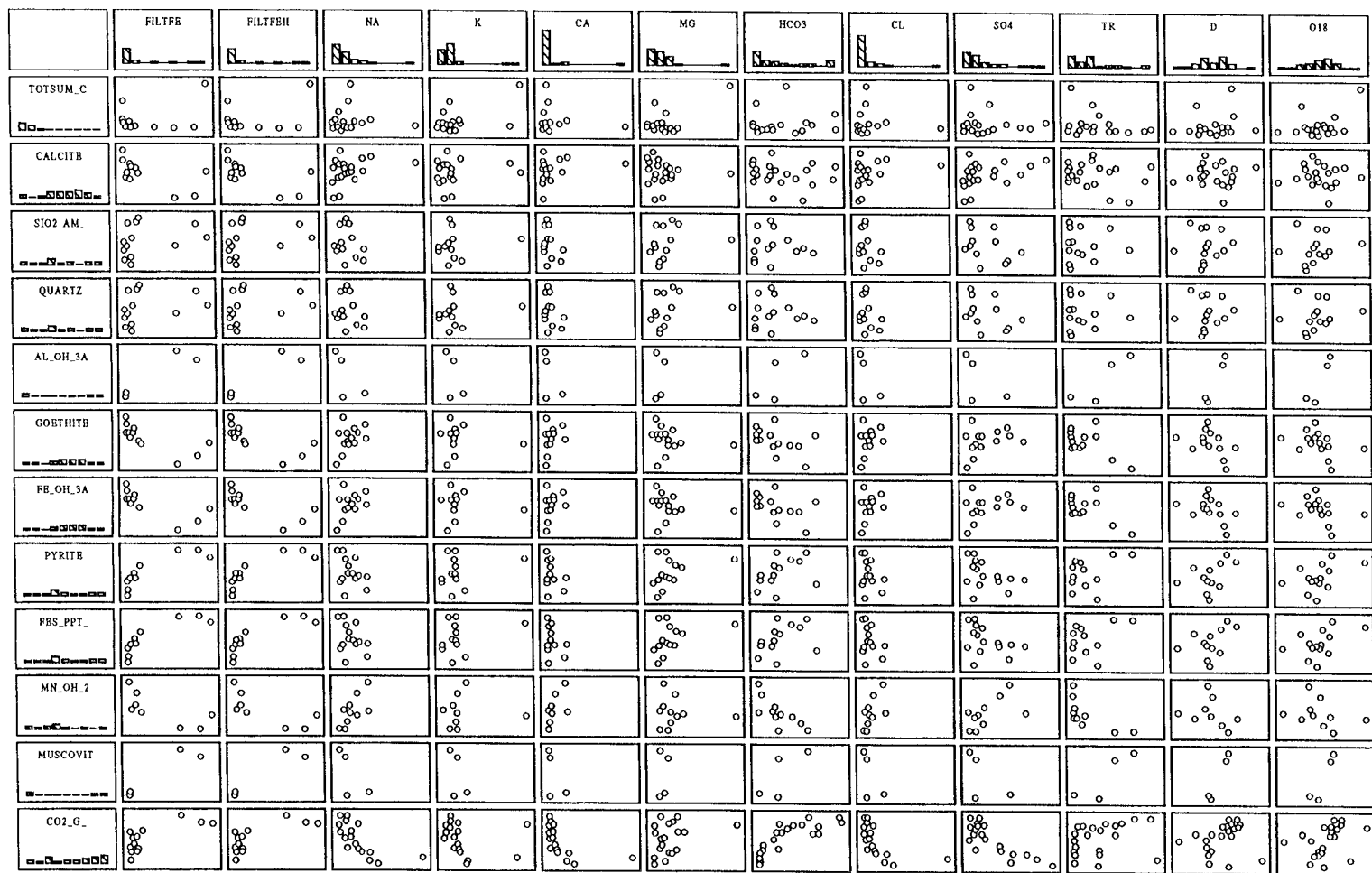




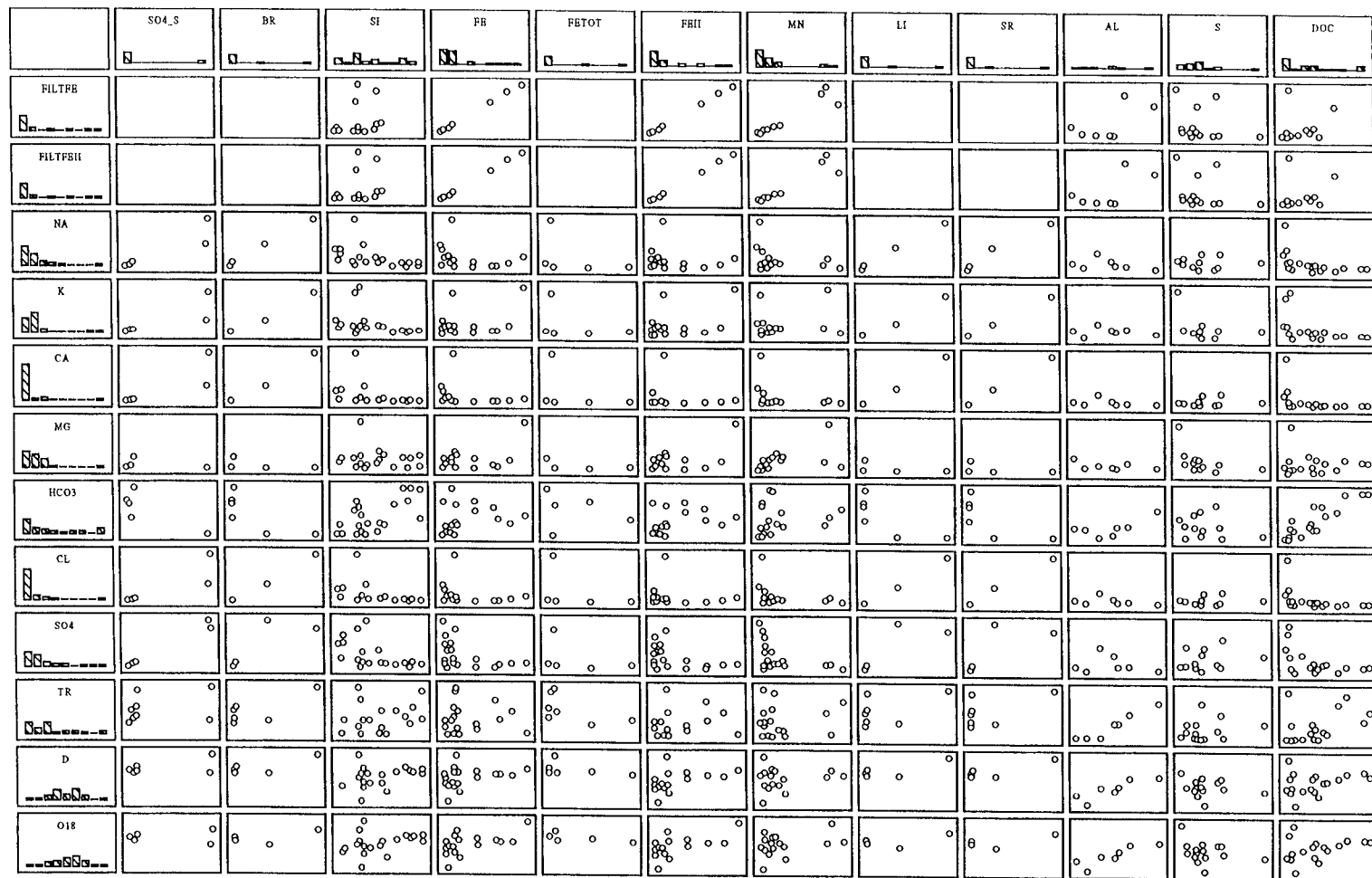
	FILTRF	FILTRFI	NA	K	CA	MG	HCO3	CL	SO4	TR	D	O18
FILTRF												
FILTRFI												
NA												
K												
CA												
MG												
HCO3												
CL												
SO4												
TR												
D												
O18												

