3 METHODS OF INVESTIGATION

3.1 SCOPE OF WORK

A literature study, which included a request for data from various mining companies, was undertaken in order to describe the contamination status at selected sites and assess various attenuation and migration mechanisms of contaminants from gold mining tailings and affected soils. All relevant information was entered into a database-linked geographic information system, which allows the evaluation and display of important features such as the spatial distribution of tailings dams, reclaimed sites, surface water systems, and residential and industrial areas. The map based on such a geographic information system, contains information gathered from topographical and geological maps and technical drawings provided by mining companies and a satellite image of the Johannesburg area (Figure E.13, Appendix E). Based on this information, a selection of appropriate sites was carried out. A total of eleven sites was identified as being suitable for the purposes of this study. Sampling was conducted at seven of the eleven sites in order to close gaps in the database. All the investigated sites were either partially or completely reclaimed for the recovery of gold, and are situated above Karoo or dolomitic aquifer systems. Therefore, the selected sites represent the most typical environmental conditions for tailings disposal sites in South Africa.

Furthermore, all sites are within one km of either residential areas or areas of agricultural land use. Most of the mine residue deposits have been present in the area for decades. The study area is situated in the Gauteng Province and stretches from Brakpan in the north to Springs in the south, with the exception of one site, being situated close to Potchefstroom in the North-West Province. The case studies comprised a visual site inspection of all sites with special reference to land use and development of residential areas. Soil profiling to depths of 2.40 m was conducted and samples were collected from the seven selected reclaimed sites and analysed with respect to geotechnical, mineralogical and geochemical parameters. The main objective of field and laboratory testing was to investigate the pathway of contaminant migration in association with acid mine drainage from the tailings dams through the unsaturated zone into the receiving groundwater system. A geochemical load index was applied in order to indicate the worst-case scenario for the study sites. This index comprises different contamination classes and thus, various risk levels for groundwater resources and land development.

3.2 FIELD SURVEY AND SITE INFORMATION

A total of 22 test pits (three per site, except at site F where four test pits were investigated) were excavated by means of a Schaeff backactor on a Mercedes Unimog truck (Figure E.8, Appendix E). The test pits were excavated to a maximum depth of 2.40 m in order to determine depth to bedrock, underlying pedological conditions and the potential presence of a perched groundwater table. Samples for analyses were taken at various depths: topsoil (< 30 cm depth) to water table or maximum test pit depth. Pedological conditions were described using the protocol of the Soil Classification Working Group (1991) and Jennings, Brink & Williams (1973). Soil profiles are presented in Appendix A.

Each site has been described according to site characteristics (e.g. area, geology, vegetation, reclamation status), geotechnical parameters, hydrogeological properties and contaminant assessment of the unsaturated and saturated zones. Additional information regarding land use in close proximity to the site was obtained from topographical maps, ortho-photographs and a satellite image of the Johannesburg area.

Table 3.1 presents a summary of the selected tailings dam sites with important features such as geology, area and status of reclamation and rehabilitation.

TABLE 3.1 - Summary of site information for the investigated sites. Note that the period of deposition is unknown except study site I: 1977–1984.

Site	Type	Lithological units	Area ha	Reclamation status %	Period of reclamation	Environ. monitoring	Rehab. Measures
A	Slime	Vryheid F. Dwyka F.	50	50	Until 1996	n. a.	Paddocked
B	Slime	Dwyka F. Oaktree F.	47	90	Late 1980s	None	Paddocked
C	Slime	Dwyka F.	28	100	1977 – mid 1980s	None	All slime removed
D	Slime	Dwyka F.	71	100	1977 – mid 1980s	None	All slime removed
E	Slime	Dwyka F. Oaktree F.	70	90	Early 1990s	Surface water	Outstanding
F	Slime	Vryheid F. Dwyka F.	120	95	Late 1980s – early 1990s	None	Slime removal ongoing
G	Slime	Vryheid F.	13	95	1994 – 1995	None	Partly paddocked
H	Slime	Oaktree F.	4	100	1940s	Surface and groundwater	n. a.
I	Slime	Post-Karoo dolerites, Monte Christo / Oaktree F.	1400	30	1996	Surface and groundwater	Partly paddocked
J	Slime	Dwyka F.	117	85	1985 – present	Surface water	Partly paddocked
K	Slime	Karoo S., Malmani SG	111	15	Started in 1997	Surface and groundwater	n. a.

Note: Lithostratigraphical units are shown in Table 2.1. Reclamation status refers to the percentage volume of tailings removed. Abbreviations: F = Formation and SG = Subgroup and S = Supergroup.

Data are generally lacking for the period of deposition due to changing ownership of the sites.

3.3 SAMPLING AND LABORATORY TESTING

Soil samples were analysed for various chemical constituents in order to determine the degree of contamination. The following elements were measured in the fine grain size fraction: Fe_2O_3 (total), MnO, As, Ba, Co, Cr, Cu, Mo, Ni, Pb, Rb, Th, U, V, Zn and Zr. The grain size fraction < 75 μm was chosen for total element analyses, since the majority of trace elements is concentrated in the clay-silt particle size range (Förstner & Kersten, 1988; Förstner, 1989; Labuschagne, Holdsworth & Stone, 1993). Geochemical element analyses of the solid phase were conducted using the simultaneous X-ray fluorescence spectrometry (XRF) technique. Trace and major element analyses on gold mine tailings were obtained from Rösner (1996), who selected five different tailings dams in the East Rand for sampling at various depths. Additionally, information on the geochemistry of tailings was supplemented with data from Adamson (1973) and Blight & Du Preez (1997). Soil extraction tests were conducted on a limited number of samples, generally characterised by high pollution loads. Subsequently, leachate samples were measured by means of inductively coupled plasma mass spectrometry (ICP-MS) with the exception of U, which was measured spectrophotometrically (as U_3O_8) after solvent extraction. Mineralogical analyses were carried out on a limited number of tailings samples by means of X-ray diffraction (XRD) and additional data for soils affected by gold mine tailings were provided by Joubert (1998).

Standard foundation tests were conducted on a large number of soil samples to determine the general geotechnical properties such as hydraulic conductivity and size of the clay fraction.

Standard quality control was applied during analytical testing (e.g. standards and blank samples, double-measurements). The accuracy or correctness of data obtained from other sources (e.g. mining companies) was estimated using the electro-neutrality (EN) equation, which is based on the percentage difference defined as follows (American Public Health Association, 1995):

$$EN(\%) = \frac{\sum cations - \sum anions}{\sum cations + \sum anions} \cdot 100 \qquad [3.1]$$

where cations and anions refer to equivalent concentrations. The sum (expressed as milliequivalents per litre) of positive and negative charges in water must balance and are calculated from the cations Na^+, K^+, Mg^{2+} and Ca^{2+}, and the anions Cl^-, HCO_3^-, SO_4^{2-} and NO_3^-. Results with deviations larger than 5 per cent would require an examination of sampling and analytical procedures (which is not possible for data obtained from other sources) and should be considered with caution.

Geochemical background values typical of pristine topsoils, overlying rocks of the Vryheid Formation (i.e. sandstone and shale) and Malmani Subgroup (i.e. dolomites) were obtained from Aucamp (1997) and Elsenbroek & Szczesniak (1997).

The various analytical tests conducted in this study are summarised in Table 3.2 and the methods are discussed in detail in the next paragraphs.

TABLE 3.2 - Summary of analytical tests applied in this study.

Number of samples	**Tailings**	**Soil**	**Method**	**Testing facility**
Total element analyses	36	81	X-ray fluorescence spectrometry (XRF)	University of Pretoria Council for Geoscience
Mineralogical composition	16	-	X-ray diffraction (XRD)	Council for Geoscience
Soil extraction tests	13	16	NH_4NO_3 / Inductively coupled plasma mass spectrometry (ICP-MS)	Council for Geoscience / Anglo American Research Laboratories
Soil paste pH	-	58	American Society for Testing Materials (1990)	Council for Geoscience
Geotechnical properties	-	59	Standard foundation tests	Council for Geoscience

3.3.1 X-ray fluorescence spectrometry (XRF)

The fine grain size fraction of 81 soil samples from the investigated sites were analysed using a Philips PW 1606 simultaneous XRF at the Council for Geoscience in Pretoria. Tailings samples were analysed by Rösner (1996) using the ARL 8420 wavelength dispersive XRF of the University of Pretoria. In addition, the trace elements As and Sn were determined at the University of Erlangen-Nürnberg (Germany) using the Philips PW 2400 XRF.

Samples were dried and roasted at 950°C to determine loss on ignition (LOI). Major element analyses were conducted on fused beads according to the description of Bennet & Oliver (1992), using 1 g of a pre-roasted sample and 6 g of a lithium-tetraborate flux mixed in a crucible (i.e. 5 per cent Au, 95 per cent Pt) and fused at 1050°C in a muffle furnace with occasional swirling. The glass disk is poured into a pre-heated Pt/Au mould and then bottom surface analysed. Trace elements were analysed on pressure powder pellets using a saturated Movial solution as binder. The spectrometer was calibrated with certified reference materials. A parameter program was used for matrix correction of major elements and Ba, Cl, Co, Cr, Sc, S and V.

Standard deviation and detection limits for major and trace elements are listed in the Tables 3.3a and 3.3b:

TABLE 3.3a - Detection limits and standard deviations for major elements using XRF technique (after Bennett & Oliver, 1992).

Major Element	**Standard deviation**	**Detection limits %**
SiO_2	0.4	0.02
TiO_2	0.03	0.0032
Al_2O_3	0.3	0.01
Fe_2O_3	0.3	0.0097
MnO	0.0065	0.0013
MgO	0.1	0.0118
CaO	0.07	0.01
Na_2O	0.11	0.0265
K_2O	0.06	0.005
P_2O_5	0.08	0.01
Cr_2O_3	0.0053	0.0006
NiO	0.01	0.0013
V_2O_5	0.0018	0.0008
ZrO_2	0.005	0.0009
CuO	0.0037	0.0003

TABLE 3.3b - Detection limits and standard deviations for trace elements using XRF technique (after Bennett & Oliver, 1992).

Trace Elements	**Standard deviation**	**Detection limits mg/kg**
As	n. a.	10
Ba	50	50
Cl	100	11
Cu	3	2
Cr	40	15
Ga	2	2
Mo	1	1
Nb	3	2
Ni	6	3
Pb	3	3
Rb	5	3
S (in %)	0.02	0.01
Sc	5	1
Sn	n. a.	2
Sr	4	3
Th	6	5
U	6	3
V	10	1
Y	3	3
Zn	5	4
Zr	6	10

3.3.2 Soil extraction tests and inductively coupled plasma mass spectrometry

Various leaching methods have been discussed in Förstner (1995) to estimate the concentration of an element in the easily soluble and exchangeable fraction. In this study simple salt solutions (i.e. 1 M NH_4NO_3) were used to estimate the mobility of trace elements in the fine fractions (particle size < 75 μm) of soils.

The NH_4NO_3 soil extraction method is an accepted method in the Federal Environmental Agency of Germany for conducting hazard assessments (Umweltbundesamt, 1996), and is likely to become an internationally recognised soil leaching method for environmental studies. Schloemann (1994) and Utermann, Gäbler, Hindel, Kues, Mederer & Pluquet (1998) described the method in detail, where the extracted solution stabilises in the acid range, thus ensuring that the leached element remains in solution. This method is simple to handle and rapid. The results of the soil extraction methods using salt solutions such as NH_4NO_3 can be correlated with the amount of ions held on charged soil surfaces (e.g. clays and organic material, oxides) and with the concentration of these ions in the soil solution (Davies, 1983).

In this study, extracted concentrations were compared to the total concentration in the solid phase and to threshold values for NH_4NO_3 leachable trace elements, after Prüeß, Turian & Schweikle (1991). Concentrations higher than the set threshold concentrations (TC) can result in a limitation of the soil function, according to Prüeß et al. (1991). The threshold concentrations for extractable elements in soils are listed in Table 3.12.

Steffen, Robertson and Kirsten (1988) reported that extraction tests are not necessarily representative of the quantities of trace elements that will be leached out of the deposits in a single rain storm, but only of the quantities which are potentially available.

3.3.2.1 Method for the soil extraction test

The NH_4NO_3 (1 Molar) extraction method according to Schloemann (1994) was applied and is described in the following:

1. Sieve the air-dried soil sample through a 75 μm nylon or stainless steel sieve and discard the coarse fraction.
2. Weigh out 20 g air-dried sieved soil into an acid cleaned 250 ml Erlenmeyer flasks.
3. Ad 50 ml NH_4NO_3 (1 mol/l NH_4NO_3) solution to the soil sample.
4. Closed Erlenmeyer flasks containing the soil and salt solution were shaken for 2 hours at 20 °C on a horizontal shaking table.
5. Filtrate supernatant solution through a 150 mm diameter Whatmann no. 40 ash-less filter paper into acid-cleaned 100 ml polyethylene bottles.
6. Stabilise the extract by adding 0.5 ml conc. HNO_3 (i.e. 65 per cent).
7. Determine trace elements except U by means of the ICP-MS. Uranium is determined spectrophotometrically as U_3O_8 after solvent extraction.

3.3.2.2 Inductively coupled plasma mass spectrometry (ICP-MS)

Leachate samples from extraction tests and seepage samples collected from shallow groundwater were analysed for trace elements using a FISON simultaneous ICP-MS at Anglo American Research Laboratories in Johannesburg.

The samples were poured in pre-cleaned polyethylen bottles, instantly acidified (i.e. diluted HNO_3) to prevent precipitation, and stored in a refrigerator. All samples were submitted to the laboratory within 24 h for analysis. The detection limits for the ICP-MS are listed in Table 3.4:

TABLE 3.4 - Detection limits for ICP-MS
(after Heinrichs & Herrmann, 1990).

Element	**Detection limit** µg/l
Fe	0.8
Mn	0.05
Al	0.2
As	0.5
Ca	5.0
Co	0.02
Cr	0.03
Cu	0.03
Na	0.05
Ni	0.03
Pb	0.03
Sn	0.04
Th	0.03
U	0.03*
V	0.04
Zn	0.09

Note: *U was measured spectrophotometrically (as U_3O_8) after solvent extraction.

3.3.3 X-ray diffraction (XRD)

The mineralogical composition of sixteen tailings samples was analysed using a Siemens D 5000 XRD at the Council for Geoscience in Pretoria. Results are semi-quantitative to ± 20 per cent or better, depending on the crystallinity of the mineral present and the sample preparation method. Detection limit for a mineral is 1-3 per cent depending on background noise and peak resolution of the diffractogram pattern, as well as sample preparation. The XRD is run with a secondary monochromator[5] and a copper X-ray tube (λ = 1.54 Å), which is appropriate for general sedimentological studies. Data evaluation was supported by using the software package *Diffrax Plus*.

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[5] The use of a secondary monochromator would reduce the fluorescence of a high Fe content with Cu radiation, decreasing the background noise, and thereby limiting the interference with mineral identification and percentage estimation.

For the purpose of this study, about 1-3 g of representative sample material was milled under alcohol to about 5-10 µm (talcum powder size) using a mortar and pestle. Subsequently, the sample was mounted in a sample holder in such a way to minimise preferred orientation for XRD analysis.

3.4 GEOTECHNICAL PROPERTIES

Standard foundation tests were conducted to characterise basic geotechnical parameters such as:

- Distribution of soil types across the investigated sites;
- Grain size distribution;
- Atterberg limits (i.e. soil plasticity);
- Hydraulic conductivity (i.e. derived from geotechnical parameters).

3.4.1 Estimation of hydrogeological conditions from geotechnical data

The amount of water that can be stored in a certain volume of soil and the rate of water movement (flux) through that soil, depend on various parameters such as soil texture (i.e. particle size distribution), soil structure (i.e. aggregation of soil particles), density and shape of grains and the presence of preferential flow paths. Water may occupy both interstructural and textural voids (between the particles). At high moisture contents, water-flow through the voids may be the dominant transport process, but becomes rapidly less important as the soil moisture decreases and matrix flow dominates. Generally, the coarser and/or better sorted the solid particles, the larger the intervening voids and the easier it will be for draining water to pass through. As a result, sandy soils tend to be freely draining and permeable, while clayey soils are both slower to absorb and to drain water.

Clay minerals (or grains < 0.002 mm) are the most important particle size fraction in determining the physical and chemical properties of soil. The silt and sand fractions mainly comprise quartz and other primary minerals that have undergone little chemical alteration while the clay material, in contrast, results from chemical weathering, forming secondary minerals with a great variety of properties (Wild, 1988 and White, 1989).

One difference is that the clay minerals often consist of plate-like sheets and have a much larger specific surface than other particle types such as silt and sand. Most clays have negatively charged surfaces and are balanced externally by cations which are not part of the clay structure and which can be replaced or exchanged by other cations. The latter process is known as *cation exchange capacity* (CEC) and plays a major role in contaminant attenuation processes. Some clay minerals (e.g. montmorillonite) have only weak bonds between the adjacent sheets, and the internal surfaces may also be available for taking part in reactions such as the retention and release of nutrients, salts and contaminants (Rose, 1966). Moisture can enter between these sheets, causing them to shrink or to swell. Many clayey soils swell under moist conditions and shrink and desiccate under dry conditions, which could influence the porosity and other hydraulic properties of the soil. Typical properties of common clay minerals are shown in Table 3.5.

TABLE 3.5 - Typical values of some properties of common clay minerals (White, 1989 and Holtz & Kovacs, 1981).

Parameter	**Kaolinite**	**Illite**	**Chlorite**	**Montmorillonite**	**Vermiculite**
Thickness (nm)	50 – 2000	30	30	3	n. a.
Diameter (nm)	300–4000	1000	1000	100-1000	n. a.
Specific surface (m^2/kg)	15	80	80	800	n. a.
CEC ($cmol_{(+)}/kg$ soil)	3 - 20	10 - 40	n. a.	80 - 120	100 – 150
Plasticity	Low	Medium	Medium	High	Medium
Swelling/ Shrinking	Low	Medium	Low	High	n. a.

Note: CEC means cation exchange capacity, (+) refers to cations. 1 $cmol_{(+)}/kg$ equals 1 meq/100 g soil.

Table 3.6 summarises the clay contents of the investigated study sites, which were determined by geotechnical testing.

TABLE 3.6 - Clay contents in soil samples of the study sites (n=59).

Site	**MIN** %	**MAX** %	**AVG** %	**Number of samples (n)**
A	7.5	15.8	10.9	8
B	29.8	63.7	40.2	8
C	19.5	48.7	34.7	9
D	14.4	51.4	33.3	9
E	33.3	46.6	38.6	7
F	13.7	44.2	31.9	12
G	14.1	31.5	22.5	6
All sites	7.5	63.7	30.7	59

The clay contents of the investigated soils show a varying spatial distribution as a result of different soil types, parent rock material and weathering conditions, characterised by seasonal changes. In addition, Table 3.7 shows a list with the percentage portion of the clay content in the soils of the study sites. Nearly 50 per cent of all soil samples contain more than 30 per cent clay. No relationship was found between clay content and sample depth, nor between clay content and element concentrations.

TABLE 3.7 - Classification of clay contents according to the percentage portion in soil samples of the study sites.

Clay amount	**> 10** %	**> 10–20** %	**> 20–30** %	**> 30–40** %	**> 40– 50** %	**> 50–60** %	**> 60** %
Number of samples	3	12	11	18	11	3	1
Percentage of samples	5.1	20.3	18.6	30.6	18.6	5.1	1.7

Geotechnical data allowed the estimation of hydraulic conductivities in 23 soil samples, which are listed Table 3.8 (approach after Mathewson, 1981 and Tavenas, Jean, Leblond & Leroueil, 1983). The methodology for these approaches is discussed in paragraph 3.5.3.

TABLE 3.8 - Estimated hydraulic conductivties of soils from the study sites (n=23) according to the methods of Mathewson (1981) and Tavenas et al. (1983).

Site	**MIN** m/s	**MAX** m/s	**Number of samples (n)**
A	1×10^{-9}	1×10^{-9}	3
B	1×10^{-10}	1×10^{-10}	2
C	1×10^{-11}	9.5×10^{-8}	4
D	7.5×10^{-10}	7×10^{-9}	4
E	8×10^{-10}	9×10^{-9}	4
F	6×10^{-10}	6×10^{-9}	4
G	1×10^{-9}	1×10^{-9}	2
All sites	1×10^{-11}	9.5×10^{-8}	23

Estimated hydraulic conductivities range from impermeable (i.e. 1×10^{-11} m/s) to a very low hydraulic conductivity (i.e. 9.5×10^{-8} m/s), which is typical for soils having a high clay content. Since more than half of all investigated soil samples showed a clay content > 30 per cent, the unsaturated zone can be regarded as nearly impermeable. However, high salinity in groundwater samples and the lack of correlation between contaminant concentrations and clay contents in soils (Table B.9, Appendix B) collected on and around the investigated sites indicate the presence of alternative flow mechanisms that bypass the soil matrix, known as preferential flow. Such conditions would result in high hydraulic conductivities and an insufficient contact time between the aqueous and solid phase (soil matrix) to adsorb contaminants.

3.4.2 Soil types and properties

The reclaimed sites are underlain by various soil types, which were described in the field according to the protocol of the *Soil Classification System for South Africa* (Soil Classification Working Group, 1991). The classification comprises the following instructions amongst others:

- Identify major horizons within the soil profile;

- The horizon needs to be within the first 1.5 m of the soil profile. In this study soils were described up to the bottom of the test pit (max. 2.40 m depth).
- Use the key in the soil classification protocol to identify the soil type.

Each soil type is distinguished by a name such as *Shortlands* and differs from other soil types on the basis of certain pedological criteria. It is important to note that the different horizons have not been formed separately, but as a result of a combination of processes acting on the entire soil profile. Thus, horizons of the same soil type can vary considerably in terms of different physical characteristics such as clay content and hydraulic conductivity. Each soil is therefore described according to general geotechnical properties by means of field testing methods, which are outlined below (Jennings et al., 1973):

- Moisture;
- Colour;
- Consistency;
- Structure;
- Soil type;
- Origin.

It is important to note that soils of the study area are generally characterised by low organic matter contents as a result of poor vegetation and extensive weathering. The following soil types were encountered at the investigated sites and are summarised in Table 3.9:

TABLE 3.9 - Soil types occurring at the investigated sites.

Soil type	**Description**	**Occurrence at the study sites**
Arcadia	Vertic A horizon	Site I
Avalon	Orthic A horizon, a yellow-brown apedal B horizon and a soft plinthic B horizon	Test pits A/2, A/3, C/1, G/1, G/2, G3
Glencoe	Orthic A horizon, a yellow brown apedal B horizon and a hard plinthic B horizon	Test pit A/1
Katspruit	Orthic A horizon above a water table (G horizon)	Test pit E/1
Rensburg	Vertic A horizon above a water table (G horizon)	Test pits E/2, E/3

TABLE 3-9 continued.

Shortlands	Orthic A horizon above a red structured B horizon.	Test pits B/1, B/2, C/2, D/1, F/1, F/2, F/3
Willowbrook	Melanic A horizon above a water table (G horizon)	Test pits B/3, C/3, D/2, D/3

According to the Soil Classification System of South Africa, an orthic A horizon is a surface horizon without significant organic matter or clay content. A melanic A horizon is a dark coloured soil unit with strongly developed structure without slickensides, while a vertic A horizon is a soil unit with strongly developed structure with slickensides (only in combination with smectites). A yellow brown apedal soil horizon is a soil unit with a diagnostic yellow colour in the wet state, and has a structure that is weaker than moderate blocky or prismatic when wet. A red structured B horizon is diagnostically red when wet with strongly developed soil structure. A soft plinthic B horizon has undergone localised accumulation of iron and manganese oxides and has a loose to slightly firm consistency in the non-concretionary parts of the horizon, while a hard plinthic B horizon consists of an indurated zone of accumulated iron and manganese oxides. A G-horizon is saturated with water for long periods, and is dominated by grey colours on micro-void and ped surfaces (Soil Classification Working Group, 1991). The most common soil types encountered at the study sites are *Shortlands* and *Avalon*.

The Shortlands soil type usually forms under warm temperate and subtropical climatic conditions. The parent rock of this soil type is generally characterised by a high Fe and basic mineral content, resulting in the accumulation of Fe, Si and Al (termed *fersialization*) in the overlying soil. The high Fe content causes the typical red colour of this soil type (Figure E.2, Appendix E). Kaolinite is the most likely clay mineral occurring in this soil type.

The Avalon soil type form is characterised by the accumulation of sesquioxides (e.g. haematite occurring in ferricretes, Figure E.11) as a result of the removal of silica and bases by fersialization (Driessen & Dudal, 1991), i.e. relative accumulation of Fe and Al in the soil. Thus, sufficient Fe must be present, originating from the parent material or introduced by seepage (e.g. acid mine drainage). The soil typically has a pH ranging from 5 to 8 and high hydraulic conductivity. A short dry period is an important prerequisite for the formation of this soil type. The change from wet (i.e. summer) to dry (i.e. winter) climatic conditions results in the segregation of Fe and Mn concretions (e.g. ferricrete).