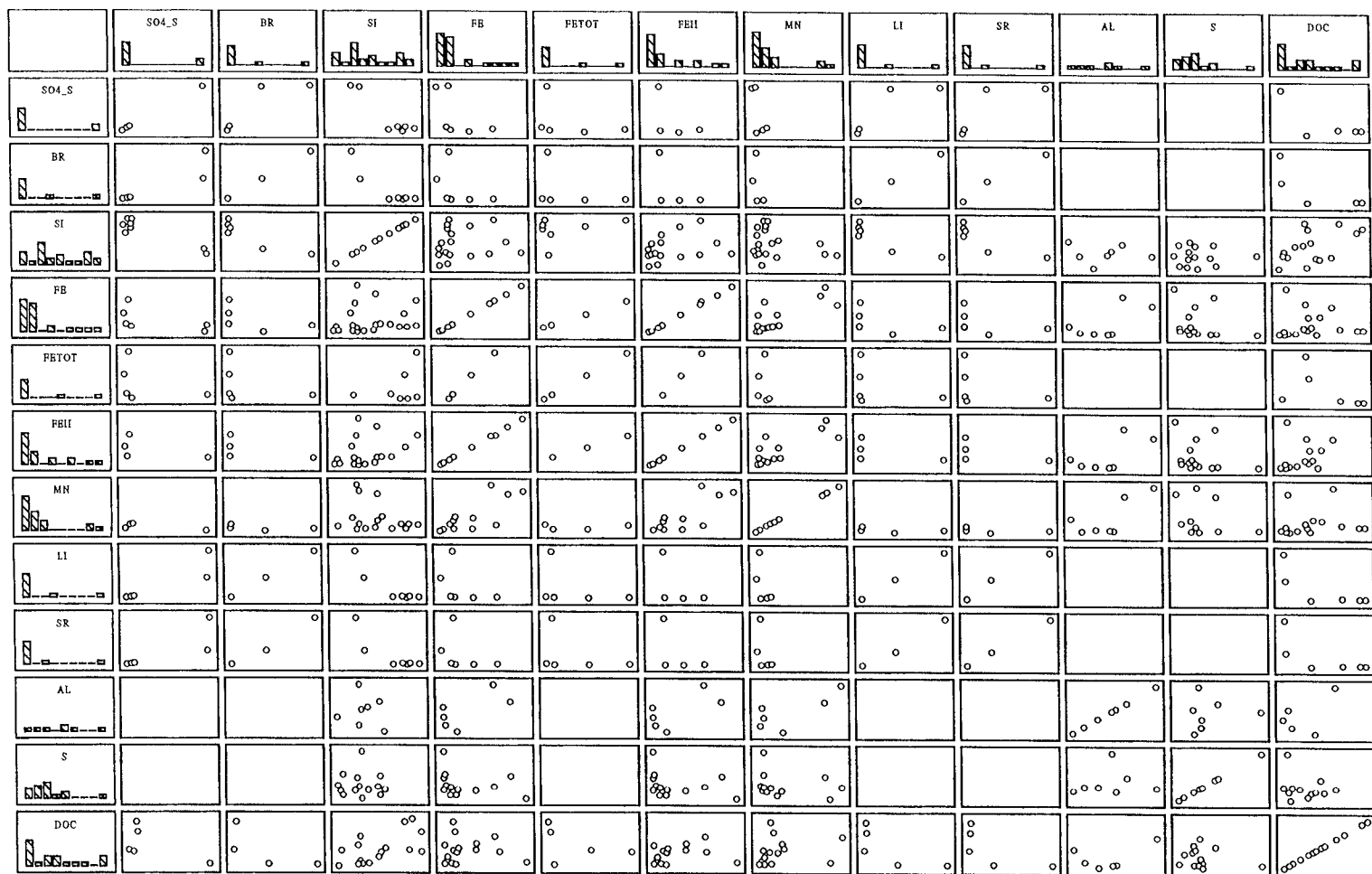


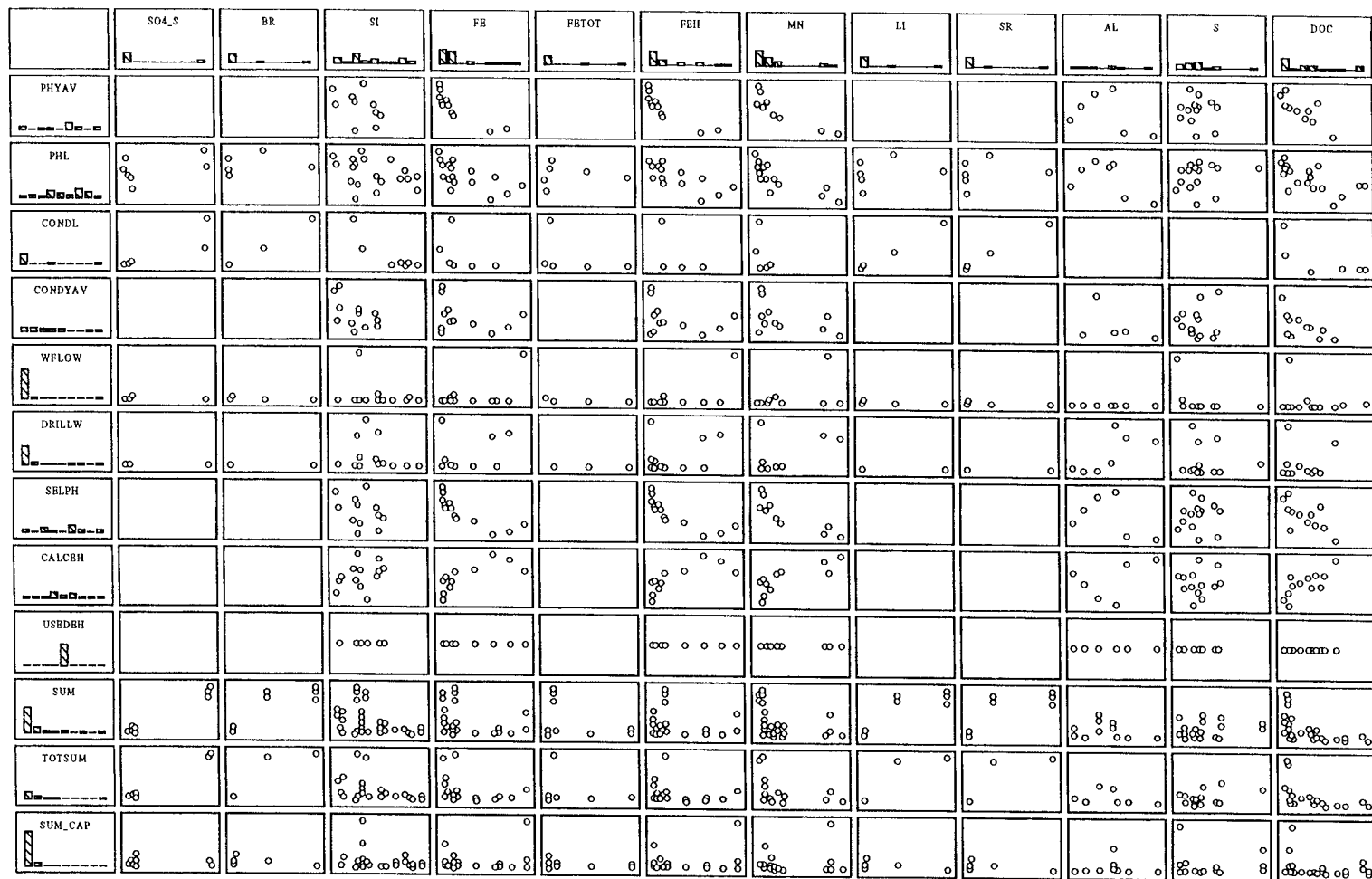


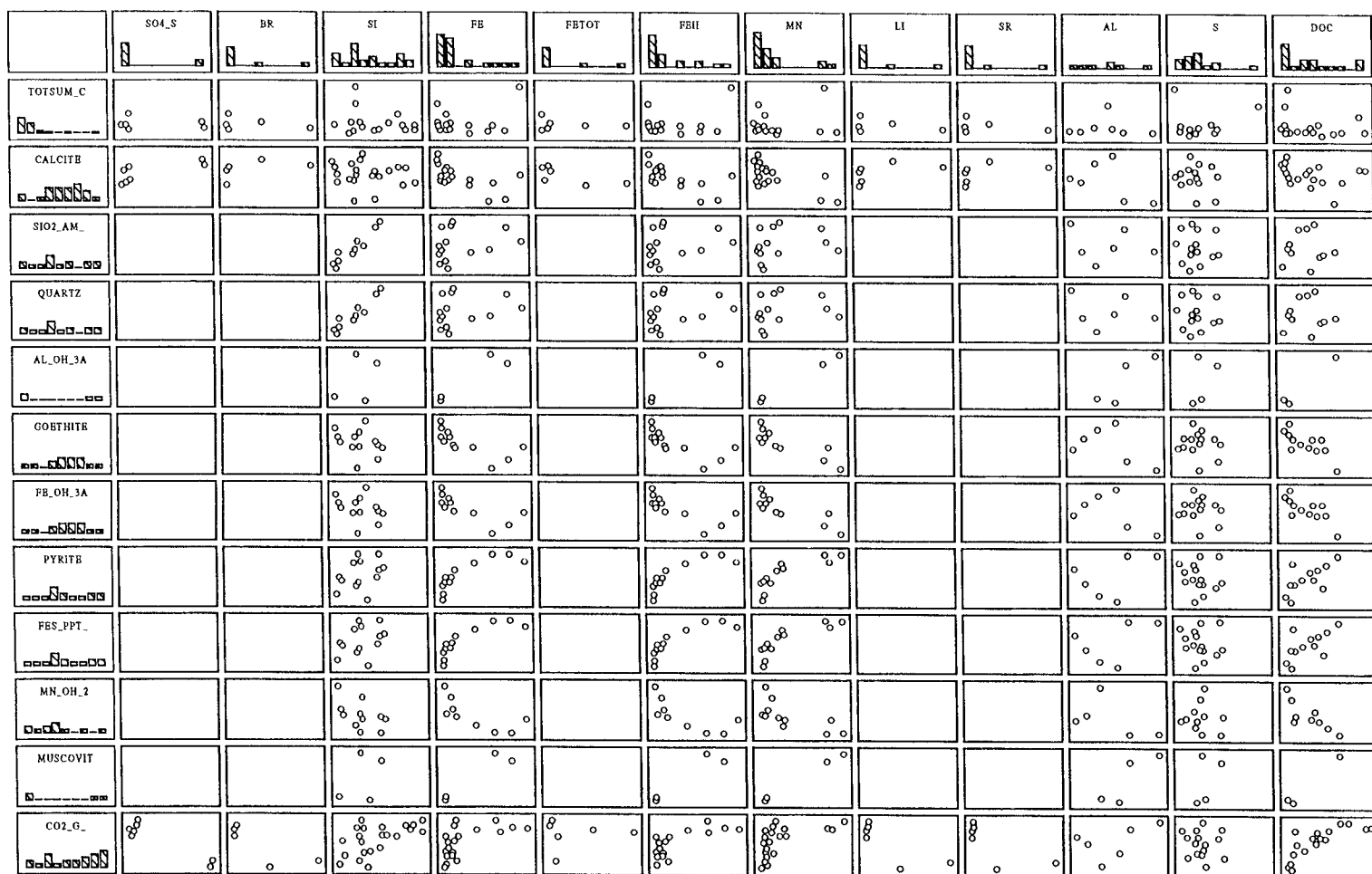


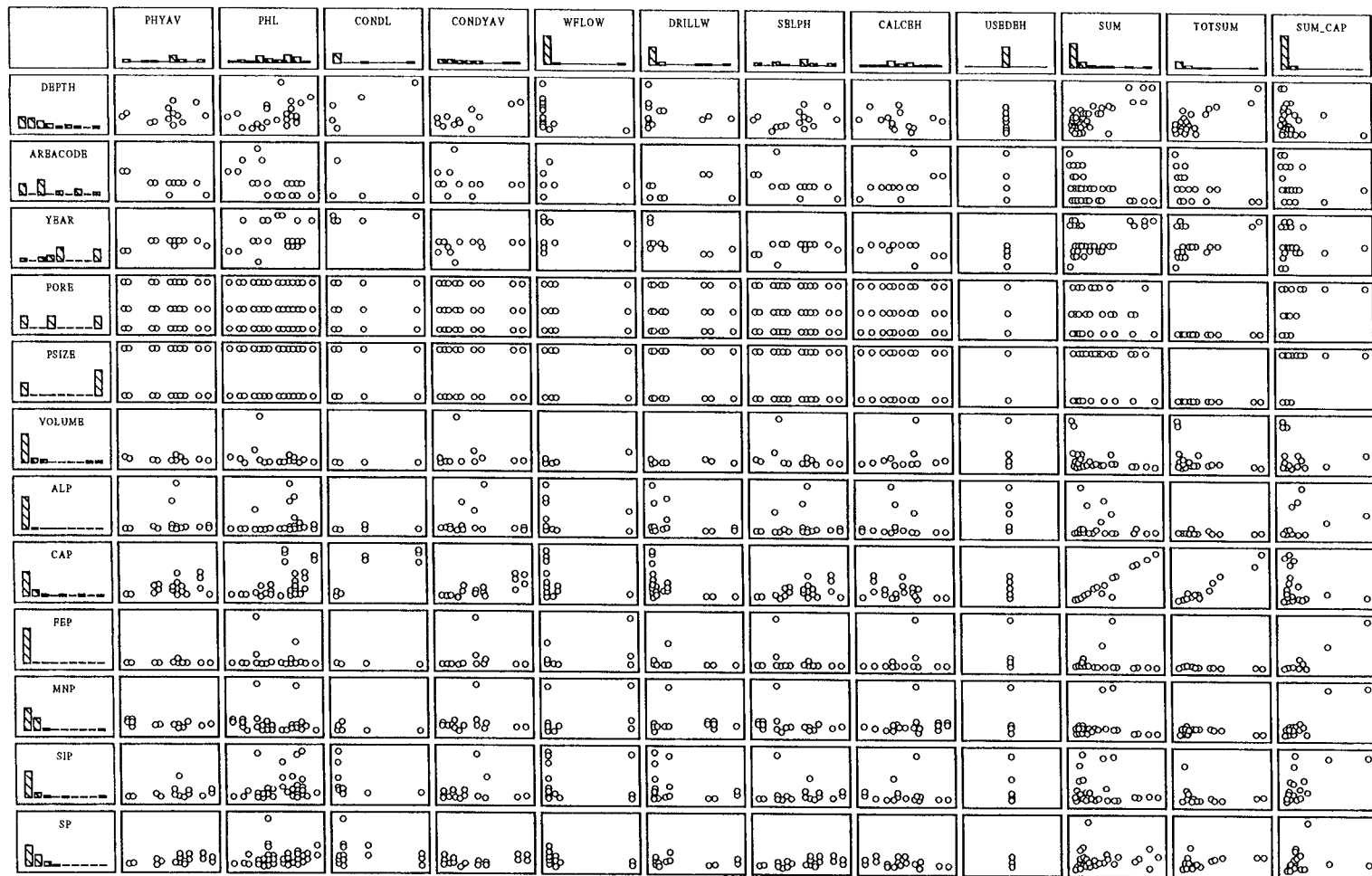


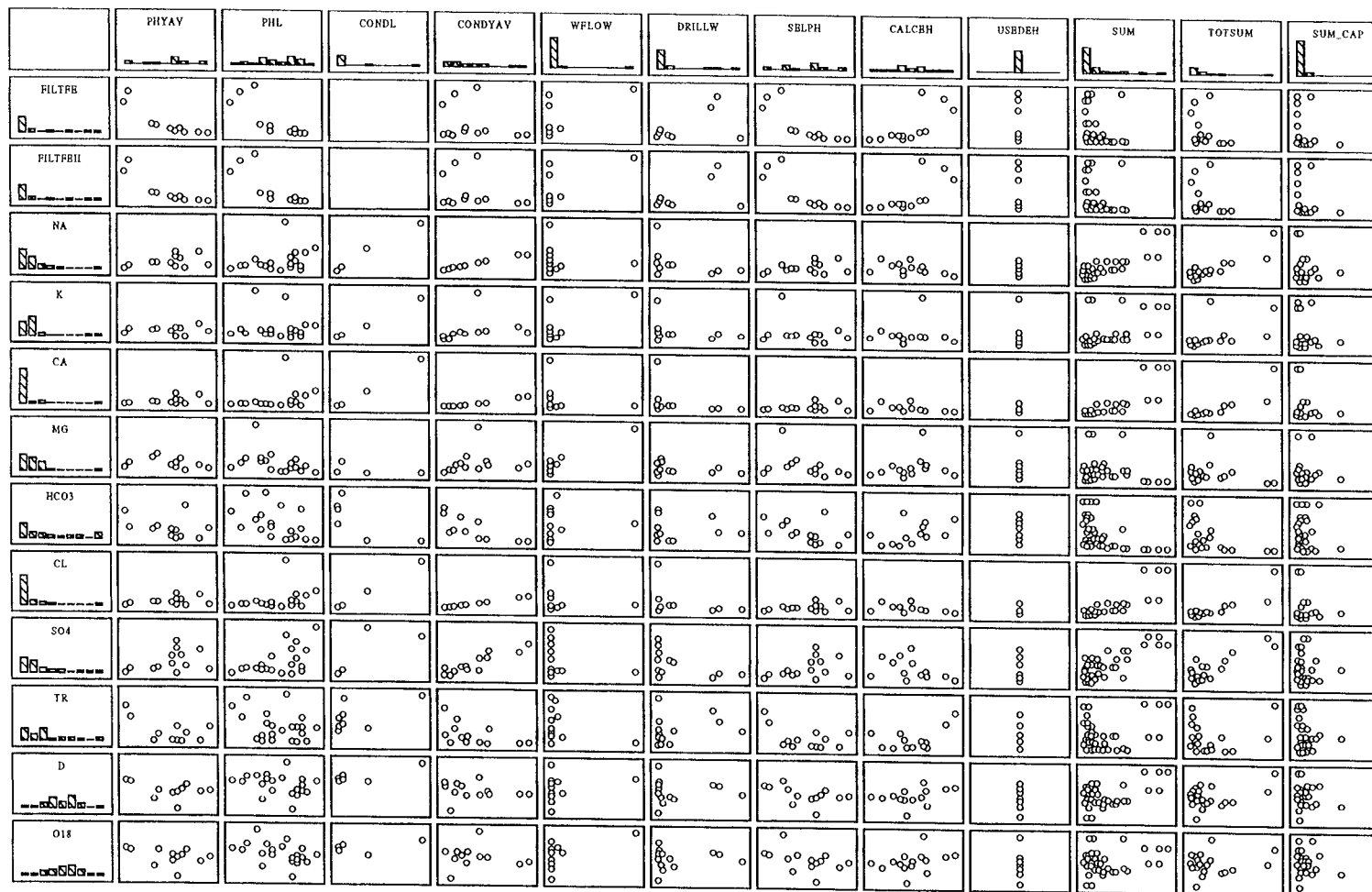




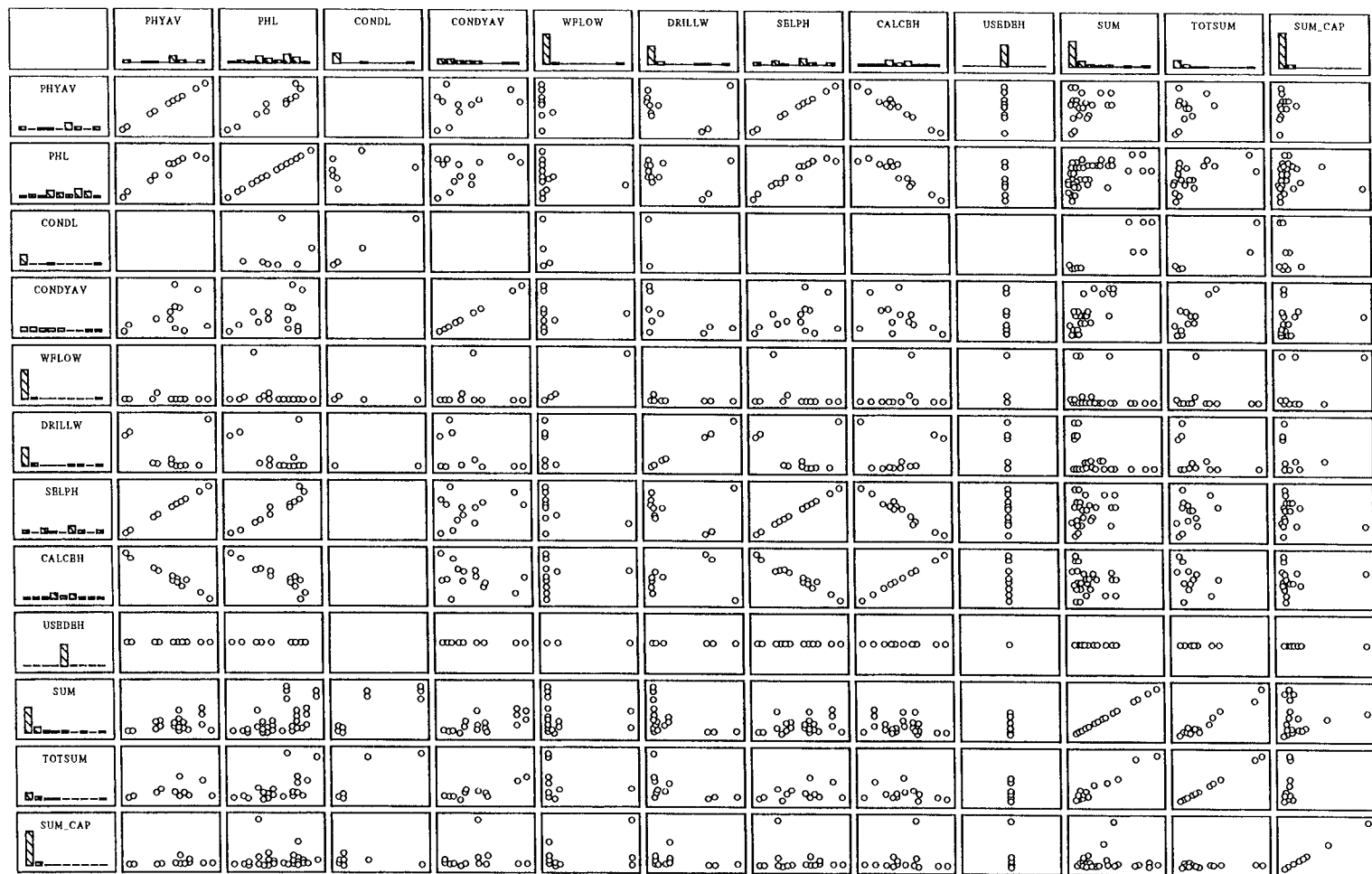


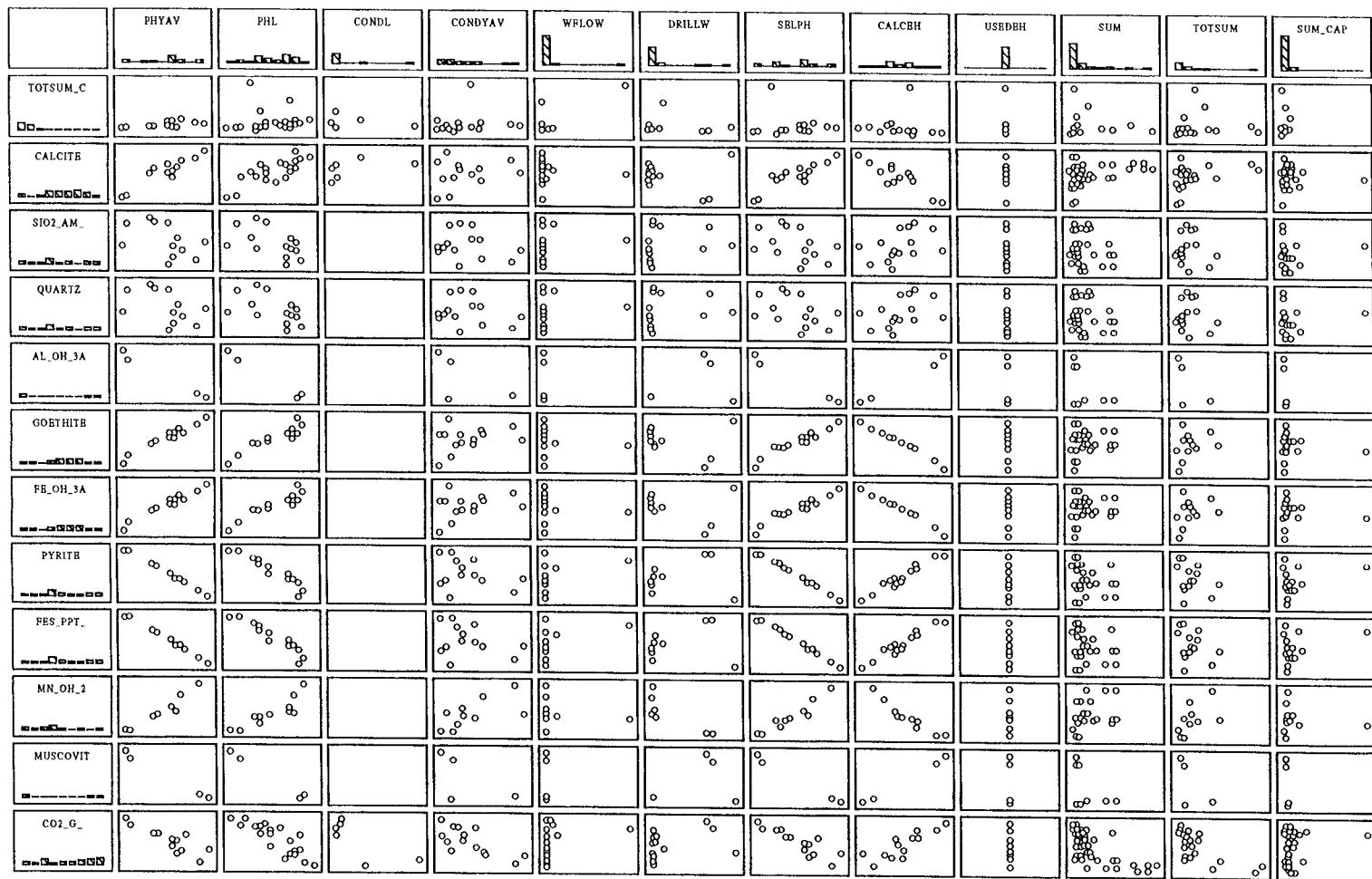


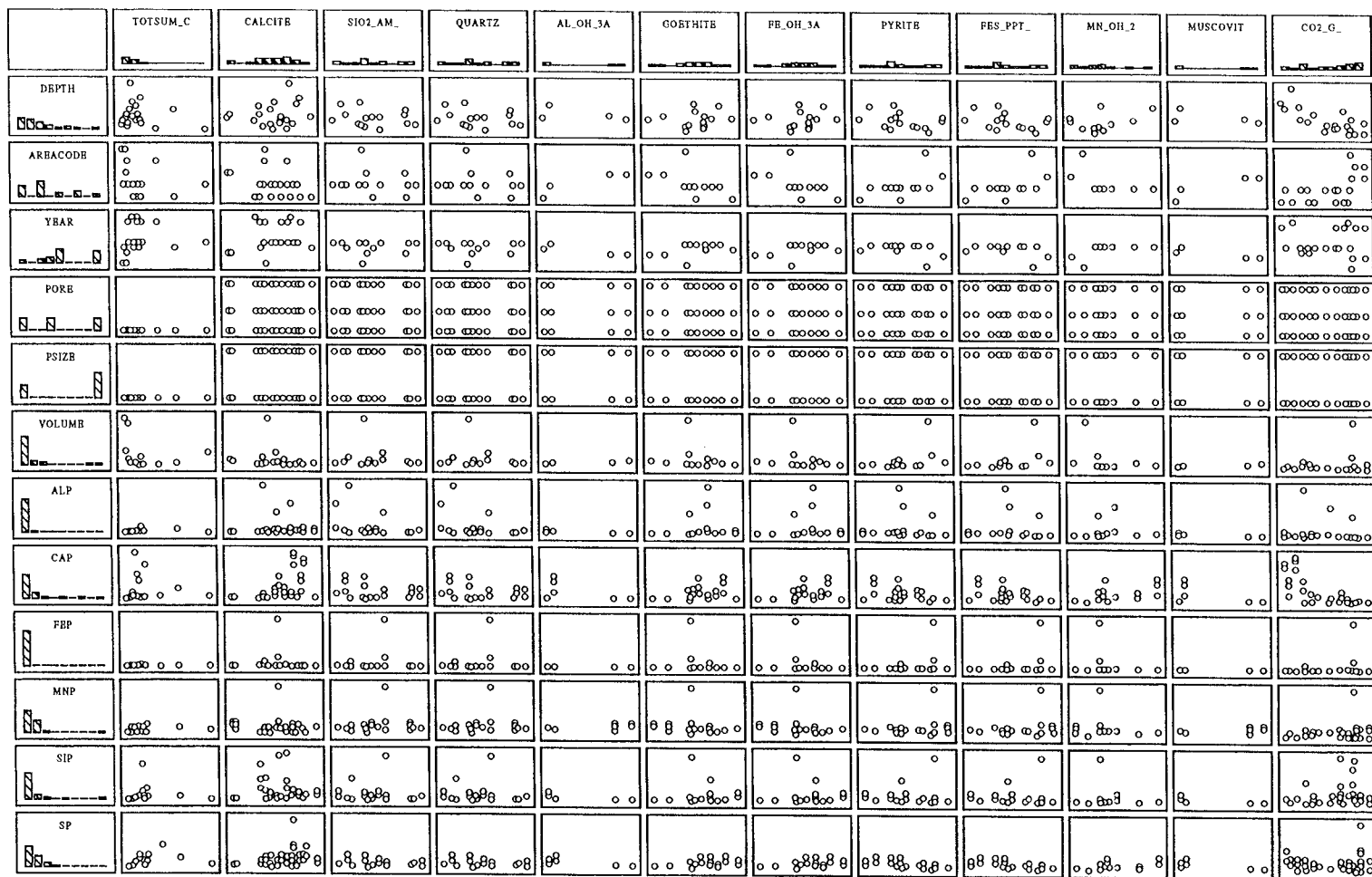


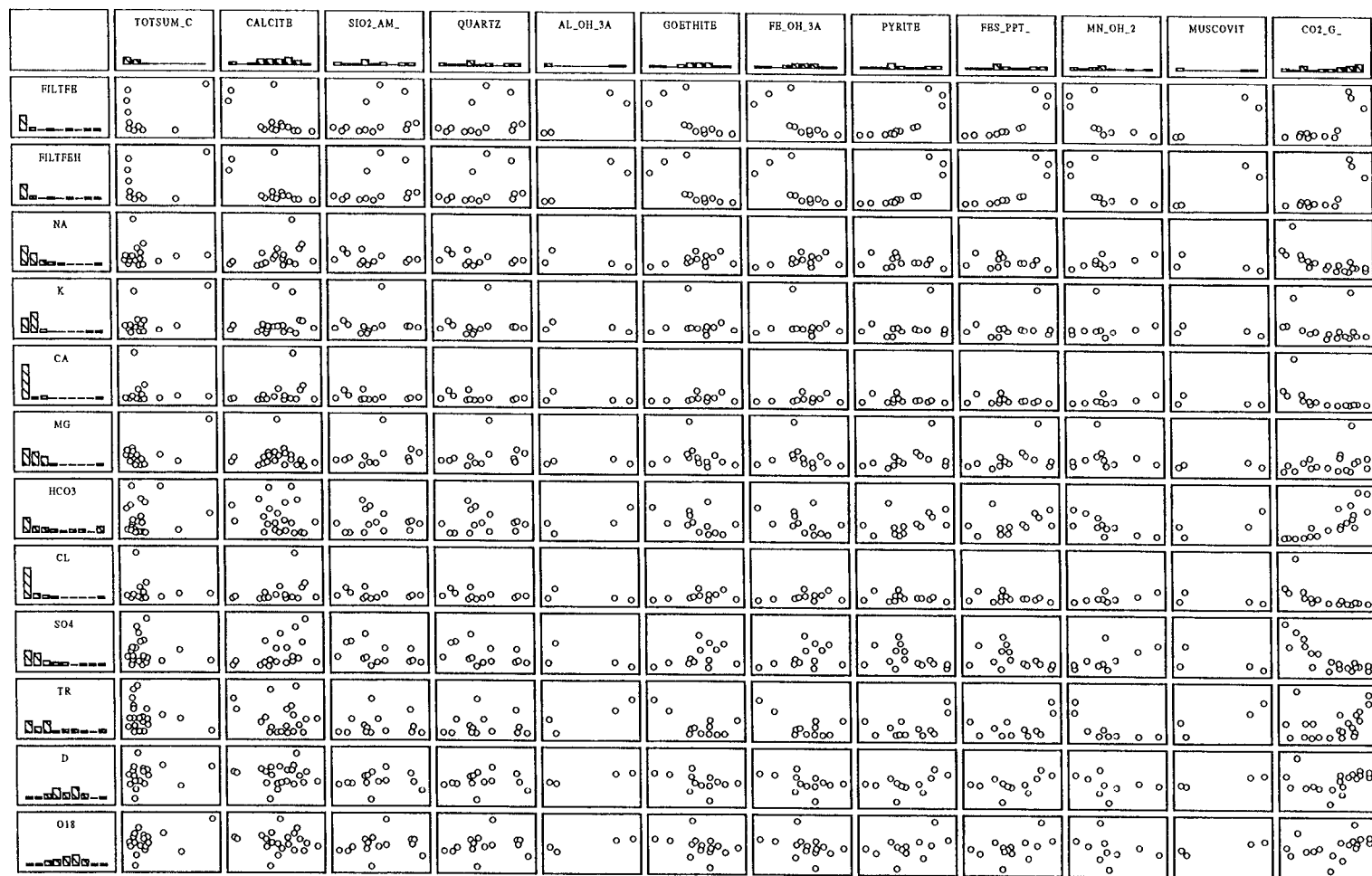


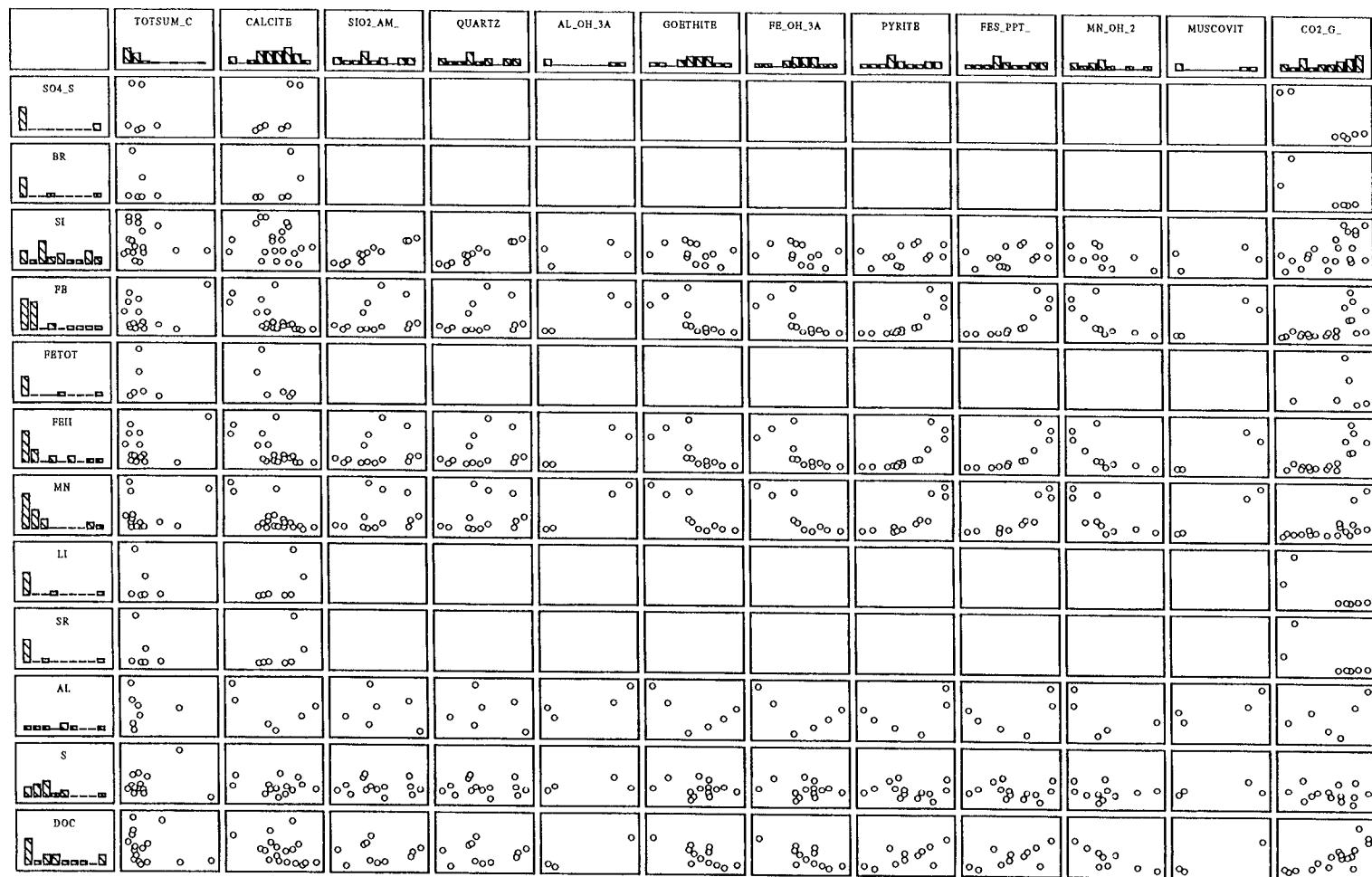


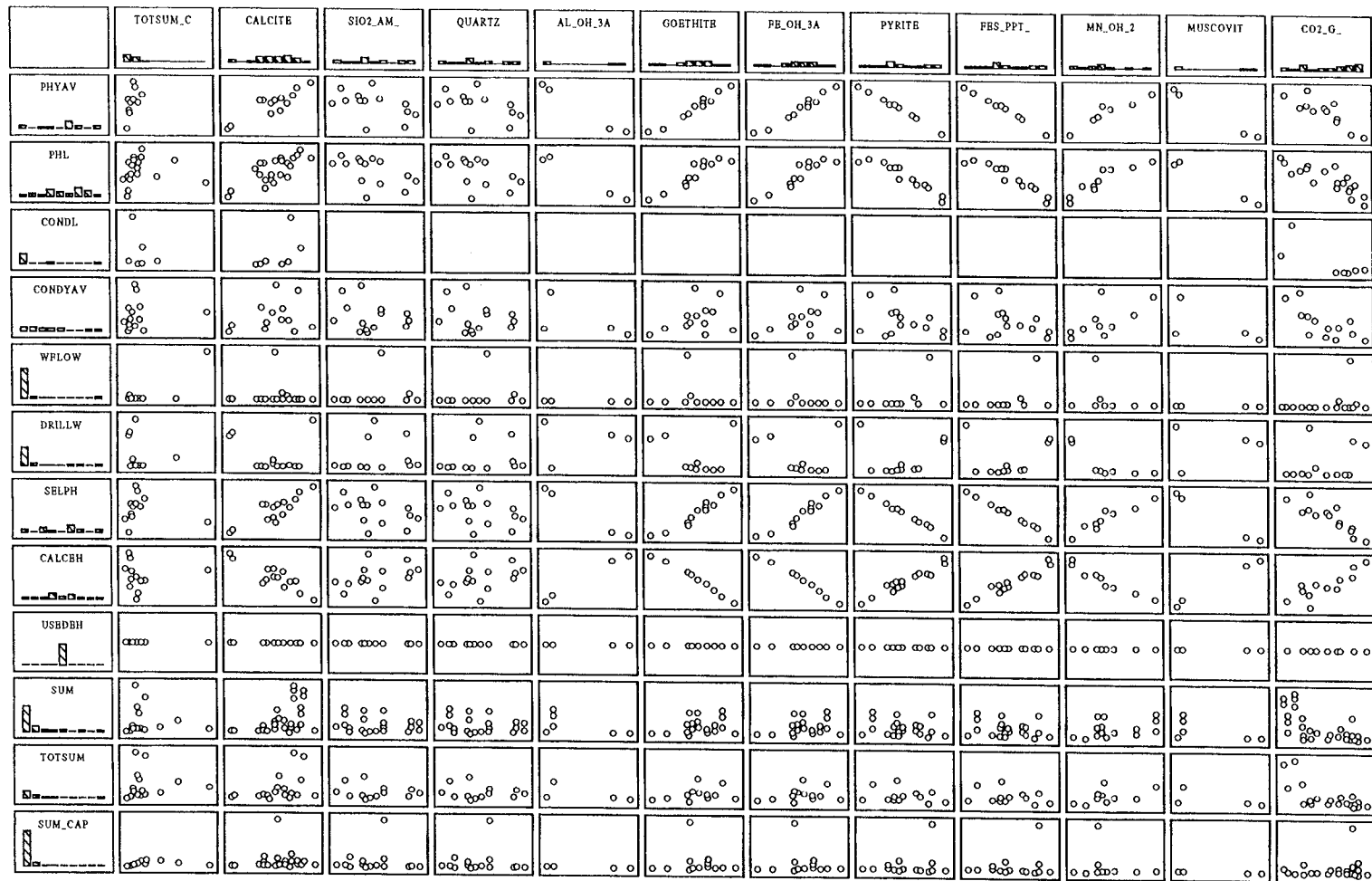


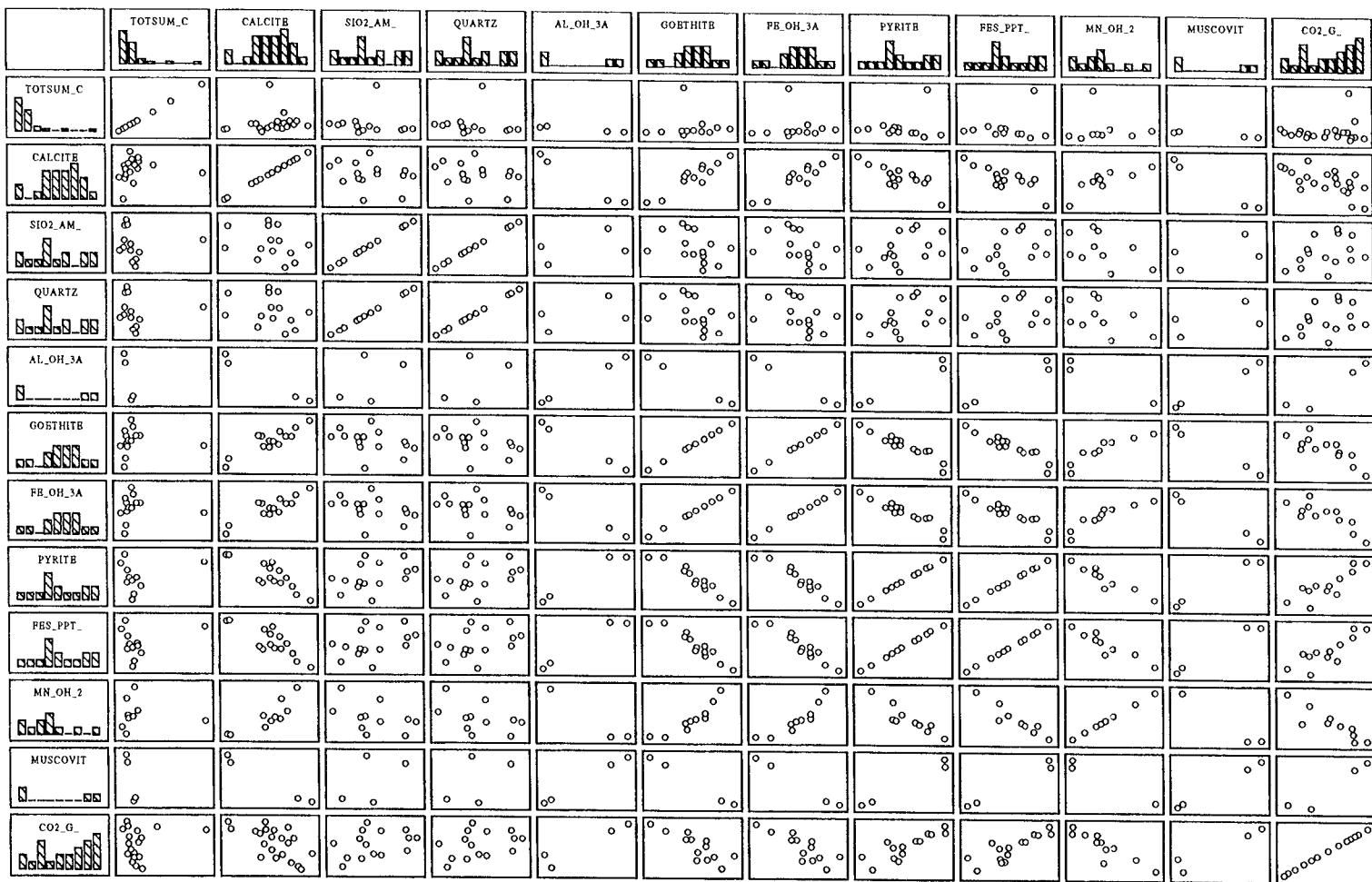












APPENDIX 2: RESULTS FROM THE STATISTICAL CALCULATIONS OF THE COLLOID MATERIAL

The results of the statistical treatment of GEOTAB groundwater and colloid data are divided into 3 data sets: All observations, Non-saline observations (Cl < 2000 mg/L) and Saline observations (Cl > 2000 mg/l). The abbreviations in the tables are:

Valid N = Number of observations

Median = Median value

Minimum = Minimum value

Maximum = Maximum value

Lower Quartile = Lower quartile value

Upper Quartile = Upper quartile value

DEPTH = Average depth of measurement (m)

YEAR = Year of sampling or measurement

PORE = Filter pore size (μm)

VOLUME = Groundwater volume passed through filter (ml)

ALP = Measured Al content on the filter ($\mu\text{g/l}$)

CAP = Measured Ca content on the filter ($\mu\text{g/l}$)

FEP = Measured Fe content on the filter ($\mu\text{g/l}$)