Furthermore, although soils in the study area show clay contents up to 60 per cent (nearly 50 per cent of all soil samples contain more than 30 per cent clay) contaminant retention is reasonably low as a result of potential preferential flow (also known as *macro-pore flow*) conditions throughout the unsaturated zone. Preferential flow is caused by large pores represented by structural cracks, fissures or paths of plant-root systems (Ward & Robinson, 1990). Cracks or fissures can be a result of alternate swelling and shrinking of expanding clays as well as cracking in the subsoil. Cracking at varying depths (up to 2 m) was observed in test pits of study site E and could be caused due to high salt loads in the soil and the alleviation of the former load, i.e. the reverse process of consolidation. In addition, preferential flow paths might be also associated with relatively impermeable ferricrete layers overlying less impermeable soil and enforcing lateral instead of vertical flow. Notably is that the dry density of the soils within the study area is higher if compared to natural soils, indicating consolidation and compaction.

Joubert (1998) investigated clay types at study site F by means of X-ray diffraction and confirmed the dominance of kaolinite $Al_2[(OH)_4/Si_2O_5]$ in all samples (n=10). Kaolinite is typically formed in highly weathered soils such as those occurring in the study area and is often associated with sesquioxides (Brink, 1985). In addition, minor amounts of the clay mineral palygorskite $(Mg,Al)_2Si_4O_{10}(OH) \cdot 4\ H_2O$ were found.

3.5 DATA EVALUATION

3.5.1 Correlation coefficients

Correlation coefficients were calculated for elements occurring in tailings and soils using the Pearson approach, in order to identify trends between two variables (bivariate). A trend is a change of the property *x* corresponding to a change of property *y*. The goodness of the trend can be estimated from the product moment correlation coefficient, *r*.

The coefficient ranges from + 1 to –1, where these extremes denote a perfect linear relationship between *x* and *y*; which may be direct or indirect, respectively, i.e. on a rectangular *x, y* coordinate system, where all points are located strictly on an ascending ($r = +1$) or descending ($r = -1$) straight line. If *r* deviates from + 1, the linear relationship becomes progressively blurred. A trend of increasing *x* associated with increasing *y*, is termed positive. If r is negative, there the correlation is negative. The value $r = 0$ indicates that there is no trend at all. In this case the plotted points are distributed randomly in the coordinate system (Marsal, 1987).

The correlation coefficient *r* can be expressed as:

$$r = \frac{n(\Sigma xy) - (\Sigma x)(\Sigma y)}{\sqrt{[n\Sigma x^2 - (\Sigma x)^2][n\Sigma y^2 - (\Sigma y^2)]}} \qquad [3.2]$$

3.5.2 Geochemical background values

Many soils, surface and groundwaters contain natural concentrations of chemical constituents that exceed soil quality or drinking water standards (Thornton, 1983; Runnels, Sheperd & Angino, 1992). To determine the extent of contamination by toxic elements in a soil or aquatic system, it is necessary to define this natural level (or pre-civilisation value) in a comparable pristine area, and then to subtract it from present values, thereby deriving the total accumulation by anthropogenic impacts (Förstner, 1983). A common approach (Turekian & Wedepohl, 1961; Banks, Reiman, Royset, Skarphagen & Saether, 1995 and Lahermo, Mannio & Taravainen, 1995) is to sample river sediments upstream (similar geology) of the contamination source, where the water quality is presumably unaffected, and to compare the sediment concentrations with samples collected directly downstream of the contamination source. However, especially in highly populated and industrialised areas, it is difficult to identify a sampling point in a river or aquifer where the sediment quality seems to be unaffected by human activities.

In this study, test pits from the reclaimed sites were excavated in soils or alluvial sediments derived mostly from sandstone and shales of the Vryheid Formation, diamictite and shales of the Dwyka Formation, or dolomites of the Malmani Subgroup. The trace element concentrations of the soils retrieved from the investigated sites were compared to average trace element concentrations from topsoils overlying identical bedrock conditions, having the same grain size (< 75 µm). The method to calculate an accumulation factor (i.e. geochemical load index) is discussed in the paragraph 3.5.5. Background concentrations for topsoils of the Malmani Subgroup were obtained from Elsenbroek & Szczesniak (1997) and of the Vryheid Formation from Aucamp (1997). Table 3.10 summarises the average background values for the Vryheid Formation and Malmani Subgroup.

TABLE 3.10 - Average background values and their standard deviations in topsoils obtained from the Vryheid Formation, Karoo Supergroup (n= 21) and Malmani Subgroup, Transvaal Supergroup (n= 4248, particle size <75 µm). Vryheid Formation data from Aucamp (1997) and Malmani Subgroup data from Elsenbroek & Szczesniak (1997).

Parameter	Vryheid Formation mg/kg	Standard deviation	Malmani Subgroup mg/kg	Standard deviation
As	22	4	18	17
Co	14	3	15	40
Cr	130	32	268	958
Cu	35	7	31	81
Fe_2O_3 (in %)	4.4	1.1	6.11	3.1
MnO (in %)	0.08	0.03	0.7	0.7
Mo	23	5	13	8
Ni	45	10	57	107
Pb	15	8	5	25
Th	13	3	18	5
U	n.d.	-	n.d.	-
Zn	103	55	50	79

3.5.3 Estimation of hydraulic conductivities in soils

Saturated hydraulic conductivities in soil samples were estimated according to the methods of Tavenas et al. (1983) or by means of comparison to the permeability, after

Mathewson, (1981). Figure 3.1 shows the procedure after Tavenas et al. (1983) to estimate saturated hydraulic conductivity in a fine-grained soil:

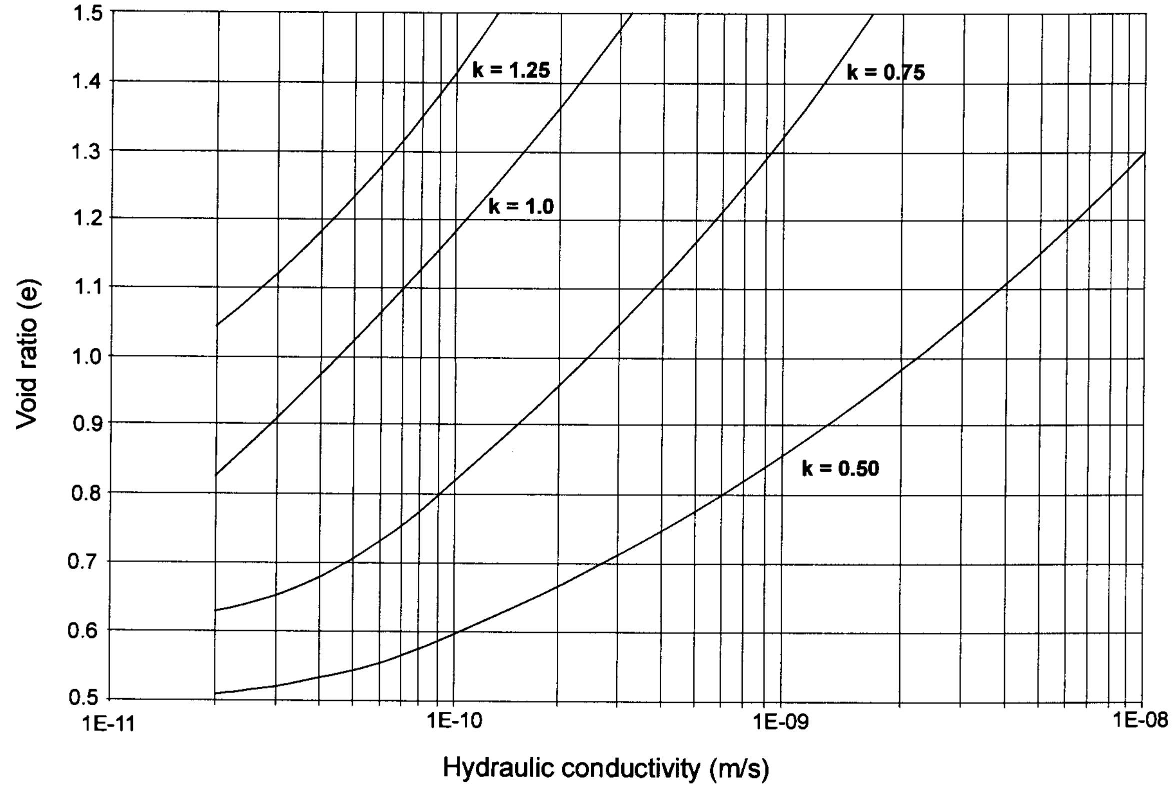

FIG. 3.1 - Estimation of saturated hydraulic conductivity (m/s) in a fine-grained soil (after Tavenas et al., 1983).

Tavenas et al. (1983) established a relationship between saturated hydraulic conductivity and the void ratio, clay fraction and plasticity index (PI) values. The soil properties are plotted on Figure 3.1 and the saturated hydraulic conductivity can be then obtained. The k value represents the sum of clay fraction and the plasticity index (example: a soil with a clay content of 45 % and a PI of 10 would have a k value of 0.45 + 0.1 = 0.55). The latter is one parameter of the Atterberg limits. Table 3.11 presents the US soil classes (U.S.C.S.) and associated hydraulic conductivities, published by Mathewson (1981).

TABLE 3.11 - Estimated hydraulic conductivity value from soil type (after Mathewson, 1981).

Soil type		**U.S.C.S symbol**	**Soil description**	**Hydraulic conductivity m/s**
COARSE-GRAINED SOIL (< 50 % of material pass 0.075 mm)	Gravel	GP	Poorly-graded gravel, little fines	$10^{-5} – 10^{-2}$
		GW	Well-graded gravel, little fines	$10^{-4} – 10^{-2}$
		GM	Silty gravel, gravel-silt-sand mixture	$10^{-8} – 10^{-5}$
		GC	Clayey gravel, gravel- sand-clay mixture	$10^{-9} – 10^{-6}$
	Sand	SP	Poorly-graded sand, little fines	$10^{-6} – 10^{-3}$
		SW	Well-graded sand, gravelly sand, little fines	$10^{-7} – 10^{-3}$
		SM	Silty sand, sand-silt mixture	$10^{-8} – 10^{-6}$
		SC	Clayey sand, sand-clay mixture	$10^{-9} – 10^{-7}$
FINE-GRAINED SOIL (> 50 % of material pass 0.075 mm)	Silt	ML	Inorganic silt and very fine sand, silty or clayey fine sand, clayey silt with slight plasticity	$10^{-9} – 10^{-6}$
		MH	Inorganic silt, micaceous or diatomaceous fine sandy or silty soil, elastic silt	$10^{-9} – 10^{-7}$
	Clay	CL	Inorganic clay with low-medium plasticity, gravelly or sandy clay, lean clay	$10^{-10} – 10^{-8}$
		CH	Inorganic clay with high plasticity, fat clay	$10^{-11} – 10^{-9}$

3.5.4 Short-term impact

The short-term or current contamination impact was investigated using the trace element mobility coefficient (MOB) and threshold excess ratio (TER), which can be expressed as:

$$TER = \frac{ExC}{TC}, \qquad [3.3]$$

where TER is the threshold excess ratio for an element, ExC is the NH_4NO_3 extractable concentration and TC is a given set soil threshold value or soil quality standard, after Prüeß et al. (1991). A concentration that is exceeding the soil threshold value can limit the functioning of the soil. Table 3.12 indicates which of the soil functions are most threatened by a certain contaminant in order to assist in the decision on appropriate counter measures.

TABLE 3.12 - Recommended maximum NH_4NO_3 extractable threshold concentration in soils (after Prüeß et al., 1991).

Element	**Threshold concen.** mg/l	**Soil functions and ranking of concerns**				
		Pollutant buffer with regard to plants for human consumption	Pollutant buffer with regard for animal consumption	Habitat for plants	Habitat for soil organisms	Pollutant filter with regard to groundwater
As	0.1	PC	X	C	X	C
Co	0.5	X	C	C	X	X
Cr	0.1	X	X	X	PC	C
Cu	2	X	C	C	PC	C
Mo	1	X	PC	C	X	X
Ni	1	X	X	C	X	X
Pb	2	PC	C	X	C	C
V	0.1	C	C	INV	X	X
Zn	10	X	X	C	X	X
U	0.04	X	X	X	X	INV

Note: Abbreviations for the ranking of concern if the threshold concentrations are not greatly (e.g. several times) exceeded: PC = primary concern, C = concern, INV = further investigations needed to assess risk. Limited soil functioning only if the threshold concentrations are greatly exceeded: X.

In addition, the mobility of trace elements was derived by comparing the extractable ratio of an element to the total concentration:

$$MOB(in\%) = \frac{ExC}{TotC}, \qquad [3.4]$$

where MOB represents the percentage mobility of an element, ExC is the NH_4NO_3 extractable fraction and TotC is the total trace element concentration measured in soil samples. The MOB coefficient gives an indication of the contaminant concentration, which could be remobilised and is thus potentially bio-available.

In this study, TER and MOB were only applied experimentally to samples of the site F, and were then extrapolated to the other samples from the remaining sites A-G. Site F generally showed the highest pollution potential of all investigated sites with respect to uranium.

3.5.5 Long-term impact

The potential future impact of contamination (worst-case scenario) was assessed by using the geochemical load index (I_{geo}), introduced by Müller (1979).

$$Igeo = \log_2 \frac{Cn}{Bn \cdot 1.5}, \qquad [3.5]$$

where C_n is the measured concentration of the element n in the soil/sediment and B_n is the average geochemical background value obtained from literature. The safety factor 1.5 is used to compensate for variation in the background data. The hazard rating comprises six different contamination classes (i.e. I-VI), which are shown in Table 3.13:

TABLE 3.13 - Classification of contamination by using the geochemical load index (after Müller, 1979).

Geochemical Load Index (I_{geo})	Contamination class	Level of contamination
> 0-1	I	Non-polluted to moderately contaminated
> 1-2	II	Moderately contaminated
> 2-3	III	Moderately to highly contaminated
> 3-4	IV	Highly contaminated
> 4-5	V	High to excessively contaminated
> 5	VI	Excessively contaminated

The application of this index reflects a worst-case scenario, assuming that the total concentration of contaminants contained in the solid phase can be remobilized and hence, is potentially bio-available. The geochemical load index is also intended to represent the long-term capacity of the soil to retain contaminants. If the geochemical load index is exceeded (i.e. greater than 1 or beginning with contamination class II), it can be assumed that further damage to the environment is occurring and the general quality of the environment is deteriorating. However, it is important to note that such an excess does not necessarily mean that damage has to occur right now. It is a calculation that damage, however defined, will occur at some time, possibly now, possibly later, if the introduction of contamination into the soil system is not reduced.

The index was successfully applied in various environmental studies in Germany, such as in the water quality monitoring program of the rivers Rhine and Elbe and in sludge deposits of the Hamburg harbour area (Förstner & Müller, 1974 and Förstner, 1982), and also recently in the geochemical mapping program of topsoils in the city of Berlin and in the Czech Republic, conducted by the German Federal Environmental Agency (Birke, 1998).

4 CASE STUDIES

4.1 CASE STUDY A

4.1.1 Site location and drainage

Site A is situated approximately 1 km east of the suburbs of Benoni and covers an area of approximately 50 ha (Figure 1.1). A residential area is located on the south-western border of the site. The site is located at an altitude of ±1630 m above sea level. Surface drainage direction is towards a non-perennial stream in the north. Test pit locations are shown in Figure D.1[6].

4.1.2 Reclamation and rehabilitation status

Approximately 50 per cent of the tailings material has been removed. No vegetation has been established on site except a poor grass cover and some trees on top of the toe wall (Figure E.1). The oxidised zone in the remaining toe wall is clearly visible and reaches up to a depth of approximately 5 m. Paddocks were constructed to prevent storm water surface run-off from the site.

4.1.3 Geological conditions

Site A is underlain by sedimentary rocks of the Vryheid Formation (sandstone and shale), in the southern part of the site. The Dwyka Formation (diamictite and shale) underlies the northern portion of the site.

4.1.4 Soils

The soils of the site A are mainly represented by the Avalon (test pits A/2 and A/3) and Glencoe (test pit A/1) soil type. No perched water table was encountered during the excavation of the test pits. Table 4.1 presents a summary of soil parameters of site A.

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[6] The Figures are contained in the relevant Appendixes (e.g. D and E).

TABLE 4.1 - Summary of soil parameters for study site A.

	Field test results							
	Geotechnical Parameters				**Derived from geotechnical parameters**			**Soil pH**
Soil units	Clay content %	Plasticity index PI	Dry density kg/m^3	Specific gravity	Void ratio	Saturated permeability m/s	U.S.C.S. soil group	
Colluvium	8.5-15.8	2.89-6.90	1752.9-1816.1	2.72	0.50-0.55	1 x 10^{-9}	ML, SC, GM	3.1-6.5
Nodular ferricrete	11.31	5.11	-	-	-	-	SC	6.9
Hardpan Ferricrete	7.5-12.10	5.94-8.90	1551.43	2.81	0.81	1 x 10^{-9}	ML, SC	4.4-6.9

Note: U.S.C.S. symbols are explained in Table 3.11.

Vertical preferential flow occurs in the clayey sand, colluvial topsoil unit (0.20-0.55 m thickness) which is characterised by cracks. The yellow-brown coloration along the cracks indicates the movement of overlying tailings within these structures. Lateral preferential flow may occur at depths between 0.45-0.60 m on the interface between the pebble marker (very loose consistency) and the underlying nodular ferricrete horizon (test pits A/2 and A/3) or the hardpan ferricrete unit present in A/1 at 0.60 m depth. The hardpan ferricrete unit shows zones of moist, brown, loose clayey sand within the matrix of very densely cemented clayey sand, through which preferential flow might occur. The ferricrete units in the profile might cause a perched water table during the wet season.

4.1.5 Assessment of contamination

4.1.5.1 Trace element concentration in soil

Soil pH conditions are favourable for metal leaching, with the lowest pH of 3.1. All test pits show a significant positive trend for Fe with depth, where the concentration progressively increases from 3.5 per cent in the topsoil up to 18 per cent at 1.0 m depth. The average concentration of Fe is significantly higher than the average background value of 4.4 per cent. This indicates the release of Fe during the pyrite oxidation process and its downward migration through the subsoil.

Chromium generally accumulates in subsoil and exceeds the average background value of 130 mg/kg (i.e. Vryheid Formation) in all samples. However, the bulk of the Cr seems to be associated with Fe_2O_3, reflected in a very good positive correlation ($r = 0.83$). Arsenic in contrast accumulates in the topsoil to concentrations which exceed considerably (5-fold) the average background value of 22 mg/kg (i.e. Vryheid Formation). The high concentrations of As found in the topsoils and the negative correlation with depth strongly suggest the leaching of As from reclaimed tailings into the topsoil, whereupon As seems to be immediately immobilised on solid surfaces such clay minerals or the formation of iron arsenates. The low remobilization and thus mobility of As has been confirmed by extraction tests. In two samples, measured U concentrations of 12 and 7 mg/kg are very low compared to site F, but still above the average background value. A list of geochemical soil data is provided in Table B.1 (Appendix B).

4.1.5.2 Short-term impact

The application of the extrapolated extractable fraction resulted in no excess of the threshold value in case of As, whereas Co, Ni and Zn exceed the threshold in all samples, resulting in a limited soil function. Chromium, Cu, Pb and Zn showed excess in only one sample. It is important to note that the extractable fraction of Ni is 28-fold higher than the recommended threshold value. The high excess of Ni can be explained by a high mobility (MOB of 51 per cent). In contrast, Zn reaches only a maximum of 3-fold excess.

4.1.5.3 Long-term impact

Soil samples of the test pits A/1, A/2 and A/3 show moderate contamination (class II) for As, Fe, Cr, Cu and Pb using the geochemical load index. Only U occurs in concentrations which are classified as moderately to highly contaminated (class III) due to a average background value of 1 mg/kg.

4.1.6 Discussions and conclusions

The low clay content in connection with a generally low organic matter and the presence of preferential flow paths might cause ongoing groundwater contamination. In addition, the most common clay mineral is likely to be kaolinite, characterised by its very low cation exchange capacity. Dry density values (i.e. colluvium) are very high compared to natural soils and only very few plant would grow satisfactorily under such conditions.

The fact that nearly half of the tailings material still remains on site, which provides a source for continuing pyrite oxidation, gives rise to concern. Furthermore, paddocks, which are situated on the reclaimed portion of the site are inappropriate as they increase rainfall infiltration and, thus enhance contaminant migration. Nickel mobility is very high in the soil and could complicate efforts to establish a self-sustaining vegetation cover as the soil function is limited.

In this condition, the soil is not fit for revegetation. It is recommended to remove the paddocks and to cover the remaining tailings material by an impermeable layer. Such a soil cover would prevent the dispersion of fine tailings material by wind and would minimise rainfall infiltration into the tailings and, thus the generation of acid mine drainage. In addition, lime should be added to the topsoil to neutralise acids. The introduction of fertilisers would improve growth conditions on the reclaimed portion of the site. In addition, lime would also enhance the attenuation capacity of the soil.

4.2 CASE STUDY B

4.2.1 Site location and drainage

Site B is situated to the south-east of Springs in close proximity to a residential area on its eastern border (Figure 1.1). The site covers an area of approximately 47 ha and is located at an altitude of ±1615 m above sea level. Surface drainage corresponds to the topographical gradient towards a wetland system in a south-westerly direction. A small squatter camp has been established in immediate vicinity to the reclaimed site. The location of the test pits is shown in Figure D.2.

4.2.2 Reclamation and rehabilitation status

About 90 per cent of the tailings material has been removed (Figure E.2). The remaining tailings will be reclaimed by the year 2003. The site shows vegetation, consisting of a poor developed grass cover and some trees. Paddocks were constructed to prevent storm water surface run-off.

4.2.3 Geological conditions

The southern section of site B is located on dolomites of the Oaktree Formation, Malmani Subgroup, whereas the northern part is situated on Dwyka Formation (diamictite and shale).

4.2.4 Soils

The soils of site B are represented by the Shortlands (test pits B/1 and B/2) and Willowbrook (test pit B/3) soil type. The photographed soil profile is presented in Figure E.3. The soil parameters of study site B are summarised in Table 4.2:

TABLE 4.2 - Summary of soil parameters for study site B.

	Field test results							
	Geotechnical Parameters				**Derived from geotechnical Parameters**			**Soil pH**
Soil units	Clay content %	Plasticity index PI	Dry density kg/m^3	Specific gravity	Void ratio	Saturated permeability m/s	U.S.C.S. soil group	
Colluvium	29.80-63.66	11.28-19.63	1619.22-1695.54	2.45-2.48	0.46-0.51	1×10^{-10}	CL, MH	3.53-6.63
Nodular ferricrete	19.02-50.24	8.21-25.67	-	-	-	-	SC, MH	5.7-6.7

Note: U.S.C.S. symbols are explained in Table 3.13.

The open-structured nature of the topsoil unit may facilitate preferential vertical infiltration, although the abundance of gypsum crystallisation could close up pores and reduce vertical infiltration rates. Lateral preferential flow may occur at the contact of the nodular ferricrete unit with the overlying soil (between 1.40-1.90 m) as ferricrete formation entails precipitation of colloidal Fe-oxides that may close pores to reduce vertical permeability. The nodular ferricrete units in the base of the profiles suggest the presence of either a seasonal perched water table or very high moisture saturation.

4.2.5 Assessment of contamination

4.2.5.1 Trace element concentration in soil

Iron shows in all test pits a positive correlation with depth of the profile (maximum depth 2.10 m). Total concentrations of Fe exceed in nearly all samples the relevant average background value of 6.1 mg/kg, suggesting the release of Fe during pyrite oxidation. Chromium is likely to bound onto Fe-oxides reflected in a significant positive correlation with Fe_2O_3 ($r = 0.83$).

Arsenic, Ni (except in test pit B/3) and Zn, in contrast, tend to accumulate in the upper soil unit and exceed in a number of samples the average background value. It is suggested that these elements have migrated from the tailings into the topsoil, where they became readily immobilised. A further downward migration of Ni and Zn is likely since low pH conditions enhance leaching. A list of geochemical soil data is provided in Table B.1.

4.2.5.2 Short-term impact

The application of the extrapolated extractable fraction would result in a high threshold excess ratio for Co (38-fold), Ni (88-fold) and Zn (53-fold). No significant excess, in contrast, was found for Cr, Pb and Fe.

4.2.5.3 Long-term impact

Site B is moderately contaminated (class II) by Ba, Cu and Ni, moderately to highly contaminated by Co and highly contaminated by Pb using the geochemical load index.