MNP = Measured Mn content on the filter ($\mu\text{g/l}$)

SIP = Measured Si content on the filter ($\mu\text{g/l}$)

SP = Measured S content on the filter ($\mu\text{g/l}$)

FILTFE = Fe content of the filtrate ($\mu\text{g/l}$)

FILTFEII = FeII content of the filtrate ($\mu\text{g/l}$)

NA = Measured Na in groundwater (mg/l)

K = Measured K in groundwater (mg/l)

CA = Measured Ca in groundwater (mg/l)

MG = Measured Mg in groundwater (mg/l)

HCO3 = Measured HCO_3 in groundwater (mg/l)

CL = Measured Cl in groundwater (mg/l)

SO4 = Measured SO_4 in groundwater (mg/l)

TR = tritium (units)

D = deuterium (per mik)

O18 = oxygen-18 per mil)

SO4_S = SO_4 -S in groundwater (mg/l) (Sulphur by ICP-AES)

BR = Measured Br in groundwater (mg/l)
 SI = Measured Si in groundwater (mg/l)
 FE = Measured Fe in groundwater (mg/l)
 FETOT = Measured total Fe in groundwater (mg/l)
 FEII = Measured FeII in groundwater (mg/l)
 MN = Measured Mn in groundwater (mg/l)
 LI = Measured Li in groundwater (mg/l)
 SR = Measured Sr in groundwater (mg/l)
 AL = Measured Al in groundwater (mg/l)
 S = Measured S in groundwater (mg/l)
 DOC = Measured dissolved organic compounds (DOC) in groundwater (mg/l)
 PHYAV = Average pH measured at surface
 PHL = Laboratory pH
 SELPH = Selected pH used in PHREEQE calculations
 CONDL = Electrical conductivity measured at the laboratory (mS/m)
 CONDYAV = Average electrical conductivity measured at surface (mS/m)
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 DRILLW = Drill water content (%)
 CALCEH = Calculated Eh according to method suggested by Grenthe et al. (1992).
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 SUM_CAP = Sum of non-Ca colloids/filter
 TOTSUM_C = Sum of non-Ca colloids/observation
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 QUARTZ = Saturation index, log
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 GOETHITE = Saturation index, log
 Fe(OH)₃a = Saturation index, log
 PYRITE = Saturation index, log
 FeS(ppt) = Saturation index, log
 Mn(OH)₂ = Saturation index, log
 MUSCOVIT = Saturation index, log
 CO₂(g) = Calculated partial pressure of carbon dioxide

All observations

	Code	Valid	Median	Minimum	Maximum	Lower	Upper
	Description	N				Quartile	Quartile
DEPTH		84	350.50	50.00	1562.50	196.38	625.78
YEAR		84	89.00	85.00	94.00	88.00	93.00
PORE	Filtre size	84	0.20	0.05	0.40	0.05	0.40
VOLUME		84	380.00	140.00	5400.00	250.00	675.00
ALP	Colloid	80	1.20	0.06	89.00	0.40	3.89
CAP	Colloid	78	132.59	2.10	2330.23	56.44	356.00
FEP	Colloid	80	4.85	0.03	848.00	1.70	10.50
MNP	Colloid	79	0.60	0.03	6.50	0.10	0.80
SIP	Colloid	78	2.73	0.10	49.76	1.00	5.76
SP	Colloid	81	24.00	0.10	243.47	9.20	37.50
FILTFE	Colloid	48	0.25	0.03	2.67	0.06	0.77
FILTFEII	Colloid	48	0.25	0.03	2.69	0.06	0.75
NA	GW	84	1205.00	111.00	8030.00	735.00	1790.00
K	GW	84	6.80	2.40	32.80	4.60	7.60
CA	GW	84	816.50	24.00	18600.00	275.90	1470.00
MG	GW	84	42.00	2.10	219.00	19.50	55.80
HCO3	GW	84	74.50	8.50	307.00	24.00	193.50
CL	GW	84	3330.00	73.00	45500.00	1695.00	5205.00
SO4	GW	84	166.80	31.10	1010.00	112.00	370.50
TR	GW	84	7.80	0.00	26.00	0.90	11.50
D	GW	84	-85.40	-124.80	-47.40	-97.00	-75.90
O18	GW	84	-11.65	-15.80	-7.20	-12.30	-10.45
SO4_S	GW	27	46.58	20.00	317.00	37.00	51.20
BR	GW	24	8.07	0.23	312.00	1.54	69.07
SI	GW	75	5.20	4.00	7.50	4.80	6.40
FE	GW	75	0.33	0.00	2.73	0.13	0.50
FETOT	GW	24	0.40	0.33	1.73	0.33	0.76
FEII	GW	60	0.38	0.03	2.69	0.10	1.06
MN	GW	69	0.39	0.01	3.00	0.16	0.61
LI	GW	27	0.10	0.01	4.25	0.06	0.13
SR	GW	27	5.15	0.19	275.00	1.36	5.33
AL	GW	21	0.09	0.02	0.16	0.04	0.10
S	GW	48	0.62	0.01	2.50	0.43	0.91
DOC		63	5.00	0.50	15.00	1.40	7.50
PHYAV	Avearage pH	39	8.00	6.90	8.70	7.60	8.10
PHL	Laboratory pH	63	8.00	6.80	8.50	7.60	8.10
SELPH	Selected pH	45	7.90	6.90	8.70	7.40	8.05
CONDL	Laboratory conductivity	27	545.00	70.50	9160.00	156.00	660.00
CONDYAV	Average conductivity	42	1055.00	223.00	3390.00	596.00	1830.00
WFLOW	Water flow	69	138.00	61.00	115000.00	117.00	175.00
DRILLW	Drilling water	54	0.26	0.06	13.70	0.16	1.99
CALCEH	Calculated Eh	45	-308.81	-410.48	-198.99	-336.46	-274.82
USEDEH	Used Eh	45	-200.00	-200.00	-200.00	-200.00	-200.00
SUM	Sum of colloids/filtre	84	174.08	0.00	2401.30	92.65	420.75
TOTSUM	Sum of colloids/observation	28	596.85	18.10	6284.08	284.02	1360.00
SUM_CAP	Sum of non-Ca colloids/filtre	84	39.40	0.00	967.50	19.10	71.23
TOTSUM_C	Sum of non-Ca colloids/observation	28	123.85	1.50	1166.50	72.75	224.35
CALCITE	Saturation index	72	0.45	-0.45	1.06	0.21	0.63
SIO2_AM_	Saturation index	45	-0.96	-1.03	-0.86	-0.97	-0.90
QUARTZ	Saturation index	45	0.38	0.31	0.48	0.36	0.43
AL_OH_3A	Saturation index	12	-1.40	-2.30	-0.26	-2.22	-0.46
GOETHITE	Saturation index	45	1.95	-0.55	3.86	1.39	2.41
FE_OH_3A	Saturation index	45	-3.95	-6.45	-2.03	-4.50	-3.48
PYRITE	Saturation index	45	-3.30	-17.17	11.37	-6.34	4.08
FES_PPT	Saturation index	45	-6.79	-13.97	1.04	-8.26	-2.72
MN_OH_2	Saturation index	33	-5.41	-5.97	-4.35	-5.72	-5.24
MUSCOVIT	Saturation index	12	8.50	6.61	10.81	6.75	10.46
CO2_G_	Saturation index	72	-2.89	-5.01	-1.84	-4.00	-2.34