4.2.6 Discussions and conclusions

The high clay content can support contaminant attenuation mechanisms within the soil matrix. However, kaolinite is most likely the dominating clay type, having a low cation exchange capacity. In addition, paddocks are inappropriate as they increase rainfall infiltration and thus, enhance contaminant migration. High mobility of Ni and Zn in soils could complicate efforts to establish a self-sustaining vegetation cover, as these trace elements are known to be phytotoxic.

In this condition, the soil is not fit for revegetation. It is recommended to remove the paddocks and remaining (or residual) tailings material from the surface and to add lime to those portions of the topsoil, delineated as acid soils. Further soil management measures like the addition of fertilisers would improve growth conditions. In addition, lime would also enhance the attenuation capacity of the soil.

4.3 CASE STUDY C

4.3.1 Site location and drainage

The site covers an area of approximately 28 ha and is located to the south-east of Springs (Figure 1.1) at an altitude of ± 1610 m above sea level. A golf course is situated in immediate proximity to the north-eastern border of the reclaimed site. The general surface drainage direction is in southerly direction towards a canal and dam. The location of the test pits is shown in Figure D.3.

4.3.2 Reclamation and rehabilitation status

Site C has been completely reclaimed and is sparsely covered by grass vegetation. No rehabilitation measures (including paddocks) were found.

4.3.3 Geological conditions

Site C is mostly is covered by alluvial sediments deposited by a tributary of a perennial stream. Sedimentary rocks of the Dwyka Formation (diamictite and shale) underlie the alluvium.

4.3.4 Soils

Site C is covered by the following soil types: Colluvium of the Avalon (test pit C/1) type and alluvium of the Shortlands (test pit C/2) and Willowbrook (test pit C/3) types. Table 4.3 summarises the soil parameters for study site C.

TABLE 4.3 - Summary of soil parameters for study site C.

	Field test results							
	Geotechnical parameters				**Derived from geotechnical Parameters**			**Soil pH**
Soil units	Clay content %	Plasticity index PI	Dry density kg/m^3	Specific gravity	Void ratio	Saturated permeability m/s	U.S.C.S. Soil group	
Upper alluvium (< 0.60 m)	24.58-38.41	11.95-23.52	1700.96	2.40	0.41	1 x 10^{-10}- 1 x 10^{-11}	Cl	3.5-6.1
Deeper alluvium (> 0.60 m)	19.51-48.66	13.46-28.47	1738.90	2.57	0.64	1 x 10^{-9}	CH	7.4-7.7
Ferruginous colluvium	28.90-46.78	14.06-14.26	1520.08-1602.09	2.61-2.80	0.63-0.84	8 x 10^{-8}- 9.5-10^{-6}	Cl	3.8-5.0

Note: U.S.C.S. symbols are explained in Table 3.11.

Vertical preferential flow may occur between 0.10-2.10 m in the alluvial soils of test pits C/2 and C/3 as these soil units have a shattered structure (well aggregated soil). A perched water table occurs at 2.00 m in test pit C/2 that implies preferential lateral flow. Lateral preferential flow may occur at 1.20 m in the colluvial soils of test pit C/1 on the boundary between the colluvium and nodular ferricrete units.

4.3.5 Assessment of contamination

4.3.5.1 Trace element concentration in soil

Iron and Cr accumulate in the subsoil and exceed the average background value considerably. Almost neutral pH values were measured at the bottom of the test pits (about 2.40 m depth), indicating the effect of buffering minerals such as carbonates and/or fluctuations in a shallow groundwater table causing dilution effects. The neutral pH conditions at greater depths would result in metal precipitation, and thus immobilisation.

A shallow water table was encountered in test pit C/2 (Figure E.10) and a chemical analysis was conducted to assess the impact of the seepage on groundwater quality. The results are presented in Table 4.4a and 4.4b:

TABLE 4.4a - Chemical analyses of seepage water from test pit C/2 depicting macro-chemistry and other main parameters.

C/2	**pH**	**EC**	**TDS**	**Ca**	**Mg**	**Na**	**K**	**Cl**	**NO_3**	**HCO_3**	**SO_4**
		mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
	6.3	309	2214	219	147	262	5.37	336	<0.1	348	1006

Note: Accuracy of the analysis, which is outlined in paragraph 3.3: Δ -2.22 per cent (high accuracy).

The pH conditions in the seepage sample correspond to the soil pH. Total dissolved solids (TDS) are relatively high and mainly caused by high salt concentrations such as SO_4^{2-} and Cl^-. Table 4.4b represents metal and cyanide concentrations determined in the seepage sample.

TABLE 4.4b - Chemical analyses of seepage water from test pit C/2 showing metal and CN concentrations.

C/2	**As**	**Cu**	**CN**	**Fe**	**Mn**	**Ni**	**Pb**	**Zn**
	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
	<0.1	<0.01	<0.2	3.2	0.76	0.13	<0.01	0.05

Arsenic, CN, Cu and Pb concentrations are below the detection limit, therefore, there is tendency that these elements are retained from the soil. Additionally, the cyanide ion (CN^-) decomposes in aqueous solutions to cyanate (OCN^-), which is not stable and further disintegrates to CO_2 and NH_3 (Mortimer, 1987). The extraction test has shown that As, Cu and Pb have very low mobilities even under acidic conditions. The concentrations of Fe, Mn, Ni and Zn are low compared to seepage samples of the sites G and F. High alkalinity values in seepage reflect the acid neutralisation capacity of the subsoil by carbonate containing minerals. A list of geochemical soil data is provided in Table B.1.

4.3.5.2 Short-term impact

Extrapolated extractable Ni concentrations result in threshold excess, varying from 19 to 40-fold. In addition Co and Cr reach a threshold excess of up to 44 and 18-fold, respectively, indicating limited soil functioning. Zn, in contrast does not significantly exceed the threshold concentration.

4.3.5.3 Long-term impact

Site C is moderately contaminated (class II) by Cu, Fe, Ni, Mn, V and Zn while the site is highly (class IV) contaminated by Co using the geochemical load index.

4.3.6 Discussions and conclusions

The relatively high clay content may support contaminant attenuation mechanisms within the soil matrix. However, kaolinite is most likely the dominating clay type, having a low cation exchange capacity. The mobility of the phytotoxic elements Co, Cr and Ni is considerable and could endanger the present vegetation cover in the long-term.

Soil management measures like the addition of lime and fertilisers are currently not required since a vegetation cover has developed on site, but might be necessary in the future. Vadose zone monitoring (pH, redox conditions and trace element concentrations in seepage by using lysimeters) is advised, as the future contamination potential (worst-case scenario) of the soil is high.

4.4 CASE STUDY D

4.4.1 Site location and drainage

Site D is situated adjacent to a highway and in close proximity to a large township (Figure 1.1). It covers an area of approximately 71 ha and is located at an altitude of ± 1610 m above sea level. Surface run-off may occur towards a canal in northern direction. The location of the test pits is shown in Figure D.4.

4.4.2 Reclamation and rehabilitation status

Site D has been completely reclaimed and poor grass vegetation covers the entire area (Figure E.4). No rehabilitation measures were undertaken.

4.4.3 Geological conditions

Site D is mostly underlain by sedimentary rocks of the Dwyka Formation (diamictite and shale).

4.4.4 Soils

The soils of the site D are represented by the following soil types: Colluvium of the Willowbrook (test pits D/2 and D/3) and Shortlands (test pit D/1) types. Table 4.5 presents a summary of soil parameters of site D:

TABLE 4.5 - Summary of soil parameters for study site D.

	Field test results							
	Geotechnical Parameters				Derived from geotechnical parameters			Soil pH
Soil units	Clay content %	Plasticity index PI	Dry density kg/m^3	Specific gravity	Void ratio	Saturated permeability m/s	U.S.C.S. soil group	
Colluvium	14.37-42.23	6.09-14.55	1566.60-1684.51	2.61-2.64	0.55-0.69	9×10^{-9}	CL, SC	3.5-6.8
Ferruginous colluvium	20.93-51.44	9.36-30.32	1553.81-1644.29	2.68-2.69	0.64-0.72	7.5×10^{-10}	CL, SC	6.3-6..8

Note: U.S.C.S. symbols are explained in Table 3.11.

The soils exhibit an open soil structure to a maximum depth of 1.40 m. This structure might facilitate preferential vertical infiltration, although the abundance of gypsum crystallisation could close up the pores and, thus reducing vertical infiltration rates. Lateral preferential flow may occur at 1.10 m on the boundary between the colluvium and nodular ferricrete units, as ferricrete formation entails precipitation of colloidal Fe-oxides that may close pores to reduce vertical permeability.

Lateral preferential flow may also occur at 1.40 m at test pit D/3 on the boundary between the ferruginous colluvium and the residual shale. Cracking occurs from 0.80-2.30 m and might allow vertical preferential flow. A hard pan ferricrete unit occurs at 1.60 m in test pit D/1 and lateral preferential flow may be induced on this layer.

4.4.5 Assessment of contamination

4.4.5.1 Trace element concentration in soil

Iron shows low concentrations in the topsoils (3.4-8.9 per cent), but an accumulation in the subsoil (6.8-12.0 per cent). Test pit C/3 shows generally much lower Fe and As concentrations than in the two other test pits. The pH value in all topsoil samples varies from 3.5-3.8, thus indicating favourable leaching conditions for metals. A list of geochemical soil data is provided in Table B.1.

4.4.5.2 Short-term impact

Extrapolated extractable concentrations for Co would result in threshold excess of 28 to 53-fold and 24 to 72-fold for Ni, respectively. In contrast, Cr and Cu exceed the threshold concentration only by a factor of 6 to 10 and 2 to 4, respectively. Extractable concentrations of Zn and Pb are negligible.

4.4.5.3 Long-term impact

Site D is moderately contaminated (class II) by As, Cr, Fe while the site is moderately to highly contaminated (class III) by Ni and Pb. The site is highly contaminated (class IV) by Co, U and V using the geochemical load index.

4.4.6 Discussions and conclusions

The relatively high clay content may support contaminant attenuation mechanisms within the soil matrix. However, kaolinite is most likely the dominating clay type, having low expanding capabilities and thus, a low cation exchange capacity. A high mobility of the phytotoxic elements Co and Ni could endanger the present vegetation cover in the long-term.

Liming is not recommended as the primary pollution source was removed and a grass cover has developed on site, but might be necessary in the future. However, vadose zone monitoring is advised as the future contamination potential (worst-case scenario) of the soil is high.

4.5 CASE STUDY E

4.5.1 Site location and drainage

Site E is situated approximately 1 km to the north of the outskirts of Springs (Figure 1.1) and is bordered by a dam on its western side. An industrial area is located on the eastern border of the reclaimed site. The reclaimed site E covers an area of approximately 70 ha and occurs at an altitude of ±1585 m above sea level. Surface drainage occurs towards a drainage canal in a southerly direction. This canal feeds a dam further downstream. The location of the test pits is shown in Figure D.5.

4.5.2 Reclamation and rehabilitation status

The site has been almost completely reclaimed, with about 90 per cent of the tailings being removed. The site shows a poor grass cover and paddocks systems were established to prevent storm water surface run-off from the site (Figure E.5).

4.5.3 Geological conditions

Site E is mostly underlain by alluvial sediments deposited by a tributary of a perennial stream. The alluvium is underlain in the northern section of the site by sedimentary rocks of the Dwyka Formation (diamictite and shale) and by dolomitic rock (low chert content) of the Oaktree Formation in the southern portion of the site. A dolerite sill occurs in the central portion of the site.

4.5.4 Soils

The soils of site E are represented by the following soil types: Alluvium of the Rensburg (test pit E/2 and E/3) type and colluvium of the Katspruit (test pit E/1) type. A summary of the soil parameters is presented in Table 4.6.

TABLE 4.6 - Summary of soil parameters for study site E.

	Field test results							
	Geotechnical Parameters				**Derived from geotechnical Parameters**			**Soil pH**
Soil units	Clay content %	Plasticity index PI	Dry density kg/m^3	Specific gravity	Void ratio	Saturated permeability m/s	U.S.C.S. soil group	
Alluvium	35.02-46.55	22.27-38.38	1484.87-1535.06	2.68	0.75-0.80	9×10^{-9}- 8×10^{-10}	CH, CL	5.1-8.3
Colluvium	33.29-40.06	21.28-23.05	1535.78-1775.29	2.55-2.70	0.44-0.76	1×10^{-10}- 9.5×10^{-9}	CL	6.7–7.0

Note: U.S.C.S. symbols are explained in Table 3.11.

In the clayey alluvial soils of test pits E/2 and E/3, cracking occurs between 0.60 m and a maximum of 1.50 m and these features may be preferential vertical flow paths. Both test pits refused on an alluvial boulder layer (at 1.50 m in test pit E/2 and 1.30 m in E/3). A perched water table occurs at 2.00 m in the colluvial soils of test pit E/1 that implies preferential lateral flow. Vertical preferential flow may occur between 0.50 m and 2.00 m in test pit E/1 as these soils are slickensided.

4.5.5 Assessment of contamination

4.5.5.1 Trace element concentration in soil

Iron, Cr and V show increasing concentrations with depth, thus indicating the downward migration through the subsoil, whereas Pb and Zn tend to accumulate in the topsoil. A list of geochemical soil data is provided in Table B.1.

4.5.5.2 Short-term impact

No information on extractable metal concentrations of the soil is available. Leaching tests, conducted by the mining company, indicated high SO_4^{2-} concentrations even after the fourth extraction, exceeding the recommended maximum concentration of 600 mg/l (South African Bureau of Standards, 1984). The high SO_4^{2-} concentrations in the topsoil are a result of pyrite oxidation and the subsequent leaching from the reclaimed tailings. However, pH conditions in the top and subsoil are fairly neutral and would not allow significant contaminant mobilisation.

4.5.5.3 Long-term impact

Site E is moderately contaminated (class II) by Fe, Co, Pb and V, while the site is moderately to highly contaminated (class III) by Pb and V using the geochemical load index. Seepage is characterised by high SO_4^{2-} concentrations as a result of pyrite oxidation. A dolomitic aquifer system, which is in particular vulnerable to pollution (high flow velocities along fissures, fractures and cracks), underlies the southern portion of this site.

4.5.6 Discussions and conclusions

The relatively high clay content may support contaminant attenuation mechanisms within the soil matrix. However, kaolinite is most likely the dominating clay type, having low expanding capabilities and thus, a low cation exchange capacity. In addition, the site is partially situated on dolomite.

However, the remaining tailings should be removed and vadose zone monitoring is advised as the future contamination potential (worst-case scenario) of the soil is moderately to high. In addition, extraction tests for metals should be conducted. Liming is not required since the soil pH conditions are in a normal range.

4.6 CASE STUDY F

4.6.1 Site location and drainage

Site F is situated approximately 1 km south of the outskirts of Springs (Figure 1.1) adjacent to a highway and bordered to the east by a small township. Site F consists of two reclaimed tailings dams, which were located next to each other. The reclaimed sites cover a total area of approximately 120 ha. The site is located at an altitude of ±1585 m above sea level. Surface drainage is towards a perennial stream in the east. The location of the test pits is shown in Figure D.6.

4.6.2 Reclamation and rehabilitation status

Both sites have been reclaimed, but small volumes of tailings material still remain on site and indicate the footprint of the former deposit. Some poor vegetation has been developed on site. The mining company is currently in the process of removing these residual tailings (Figure E.6).

4.6.3 Geological conditions

The larger portion of the site is underlain by rocks of the Vryheid Formation (sandstone and shale); whereas the remainder in the south-eastern section is underlain by sedimentary rocks of the Dwyka Formation (diamictite and shale).

4.6.4 Soils and groundwater

The soils of site F are represented by the Shortlands soil type. Table 4.7 presents a summary of soil parameters:

TABLE 4.7 - Summary of soil parameters for study site F.

	Field test results							
	Geotechnical Parameters				**Derived from geotechnical Parameters**			**Soil pH**
Soil units	Clay content %	Plasticity index PI	Dry density kg/m^3	Specific gravity	Void ratio	Saturated permeability m/s	U.S.C.S. soil group	
Colluvium	13.70-44.22	6.64-14.33	1661.44-1711.02	2.51-2.78	0.47-0.67	6 x 10^{-9} to 1 x 10^{-10}	CL	3.7-6.7
Nodular ferricrete	19.72-39.56	6.21-15.89	1739.57	2.72	0.56	6 x 10^{-10}	CL	4.7

Note: U.S.C.S. symbols are explained in Table 3.11.

This study site is characterised by an average clay content of about 32 per cent, thus providing potential contaminant attenuation within the soil matrix. However, Joubert (1998) conducted mineralogical analyses using X-ray diffraction on soil samples (n=28) and found that kaolinite is the dominating clay type, which has a low cation exchange capacity. Furthermore, Fe oxide was detected in all samples, mostly as ferricrete. Such ferricrete layers can adsorb significant amounts of metals.

The soils are open structured between a minimum depth of 0.05 m (test pit F/3) and a maximum of 2.40 m (test pit F/4). The open structured nature of this soil unit should facilitate preferential vertical infiltration, although the presence of gypsum (observed at various depths) could close up the pores to reduce vertical infiltration rates. Lateral preferential flow may occur at the contact of the nodular ferricrete unit with the overlying soil (at a minimum of 1.00 m in test pit F/3 and a maximum of 2.20 m in test pit F/2) as ferricrete formation entails precipitation of colloidal Fe oxides that may close pores to reduce vertical permeability. No perched water tables were encountered but the basal nodular ferricrete unit present in most of the test pits is indicative of seasonal high moisture contents or a perched aquifer at the base of the profiles.

The site is underlain by a dolomitic aquifer. Repeated collapse during drilling, the recirculation of air during borehole development and the high transmissivity calculated from pumping tests indicate the presence of karstified features at shallow depth in this area. In addition, the high transmissivity of the dolomitic aquifer results in the immediate down gradient migration of contaminants away from the site, towards a perennial stream in the east. A hydrocensus revealed the presence of 18 boreholes in close proximity to the site, which are used for irrigation of gardens and swimming pools. One monitoring borehole has been drilled on site and shows a groundwater yield of approximately 5 l/s. The water table was determined at approximately 11 m below surface. No groundwater quality data were available.

4.6.5 Assessment of contamination

4.6.5.1 Trace element concentration in soil and groundwater

The site shows relatively low concentrations of Mn, Co, Pb, Zn compared to the average background value for the Vryheid Formation. However Fe shows higher concentrations than the averaged background, indicating the potential release of Fe during the pyrite oxidation process. Test pit F/1 shows the highest concentrations of As, Cu, Ni, U and Zn at a depth of about 70 cm, which is characterised as a sandy clay with abundant gypsum crystals.

Arsenic and Ni show 8-fold higher concentrations than the averaged background, however their mobility is very low. Very high concentrations of U were found in six of sixteen samples. Of those, three samples showed concentrations greater than 700 mg/kg, two of them collected from the topsoil. It is likely that the high U concentrations (which are in the range of gold/uranium ore, Table 2.2) in the soil emanate from the deposition of radioactive material generated prior to the tailings disposal by a uranium processing plant (according to staff information of the former site operator).

Measured groundwater quality on site is presented in Table 4.8:

TABLE 4.8 - Groundwater quality at site F, measured in January, April and August 1996. Data obtained from mining company. Recommended maximum limit (RML) according to Aucamp & Vivier (1987).

Sampling date (mg/l)	**pH**	**TDS**	**Alk**	**Ca**	**Mg**	**Na**	**K**	**Cl**	**SO_4**	**NO_3**	**CN**
Jan. 1996	6.7	2274	158	314	123	132	15	165	869	64	<1
Apr. 1996	7.0	1328	162	184	69	102	0.1	216	729	3.5	<0.5
Aug. 1996	7.3	1502	n. a.	112	11	100	7.7	176	712	n. a.	n. a.
RML	6-9	-	300	150	70	100	200	250	200	6	0.2

Note: n. a. means information not available. Accuracy of the analysis, which is outlined in paragraph 3.3: January 1996: Δ = 14.6 per cent; April 1996: Δ = -5.0 per cent and August: Δ = -28.4 per cent.

These results indicate that groundwater underneath the reclaimed site shows a poor quality and does not conform with specified drinking water limits of South African Bureau of Standards (1984) with regard to Ca^{2+}, Mg^{2+}, SO_4^{2-} and NO_3^-. The groundwater shows a predominant Mg-Ca-SO_4 character, which becomes more pronounced with increasing values for total dissolved solids (TDS). The pH is fairly neutral, although high concentrations of total dissolved solids occurred in January 1996 indicating the acid neutralisation capacity of the groundwater (most likely of dolomitic nature) in this particular area. Lower concentrations of total dissolved solids and earth alkali metals in April 1996, in contrast, can be explained with dilution effects as a result of rainfall recharge to the aquifer. No information regarding heavy metals in the groundwater was available. A list of geochemical soil data is provided in Table B.1.

4.6.5.2 Short-term impact

The extractable concentrations of trace elements of all soil samples were determined and extrapolated to the other sites A-G. The calculation of threshold excess revealed that Co, Ni and U exceed significantly the threshold concentration. Co reaches an excess of up to 40, Ni of 72.5 and U of 118.75. The latter due to a very high mobility of 6.4 per cent at pH values between 3-4. Low threshold excess was found for Cu, Pb and Zn due to relatively low total concentrations. However, the mobility as shown in the extraction tests is fairly high for these elements.

The soil pH indicates an pH increase with depths from 4.4 to 6.3 and 4.5 to 5.2, respectively in two test pits, whereas the two other test pits showed lower pH values even at greater depths (maximum depth 2.4 m). The fluctuating soil pH conditions might be a result of the spatial variation of buffer minerals within the soil or a localised perched water table.

4.6.5.3 Long-term impact

Site F is moderately contaminated (class II) by Mn, Co and Th while the site is moderately to highly contaminated (class III) by As and Ni, although As seems to have a very low mobility. In addition, U excessively contaminates (class VI) the site. Important to note is that U showed a high mobility, thus becoming easily bio-available to organisms and plants. Groundwater in general is of poor quality, mainly reflected by high total dissolved solid values.

4.6.6 Discussions and conclusions

The mining company is currently in the process of removing residual tailings. However, high mobility of Co and Ni could complicate efforts to establish a self-sustaining vegetation cover, as these metals are known to be phytotoxic. The topsoil shows favourable leaching conditions and soil management measures such as liming would be required to neutralise acids and to improve growth conditions. Lime would also enhance the attenuation capacity of the soil.

Uranium concentrations are unacceptably high and a detailed risk-based site investigation would be necessary to assess the radiological impact on groundwater and plants, and subsequently, the degree of site rehabilitation required. In addition, vadose zone and groundwater monitoring is advised as the future contamination potential (worst-case scenario) of the soil is very high and groundwater underneath the reclaimed site is of poor quality.

4.7 CASE STUDY G

4.7.1 Site location and drainage

The site situated approximately 4 km north-east of the outskirts of Nigel (Figure 1.1). The site covers an area of approximately 13 ha and is located at an altitude of ±1610 m above sea level. Surface drainage direction is towards a canal in western direction. Agricultural activities take place in immediate vicinity of the site. The location of the test pits is shown in Figure D.7.

4.7.2 Reclamation and rehabilitation status

The reclamation of tailings dam site G has been completed, except some waste rock material at the south-eastern border. However small volumes of residual tailings material indicate the presence of the former deposit. Vegetation is poorly developed and consists of isolated trees and grass.

4.7.3 Geology

The reclaimed site G is underlain by sedimentary rocks of the Vryheid Formation (sandstone and shale).

4.7.4 Soils

The soils of the site G are represented by the Avalon soil type. Table 4.9 summarises the soil parameters for site G:

TABLE 4.9 - Summary of soil parameters for study site G.

	Field test results							
	Geotechnical parameters				**Derived from geotechnical Parameters**			**Soil pH**
Soil units	Clay content %	Plasticity index PI	Dry density kg/m^3	Specific gravity	Void ratio	Saturated permeability m/s	U.S.C.S. soil group	
Colluvium	14.10-24.69	5.78-8.27	1786.21	2.64	0.48	1 x 10^{-9}	SC	4.0-4.8

Table 4.9 continued.

Nodular ferricrete	22.63-31.45	7.60-9.48	1782.53	2.68	0.50	1 x 10^{-9}	SC, CL	6.3-6.9

Note: U.S.C.S. symbols are explained in Table 3.11.

Lateral preferential flow may occur on the hardpan ferricrete unit that caused refusal in all the test pits between 1.10 m and 1.50 m. A perched water table occurs between 0.95-1.30 m in all test pits, which could indicate the presence of preferential flow paths.

4.7.5 Assessment of contamination

4.7.5.1 Trace element concentration in soil

Most of the elements show no geochemical pattern except As, having the highest concentrations in the topsoil. Cobalt, Cu and Th exceed the average background value. Chromium, Pb, V and Zn, in contrast show lower concentrations than the average background. A list of geochemical soil data is provided in Table B.1.

4.7.5.2 Short-term impact

Low soil pH values (4.0-4.8) indicate favourable leaching conditions for metals, which is also reflected in one chemical analysis on seepage water from test pit G/2. The results of the chemical analysis are shown in Tables 4.10a and 4.10b.

TABLE 4.10a - Chemical analyses showing macro-chemistry and other parameters of seepage water in test pit G/2.

G/2	**pH**	**EC** mS/m	**TDS** mg/l	**Ca** mg/l	**Mg** mg/l	**Na** mg/l	**K** mg/l	**Cl** mg/l	**NO_3** mg/l	**HCO_3** mg/l	**SO_4** mg/l
	4.9	670	6802	525	257	227	154	207	<0.1	8	4760

Note: Accuracy of the analysis, which is outlined in paragraph 3.3: Δ = -26.4 per cent (high inaccuracy).

TABLE 4.10b - Chemical analyses showing various metal and CN concentrations of seepage water from test pit G/2.

G/2	**As** mg/l	**Cu** mg/l	**CN** mg/l	**Fe** mg/l	**Mn** mg/l	**Ni** mg/l	**Pb** mg/l	**Zn** mg/l
	0.12	0.1	<0.2	431	359	4.4	0.03	0.3

It is apparent that high total dissolved solid values and elevated metal concentrations are caused by leaching from tailings.

4.7.5.3 Long-term impact

Site G is moderately contaminated (class II) by Co using the geochemical load index.

4.7.6 Discussions and conclusions

Soil management measures are not required as the tailings were removed and a grass cover has developed.

The following site information (study sites H-K) was obtained from a literature survey and supplemented with data from mining companies. Only limited extraction test or geochemical soil data were available, thus final conclusions are premature. However, for some of these sites radiological and groundwater data became available and supplemented the following site assessments. The structure of the following paragraphs varies depending on the availability of site-specific data. Those data, which are not referenced, were provided by mining companies and are confidential.

4.8 CASE STUDY H

4.8.1 Site location and drainage

The site is located in the North-West Province, west of Potchefstroom and is isolated from industrial or residential development. The reclaimed investigated site covers an area of approximately 4 ha. The altitude varies between 1373-1560 m above sea level. Predominant drainage mechanism of the site is sheet-wash in southerly direction. The drainage has resulted in portions of a floodplain being covered by fine slimes material originating from the tailings dams. In addition, the residual tailings are heavily eroded and fine material has spread onto farmland in the direction of prevailing wind. The quantity of residual tailings material is estimated at 2 million tons for the entire site (Aucamp, 1997). The position of the auger holes is shown in Figure D.8.

4.8.2 Reclamation and rehabilitation status

The site is not completely reclaimed and significant quantities of residual tailings material remain on site. Three other tailings dams and a waste rock dump are situated within a radius of one km of the investigated site. Vegetation on site comprises poorly developed grassveld and small shrubs. Exotic trees (including *Eucalyptus*) occur around the old mine operations and on top of the disposed material (Aucamp, 1997). The Department of Minerals and Energy currently is developing a rehabilitation plan for the site.

4.8.3 Geology

The entire area is underlain by dolomite (low chert content) of the Oaktree Formation (Aucamp, 1997).

4.8.4 Soil

Three different dolomite residuum soil and rock units were identified. A chert-rich residual dolomite occurring in auger holes H/1 and H/2, a shale horizon occurring in auger hole H/1 and a ferricrete-rich and chert-poor soil horizon in auger holes H/2 and H/3. A geotechnical description of the soil is not possible since these data are not available. However, soil profiling in three auger holes revealed three general soil and rock units, consisting of a chert-rich topsoil, underlain by a sandy clay with a shale horizon and a ferricrete and chert-poor horizon (Aucamp, 1997).

The sandy topsoil, which is mixed with varying amounts of tailings would allow a rapid recharge. This would be enhanced by chert gravel, found in auger hole H/1 up to a depth of approximately 1.50 m. The clayey sand (in auger holes H/2 and H/3) at the interface to the overlying soil is, in contrast, relatively impermeable, enhancing lateral preferential flow. Additionally, lateral preferential flow may occur at the boundary between the sandy clay and nodular ferricrete units (Aucamp, 1997).

4.8.5 Assessment of contamination

4.8.5.1 Radioactive contamination in tailings, soils and sediments

A survey was conducted in the mid 1990s by the Council for Nuclear Safety on the entire site, which covers approximately 43 ha, to assess risks associated with dispersing radioactive material contained in the fine tailings. The results clearly indicate elevated levels of contamination on the mine site, adjoining farmland and in stream sediments significantly above background values. Background values for pristine or uncontaminated soils in the Highveld region vary from 15-75 Bq/kg (or 1.2-5.9 mg/kg) of ^{238}U. Each of its radioactive decay products including ^{226}Ra and

^{232}Th, show a similar range. ^{40}K is a widely distributed naturally occurring radionuclide usually found at levels of a few hundred Bg/kg in soils (Council for Nuclear Safety, 1996). Table 4.11 lists the results for different radionuclides:

Table 4.11 - Solid samples collected in November 1995 around study site H. Samples 1, 3, 5 and 6 were taken from material that has been eroded from the tailings dams.

Sample No.	**^{40}K** Bq/kg	**^{226}Ra** Bq/kg	**^{232}Th** Bq/kg	**^{238}U** Bq/kg	**Sampling medium**
1	770	285	60	200	Fines accumulated in erosion channels
3	590	740	45	1300	Fines in water collected in sump
5	935	1360	100	<400	Sediment on nearby farm land
6	830	200	45	<400	Fines accumulated in erosion channels
7	200	220	100	7100	Sample taken within the vicinity of the site
10	440	145	30	<400	Average tailings sample
11	260	20	20	<100	Background sample taken some distance away on a ploughed field

The contamination depth in soils varies within the topsoil and the deposits (eroded tailings dam material) were particularly thick in the streambed. The Council for Nuclear Safety (1996) concluded that although soils on the nearby farmland and stream sediments are contaminated with radionuclides, the level of activity is relatively low and does not cause an immediate radiation hazard. However, in the long-term the material may disperse and accumulate in certain areas (such as areas situated in the prevailing wind direction) and could cause an unacceptable long-term risk. Further investigations including water analyses are ongoing until a final rehabilitation plan is in place.

4.8.5.2 Groundwater

Two sets of groundwater measurements from one borehole on site are available and are presented in Table 4.12.

Table 4.12 - Groundwater chemistry of the site H (Aucamp, 1997).

Sampling date	**pH**	**EC** mS/m	**NO_3** mg/l	**SO_4** mg/l	**HCO_3** mg/l	**Na** Mg/l	**K** mg/l	**Ca** mg/l	**Mg** mg/l
June 1998	7.7	338	145.5	2247	329.4	63.2	26.2	521	307
July 1998	7.7	328	-	2552	62.3	66.8	30.3	527	310

Note: Accuracy of the analysis, which is outlined in paragraph 3.3: June 1998: $\Delta = 0.1$ per cent (very high accuracy) and July 1998: $\Delta = -4.1$ per cent (high accuracy).

High concentrations of SO_4^{2-} indicate the impact of acid mine drainage on groundwater quality. High concentrations of Ca^{2+} and Mg^{+} are a result of the dissolution of dolomitic rock causing neutral pH conditions.

4.8.5.3 Short-term impact

Extraction tests, using the 2 mm particle size fraction, were conducted to assess the current contamination impact. Extractable concentrations of Ni, Zn and Cd in the soil increase where nodular ferricrete is more distinctly developed in the soil. The extractable concentrations of Cr and Cu do not reflect a clear geochemical pattern. It can be concluded that Cd poses a hazard in the ferricrete, reflected by a threshold excess of almost 10. Copper, Ni and Zn pose a hazard in both the ferricrete-poor soil and the ferricrete. Mercury does not pose a hazard, because the mobile portion in soils is very low (Aucamp, 1997).

No hazard rating could be established for this site. However, high concentrations of SO_4^{2-} in groundwater indicate the migration of acid mine drainage into the aquifer. Residual tailings material on the surface would provide a long-term source for further pyrite oxidation and contaminant remobilization.

4.8.6 Discussions and conclusions

The dispersion of tailings material and subsequent accumulation on farmland and in stream sediments as a result of wind erosion, is a concern in terms of agricultural land use. In addition, a variety of trace elements such as Co, Cu, Ni and Zn are extractable and could complicate efforts to establish a self-sustaining vegetation cover on site.

The high levels of trace elements and radionuclides would require a detailed risk-based site investigation (site is here considered as the entire area of approximately 43 ha) as the long-term risks for the groundwater and surface exposure routes (plants, animals, humans and environment) are currently unknown. In addition, vadose zone (pH, redox conditions and trace element concentrations in seepage by using lysimeters) and groundwater monitoring is advised as the future contamination potential (worst-case scenario) of the site seems to be very high and groundwater underneath the reclaimed site shows a poor quality.

The residual tailings material should be covered using an impermeable cover to prevent further wind dispersion of fine tailings and to reduce seepage and, thus minimising acid mine drainage. Soil management measures would be required to neutralise acids and improve growth conditions.

4.9 CASE STUDY I

4.9.1 Site location and drainage

The site is located adjacent to the R23 (Old Heidelberg Road) between Brakpan and Heidelberg (Figure 1.1). A township is situated less than 2 km east of the tailings dam and farming is taking place in the immediate surroundings of the deposit. A perennial stream flows through a wetland system in a north-westerly direction at a distance of less than one km from the western boundary of the site. The area slopes gently in a westerly direction towards the wetland system. Surface run-off is controlled and limited by a drainage collection system surrounding the tailings dam.

4.9.2 Reclamation and rehabilitation status

The tailings dam comprises a southern compartment, which is currently reclaimed and retreated, and a northern compartment (active dam) where gold mine tailings are currently disposed at a rate of approximately 950000 tons/day, using the cycloned deposition approach. Disposal activities commenced in 1985 and will be completed in 2005. The maximum dam wall height of the current active dam is over 60 m and the target height is anticipated to be approximately 85 m above lowest ground level. The current active dam covers an area of 870 ha, whereas the entire affected area, which includes the reclaimed portion, is approximately 1400 ha. No vegetation occurs on the reclaimed site (southern portion) due to the ongoing reclamation operation. A portion of the active slope wall was grassed in order to prevent wind erosion (Figure E.7).

4.9.3 Geology

The tailings dam is mainly underlain by andesitic lava of the Ventersdorp Supergroup, quartzite of the Black Reef Formation, dolomitic rocks of the Oaktree and Monte Christo Formations, sandstone and mudstone of the Dwyka and Vryheid Formations and post-Karoo dolerite intrusions. However, doleritic and dolomitic rocks cover the largest portion of the area.

4.9.4 Field work

The tailings dam is surrounded by monitoring boreholes, which are sampled on a quarterly basis in order to determine the groundwater quality (total dissolved solids, pH, EC, alkalinity, total hardness, major cations and anions, CN, As, Fe and Mn) at various depths and distances away from the tailings dam. An extensive geotechnical study was launched as part of the feasibility study for the northern tailings dam in the mid 1980s. This comprised core drilling, the excavation of a large number of test pits and a soil survey in the area now covered by the tailings dam. Pump tests have been conducted to assess the hydrogeological properties of the aquifer underneath the site. As a result, detailed geological and hydrogeological information was available and incorporated into a numerical groundwater model.

4.9.5 Soils

The following soil types occurred on the active dam site and were identified during the geotechnical feasibility study.

- Red, apedal, medium textured soils associated with chert and mostly represented by Msinga soil type.
- Small areas are covered by yellow, brown, apedal, medium textured soils associated with chert and Karoo sediments.
- Black and dark-colored, structured, medium to heavy textured soils associated with dolerite and mostly represented by Rydalvale and Rosehill soil types.

Table 4.13 summarises geotechnical soil parameters for study site I, obtained from geotechnical reports of the mining company:

TABLE 4.13 - Summary of soil parameters for study site I.

	Field test results							
	Geotechnical Parameters				**Derived from geotechnical Parameters**			**Soil pH**
Soil units	Clay content %	Plasticity index PI	Dry density kg/m³	Specific gravity	Void ratio	Saturated permeability m/s	U.S.C.S. soil group	
Alluvium	15-69 (45.2)	12-44 (31)	-	-	-	0.2-3.1 x 10^{-5}	CH, CL, SC, GC	3.7-5.7
Colluvium	19-68 (44.5)	7-50 (21)	-	-	-	0.2-3.1 x 10^{-5}	CL, CH	-
Residual dolerite	5-64 (33.3)	2-35 (20)	-	-	-	10^{-5}-10^{-7}	GC, CH, CL	-
Residual dolomite	9-62 (44.6)	4-52 (27)	-	-	-	10^{-5}-10^{-7}	CH, CL	-

Note: U.S.C.S. symbols are explained in Table 3.11. Values in brackets are averaged values.

The surficial colluvial, alluvial and residual soils have hydraulic conductivities in the range of between 0.2 and 3.1 x 10^{-5} m/s. The deeper residual soils and weathered bedrock showed varying hydraulic conductivities of between 10^{-5}-10^{-7} m/s. Unweathered to slightly weathered bedrock indicate a permeability in the order of 10^{-8}

m/s. A soil survey conducted at the reclaimed dam indicates the presence of soils of the Arcadia soil type.

4.9.6 Aquifer properties

Groundwater flow occurs under unconfined to semi-confined conditions. Groundwater levels are shallow (mean between 1-2 m) and a significant groundwater mound has developed underneath and in close proximity to the tailings dam. The groundwater mound seems to be better developed where dolerite rocks, showing a lower permeability than dolomite, and clayey and silty weathered formations, is present. These areas are generally wet due to seepage.

Farther away from the tailings dam, groundwater levels seem to reflect the topographical gradient towards the west. However, groundwater drainage takes place radially, in a westerly, north-westerly and northerly direction towards two rivers and with an average hydraulic gradient < 2 per cent.

Many boreholes close to the tailings dam are artesian, indicating that the tailings dam is hydraulically connected with deeper rock fracture systems underlying superficial soils and highly weathered bedrock. Monitoring boreholes drilled into the shallow and deeper aquifer system revealed slightly higher groundwater levels in the shallow boreholes (compared to shallow boreholes farther way) and thus indicating that seepage originated from the tailings dam.

4.9.7 Assessment of contamination

4.9.7.1 Chemical properties in soils and groundwater

Geochemical analyses of soil samples collected at various depths on the reclaimed portion of the site yielded low pH values ranging between 3.7-5.7, indicating the effect of acid mine drainage, despite relatively low SO_4^{2-} concentrations varying between 370-760 mg/kg. Relatively low SO_4^{2-} concentrations suggest leaching of the subsoil. Trace element concentrations and pH values in soils are shown in Table 4.14:

TABLE 4.14 - Metal and pH ranges in soils of the reclaimed (southern) portion of site I.

Site I	**pH**	**Cu** mg/kg	**Fe** mg/kg	**Mn** mg/kg	**Zn** mg/kg
	3.8-5.7	0.6-1.7	22.5-44.9	5.4-23.3	0.8-5.4

The calculation of sodium adsorption ratios (SAR) showed low ratios and thus, there is no indication that the soil is becoming brackish. Groundwater quality data for the shallow and deeper boreholes are listed in Table 4.15a/b:

TABLE 4.15a - Range of groundwater quality in shallow boreholes of site I.

Shallow groundwater	**pH**	**EC** mS/m	**TDS** mg/l	**Ca** mg/l	**Mg** mg/l	**Cl** mg/l	**SO_4** mg/l	**CN** mg/l	**Fe** mg/l	**Mn** mg/l
Apr. 1995										
MIN	6.2	109	1098	45	6	30	356	< 0.5	n. a.	< 0.1
MAX	7.8	707	7560	539	780	958	3735	5.0	n. a.	4.3
AVG	7.1	301	3585	321	173	247	1557	1.6	n. a.	1.0
Aug. 1995										
MIN	6.1	83	540	74	58	40	264	n. a.	0.3	< 0.1
MAX	7.7	1220	8892	651	1025	2245	5057	n. a.	66.0	23
AVG	7.0	357	2350	389	268	309	1690	n. a.	14.8	5.0
Nov. 1995										
MIN	6.3	107	612	1	4	48	383	n. a.	< 0.1	< 0.1
MAX	8.1	427	4410	526	744	1783	4153	n. a.	3.9	21.0
AVG	6.6	266	2404	309	290	310	1928	n. a.	0.6	2.6

Table 4.15a continued.

Apr. 1996										
MIN	6.3	107	612	20	121	57	302	n. a.	< 0.1	< 0.1
MAX	8.1	1038	8932	436	776	1801	4295	n. a.	2.6	24.7
AVG	6.6	306	2822	222	283	288	1783	n. a.	0.3	4.0

Note: n. a. means not available. Cyanide (CN) concentrations refer to free CN.

TABLE 4.15b - Range of groundwater quality in deeper boreholes of site I.

Deeper groundwater	**pH**	**EC** mS/m	**TDS** mg/l	**Ca** mg/l	**Mg** mg/l	**Cl** mg/l	**SO_4** mg/l	**CN** mg/l	**Fe** mg/l	**Mn** mg/l
Apr. 1995										
MIN	6.8	31	270	22	4	8	31	< 0.5	n. a.	< 0.1
MAX	8.4	785	6510	601	863	1108	3751	4.2	n. a.	29.3
AVG	7.3	269	2402	291	119	217	1418	0.7	n. a.	4.4
Aug. 1995										
MIN	6.2	35	256	73	19	8	35	n. a.	n. d.	< 0.1
MAX	7.9	951	7008	757	796	1516	4015	n. a.	73.0	28.0
AVG	7.0	310	2278	381	216	244	1456	n. a.	14.0	5.0
Nov. 1995										
MIN	6.6	27	188	27	26	12	43	n. a.	< 0.1	< 0.1
MAX	8.5	749	7414	1448	750	1166	3254	n. a.	10.7	21.0
AVG	6.0	262	2555	329	219	180	1123	n. a.	1.1	2.6
Apr. 1996										
MIN	6.6	27	188	27	26	12	43	n. a.	< 0.1	< 0.1
MAX	7.9	622	4874	426	444	1026	2919	n. a.	65.0	4.5
AVG	6.8	236	2058	193	220	190	1230	n. a.	5.8	1.2

Note: n. a. means not available and n. d. means below detection limit. Cyanide (CN) refers to free CN.

The high concentrations of total dissolved solids (TDS) in shallow and deeper groundwater are a result of seepage from the pond, containing high loads of SO_4^{2-} and Cl^-. Considerable concentrations of Ca^{2+} and Mg^{2+} are caused by the dissolution of dolomitic aquifer material, leading to fairly neutral pH conditions. There is no distinctive trend indicating a polluted shallow aquifer and a less polluted groundwater deeper system.

4.9.7.2 Radioactive contamination in surface and groundwater

A limited number of groundwater and surface water samples were analysed for radionuclides, indicating that surface water systems show far higher radioactivity than groundwater samples. However, concentrations and activities are low and within recommended concentrations of DWAF for domestic use (1996a) and agricultural use (1996d).

4.9.7.3 Short-term impact

Generally, samples from the shallow boreholes show higher total dissolved solid, SO_4^{2-}, Na^+ and Cl^- concentrations than those obtained from the deeper monitoring boreholes. As a result, most of the groundwater samples obtained during the monitoring survey show a predominant Ca-Mg-SO_4 signature, which is typical of water affected by acid mine drainage.

However, deeper monitoring boreholes further away from the tailings dam show a better groundwater quality than samples from the shallow boreholes in the area between the tailings dam and the main drainage features, due to natural dilution effects and attenuation processes.

The shallow and deep boreholes in close proximity to the tailings dam exceed the crisis limits for SO_4^{2-} of 1200 mg/l, whilst those further away show concentrations which fall between maximum permissible of 600 mg/l and the crisis limit of 1200 mg/kg (South African Bureau of Standards, 1984). Heavy metal analyses were conducted on a random basis. Elevated concentrations of As, Cd, Co, Fe, Mn and Ni were found at almost neutral pH values (pH varies between 5.4-7.4) and indicate seepage draining from the tailings dam into the aquifer. It is important to note that similar contaminants were also found in elevated concentrations in soil samples at reclaimed sites (study sites A-G). Surface water samples taken along the adjacent river showed high concentrations of SO_4^{2-} at fairly neutral pH conditions (6.0-7.6), indicating acid mine drainage.

4.9.7.4 Long-term impact

A numerical groundwater model was applied to estimate the degree of future contamination. The model supported the assumption of groundwater drainage radially away from the tailings dam and towards the surface drainage features. The model was run for 50 years, which represents 40 years after final rehabilitation and closure (scheduled for the year 2005). A groundwater risk assessment using Monte-Carlo simulations indicates a low impact on surface water resources downstream of the tailings dam. Salts contained in seepage from the tailings dam represent less than 0.5 per cent of the total salt load of the nearby river. It is estimated that salts in seepage would contribute less than 2.5 per cent to the total salt load after 50 years.

In conclusion, groundwater in close proximity to the tailings dam has been polluted by seepage from the tailings dam, but groundwater quality further away suggests that drainage features such as the river have a much larger impact than the tailings dam.

4.9.8 Discussions and conclusions

The development of a groundwater mound underneath the active dam is of major concern as contaminants can migrate from the pond directly into the dolomitic aquifer without or insufficient attenuation. The Ca-Mg-SO_4 signature of groundwater underneath and in close proximity to the active tailings dam clearly indicates such migration. High hydraulic conductivities along rock fractures and fissures would allow rapid dispersing of contaminants, and thus causing risks in the long-term.

However, a detailed site investigation was conducted and a water quality monitoring program is ongoing. New monitoring data should be implemented into the present groundwater model (as part of the risk management), which allows verifying and refining predicted scenarios. It is anticipated that after decommissioning of the slurry disposal (in the year 2005) on the active dam site, the surface will be covered with an impermeable soil layer to prevent wind erosion (which is currently significant) and to minimise rainfall infiltration.

It must be noted that portions of the slope wall are already covered with soil (Fig. E.7, Appendix E) and the mining company is currently assessing various cover options. It is recommended to investigate the reclaimed portion of the site in more detail, as the presence of phytotoxic elements is most likely, complicating efforts to establish a self-sustaining vegetation cover on this portion of the site. Soil management measures are required to improve growth conditions and to enhance the attenuation capacity of the soil.

4.10 CASES STUDY J

4.10.1 Site location and drainage

Site J is located south of Brakpan (Figure 1.1), in the immediate vicinity of a wetland system on its western border and covers an area of approximately 117 ha. A large township is located approximately 2 km from the eastern border of the site. The wetland extends from the western border of site J along a non-perennial stream, and terminates at the confluence with another perennial stream, which eventually drains into the Vaal dam. Drainage direction is south-westerly towards the wetland system.

4.10.2 Reclamation and rehabilitation status

The site is currently in the process of reclamation and a small quantity of tailings remains on surface. Vegetation, comprising some trees is poorly developed.

4.10.3 Geology

The tailings dam area is underlain by sedimentary rocks of the Dwyka Formation (diamictite and shale).

4.10.4 Field work

A geochemical pollution study was conducted in the wetland system next to the tailings dam, indicating acid mine drainage escape from this study site. Soil samples were taken at five different sampling points down-gradient of the site. Twenty one vibracore holes were drilled up to a maximum depth of 2 m along a traverse approximately 300 m long and adjacent to site J. From these boreholes, sediment and water samples were obtained and analysed for their contamination levels. Furthermore, a surface water sampling point of Rand Water is located downstream of the tailings dam and monitored for its water quality.

In addition, the operating mining company of the tailings dam drilled one borehole on the north-eastern border of the site to monitor groundwater quality and to conduct aquifer testing. The borehole remained dry even at a depth of 40 m. As a result, no groundwater data are available for this site. However, a hydrocensus conducted by the operator resulted in groundwater quality data for one borehole up-gradient of the site.

4.10.5 Soils

Information of the soils underlying the tailings dam area were obtained from a borehole log of the proposed monitoring borehole, drilled on the north-eastern border of site J. The borehole log indicates a clayey, sandy material, which is considered to be the weathering product of the Karoo sediments underneath. The thickness of the clay layer in that particular borehole profile is approximately 7 m.

Soils down-gradient of the site consist of a mixture of soil and yellow oxidised tailings underlain by a red to dark brown clayey soil (due to ferric oxides) with abundant ferruginous concretions. The clay consists predominantly of kaolinite at shallow depths and montmorillonite at greater depths.

Owing to a lack of geotechnical data, no hydrogeological characterisation of the unsaturated zone was possible.

4.10.6 Aquifer conditions

No water table was encountered up to a maximum drilling depth of 40 m. In addition, no aquifer information was available for the borehole approximately 300 m up-gradient of site J. It is known that in close proximity to site J, the groundwater table has been lowered to allow for underground mining. The water level has been maintained at approximately 1600 below ground surface at a pumping rate of approximately 70 Ml/d. Pumping ceased in 1991 and since then the water level in the mine has been rising.