Non saline observations

	Code	Valid	Median	Minimum	Maximum	Lower	Upper
	Description	N				Quartile	Quartile
DEPTH		36	274.38	70.00	800.90	70.00	440.50
YEAR		36	93.00	87.00	94.00	88.50	93.00
PORE	Filtre size	36	0.20	0.05	0.40	0.05	0.40
VOLUME		36	250.00	140.00	800.00	220.50	355.00
ALP	Colloid	32	1.35	0.06	13.00	0.68	2.48
CAP	Colloid	30	78.12	27.50	251.58	50.00	122.99
FEP	Colloid	32	6.02	0.03	60.00	3.51	14.56
MNP	Colloid	31	0.68	0.03	1.50	0.08	1.10
SIP	Colloid	30	5.03	0.60	49.76	1.40	9.43
SP	Colloid	33	29.30	0.48	243.47	13.00	42.00
FILTFE	Colloid	15	0.12	0.03	2.30	0.04	1.67
FILTFEII	Colloid	15	0.12	0.03	2.30	0.03	1.67
NA	GW	36	674.00	111.00	860.00	271.50	750.50
K	GW	36	4.35	2.40	7.40	3.10	4.75
CA	GW	36	192.50	24.00	440.00	107.00	339.90
MG	GW	36	21.00	4.60	64.00	8.40	49.80
HCO3	GW	36	211.50	61.00	307.00	96.00	301.50
CL	GW	36	1455.00	73.00	2010.00	528.00	1710.00
SO4	GW	36	112.00	31.10	180.00	63.50	144.00
TR	GW	36	11.50	0.10	24.00	6.95	15.00
D	GW	36	-76.95	-124.80	-67.90	-82.40	-74.55
O18	GW	36	-10.60	-15.80	-9.80	-10.95	-10.00
SO4_S	GW	21	46.58	20.00	51.20	29.00	46.89
BR	GW	18	5.20	0.23	8.14	0.67	8.14
SI	GW	36	6.60	4.80	7.50	5.10	7.10
FE	GW	36	0.36	0.04	2.23	0.23	1.38
FETOT	GW	21	0.37	0.33	1.73	0.33	0.76
FEII	GW	24	0.75	0.04	2.23	0.08	1.70
MN	GW	33	0.43	0.10	3.00	0.15	0.50
LI	GW	21	0.10	0.01	0.13	0.02	0.11
SR	GW	21	5.12	0.19	5.33	0.32	5.15
AL	GW	12	0.10	0.04	0.16	0.07	0.14
S	GW	15	0.71	0.46	1.20	0.63	1.10
DOC		30	8.25	1.40	15.00	5.00	14.00
PHYAV	Avearage pH	15	8.00	6.90	8.70	7.00	8.20
PHL	Laboratory pH	24	7.90	6.80	8.20	7.30	8.20
SELPH	Selected pH	15	8.00	6.90	8.70	7.00	8.20
CONDL	Laboratory conductivity	21	545.00	70.50	660.00	143.00	547.50
CONDYAV	Average conductivity	15	453.00	223.00	680.00	255.00	596.00
WFLOW	Water flow	33	156.00	100.00	8700.00	138.00	2812.50
DRILLW	Drilling water	24	0.23	0.06	13.70	0.17	9.50
CALCEH	Calculated Eh	15	-320.20	-410.48	-198.99	-326.48	-222.67
USEDEH	Used Eh	15	-200.00	-200.00	-200.00	-200.00	-200.00
SUM	Sum of colloids/filtre	36	133.70	0.00	382.46	72.49	174.08
TOTSUM	Sum of colloids/observation	12	454.70	77.04	777.84	187.94	524.40
SUM_CAP	Sum of non-Ca colloids/filtre	36	41.14	0.00	256.47	20.65	85.99
TOTSUM_C	Sum of non-Ca colloids/observation	12	119.45	49.54	475.99	75.07	231.99
CALCITE	Saturation index	36	0.37	-0.45	1.06	0.08	0.62
SIO2_AM_	Saturation index	15	-0.96	-0.97	-0.88	-0.96	-0.95
QUARTZ	Saturation index	15	0.38	0.36	0.46	0.38	0.39
AL_OH_3A	Saturation index	9	-0.66	-2.30	-0.26	-2.30	-0.46
GOETHITE	Saturation index	15	2.35	-0.55	3.86	0.25	2.39
FE_OH_3A	Saturation index	15	-3.54	-6.45	-2.03	-5.64	-3.50
PYRITE	Saturation index	15	-6.34	-17.17	11.37	-8.41	11.24
FES_PPT_	Saturation index	15	-8.26	-13.97	1.04	-9.58	0.73
MN_OH_2	Saturation index	9	-5.97	-5.97	-5.32	-5.97	-5.64
MUSCOVIT	Saturation index	9	10.10	6.61	10.81	6.61	10.46
CO2_G_	Saturation index	36	-2.39	-3.97	-1.84	-2.95	-2.18

Saline observations

	Code	Valid	Median	Minimum	Maximum	Lower	Upper
	Description	N				Quartile	Quartile
DEPTH		49	460.49	50.00	1562.50	240.50	691.06
YEAR		49	89.00	85.00	94.00	88.00	89.00
PORE	Filtre size	49	0.20	0.05	0.40	0.05	0.40
VOLUME		49	485.00	140.00	5400.00	375.00	1050.00
ALP	Colloid	49	0.70	0.10	89.00	0.30	4.25
CAP	Colloid	48	302.50	2.10	2330.23	105.50	612.50
FEP	Colloid	48	4.35	0.10	848.00	1.01	7.80
MNP	Colloid	48	0.60	0.08	6.50	0.20	0.80
SIP	Colloid	48	1.30	0.10	47.00	0.60	4.07
SP	Colloid	48	23.00	0.10	108.15	6.70	35.00
FILTFE	Colloid	33	0.25	0.04	2.67	0.07	0.47
FILTFEII	Colloid	33	0.25	0.04	2.69	0.07	0.45
NA	GW	49	1680.00	860.00	8030.00	1300.00	2200.00
K	GW	49	7.30	4.80	32.80	7.00	9.00
CA	GW	49	1400.00	370.00	18600.00	990.00	1890.00
MG	GW	49	49.00	2.10	219.00	24.00	62.50
HCO3	GW	49	27.00	8.50	298.00	11.00	70.00
CL	GW	49	5000.00	2010.00	45500.00	3820.00	6330.00
SO4	GW	49	330.00	106.00	1010.00	210.00	519.00
TR	GW	49	3.00	0.00	26.00	0.40	8.00
D	GW	49	-94.30	-108.90	-47.40	-99.60	-85.00
O18	GW	49	-11.90	-13.90	-7.20	-13.00	-11.65
SO4_S	GW	7	309.00	51.20	317.00	309.00	317.00
BR	GW	6	221.00	130.00	312.00	130.00	312.00
SI	GW	40	5.00	4.00	7.50	4.20	5.45
FE	GW	40	0.26	0.00	2.73	0.08	0.44
FETOT	GW	4	0.43	0.37	0.43	0.40	0.43
FEII	GW	36	0.29	0.03	2.69	0.11	0.46
MN	GW	37	0.31	0.01	2.57	0.20	0.68
LI	GW	7	1.71	0.13	4.25	1.71	4.25
SR	GW	7	80.70	5.12	275.00	80.70	275.00
AL	GW	9	0.06	0.02	0.09	0.02	0.07
S	GW	33	0.50	0.01	2.50	0.18	0.69
DOC		34	1.70	0.50	11.00	1.10	5.30
PHYAV	Avearage pH	24	7.95	7.50	8.50	7.75	8.05
PHL	Laboratory pH	39	8.00	7.30	8.50	7.60	8.10
SELPH	Selected pH	30	7.90	7.30	8.50	7.50	8.00
CONDL	Laboratory conductivity	7	3320.00	660.00	9160.00	3320.00	9160.00
CONDYAV	Average conductivity	27	1540.00	860.00	3390.00	1080.00	1890.00
WFLOW	Water flow	37	135.00	61.00	115000.00	96.00	144.50
DRILLW	Drilling water	30	0.40	0.08	2.60	0.14	0.81
CALCEH	Calculated Eh	30	-307.24	-381.82	-269.47	-336.46	-277.63
USEDEH	Used Eh	30	-200.00	-200.00	-200.00	-200.00	-200.00
SUM	Sum of colloids/filtre	49	337.40	0.82	2401.30	120.10	773.00
TOTSUM	Sum of colloids/observation	17	1011.50	18.10	6284.08	161.88	1956.30
SUM_CAP	Sum of non-Ca colloids/filtre	49	35.90	0.50	967.50	11.10	63.80
TOTSUM_C	Sum of non-Ca colloids/observation	17	115.80	1.50	1166.50	71.40	199.15
CALCITE	Saturation index	37	0.41	0.14	0.86	0.33	0.63
SIO2_AM_	Saturation index	30	-0.95	-1.03	-0.86	-1.00	-0.88
QUARTZ	Saturation index	30	0.39	0.31	0.48	0.33	0.46
AL_OH_3A	Saturation index	3	-2.14	-2.14	-2.14		
GOETHITE	Saturation index	30	1.90	1.32	3.17	1.42	2.44
FE_OH_3A	Saturation index	30	-3.99	-4.58	-2.72	-4.47	-3.46
PYRITE	Saturation index	30	-3.24	-13.67	6.72	-5.26	2.65
FES_PPT_	Saturation index	30	-6.66	-12.26	-1.14	-7.96	-3.53
MN_OH_2	Saturation index	24	-5.39	-5.72	-4.35	-5.50	-4.95
MUSCOVIT	Saturation index	3	6.89	6.89	6.89		
CO2_G_	Saturation index	37	-3.74	-5.01	-1.84	-4.58	-2.85

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Maria Malmström¹, Steven Banwart¹, Lara Duro², Paul Wersin³, Jordi Bruno³

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January 1995

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Lars Werme, Joachim Eriksson
Swedish Nuclear Fuel and Waste Management Co, Stockholm, Sweden
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Marie Wiborgh (ed.)

Kemakta Konsult AB, Stockholm, Sweden
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² Studsvik Nuclear AB, Nyköping, Sweden
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Bill Wallin

Geokema AB, Lidingö, Sweden

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Swedish Nuclear Fuel and Waste Management Co., Stockholm

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April 1995

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Luis Moreno, Björn Gylling, Ivars Neretnieks
Department of Chemical Engineering and Technology, Royal Institute of Technology, Stockholm, Sweden

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Yvonne Ohlsson, Ivars Neretnieks
Department of Chemical Engineering and Technology, Royal Institute of Technology, Stockholm, Sweden
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Ove Landström¹, Eva-Lena Tullborg²
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² Terralogica AB
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Roland Pusch
Clay Technology AB
February 1995

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Paul R La Pointe¹, Peter Wallmann¹, Sven Follin²
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September 1995

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Kaj Ahlbom¹, Olle Olsson¹, Stefan Sehlstedt²
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² MRM Konsult AB
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Björn Lagerblad, Jan Trägårdh
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Kristina Skagius¹, Anders Ström², Marie Wiborgh¹
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November 1995

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