4.10.7 Assessment of contamination

4.10.7.1 Chemistry of surface water

The average concentrations of surface water samples from Rand Water, collected approximately 1.5 km downstream of site J are shown in Table 4.16.

Table 4.16 - Average values for selected water quality parameters measured by Rand Water approximately one km downstream site J. Measurements were taken in the period from October 1991 until September 1992.

Parameter	**pH**	**EC** mS/m	**Hardness** as $CaCO_3$	**Ca** mg/l	**Mg** mg/l	**Na** mg/l	**K** mg/l	**SO_4** mg/l
	6.4	220	1907	559	124	242	42	1797

It is evident that the high SO_4^{2-} content suggests the release of acid mine drainage and associated contaminants (e.g. Fe, Mn and Ni) from study site J. Table 4.17 presents a selection of average metal concentrations at the same sampling point. Cobalt, Mn and Ni exceed the crisis limit value significantly and indicate the high mobility of these elements under nearly neutral pH conditions. Soil and sediment samples downstream of the tailings dam contain significant concentrations of metals.

Table 4.17 - Average metal concentrations at a Rand Water sampling point approximately one km downstream of site J. Measurements were taken in the period of October 1991 until September 1992. Crisis limit according to Aucamp & Vivier (1987).

Parameter	**As** mg/l	**Co** mg/l	**Cr** mg/l	**Cu** mg/l	**Fe** mg/l	**Mn** mg/l	**Ni** mg/l
Sampling	0.0008	2.8	0.003	0.13	0.01	14	13.4
Crisis limit	0.6	1	0.4	2	2	2	1

Table 4.18 represents data from four different water quality sampling points, which are approximately 2, 2.5, 5 and 7 km downstream of tailings dam site J.

TABLE 4.18 - Surface water quality with increasing distances downstream of tailings dam site J. All concentrations in mg/l.

Distance Downstream	**pH**	**Eh** mV	**EC** mS/m	**SO_4**	**HCO_3**	**Ca**	**Co**	**Fe**	**Mn**	**Zn**
2 km (a)	5.8	94.4	320	1517	18.9	564	4	< 1	21	16
2.5 km (b)	5.4	118.7	310	1428	15.7	561	4	< 1	22	13
5.0 km (c) before confluence	6.6	48.7	210	759	91.1	289	< 1	< 1	6	< 5
7.0 km (d) after confluence	7.4	5.8	200	893	116.3	273	< 1	< 1	< 1	< 5

The table above indicates the improvement in water quality as the distance to the tailings dam from the study site increases. Although the pH becomes fairly neutral at sampling point (d), SO_4^{2-} concentrations still exceed the maximum allowable concentration of 600 mg/l (South African Bureau of Standards, 1984). Thus, water treatment needs to be considered before using the water for domestic or agricultural purposes. High Ca^{2+} concentrations might be caused by lime treatment of slime, which causes a rise of pH. Metal concentrations of Mn and Zn decrease significantly farther downstream as a result of dilution effects. Redox conditions (Eh values) indicate a slight oxidising milieu. Rivers and streams show generally Eh values in the order of 400 mV.

4.10.7.2 Radioactive contamination

A radiometric survey has shown that significant amounts of U and Th are leaving the tailings dam site J and entering the wetland system, where both seem to be partially adsorbed by peat. Total α-activity was about 2 Bq/l downstream of tailings dam site J, (corresponding to sample (a) in Table 4-18) and 0.4 Bq/l approximately 5 km downstream (similar to sampling point (c) in Table 4-18), thus indicating a significant decrease of concentration caused by dilution and adsorption on organic material such as peat.

4.10.7.3 Short-term impact

Peat samples downstream of the tailings dam at site J contain very high concentrations of trace elements as a result of adsorption. Results for some trace elements are shown in Table 4.19:

TABLE 4.19 - Trace element concentrations in a peat sample near site J.

Element	**Cd** mg/kg	**Co** mg/kg	**Cu** mg/kg	**Pb** mg/kg	**Zn** mg/kg	**Th** mg/kg	**U** mg/kg
	25	946	438	261	931	110	195

It is apparent from Table 4.20 that all considered trace elements occur in anomalous concentrations in soil samples affected by seepage from tailings dam site J.

TABLE 4.20 - Trace element concentrations of soil and sediment samples in close proximity to site J (n = 53).

Element	**As** mg/kg	**Co** mg/kg	**Cu** mg/kg	**Cr** mg/kg	**Ni** mg/kg	**Pb** mg/kg	**Zn** mg/kg
MIN	1	15	4	35	42	17	23
MAX	2040	6117	1071	713	17844	247	10516
AVG	455	582	274	340	1882	69	1095
Standard deviation	523	1096	290	194	2948	55	1744
Factor above average background	21	42	7.9	2.6	41.8	4.6	10.6

Affected soils, sediments, peat and the wetland systems act as a metal pool downstream of the tailings dam, accumulating considerable amounts of metals, because heavy metals seem to be immobilised under prevailing pH (fairly neutral) conditions and the presence of organic material. Sulphate-reducing bacteria in the peat may also lead to precipitation of chalcophile elements from solution as sulphides.

4.10.8 Discussions and conclusions

The extremely high metal concentrations in soils and sediments down-gradient of the study site J pose a long-term environmental risk. Phytotoxic elements such as Co and Ni in soils and sediments would complicate efforts to establish a self-sustaining vegetation cover, even if only a minor portion of the total element concentration is mobile.

A detailed risk-based site investigation is recommended, which should extend to the affected soils and sediments down-gradient of the site. In addition, vadose zone monitoring is advised, as the future contamination potential (worst-case scenario) of affected soils and sediments is considerably high.

4.11 CASE STUDY K

4.11.1 Site location and drainage

Site K is situated north of Springs in immediate vicinity of a large tailings dam (Figure 1.1). The tailings dam covers an area of approximately 111 ha and is situated at an altitude of ± 1600 m above sea level. Surface drainage follows the topographical gradient, which is reflected by a gentle slope towards the north.

4.11.2 Reclamation and rehabilitation status

The tailings dam was used for the disposal of slurry during the period 1969 to 1994. Since 1994 the tailings dam has been in the process of reclamation. The current reclamation status is estimated with 15 per cent of the total volume having been reclaimed. No vegetation occurs on site K.

4.11.3 Geology

The northern part of the tailings dam is underlain by dolomite, whilst the southern part is underlain by Karoo sedimentary rocks (the formation is not known). The thickness of Karoo rocks to the south of the tailings dam varies from 6-15 m. Two NW-SE trending dolerite dykes occur below the tailings dam. A third narrower dyke occurs towards the west of the site. A dolerite sill with a thickness between 10-20 m, occurs at depths of about 20-40 m below the site, and outcrops to the north of the dam.

4.11.4 Field work

A number of boreholes have been drilled in order to monitor the groundwater quality (on a quarterly basis) affected by acid mine drainage released from the tailings dam as well as to abstract contaminated groundwater down-gradient of the site.

4.11.5 Soils

The tailings dam is directly underlain by a zone of transported and residual clayey soils with a thickness of up to 5 m. In order to assess the role of the perched aquifer system, a soil field survey was launched and a number of auger holes were drilled to depths between 2-5 m. Clayey sands dominate the unsaturated zone to a depth of a few metres.

4.11.6 Aquifer conditions

A perched aquifer was encountered at a general depth of 3 m. A permeability test on a sample taken from a depth of 1.8 m above the perched aquifer indicated a very low hydraulic conductivity in the order of 10^{-10} m/s.

Furthermore, three different aquifers have been identified in the course of a detailed groundwater study:

- Perched water tables, occurring above shallow ferricrete or clay horizons at depths between 3-5 m below surface.
- Semi-confined weathered and fractured aquifer, occurring at depths of about 20-30 m. The base of the aquifer comprises less fractured dolomitic rocks. The semi-confined aquifer is hydraulically connected to the underlying fractured aquifers within the dolomite. Preferential flow paths are associated with zones of highly weathered/residual dolomite (wad) and highly fractured zones along dyke contacts and faults. Preferential flow paths, which are characterised by higher permeabilities, are likely to be the main zones of contaminant transport.
- Confined fractured aquifers, occurring at depths below 30 m in fractured zones within the unweathered hard rock dolomite, as well as along dyke and sill contact zones. Due to recharge from the semi-confined aquifers above the deeper aquifer, groundwater contamination is likely.

The presence of north-striking dykes and fractures (zones of higher permeability) results in preferred contaminant migration towards a stream channel to the north. Different groundwater tables around the north-west corner of the tailings dam suggest a compartmentalisation of the semi-confined aquifer by dyke systems.

4.11.7 Assessment of contamination

4.11.7.1 Short-term impact

The impact and extent of soil contamination underneath the study site is unknown since only a minor portion has been reclaimed. However, only limited groundwater quality data were available.

High SO_4^{2-} concentrations (range of 1000-2800 mg/l) in abstraction and monitoring boreholes around and on the site indicate the impact of acid mine drainage released from the tailings dam. The pH values are neutral to slightly alkaline with an average value of 7.9 in groundwater samples collected beneath the tailings dam, indicating the presence of buffer minerals and the effect of recharge. Cobalt, Cu, Fe, Ni, Mn, Zn concentrations measured from groundwater samples and sampled from piezometers in the dolomitic aquifer are below the recommended maximum limit (South African Bureau of Standards, 1984) for domestic use. Sulphate concentrations range between 200-600 mg/l with a maximum concentration of > 2000 mg/l in one groundwater sample.

However, analyses conducted on effluent samples collected around the study site show significantly higher concentrations with respect to Co, Cu, Fe, Ni and Mn than those sampled in the dolomitic aquifer. The pH is extremely low (around 2) resulting in the dissolution of metals. Total dissolved solids concentrations are extremely high in these samples and reach a mean of 14600 mg/l reflecting high salt loads. Hence, the fairly neutral pH in the groundwater is caused by the high buffer capacity of the dolomitic groundwater in the area. Metal mobility is relatively low under these pH conditions, despite high salt concentrations.

4.11.7.2 Long-term impact

A numerical groundwater model was used to assess the future contamination impact. The modelling exercise has indicated very small changes in groundwater quality after complete reclamation of the tailings dam. However, potential effects of removal, including remobilization of contaminants or the seepage from residual paddocks, have not been included in the model. The impact of remobilization of contaminants is considered to be likely to be short-term only, based on the results of the model. Additionally, the effect of paddocks on groundwater quality is not likely to result in a considerable redistribution of contaminants.

4.11.8 Discussions and conclusions

Vadose zone monitoring on the reclaimed portion and groundwater monitoring up- and down-gradient of the study site is advised as the future contamination is likely to be high. This is already indicated by contaminated groundwater underneath the site.

The removal of the remaining tailings might improve water quality in the long-term, but a risk-based site investigation is recommended once the reclamation is completed. New monitoring data should be integrated into the present groundwater model to verify and to refine predicted scenarios.

5 ENVIRONMENTAL IMPACT ASSESSMENT

5.1 INTRODUCTION

The unsaturated zone is considered to be both a barrier (geochemical and physical) and a pathway between the primary contamination source (i.e. tailings dam) and the receiving aquifer. Consequently, the properties of the unsaturated zone define the degree of aquifer vulnerability. Water movement and contaminant attenuation conditions have the potential to mitigate the contamination of the groundwater system. However, once this barrier has become contaminated, it can also act as a source for ongoing groundwater contamination. Figure 5.1 illustrates the various contaminant pathways of tailings impoundments.

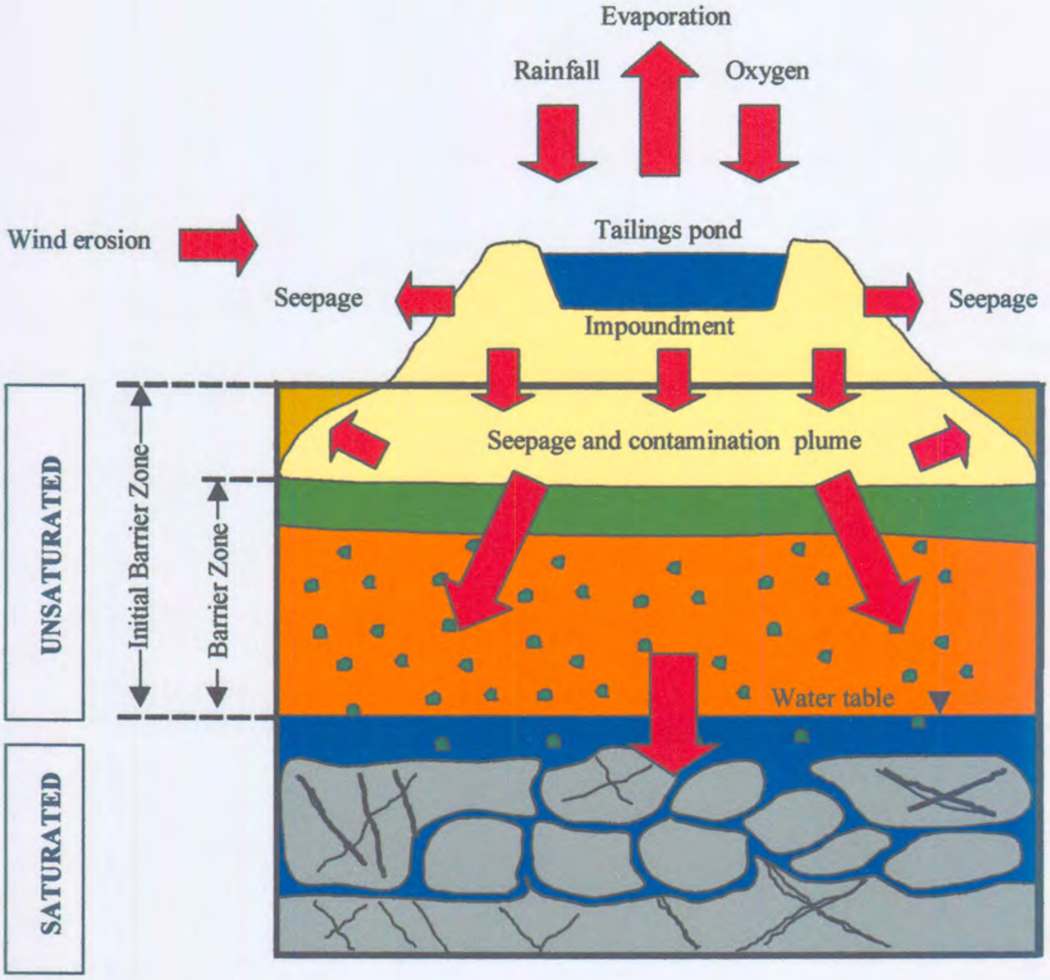

FIG. 5.1 - Conceptual model of a cross-section of a tailings impoundment depicting various contaminant pathways (modified after Parsons & Jolly, 1994).

In most mining operations it is common practice to discharge all contaminated effluents from the ore processing facility to the tailings impoundment. Therefore, minimising the quantity of ponded water escaping from the impoundment will be necessary to avoid pollution, and forms an integral part of best practice environmental management.

5.2 CHARACTERISATION OF THE PRIMARY CONTAMINATION SOURCE

The major and trace element chemistry of five different tailings dams, situated in the East Rand area south-east of Johannesburg is shown in Tables 2.6a and 2.6b. Samples were collected at different gold mine tailings dams from the oxidised zone up to a maximum depth of approximately one metre. It was found that gold mine tailings contain significant concentrations of trace elements. In addition, the mobility of various trace elements in 13 tailings samples was investigated. The extractable concentrations and the relevant threshold value for soils are presented in Table 5.1.

TABLE 5.1 - Extractable elements in gold mine tailings (1 M NH_4NO_3 soil extraction method). Samples were obtained from five different tailings dams in the East Rand area (n=13). Extraction test data for tailings are summarised in Table B-4. Threshold values for soils after Prüeß et al. (1991).

Element mg/l	**25th percentile value**	**75th percentile value**	**Threshold for soils**
As	n. d.	n. d.	0.1
Ca	860	1770	n.a.
Co	1	17.5	0.5
Cr	n. d.	2.25	0.1
Cu	2.5	12.52	2
Fe	2.5	55	n.a.
Mg	72.5	802.5	n.a.
Mn	2.5	27.5	n.a.
Ni	2.5	57.5	1
Pb	n. d.	0.5	2
S	1257.5	4837.5	n.a.
U	n. d.	n. d.	0.04
Zn	1.5	27.5	10

Note: n.d. means not detectable and n.a. information not available.

The extractable portions of Co, Cr, Cu, Ni, Pb and Zn in the bulk of the tailings samples exceed the threshold value for soils. High concentrations of S (maximum > 10000 mg/l) in the leachate indicate the oxidation of sulphide minerals such as pyrite resulting in acid mine drainage. It can be summarised that all investigated reclaimed sites have shown soils with elevated concentrations of contaminants, which are typically contained in tailings material. This clearly indicates the escape of acid mine drainage and associated contaminants from the impoundment into the unsaturated and saturated zones.

In addition, Hahne et al. (1976) reported that Al is the predominant extractable cation in mine residue samples and is a prime hazard for the soils underneath mine deposits due to its phytotoxic effects.

It is interesting to note that no correlation was found between total element concentrations and sampling depth within the oxidised zone of the investigated tailings dams in the East Rand area (Rösner et al., 1998). Steffen, Robertson & Kirsten (1988) reported similar findings.

5.3 SHORT-TERM IMPACT ON THE SUBSURFACE

5.3.1 Unsaturated zone (vadose zone)

The soil underneath reclaimed tailings dams has been contaminated with various trace elements and salts. Table 5.2 summarises the trace element and Fe concentrations in soils underneath the reclaimed study sites. Average background values for the relevant geological units (Vryheid Formation and Malmani Subgroup) are listed in Table 3.10.

TABLE 5.2 - Trace element concentrations in soils underneath the study sites (n=81). Underlined values indicate more than a two-fold excess of the relevant average background value.

Element		**Study sites** (75th percentile values)						
		A	**B**	**C**	**D**	**E**	**F**	**G**
As	mg/kg	53.5	19.8	26.3	24.5	22.8	28.8	40.0
Co	mg/kg	15.0	33.0	40.5	52.5	33.3	26.3	21.3
Cr	mg/kg	346.0	351.0	252.8	192.8	303.0	208.3	129.8
Cu	mg/kg	81.5	131.3	51.0	93.8	53.5	64.3	42.25
Fe	%	10.6	13.0	10.5	9.6	7.6	8.53	4.5
Ni	mg/kg	72.5	158.0	76.3	111.8	85.5	144.0	73.0
Pb	mg/kg	23.25	13.5	18.0	21.0	21.0	10.8	4.8
Zn	mg/kg	75.0	93.8	38.8	53.3	57.8	84.5	44.0
Th	mg/kg	14.8	18.3	18.8	18.0	17.0	20.0	19.3
U	mg/kg	10.8	n. d.	8.0	n. d.	n. d.	818.0	n. d.

The 75th percentile values (Table 5.2) for some elements significantly exceed the average background value for the relevant lithological unit (Table 3.10). However, the total element concentration is a poor reflection of trace element bio-availability. Therefore, Table 5.3 summarises the results of the soil extraction compared with a given threshold concentration after Prüeß et al. (1991):

TABLE 5.3 - Threshold excess ratio of trace elements in soil samples of study site F (n = 16). Extraction test data for soils are summarised in Table B.5 (Appendix B).

	As	**Co**	**Cr**	**Cu**	**Ni**	**Pb**	**U**	**Zn**
MAX	0	40.0	12.5	3.8	77.5	0.5	1500.0	6.3
AVG	0	8.1	0	0.35	14.8	0	105.1	1.3
Number of samples with ratio > 1	0	10	1	5	11	2	3	10

Extractable concentrations of Co, Ni and Zn exceed their threshold values for soils in more than 50 per cent of the investigated samples. For each element, threshold concentrations are exceeded to the greatest extent in the topsoil samples. There is also a decrease in threshold excess with depth as a result of a reduced vertical migration (attenuation).

Furthermore, Cr, Pb and U exceed the threshold in at least one soil sample. Uranium exceeds the threshold to the greatest extent, being 1500 times above the threshold concentration in one sample at study site F. The high U concentrations emanate from radioactive waste material from an former uranium extraction plant. The radioactive material has been deposited on the site prior to the establishment of the tailings dam. Extractable As concentrations were in all instances below the detection limit of the analytical technique and as such did not exceed the threshold value of 0.1 mg/l. Detection limits for XRF and ICP-MS techniques are presented in Table 3.3a, 3.3b and 3.4, respectively.

The mobility of various elements in soil samples of study site F is shown in Table 5.4. Extractable trace element concentrations are expressed as a percentage of the total element concentration.

TABLE 5.4 - Mobility (in percentage of the total concentration) of elements in soil samples from study site F (n=16). Average values were only calculated if more than two samples showed a value larger than 0 per cent. The MIN values were in all instances below 0.1 per cent mobility. All data are summarised in Table B.6.

	As %	**Co** %	**Cr** %	**Cu** %	**Fe** %	**Ni** %	**Pb** %	**U** %	**Zn** %
MAX	-	66.7	0.5	8.3	19.3	50.7	12.5	6.4	39.1
AVG	-	14.9	-	0.9	1.8	8.6	-	-	11.4

Cobalt, Ni and Zn are the most mobile trace elements and the mobility decreases for each element with increasing soil depth due to attenuation. The elements are most mobile in the topsoil units of the test pits. This suggests that a significant portion of the Co, Ni and Zn amount in each soil sample is present in the mobile, easily soluble and exchangeable portions.

Figure 5.2 below shows the relation between soil depth and pH on a site-specific base, where the soil pH increases with increasing depth (best-fit curve). This can be a result of buffering reactions by minerals such as carbonates or alternatively, by a fluctuating shallow groundwater table, which causes mixing and dilution effects with dolomitic groundwater. It is evident that some sites show a more distinct trend (e.g. sites B, E and G) than others.

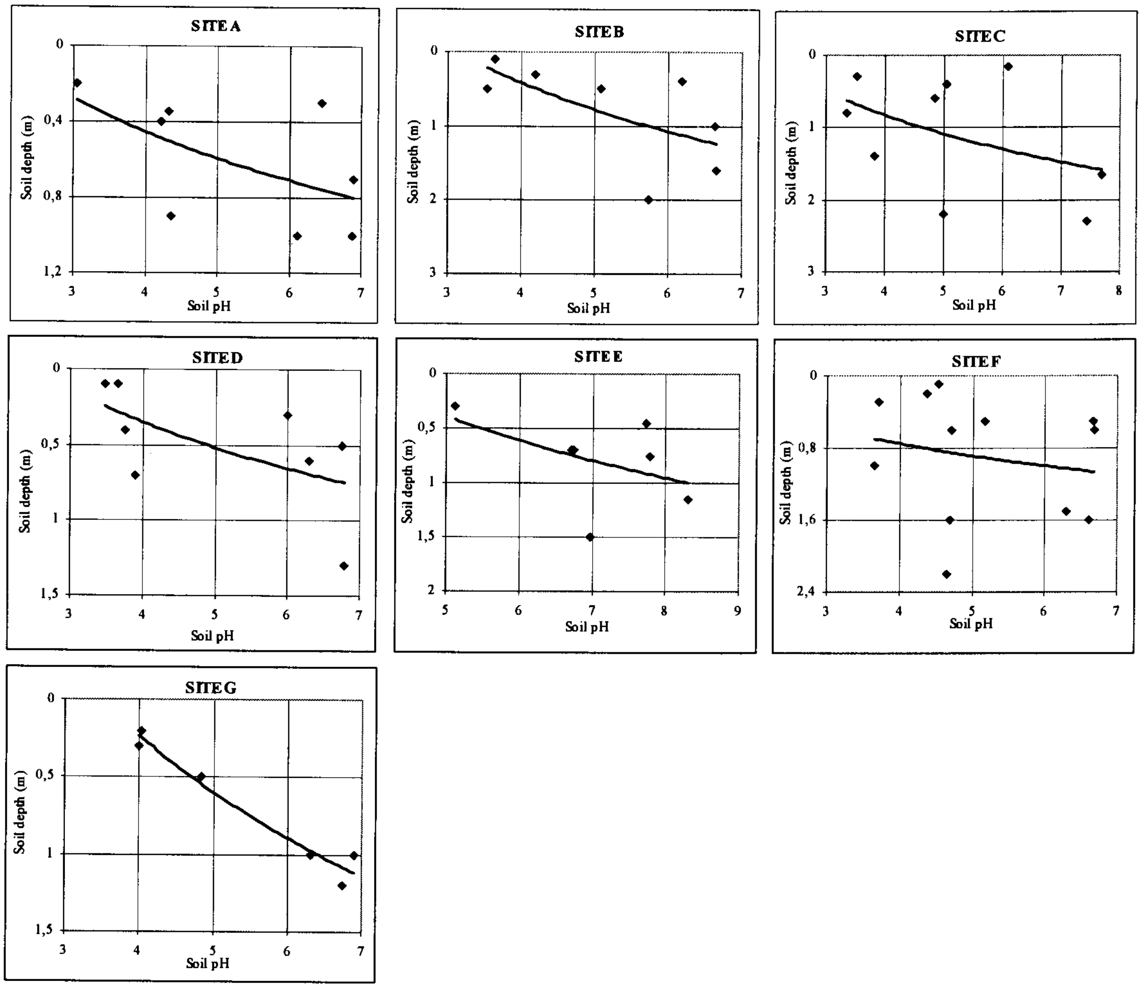

FIG. 5.2 - Relation between soil depth and soil pH on a site-specific base (all sites n=58).

Trace elements can be distinguished by their geochemical behaviour with respect to the ease of solubility and mobility. One of the master variables for dissolution reactions is the soil pH, another one the redox conditions (not determined in this study). Generally, most of the metals dissolve in the acid range (pH below 7) and precipitate under neutral to alkaline conditions (pH above 7) in soils, the exception being the amphoteric metals like Al, As, Cr, Pb, V and Zn.

Figures 5.3a to 5.3h illustrate the element mobility of Co, Cu, Ni, U, Zn, Cr, Pb and Fe as a function of the soil pH. It is evident that considerable dissolution of these elements only takes place at a pH < 4.5, predominantly occurring in the topsoil. Cobalt, Ni and Zn show increasing mobility with decreasing pH, whereas Cr, Pb and U seem to be insoluble. Furthermore, Cu shows a weak, but similar trend compared to Co, Ni and Zn. An explanation of the low mobility of Cr, Cu, Fe, Pb and U could be, that a significant portion of these trace elements appears to be contained in the residual fraction and thus, is not bio-available.

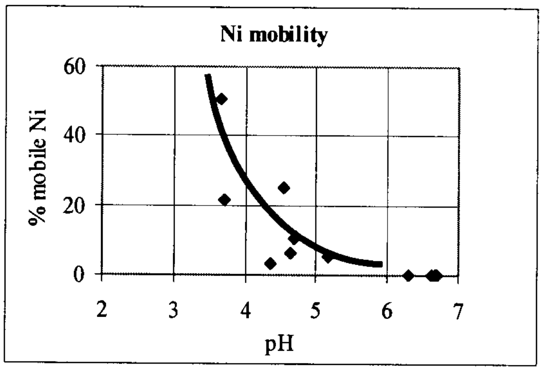

FIG. 5.3a - Ni mobility in soils

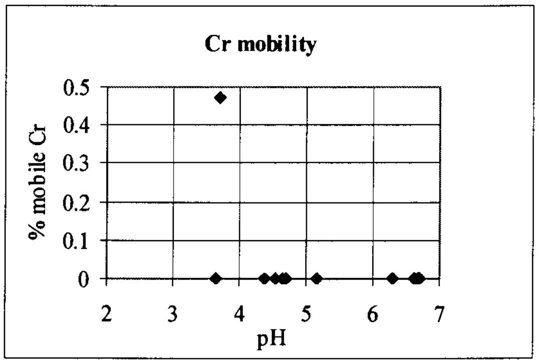

FIG. 5.3b - Cr mobility in soils

Alloway (1995) reported similar findings for Cr (Figure 5.3b), which is contained in the majority of soils and where the relatively insoluble and less mobile Cr^{3+} form predominates and generally occurs as insoluble hydroxides and oxides or even chromite ($FeCr_2O_4$). In addition, the acid character of acid mine drainage-affected soils suggests a rapid reduction from Cr^{6+} to Cr^{3+}, which can substitute Al^{3+} in clay minerals. However, soils of the study area tend to be dominated by kaolinite, which is characterised by a low cation exchange capacity. In addition, Alloway (1995) reports that above a soil pH of 5.5 complete precipitation of Cr^{3+} is likely. Brooks (1987) compared the solubility of Cr and Ni and found that Ni is clearly more mobile than Cr, corresponding to the findings of this study.

The mobility of Ni increases as the pH (Figure 5.3a) and cation exchange capacity decrease (Alloway 1995). Kabata-Pendias (1994) reported that over 60 per cent of Ni in soils may be associated with the residual fraction, approximately 20 per cent with the Fe-Mn oxide fraction and organic matter while the remainder is bound up with the carbonate fraction (Alloway 1995). However, the presence of ferruginous soils caused due to intense weathering and free drainage (Brink, 1985) in the study area suggests that a large portion of Ni is likely to be bound on Fe-hydoxides, whereas organic material is probably absent. In addition, it is well established that the Ni uptake by plants increases as the exchangeable fraction in soils increases due to the acidification caused by acid mine drainage. Hence, the concentration of Ni in plants can reflect the concentration of the element in the soil, although the relationship is more directly related to the concentration of soluble ions of Ni and the rate of replenishment of the mobile fraction (Hutchinson 1981).

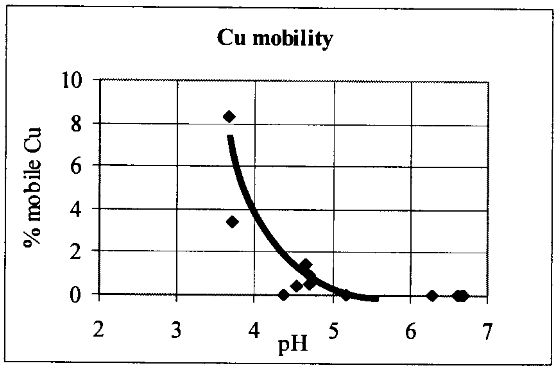

FIG. 5.3c - Cu mobility in soils

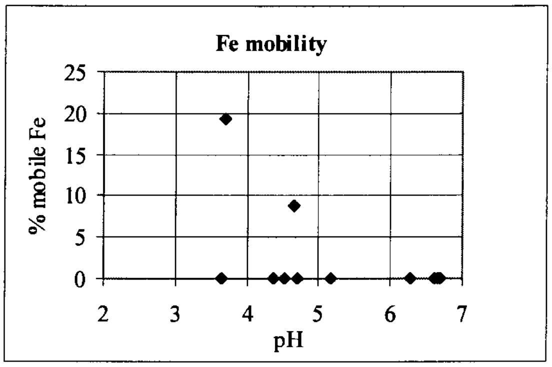

FIG. 5.3d - Fe mobility in soils

Although Cu (Figure 5.3c) is less mobile than Co, Ni and Zn it is important to note that Cu concentration levels of 1.5 to 4.5 mg/kg damage or kill roots of growing plants (Alloway, 1995).

Iron mobility (Figure 5.3d) is very low and significant mobility was only found in two soil samples at a pH < 5. The results correspond to the general immobile character of Fe, where a carrier such as colloids is required to allow migration in soil water/groundwater. It is important to note that Fe-precipitates such as Fe-oxides provide additional adsorption surfaces for other metals within the soil system. Total Fe (measured as Fe_2O_3) concentrations found in soil samples of the study area range from 3-24 weight-per cent and are often associated with the occurrence of ferricretes.

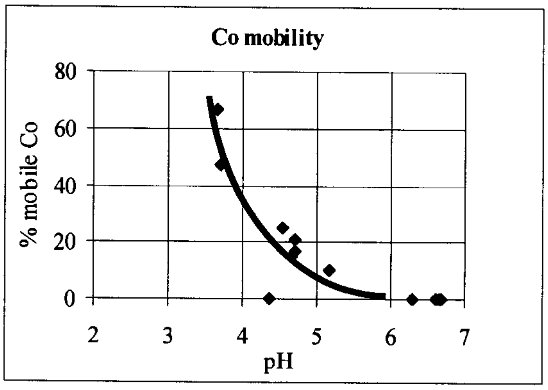

FIG. 5.3e - Co mobility in soils

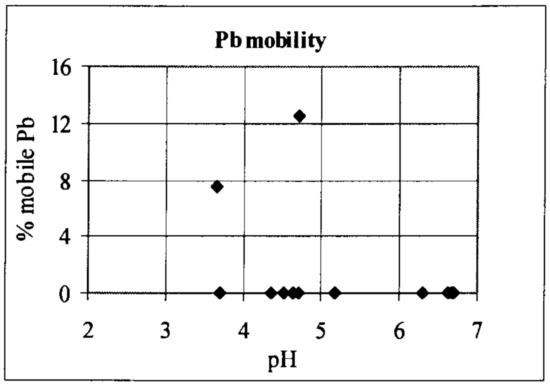

FIG. 5.3f - Pb mobility in soils

Cobalt (Figure 5.3e) shows a very high mobility (pH < 5) up to 67 per cent compared to the solid phase. This would result in a higher plant uptake and is as also reported by

Alloway (1995). Furthermore, Co is often found adsorbed onto Mn minerals such as MnO_2. A positive correlation coefficient of $r = 0.63$ ($n = 81$) was calculated for MnO versus Co, which corresponds with the above observation.

Lead (Figure 5.3f) has a very low mobility in soils, and thus accumulates within the topsoil. Similar observations were made in Finland, Canada and in the United Kingdom by Alloway (1995) who found that soils affected by mining operations show higher accumulations of Pb in topsoils than in unaffected soils, suggesting a low mobility even under acid soil conditions.

The mobility of U (Figure 5.3g) is very low, but three samples showed an elevated mobility occurring only in the topsoil. Uranium mobility is very low and occurs only under strongly acid soil conditions ($pH < 5$). In case of elevated mobility, the threshold excess value of U ranged from 62 to 1500. However, the correlation coefficient between U/As gave a positive coefficient of $r = 0.74$ ($n = 81$), which is also reflected by the immobility of As. The formation of the uranyl cation UO_2^{2+} is the most likely reason for the solubility (e.g. $(UO_2)SO_4$ is soluble) of U over a wide pH range. However, the low mobility of U found in the soil underneath reclaimed tailings deposits could be caused due to a co-precipitation (secondary mineral) with arsenate in the soil (Bowie and Plant, 1983) after U was released from the primary mineral during the operation of the tailings dam.

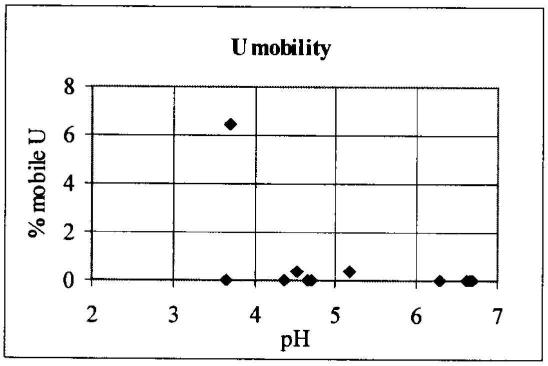

FIG. 5.3g - U mobility in soils

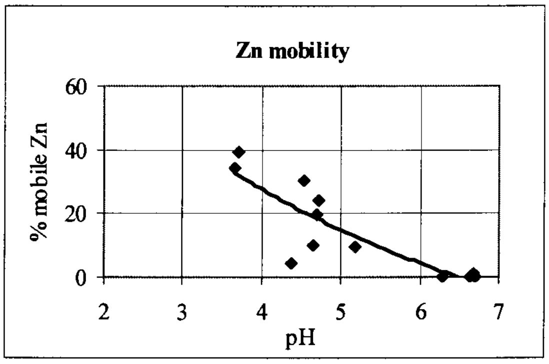

FIG. 5.3h - Zn mobility in soils

Zinc (Figure 5.3h) has a high mobility, where the solubility increases with decreasing soil pH, corresponding with the findings of Kabata-Pendias (1994) for acid soils. Figures 5.4a to 5.4h show element concentrations versus soil depth at the study sites A-G (the dotted reference line indicates the highest average background value).

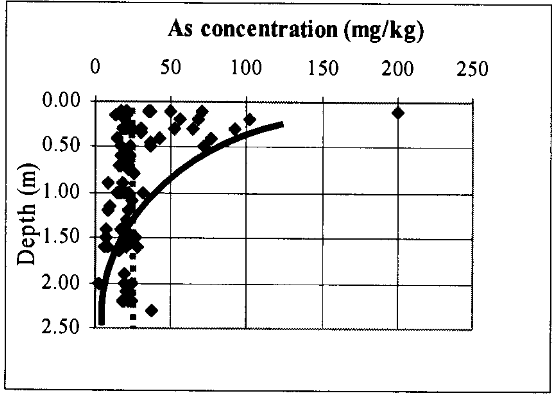

FIG. 5.4a - Total As contents in soils (n=81)

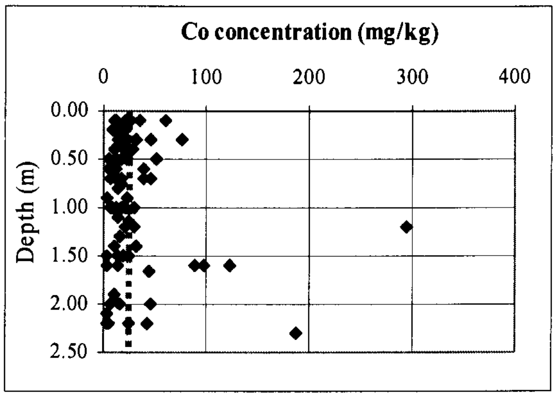

FIG. 5.4b - Total Co contents in soils (n=81)

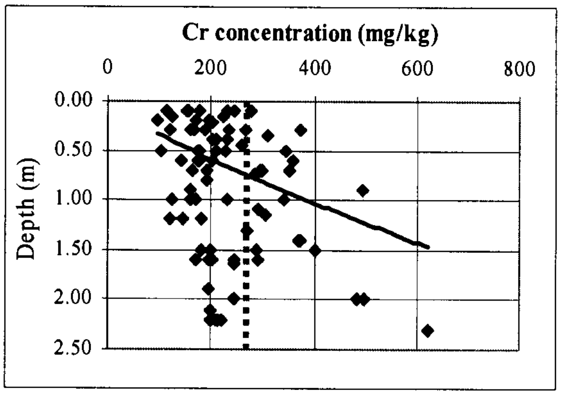

FIG. 5.4c - Total Cr contents in soils (n=81)

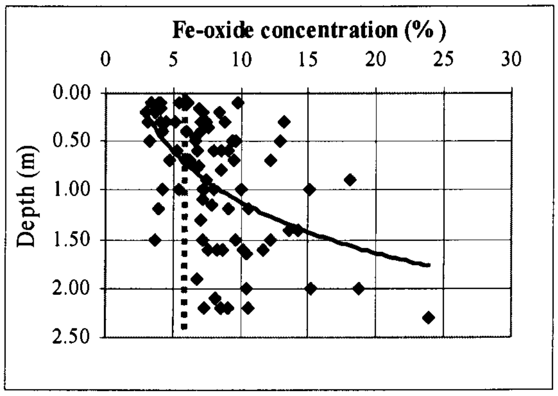

FIG. 5.4d - Total Fe contents in soils (n=81)

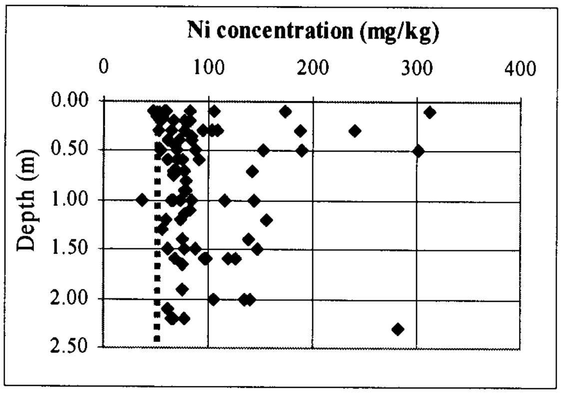

FIG. 5.4e - Total Ni contents in soils (n=81)

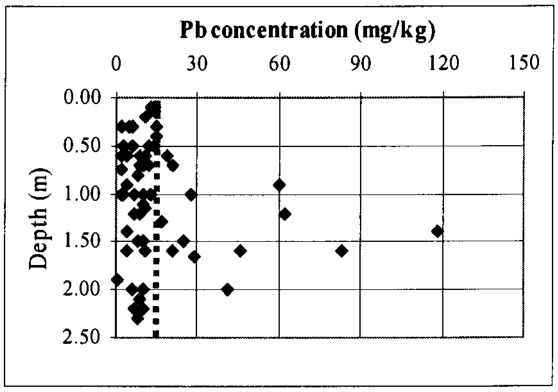

FIG. 5.4f - Total Pb contents in soils (n=81)

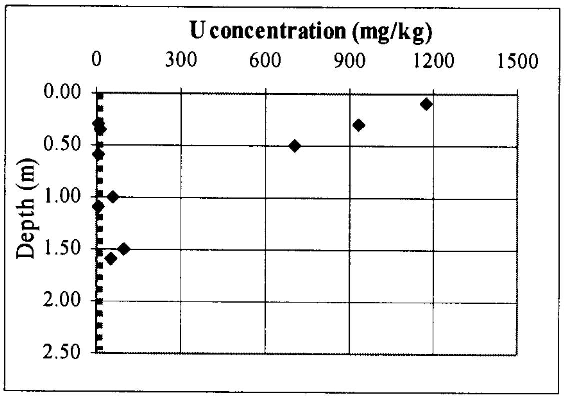

FIG. 5.4g - Total U contents in soils (n=10)

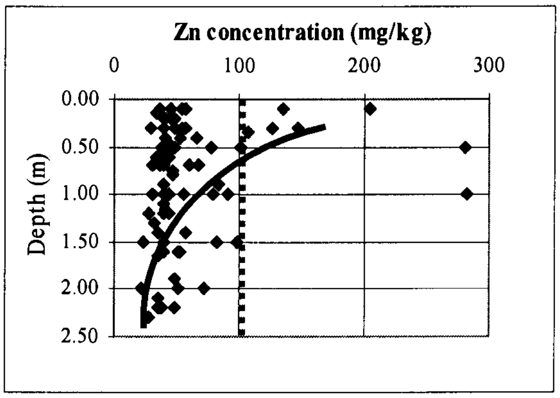

FIG. 5.4h -Total Zn contents in soils (n=81)

Chromium and Fe show a weak linear trend (using best-fit curves) by increasing concentrations with increasing soil depth. However, the trend of As and Zn suggests an exponential decrease in concentration with depth. Furthermore, Co, Ni and Pb show no trend with soil depth, although Co and Ni showed a high mobility using extraction tests. Uranium concentrations are the highest only in the upper soil units. It can be summarised that there is no clear geochemical pattern regarding a correlation between soil depth and total element concentrations as a result of heterogeneous site and soil conditions.

However, highest concentrations seem to occur generally in the upper soil units and depth related element accumulations were found, if single test pits were considered. Although, generally clay minerals have a much higher adsorption capacity for trace elements compared to coarser grain size fractions, no correlation was found between the clay content in soils and total element concentrations indicating the presence of preferential flow paths that bypass matrix flow. It is anticipated that the flow rate under such conditions is high, resulting in a contact time between seepage and solid phase, which is insufficient to allow significant sorption effects for contaminants. Trace element concentrations versus the relevant clay content in soils is shown in the following Figures 5.5a to 5.5d.

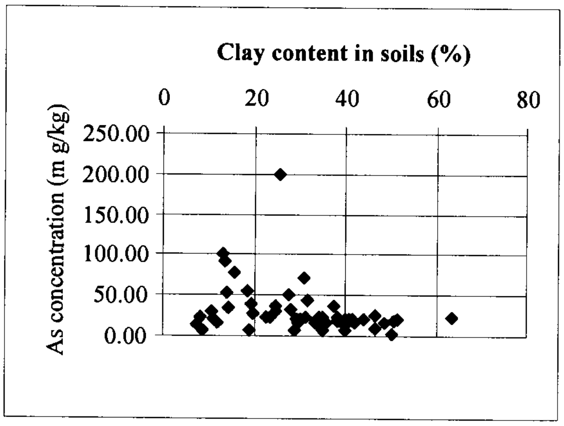

FIG. 5.5a - As content versus clay amount

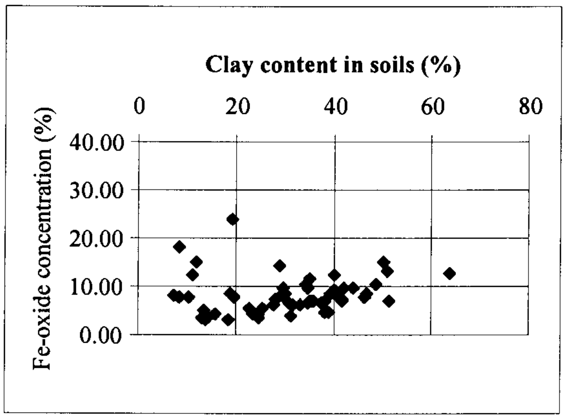

FIG. 5.5b – Fe content versus clay amount

These observations correspond to the findings of Merrington & Alloway (1993) which investigated the properties of heavy metals in soils affected by old iron mines in England.

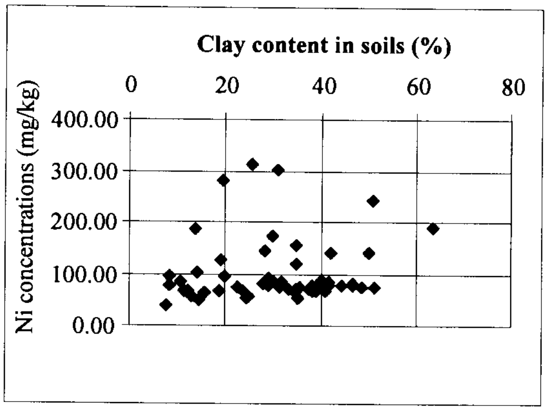

FIG. 5.5c - Ni content versus clay amount

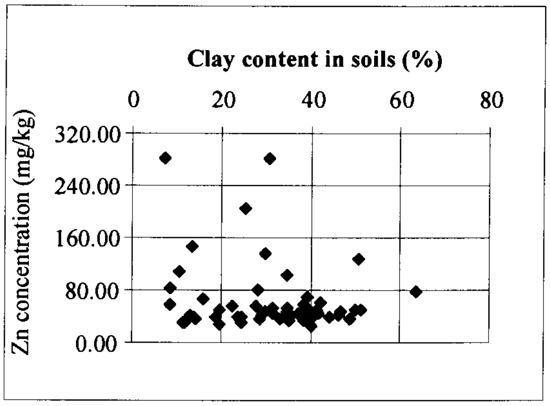

FIG. 5.5d - Zn content versus clay amount

In conclusion, trace element values exceeding threshold concentrations (i.e. Co, Cr, Cu, Ni, Zn and U) may limit soil functioning. The mobility of the trace elements is a function of soil pH. The majority of the topsoil samples were highly acidic (pH 3-4), whereas deeper samples showed generally higher pH-values (pH 5-7). The low pH value in soils underlying tailing dams is a direct result of the pyrite oxidation and the associated generation of acid mine drainage. All of the investigated trace elements are most mobile when pH < 4.5 and least mobile when a soil pH > 6 prevails. Cobalt, Ni and Zn are the most mobile trace elements and phytotoxic, corresponding to literature. In contrast, the mobility of Cr, Cu, Fe, Pb and U is lower, indicating that a significant portion of these trace elements are contained in the residual fraction (i.e. bound to silicates). Arsenic concentrations were below the lower detection limit in all soil extraction tests. The potential hazard for land development and groundwater contamination posed by the trace elements can be summarised as U>>Co=Ni=Zn>Cr=Pb>>As. This potential hazard series is only a function of the degree and frequency with which a trace element exceeds the relevant threshold value. It is also important to note that such extraction tests were only conducted at one site and further tests would be necessary to confirm and support these findings.

Furthermore, hydraulic conductivities derived from geotechnical properties (estimated from techniques of Mathewson, 1981 and Tavenas et al., 1983) and *in situ* test data indicate a low to very low vertical hydraulic conductivity (range between 10^{-7}–10^{-10} m/s) of the investigated soil profiles. Contaminants measured at greater depths would, however, require alternative migration mechanisms than percolation through the porous media, because of their retardation. Soil conditions indicating preferential flow (bypass of the soil matrix) were observed in nearly all test pits, but attempts to identify dominating contaminant migration pathways and mechanisms would be premature and further investigations are necessary.

Extraction tests on gold mining tailings (Table 5.1) have shown high S concentrations contained in the leachate. Hence, incomplete reclamation of tailings would result in an additional reservoir for acid generating processes and contaminant release.

The mitigation of acid mine drainage at greater soil depths may have various reasons such as the presence of acid neutralising minerals (e.g. carbonates) or a seasonal fluctuating groundwater table, which causes dilution (mixing with dolomitic groundwater) effects.

Finally, many countries such as the Netherlands (Ministry of Housing, Physical Planning and Environment, 1997) provide guidelines for soil quality to assess the degree of soil contamination. However, these guidelines were established in Europe and North America, where humid climate conditions, which determine different soil conditions, predominate. No soil protection guidelines are available for climatic conditions such as those experienced in the study area.

5.3.2 Saturated zone (groundwater system)

5.3.2.1 Regional groundwater quality

Groundwater collected across the study area (East Rand area) can be characterised as two distinctive groundwater types (Kafri et al. 1986 and Scott, 1995):

- Ca-Mg-HCO_3;
- Ca-Mg-SO_4.

Piper diagrams representing these groundwater types are shown in Scott (1995). Groundwater quality showing a predominant Ca-Mg-HCO_3 character frequently indicates recharged waters associated with dolomitic aquifers. Such groundwater often shows low total dissolved solid values and a high total hardness. High Na^+ levels in some samples are probably reflected by ion exchange processes (Lloyd & Heathcote, 1985) preferably from the overlying Karoo strata (Scott, 1995). Scott (1995) reported that high Ca^{2+} concentrations resulting from the dissolution of dolomite (and in some cases from lime treatment) and alkalinity may exceed drinking water standards in some areas.

In contrast, groundwater quality which is predominantly characterised by a Ca-Mg-SO_4 signature and high total dissolved solid concentrations indicates discharge areas (Palmer, 1992), but in the case of the study area, it is more likely to indicate acid mine drainage-related pollution. Although the relative proportions of cations in this groundwater remain similar to those of unaffected groundwater, the anion signature reflects the progressive dominance of SO_4^{2-} over HCO_3^- as the reaction products of the sulphide mineral (e.g. pyrite) oxidation are introduced into the groundwater system. In contrast, Scott (1995) reported that in some areas, the ratios of main elements in surface and groundwater are very similar, particularly along less polluted portions of the Blesbokspruit, indicating that surface and groundwater are closely related across parts of the study area.

It is important to note that dolomites of the Oaktree (chert-poor dolomite) and Eccles (chert-rich dolomite) Formations occur within the study area. Kafri et al. (1986) reported that these dolomites occasionally contain considerable amounts of pyrite, which could contribute to metal and SO_4^{2-} pollution in groundwater. However, it is highly unlikely that the natural pyrite content would cause a SO_4-dominated water type in dolomitic aquifers.

5.3.2.2 Groundwater quality in the study area

Groundwater quality beneath and in close vicinity of the investigated tailings dams is dominated by the Ca-Mg-SO_4 type, although all sites with relevant groundwater data (sites H, I and K) are underlain by dolomitic rocks. In general, groundwater quality seems to improve further down-gradient of the tailings dams and reclaimed sites as a result of dilution effects and the high acid neutralisation capacity of the dolomitic aquifer. However, groundwater quality in close proximity to the sites is characterised by elevated metal (e.g. As, Cd, Co, Fe, Mn and Ni) and total CN concentrations, occasionally exceeding drinking water standards in boreholes. Sulphate concentrations are often very high in the immediate vicinity to the tailings dam (generally > 2000 mg/l, but up to 4000 mg/l).

Water in perched or shallow aquifers is of very poor quality. Table 5.5 shows the seepage water quality of the case study sites C and G and in an old open mine shaft next to a tailings dam, which is situated about 1.7 km south-east of Duduza (Figure E.9). For comparison, these data are supplemented with pond water qualities from gold tailings operations in Arizona, USA.

Table 5.5 - Seepage water quality of study sites C and G, an open mine shaft and for comparative reasons of gold tailings operations in Arizona, USA from Lampkin & Sommerfeld (1981).

Parameter		**Average contents at sites C and G**	**Open mine shaft**	**Pond from gold tailings in Arizona, USA**
pH		5.5	2.0	n. a.
Total Alkalinity		178	n. a.	n. a.
Total Hardness		1760.5	1695	n. a.
TDS	mg/l	4508	4386	n. a.
EC	mS/m	489.5	516.0	n. a.
SO_4	mg/l	2883	2975	n. a.
Cl	mg/l	271.5	223	n. a.
NO_3	mg/l	< 0.1	<0.1	n. a.
F	mg/l	0.4	0.6	n. a.
Al	mg/l	1.5	86.3	n. a.
As	mg/l	0.12	< 0.10	n. a.
Ca	mg/l	372	453	176.4
CN	mg/l	< 0.2	2.2	n. a.
Cu	mg/l	0.1	1.2	21.0
Fe	mg/l	217.1	98.1	109.1
K	mg/l	79.7	43.6	1.5
Mg	mg/l	202	137	53.1
Mn	mg/l	179.9	29.2	16.1
Na	mg/l	244.5	279.0	13.4

Table 5.5 continued.

Ni	mg/l	2.3	16.9	n. a.
Pb	mg/l	0.03	0.02	n. a.
Zn	mg/l	0.2	3.3	115.6

Note: n. a. means information not available. Accuracy of the analysis, which is outlined in paragraph 3.3, for open mine shaft: Δ -18.3 per cent (high inaccuracy).

It is apparent that SO_4^{2-} as well as some metals (e.g. As, Fe, Mn, Ni, Pb and Zn) and CN exceed the recommended target water quality standards. The pH in the seepage of the old open mine shaft is extremely low and enhances metal leaching. As a result some metal concentrations such as Fe and Mn are substantially lower due to precipitation as hydroxides. Even if the topsoil would be treated with lime and fertilisers (e.g. superphosphate) in order to achieve suitable soil conditions for plant growth, it may take a long time until groundwater quality will gradually improve. Finally, it must be stressed that agricultural activities often occur in immediate vicinity to tailings dams, and the use of such affected water for agricultural (e.g. irrigation) or domestic purposes should be avoided.

5.3.2.3 Estimation of seepage and sulphate loads

A load estimation was undertaken in order to assess the annual discharge of seepage and SO_4^{2-} from the reclaimed sites in South Africa into the groundwater system. Rösner et al. (1998) have shown that the total area of land affected by reclaimed gold mine tailings amounts to approximately 13 km^2. All these sites are situated within the Vaal Barrage catchment system, which plays a major role for water supply of the Johannesburg region. Vegter (1984) estimated a net recharge value of 12.5 per cent of the annual rainfall for the Gauteng Province, which equals 80-100 mm recharge per year (Vegter, 1984).

Thus, the load estimation can be expressed as:

1. Conversion of recharge/year (i.e. 80 mm/year) into an area size. Method after Hölting (1996).

$$\text{80 mm recharge/year equals } 2.536\ l/(s \cdot km^2) \tag{5.1}$$

2. Conversion of recharge value from seconds per km^2 to year per km^2

$$2.536 \frac{l}{s \cdot km^2} = 79.97 \cdot 10^6 \frac{l}{year \cdot km^2} \tag{5.2}$$

3. Multiplication of recharge value with the relevant area size (i.e. 13 km^2)

$$79.97 \cdot 10^6 \frac{l}{s \cdot km^2} \cdot 13 km^2 = 1.0397 \cdot 10^3 \frac{Ml}{year} \tag{5.3}$$

The estimation above has shown that the accumulated volumes of seepage amount annually to approximately 1000 Ml. For the SO_4^{2-} load estimation two scenarios with concentrations of 1006 mg/l and 4760 mg/l, respectively were considered. These values were obtained from seepage analyses conducted at study sites C and G (Table 4.4a and 4.10a), respectively and also correspond to measurements at other sites (study sites H and I). The results for the two scenarios are as follows:

1. Scenario: 1046 t SO_4^{2-}/yr for all reclaimed sites (13 km^2) equals 804 kg/ha · yr
2. Scenario: 4949 t SO_4^{2-}/yr for all reclaimed sites (13 km^2) equals 3.81 tons/ha · yr

It should be noted that the process of pyrite oxidation is kinetically controlled and long-term predictions can be only made by using relevant geochemical models. Furthermore, without any site rehabilitation, a load decrease of SO_4^{2-} and associated contaminants will only occur once the source of oxidation (i.e. pyrite) has depleted. In addition, changes in land use might enhance pyrite oxidation, thus resulting in even higher contaminant loads.

It is of great concern that sites, which were reclaimed a long time ago (e.g. study site C, more than 10 years ago) still show very high SO_4^{2-} concentrations in the seepage. However, it is premature to quantify accurately the amount of seepage and subsequently the load of contaminants for all gold mine tailings dam sites in the country. Various attempts have been made in the past (Steffen, Robertson & Kirsten, 1988), but the lack of appropriate long-term water quality data around such tailings dam sites as well as lacking technical information about the sites themselves (e.g. the presence of pollution control measures) result in many unknown parameters.

5.4 LONG-TERM IMPACT ON THE SUBSURFACE

5.4.1 Unsaturated zone

The application of the geochemical load index system is a conservative approach, assuming that the total contaminant load in the solid phase could be dissolved. However, studies by Kabata-Pendias (1994) have shown that only a minor portion of heavy metals are bio-available, as they are only contained in the easily soluble and exchangeable phase. A risk assessment would be required if land development such as housing is envisaged.

The concentrations of Fe_2O_3 (total), MnO and various trace elements (i.e. As, Ba, Co, Cr, Cu, Mo, Ni, Pb, Sn, Th, U, V and Zn) were compared to background soil concentrations of similar geology by using the geochemical load index system according to Müller (1979). Based on the results of this comparison, a table listing contaminants of concern for each site was produced (Table 5.6). This methodology allows the assessment of potential future trace element loads in the investigated soil profiles. Significant contamination is reflected by contamination classes III-VI, where the average background value is considerably exceeded.

TABLE 5.6 - Hazard rating of the study sites by using the geochemical load index system (after Müller (1979).

Hazard rating of the study sites						
Study site	**Class I** None–moderately contaminated	**Class II** Moderately contaminated	**Class III** Moderately–highly contaminated	**Class IV** Highly contaminated	**Class V** High–excessively contaminated	**Class VI** Excessively contaminated
A	Ni, Zn	As, Cr, Cu, Fe, Pb, V	Sn, U	-	-	-
B	As, Cr, Fe, Mo, Th, V, Zn	Cu, Ni	Co	Pb	-	-
C	As, Cr	Cu, Fe, Ni, Mn, Pb, V	-	Co	-	-
D	Cu, Mn, Th	As, Cr, Fe	Ni, Pb	Co, U, V	-	-
E	As, Cr, Cu, Ni, Th, Zn	Co, Fe	Pb, V	-	-	-
F	Fe, Cr, Cu, Mo, V, Zn	Co, Mn , Th	As, Ni	-	-	U
G	As, Ni, Sn	Co	-	-	-	-

Table 5.6 can be summarised as follows:

- Moderately to highly contaminated sites (class III): five sites with respect to the following trace elements: As, Co, Ni, Pb, Sn, V and U. Cobalt and Ni are phytotoxic and, therefore have negative effects on plant growth (Alloway, 1995). High As concentration was only found in one case. However, As is less mobile than other metals and thus, effects are negligible.
- Highly contaminated sites (class IV): three sites with respect to Co, Pb, U and V. Vanadium is not a typical mine tailings contaminant, and enrichment caused by natural processes in association with ferricrete in soils (Figure E.11) is most likely (Németh, Molnár, Csillag, Butjás, Lukács, Pártay, Féher & Van Genuchten, 1993 and Alloway, 1995). High U concentrations were found only at one site.
- One site has been classified as an excessively contaminated site (class VI) as a result of U (measured as U_3O_8) concentrations more than 100-fold above the natural background. It is unlikely that the high U concentrations in the soil emanate from the gold mine tailings, the U concentrations are rather caused by the deposition of radioactive material deposited prior to the construction of the tailings dam.

It can be concluded that the long-term impact of typical mine tailings on soils and groundwater will mainly depend on the availability of minerals with a sufficient acid neutralisation capacity in soils. The ongoing production of SO_4^{2-} and acids are a result of sulphide mineral oxidation by residual tailings material on the surface. When the topsoil becomes highly acidic (pH around 4.5), the acidity starts to migrate into the subsoil. Once the subsoil becomes acidic, only the most acid-tolerant plants can be grown (Fenton, 1997). Acidification of the subsoil is a form of permanent soil degradation and therefore, the primary contamination source should be completely removed from the reclaimed sites in order to minimise further acid and salt generation.

5.4.2 Saturated zone

Only a limited number of tailings dams in South Africa have been investigated in detail with respect to the hydrogeological conditions, including contaminant analysis. One South African mining company applied numerical groundwater models for two tailings dam sites, situated within the study area. The model applications have shown that tailings dams continue to release seepage with high salinity for an extended time period (predictions were given for about 50 years) after decommissioning (closure) of tailings operations.

Acid and salt generation in tailings dams and the subsequent impact on the groundwater system can only be mitigated by preventing the moisture and oxygen flux into the impoundment or soil, which can be only achieved by cover systems. Such a cover would also prevent wind erosion and the subsequent dispersion of contaminants in the near surrounding of the deposit or footprint. The models have also shown that deterioration of groundwater quality occurs only in the immediate vicinity of the impoundment. Groundwater quality improves with increasing distance down-gradient of the tailings dam mainly due to dilution and sorption effects. Seepage emanating from tailings dams is, however, likely to affect water quality negatively in nearby surface water systems due to discharge from affected groundwater or surface run-off, which would have an adverse impact on water users in that particular area.

It is important to note that as a result of dewatering of underground mines, groundwater tables dropped across the study area, causing the Blesbokspruit to discharge water along permeable sections of the water course into the groundwater system. Mine closures and the reduction of mining operations resulted in a rapid groundwater table recovery in the dolomitic aquifers in the East Rand area. This might affect water courses such as the Blesbokspruit to effluent rather than influent streams in future, characterised by recharge from the aquifers (Scott, 1995). The effect of groundwater recovery cannot be regarded in isolation from the release of acid mine drainage from tailings dams.

6 RISK ASSESSMENT AND REHABILITATION MANAGEMENT

6.1 INTRODUCTION

Rehabilitation is defined as the restoration of a disturbed land area to a landform and productivity which conforms with the landform and productivity of the locality before disturbance took place.

Section 38 of the Minerals Act (50/1991) in South Africa determines the approach towards rehabilitation of the surface in a mining or prospecting environment. Within the context of the mining authorisation, rehabilitation measures must be carried out as follows:

- In accordance with the approved environmental management plan;
- As an integral part of the operations or prospecting progress;
- At the same time as the mining operations, unless otherwise determined by the Regional Director;
- To the satisfaction of the Regional Director of the Department of Minerals and Energy.

Mining plays an essential role in the South African economy. However, there are growing cost implications involved in the disposal of mining wastes due to adverse effects on soils, surface and groundwater quality. The extent of these effects depends on the physical characteristics of the disposal site, mineralogy of the ore, the metallurgical process, the method of disposal, the climate and microbiological conditions within the disposal site and in the underlying subsurface.

In the case of reclaimed gold mine tailings dams, the remediation of the subsurface (soil and groundwater) is of major importance for the mitigation of adverse effects on the aquifer and in order to enable land development. As a result, waste remediation efforts are heavily influenced by statutory and regulatory compliance and in some countries, such as the USA, waste remediation is often dictated by these regulations. Regulatory standards and guidelines are becoming increasingly prescriptive as regards procedural and technical requirements.

It is clearly preferable to prevent contamination problems at the outset by investigating contamination potentials at the mine planning stage, and deciding on the most appropriate metallurgical process and waste disposal. However, in case of contaminated sites or footprints, the success of rehabilitation measures depends on how effectively contamination has been eliminated and how sustainable the rehabilitation effort is in the long-term (Van der Nest & Van Deventer, 1996).

6.2 RISK ASSESSMENT

6.2.1 Principles and definition

With introduction of the second edition of the *Minimum Requirements for Water Monitoring at Waste Management Facilities,* DWAF (1998) has endorsed a risk-based framework within which contamination at waste facilities can be identified, assessed and environmentally soundly managed.

In this context, the term *risk* has a multitude of uses and is not to be confused with *hazard.* In the context of contaminated land or soils, the term risk is generally defined with the likelihood of human health problems occurring if no clean-up measures were taken at the site (EPA, 1997a). In terms of contaminated sites or land, risks to human health and the environment can be regarded as being comprised of three components (Young, Pollard & Crowcroft, 1998):

- Source, reflecting a toxic substance or group of toxic substances with the potential to cause harm;
- Pathway, reflecting a route by which a receptor could be exposed to, or affected by, the toxic substance(s);
- Receptor, reflecting a particular entity which is being harmed or adversely affected by the toxic substance(s).

The probability of a hazard being realised (i.e. the risk), depends on the conditions of a linkage between these components, including site-specific factors such as the contaminant concentration in the exposure medium, the bio-availability, the ease of access to the exposure pathway and the duration of exposure.

The term of *risk assessment* can be basically defined as an evaluation of the probability of harm and, in context with contaminated sites, is concerned with gathering and interpreting information on the characteristics of sources, pathways and receptors at a specific site and to understand the uncertainties inherent to the ensuing assessment of risk.

The requirements of the risk assessment define the scope of a site investigation and, thus potential remediation measures. In practice, this involves the geochemical characterisation of the contaminant(s), relevant soil properties and groundwater conditions encountered and the site factors that influence contaminant fate and migration.

Because the unsaturated zone is often a physical and geochemical barrier zone between the source of contamination and the receptor (i.e. groundwater), the understanding of the contaminant transfer between these components is an essential prerequisite to the site investigation, risk assessment and the subsequent development of rehabilitation strategy.

6.2.2 Methodology

Four key phases have to be generally assessed as part of a risk assessment according to EPA (1997b). These phases including their practical implementation are described below:

6.2.2.1 Hazard assessment

The hazard assessment is the qualitative evaluation of the adverse health effects of a toxic substance(s) in animals or humans.

Practical implementation: Collection and analysis of samples from different media (i.e. soil, air and/or water, sediments, fish, plants, animals) from and around the site of concern. Development of a *Conceptual Site Model* (method according to the EPA, 1997b) that reflects the site conditions including contaminant sources (current and future land use), release mechanisms (i.e. bio-availability), exposure pathways, migration routes, and potential human and ecological (e.g. groundwater) receptors.

6.2.2.2 Exposure assessment

The exposure assessment is the evaluation of the components, magnitude, time, and duration of actual and anticipated exposures and of doses, when known; and when appropriate, the amount of people who are likely to be exposed.

Practical implementation: Assessment of exposure pathways for contaminants from the source to the receptor via the pathway, whereas contaminants of concern have to be identified in Phase 1. As a result, a *Reasonable Maximum Exposure* (EPA, 1997a) scenario is calculated which reflects the highest level of human exposure that could reasonably be expected to occur from the contaminant identified in Phase 1. This scenario also assesses the duration of exposure time that could occur if no clean-up measures at the site take place. Such an exposure assessment ensures that the selected clean-up or remediation measure protects all people potentially affected by the contaminated site, with a focus on the most vulnerable or sensitive populations (e.g. children). These populations are also termed *critical group.*

6.2.2.3 Dose-response assessment

The dose-response assessment is the process of estimating the relation between the dose of a substance(s) and the incidence of an adverse health effect situation.

Practical implementation: Determination of the toxicity, or harmfulness, of each contaminant identified in Phase 1. It is obvious that the type and intensity of potential health problems depend on the contaminant itself and the duration of exposure. For example, a likely dose could be derived from the consumption of contaminants as a result of drinking polluted water every day for 30 years. The EPA uses two methods to evaluate effects on human health arising from exposure of toxic substances from contaminated sites. One approach calculates the probability of cancer incidents occurring as a result of exposure (e.g. 1 in 10 000 probability). The second method evaluates non-cancer health effects of contaminants from the site (EPA, 1997a).

6.2.2.4 Site risk characterisation

The site risk characterisation is the process of estimating the probable incidence of an adverse health effect to humans under various conditions of exposure, including an assessment of the uncertainties involved (usually a sensitivity analysis).

Practical implementation: Determination of the most critical risks and whether these risks are significant enough to cause negative or adverse effects on health for people living near the site. The results of the three previous Phases 1, 2 and 3 are combined, evaluated and summarised. Figure 6.1 illustrates an example for a risk-based investigation for contaminated sites.

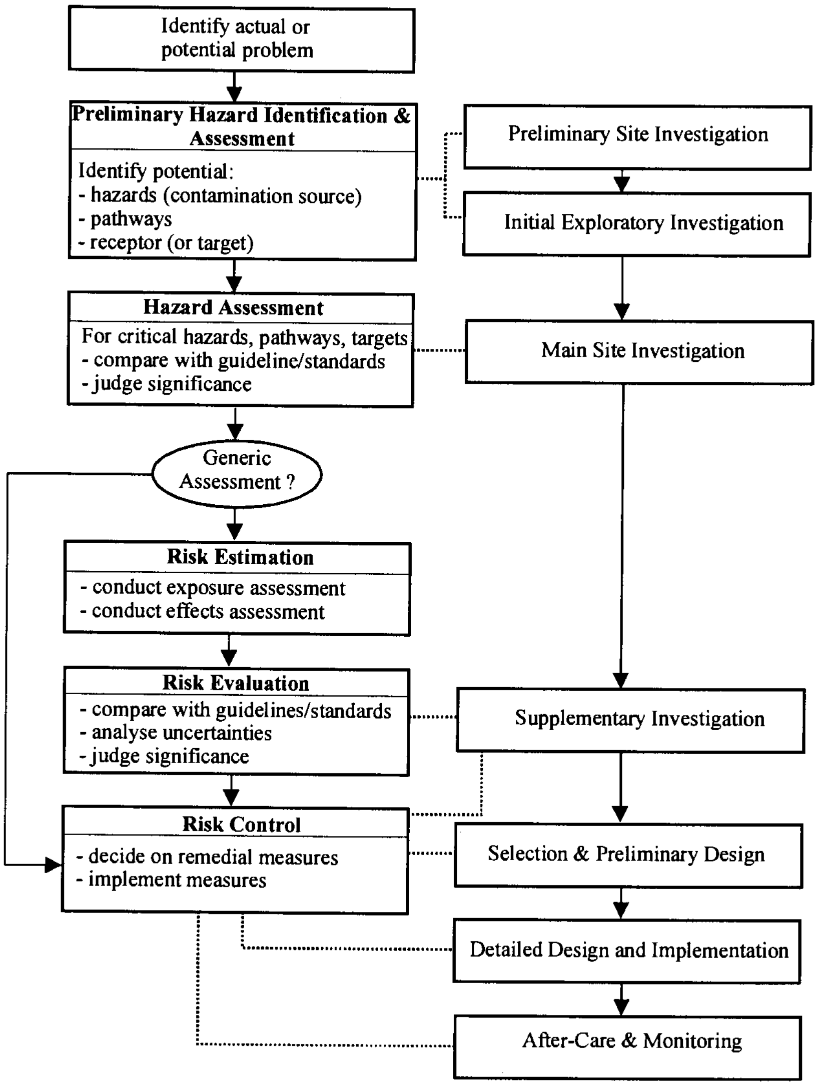

FIG. 6.1- Scheme for a risk-based site assessment (after Griffiths & Smiths, 1998)

Subsequently, a total site risk is calculated, whereby uncertainties have to be taken into account in order to prioritise decisions on the most efficient clean-up or rehabilitation measures aiming at risk reduction to acceptable levels. The latter is achieved by the application of *Hazard Ranking Systems* and a *Site Ranking Methodology*, which were introduced by the EPA (1997a). A detailed discussion of the application of risk assessment for contaminated sites is given in EPA (1997b) and Skivington (1997).

6.3 REMEDIATION APPROACHES FOR CONTAMINATED SOILS

According to Pierzynski et al. (1994), two general strategies are used to deal with soils, which are mainly contaminated by trace elements:

- Treatment technologies;
- On-site management.

Treatment technologies refer to soil that has been physically removed (*ex situ*) and processed in a certain way in an attempt to reduce the concentrations of trace elements or to reduce the extractable (bio-available) trace element concentration to an acceptable level. The TCLP (toxic characteristic leaching procedure) is a protocol used by the EPA, which dictates that materials should be leached under standard conditions. If the concentration of various substances exceeds some critical levels in the leachate, the material is classified as hazardous.

The second strategy is called on-site management (*in situ*). There are two subcategories within the on-site management option:

- Isolation;
- Reduction of bio-availability.

Isolation is one of a number of processes by which a volume of soil is solidified, resulting in prevention of any further interaction with the environment. The second subcategory consists of methods for reducing the bio-availability of trace elements in the soil. The following paragraphs will provide a brief introduction of both rehabilitation strategies and their application potential for this type of contamination.

6.3.1 Treatment technologies

Soil clean-up methods make use of the specific differences between the properties of contaminants and soil particles. Soil contamination characteristics at which clean-up may be directed are volatility of the contaminants, solubility in aqueous solution, adsorption and remobilization characteristics, size, density, shape of contaminated particles, bio-degradability and geochemical instability. The following aspects are of importance for the application of a clean-up technique:

- Soil type (properties of the inorganic and organic soil phases);
- Type and concentration levels of contaminants;
- Physical state of the contaminants (e.g. particulate pollutant, adsorbed, absorbed, liquid films around soil particles, contaminant as a liquid or solid phase in soil pores);
- Migration mechanisms of contaminants and the time interval between contamination and clean-up. Particularly in the case of in-situ treatment, it is important to know if the contaminated site is disturbed by mechanical processes or not.

Clean-up possibilities depend on the type and concentration of contaminants, which can vary significantly in the soil. Contamination caused by seepage leaving gold mine tailings dams and entering the subsurface mainly consists of:

- Acidity;
- Salts (e.g. SO_4^{2-} and Cl^-);
- Trace elements (e.g. heavy metals and heavy metal compounds and radionuclides);
- Cyanides (free and complex cyanides).

Soils contaminated with heavy metals or heavy metal compounds are in general most resistant to clean-up, because metals and metal compounds cannot be destroyed, with the exception of volatile elements such as As and Hg (Rulkens, Grotenhuis & Tichy, 1995). However, the volatilisation of As and Hg contaminants will only succeed at extremely high temperatures. In addition, heavy metals are usually found in soils accompanied by other types of contamination (e.g. organic compounds). The occurrence of organic substances can make the removal of metals from the soil substantially more complicated.

Five main principles are applied for the clean-up or decontamination of affected soils. These principles are discussed in detail by Rulkens et al. (1995) and listed below:

- Removal of contaminants by molecular separation (e.g. treatment by extraction and treatment by desorption or remobilization);
- Removal of particulate contaminants by phase separation (e.g. classification with hydrocyclones, froth flotation and jig techniques);
- Removal of contaminants by chemical/thermal destruction;
- Removal of contaminants by biodegradation (e.g. land farming and biological slurry reactors, not applicable to heavy metals);
- Removal of contaminants by biological adsorption or biological mobilisation.

A large number of clean-up techniques have been developed on the basis of these principles. However, only a few approaches are presently successfully applied in practice (Rulkens et al., 1995).

In general, mining sites in South Africa are far too large to be cleaned up using the available technology at reasonable cost. Approximately 13 km^2 of land has been affected by gold mine tailings material, which have been reclaimed. If only the top 30 cm (i.e. topsoil) of these areas were to be treated, this would imply that 3.9 million m^3 and hence, at least 5.5 million tons of material would have to be treated. This is a very conservative estimate, since treating the topsoil would not be sufficient - some contaminants have already reached the groundwater system, indicating that contamination of the subsoil or deeper parts of the unsaturated zone has been occurred.

Areas affected by wind-blown tailings or contaminated sediments in waterways downstream of these deposits have not been considered in this example. Even with an effective treatment technology available, it would be cost prohibitive to treat such large quantities of material necessary to address the problem. Therefore, present-day treatment technologies are confined to rehabilitation scenarios where only small volumes of soil are involved.

6.3.2 On-site management

The isolation approach of the on-site management includes *in situ* approaches described under the treatment technologies option. All isolation approaches aim to isolate the contaminants from the surrounding environment by encapsulating them into a nonporous matrix.

Of major interest in the context of rehabilitation of land affected by mine tailings are the methods to lower the mobility and hence, to reduce the bio-availability of trace elements. These methods include the following aspects relevant for soils affected by acid mine drainage:

- Altering soil pH;
- Increase sorption capacity;
- Precipitation of trace elements as some insoluble phase.

The influence of soil pH, cation exchange capacity, and adsorption mechanisms on trace element bio-availability are well studied and reported in soil literature (e.g. in Alloway, 1995), although generally not in association with a remediation technique.

Of all the methods for reducing trace element bio-availability, increasing the soil pH by adding lime (generally to a pH of ≥ 6.5) is probably the most common approach applied. This is a result of the general tendency for most trace elements to precipitate as hydroxides at a pH > 6.5 and of the fact that soil pH management is a routine measure of a fertility program. However, where more than one trace element is involved in the remediation (common situation), changing the soil pH may reduce the mobility of some elements whilst mobilising others such as Mo (Pierzynski et al., 1994). Fenton (1997) reported that the following quantities of lime are required in order to achieve a pH increase of 0.5:

- 2.5 tons/ha for a clay;
- 1.7 tons/ha for a silty clay loam;
- 1.5 tons/ha for a sandy clay loam;
- 1.0 tons/ha for a sandy loam.

In the study area, the majority of samples from the upper soil units had a strongly acid character (pH 3-4), whereas samples from deeper depths showed generally higher pH-values (pH 5-7). Thus, approximately 10 tons/ha lime would be required to neutralise a silty clay loam from pH 4 to pH 7. Additionally, more lime is required to maintain a certain level of soil quality. Quantities depend on climatic conditions and range from 75-1000 kg/ha/year (Fenton, 1997). However, these lime quantities are only estimations since the quantities for a specific site have to be determined on the base of the dominating clay mineralogy in the soil, and more importantly the reactivity of the liming material itself. An extensive discussion on this topic is also found in Logan (1992) and Whitney & Lamond (1999).

Bio-availability of some trace elements to plants is mainly influenced by the soil pH and cation exchange capacity, with availability decreasing at neutral to alkaline pH values and as the exchange capacity increases. It should be noted that the soil pH does not predict the amount of lime necessary to neutralise acid soil. That requires the determination of the soil reserve acidity, which is a function of the soil pH as well as the cation exchange capacity. To measure reserve acidity, the soils cations such as Ca must be extracted using a buffer solution, where the adsorbed cations exchange with cations in the buffer solution. Subsequently, the cations are filtered and the quantity of cations, which can neutralise acids (mainly Ca and to some extent Mg) determines the amount of lime required to neutralise acidity. Thus, a soil with a high cation exchange capacity will require more lime than one with a low exchange capacity.

An increase in soil cation exchange capacity in order to improve the soil retention capacity for contaminants, can be achieved for instance by:

- Adding clays with a high cation exchange capacity (e.g. montmorillonite);
- Adding organic material (e.g. manures, sludges).

Other methods are aimed to reduce the bio-availability of trace elements such as:

- Adding large amounts of Fe and Al salts (increasing adsorption capacity for oxy-anions with a subsequent reduction in their bio-availability).
- Adding hydroxides, carbonate or phosphate-containing salts can cause precipitation of the corresponding trace element-containing solid phase. If the solid phase then controls the activity of the trace element in the soil solution and this activity is lower than the initial level, the bio-availability will be reduced.

- Mixing the contaminated soil with uncontaminated material or materials such as coal fly ash, paper mill wastes, sewage sludge in order to dilute existing pollution levels (attenuation) in the contaminated soil.

Sutton & Dick (1984) discuss many of these methods in detail with respect to soil treatment. Another aspect is *phytotoxicity*, which can protect the human food chain. This phenomenon is called the *soil-plant barrier* and refers to the situation where a plant reacts phytotoxically to a trace element concentration below that which would be harmful if humans were to consume the plant as food.

Certain elements might exert a lower risk to humans because of phytotoxic reactions to plants and were discussed in paragraph 2.7. However, some elements, such as Cd, Mn, Mo and Zn, are not affected by this phenomenon, as a result of insolubility or strong retention of the element in the soil that prevent plant uptake (Pierzynski et al., 1994).

Another mechanism is the low mobility in non-edible portions of plants that prevent movement into edible portions (e.g. roots versus above ground portions), or phytotoxicity that occurs in concentrations in the edible portions of plants below a level at which they would be harmful to animals or humans. Detailed information about the effects of heavy metal pollution on plants is given in Hutchinson (1981). It is important to note that direct ingestion of contaminated soil or dust (e.g. mine tailings) bypasses the soil-plant barrier and thus poses a direct risk to human health.

6.3.2.1 Vegetation cover for reclaimed sites

A primary objective for the satisfactory rehabilitation of land affected by mine tailings is to establish a permanent self-sustaining vegetation cover (Sutton & Dick, 1984). This may have a beneficial effect, since it may reduce the amount of leachates entering the subsurface.

However, the establishment of vegetation (recultivation) on land affected by mine tailings is often hindered due to the low availability of plant nutrients and soil moisture. Another primary factor is the low pH in soils (caused by acid mine drainage and a lack of buffer minerals) which prevents the establishment of vegetation as a result of leached nutrients. In addition, incomplete reclamation often results in tailings material remaining on the surface. This residual tailings material provides an additional reservoir for acid mine drainage generation and associated contaminants and complicates rehabilitation efforts even more.

Although the acid and soluble salt amounts will decrease with time due to weathering and leaching processes, the underlying soil might remain too acid for plant growth. As a result, most of the areas covered by tailings dams which were reclaimed will remain without an appropriate vegetation cover for an extended period of time (also as a result of highly soil compaction), if exposed to weathering.

Treatment options were discussed in the previous paragraphs and amelioration could be achieved by addition of soil amendments such as lime or coal fly ash. Once the abandoned mined land shows vegetation growing on the surface, the initial regeneration of these areas towards future land development has begun. In addition, a vegetation cover on abandoned mined lands improves the aesthetics of the area. The land use capability, location, and objectives of the owner will determine the ultimate use of these areas. This would also include ecological aspects in respect of agriculture, forestry, wildlife, and recreation (Sutton & Dick, 1984).

A further option is to cover remaining tailings with soil. Such a cover or cap would prevent leachate migration and reduce the need for other cost intensive techniques such as soil clean-up or groundwater treatment. A detailed discussion is given in Daniel (1983), Hutchison & Ellison (1992) and Johnson, Cooke & Stevenson (1998).

6.4 REMEDIATION OF GROUNDWATER CONTAMINATED BY ACID MINE DRAINAGE

Most of the remediation techniques used in practice are related to organic pollution such as petroleum from leaking underground tanks. However, limitations to remediation of contaminated groundwater became apparent in the mid 1980s as data from groundwater remediation projects in the USA became available (EPA, 1989 and 1996a; Mackay & Cherry, 1989; Travis & Doty, 1990 and Kavanaugh, 1996). The most common groundwater remediation strategy in the USA has been the pump-and-treat approach (P&T technology), where contaminated groundwater is pumped to the surface, treated and returned to the aquifer. Because of growing concerns in the USA that this approach was not likely to achieve target levels in many cases, and that predictions of clean-up times had been seriously underestimated, an independent assessment of the issues was conducted by the U.S. National Research Council in 1994. A number of 77 remediated sites were investigated in the U.S. with regard to their clean-up success. The survey revealed that only 8 of the 77 sites reached the remediation clean-up level and in most cases the concentration of the target compounds in the extracted water had reached a constant level.

The low success of P&T technologies is not surprising, because even in the case of an optimal design of the P&T approach, restoration of groundwater is limited by four factors which are inherent to the problem of removing contaminants from the subsurface (Kavanaugh, 1996). These factors are:

- Compounds strongly adsorbed to aquifer solids (Mackay & Cherry, 1989);
- Highly heterogeneous subsurface environments contain zones of low permeability (e.g. clay);
- Slow mass transfer of contaminants from aquifer solids to the bulk interstitial fluid (Brusseau & Rao, 1989);
- Wide spread presence of non-aqueous phase liquids (NAPL's), particularly those that are more dense than water (Mecer & Cohen, 1990). This factor does not account for inorganic trace element pollution.

Alternative remediation techniques such as semi-reactive walls and bio-remediation approaches are not applicable to groundwater, if heavily affected by salt and heavy metal contamination. Thus, remediation efforts should focus on control of the pollution source (e.g. vegetation cover, drainage systems) and, if contamination in the subsurface occurs, on limiting the bio-availability of contaminants within the unsaturated zone. Various clean-up technologies have been reviewed by the EPA (1987a). The long-term management of the contaminated subsurface will be discussed in the following paragraphs.

6.5 LONG-TERM ENVIRONMENTAL MANAGEMENT FOR LARGE CONTAMINATED AREAS

Areas such as those covered by gold mine tailings dams, are too large to be cleaned up economically. However, since the unsaturated zone (vadose zone) underneath the mine tailings is expected to be contaminated for a long time, it is necessary to understand the mobility of contaminants and the capacity of the unsaturated zone to retain contaminants in the long term. In a number of cases, contaminants have already migrated into the groundwater system, thus causing a deterioration in groundwater quality.

The parameters, which control the balance between retention and mobility of contaminants in soils and sediments are also termed *master variables* (i.e. pH, but also redox conditions and the presence of complexing agents such as dissolved organic matter and inorganic anions) according to Salomons & Stigliani (1995).

For a *short-term risk assessment* (time period of 5-10 years) it is sufficient to understand how these master variables are associated with mobility and hence, bio-availability of contaminants. A great deal of information is available in the literature on this subject (Salomons & Stigliani, 1995).

However, there is less information available which deals with the mechanisms which determine the master variables. This is not important for short-term processes, which determine the current contamination status of soils and sediments and their immediate impact on the environment. Salomons & Stigliani (1995) found that in a number of cases the present impact may be slight; however, this may increase if the retention capacity of soils for contaminants changes or when the master variables controlling the interaction between the soil and the soil solution change. This could be a result of the consumption of minerals which provide acid neutralisation capacity. These changes are of a long-term nature and are caused by the dynamic geochemical behaviour of the master variables and the major element cycles in the soil-sediment system. Figure 6.2 illustrates the relationship between the master variables, the major element cycles and contaminants.

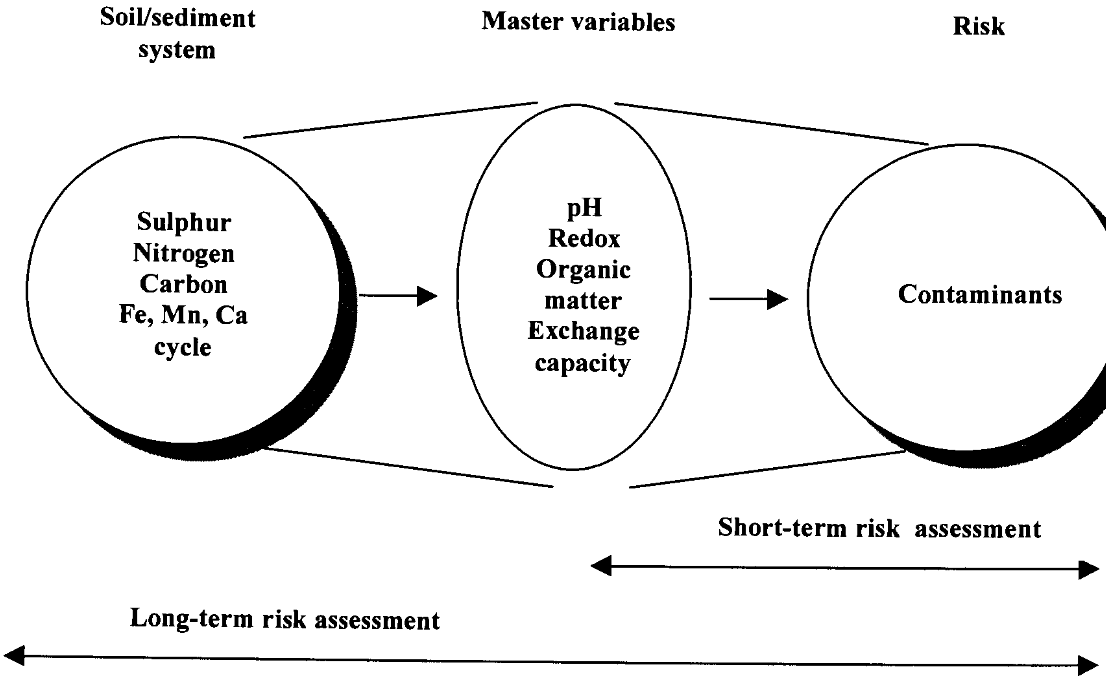

FIG. 6.2 - Association between the master variables, the major element cycles and contaminants (modified after Salomons & Stigliani, 1995).

It is important to understand that these changes in contaminant concentrations in the soil solution show a non-linear relationship, in particular for inorganic pollutants (such as heavy metals). Changes in the pH or Eh conditions can cause sharp increases in concentrations over a short time period (Salomons & Stigliani, 1995). This could be a result of changing land use (e.g. deposition and reclamation of tailings dams), continued acid deposition and changes in hydrology.

Although the previous discussion has focused on the chemical properties and behaviour of contaminants in the soil, it is important to realise that other disciplines must be taken into account for a complete understanding of this complex system and in order to be able to perform predictive long-term modelling. Hence, it is important to assess the significance of increased mobility on transport, plant uptake and impact on the soil ecosystem as part of a risk assessment. Integration of these aspects would allow one to establish eco-toxicological guidelines, sustainable agriculture, changing land-use and long-term protection of groundwater resources for certain target areas such as those affected by mining operations.

6.6 ENVIRONMENTAL MONITORING AND AFTER-CARE MANAGEMENT

How effectively contamination can be mitigated and how sustainable the rehabilitation effort is under long-term conditions determine the success of rehabilitation (Van der Nest & Van Deventer, 1996). Consequently, the only available tool to measure the success of rehabilitation at a specific site is the monitoring approach. Only monitoring reveals if improvement occurs as a direct result of the rehabilitation measure. Monitoring would also justify the use of a specific remediation method for further applications under similar conditions. Therefore, monitoring serves as a quality control tool for rehabilitation management and thus, forms an integral part of a risk assessment and after-care measures for previously contaminated and subsequently rehabilitated sites.

The type and extent of monitoring, however, would depend on the site-specific conditions and could comprise the monitoring of the vadose zone, surface and/or groundwater systems. The latter monitoring technique could for example consist of the establishment of boreholes suitable for groundwater sampling up and down-gradient of the site. Groundwater monitoring approaches are discussed in detail in textbooks such as Palmer (1992), Daniel (1993), and by Mulvey (1998), whereas vadose zone monitoring techniques are described in Everett, Wilson & Hoylman (1984).

In addition, *The Minimum Requirements for Water Monitoring at Waste Management Facilities* (DWAF, 1998) give guidance on this issue. In this context, it should be emphasised that this guideline states:

> *"It is a minimum requirement that a risk assessment, to determine the risk of water becoming polluted, be performed at all waste sites before the installation of a monitoring system. The risk assessment ensures that the design of the monitoring system is adequate..."*

Finally, the use of remote sensing techniques (e.g. using satellite images) could provide an important tool for the monitoring of reclamation activities of mining companies as well as of the nearby environment (rapid developing residential areas such as townships and illegal land use) of tailings dams.

6.7 FINANCIAL IMPLICATIONS OF REMEDIAL MEASURES

Acid mine drainage is recognised as a global pollution problem. At the 1998 Environmental Workshop of the Minerals Council in Australia 17 international companies, representing about 40 per cent of the world's mining activity, have agreed to join forces to control acid mine drainage. It is assumed that rehabilitation of acid mine drainage-related environmental damages will cost an estimated US $ 550 million in Australia and US $ 35 billion in North America (Dorfling, 1998). The cost figures for South Africa to rehabilitate existing mining facilities and to mitigate such damages are currently unknown. Table 6.1 provides a selection of cost ranges for some rehabilitation approaches.

TABLE 6.1 - Summary of selected rehabilitation measures and associated costs (after Wood, 1998).

Rehabilitation measure	**Estimated cost range**
Surface amendment	US $ 15-65/ton
Vertical slurry wall: shallow	US $ 45-95/m^2
Vertical slurry wall: deep	US $ 95-190/m^2
Excavation and disposal off-site	US $ 10-80/ton
Cover system	US $ 32-50/m^2
Containment	US $ 16-80/ton
Soil washing	US $ 80-400/ton
Physico-chemical washing	US $ 80-275/ton
Ex-situ stabilisation/solidification: inorganic	US $ 32-56/ton
Solidification: lime based	US $ 32-65/ton

Clean-up costs for contaminated soils (i.e. soil washing) range between US $ 80–400/ton (Wood, 1998). This study has shown that at least 5.5 million tons of material would have to be treated in South Africa, if only the contaminated topsoil (< 30 cm) underneath the reclaimed sites would have to be considered. Consequently, only the topsoil clean-up would cost at least US $ 440 million, assuming the lowest treatment cost scenario of US $ 80/ton. In addition, the following costs can be expected and would add to this cost scenario:

- Risk assessments for each site or certain impact areas (including radiological risks);
- Treatment of contaminated soil material underneath the topsoil unit (i.e. subsoil) and/or at higher soil clean-up costs (> 80 US $/ton)
- Groundwater remediation;
- Removal and treatment of contaminated sediments in waterways;

- Remediation of existing gold mine tailings dams (e.g. cover systems) to prevent wind erosion and, thus mitigate the generation of acid mine drainage;
- Revegetation of reclaimed sites.

It is obvious that these rehabilitation costs cannot be afforded either by the South African government or by the mining industry. It is also questionable if the predicted costs for Australia and North America will ever be spent. Thus, rehabilitation (including treatment of contaminated soils and groundwater) of large-scale polluted sites is uneconomical and this should only be applied at highly contaminated sites or areas determined by a risk assessment as high risk areas.

6.8 ENVIRONMENTAL MANAGEMENT MEASURES REQUIRED FOR THE INVESTIGATED SITES

One of the findings of this research study is the recommendation of site-specific environmental measures for the study sites A-K (field testing only at sites A-G). It must be noted that this study serves as a hazard assessment or Phase 1 of a risk assessment and, thus recommendations are of preliminary nature. Table 6.2 summarises the environmental management measures, which are suggested for the investigated sites.

It was found that paddock systems are inappropriate in pollution prevention and should be removed from the reclaimed sites. In addition, residual tailings, which remain in excessive volumes on site, should be covered with an impermeable layer (assuming that the cover option is associated with lower costs compared to the removal), in order to reduce wind-erosion and infiltration of rainwater into the tailings dam.

TABLE 6.2 - Summary of environmental management measures recommended for the study sites A-K.

Site	Monitoring		Soil management	Site Summary
	Vadose Zone	Groundwater		
A	O	No	R	Topsoil pH 3-4, metal contents above background, site is not entirely reclaimed.
B	R	R	R	Topsoil pH about 3-4, metal contents such as Co and Ni above background, site is partially situated on dolomite.
C	R	O	O	Topsoil pH about 3-4, acidity tends to migrate into the subsoil, metal contents of above background. The site has a vegetation cover.
D	R	O	O	Topsoil pH 3-4, metal contents such as Co above background. The site has a vegetation cover.
E	O	O	No	Topsoil pH is fairly neutral, metal contents of Ni and Co above background, but with low mobility, remove residual tailings material; site is partially situated on dolomite.
F	R	R	R	Topsoil pH 3-5, metal contents such as Co and Ni above background, very high U content, removal of residual tailings is ongoing.
G	No	No	No	Topsoil pH varies from 4 to neutral, metal contents such as Co above background, remove paddocks.
H	O	R	O	Rehabilitation plan is currently developed; cover residual tailings; site is partially situated on dolomite
I	R	R	R	Groundwater risk assessment including monitoring is ongoing; site is partially situated on dolomite; this site is the largest gold mining tailings dam in South Africa
J	R	No	O	Remove residual tailings material
K	R	R	O	Groundwater investigations are ongoing; site is partially situated on dolomite; geochemical soil data lacking

Note: Soil management comprises liming of those portions of the topsoil identified as acid soils and the addition of fertilisers and organic material. In cases of excessively polluted soils excavation should be considered. R means "required", O means "optional" or future requirement, No means "no measures necessary". Grey shaded are those sites, where no field and laboratory tests during this study were conducted.

It is apparent that most of the sites require monitoring of the vadose or unsaturated zone, as this zone represents the physical and geochemical barrier zone between the source (i.e.

contaminated site) and the receptor (i.e. groundwater system) and, thus is of major significance for the assessment of contamination. Additionally, sites with a perched water table or dolomitic aquifer should be monitored with respect to groundwater quality. A risk assessment is only recommended if significant contamination of soil and/or groundwater is indicated. It should be mentioned that for some of the investigated sites, environmental management measures such as removal of residual tailings, soil analyses, liming of acid soils, groundwater monitoring and numerical groundwater modelling were already undertaken by the operating mining company, although such measures were previously not mandatory.

The EPA (1997b), other EPA sources and Skivington (1997) provide additional information on risk assessment and their practical implementation.

7 DISCUSSION AND CONCLUSIONS

7.1 DISCUSSION

Acid mine drainage is recognised as a global pollution problem and is related to the generation of large volumes of mine waste such as the tailings produced as a result of intensive mining activities for gold, coal and other mineral resources in South Africa. To date, more than 270 tailings dams have been constructed to store these fine-grained tailings. Most of the tailings dams are situated south of Johannesburg within the highly populated Gauteng Province (currently 7.7 million people increasing to estimated 8.5 million in the year 2000). Up to 1998 a total of 70 tailings dams were reclaimed throughout the Johannesburg region in order to extract the gold, still present in economically viable concentrations (currently approximately 0.40 g Au/ton). Once the tailings material has been completely reclaimed, the land has a certain potential for development. However, it is important to realise that the reclaimed tailings material leaves a contaminated subsurface below an area, also known as the footprint.

It is known that gold mine tailings, originating from sulphide-bearing ore, are prone to the generation of acid mine drainage. It is estimated that the remediation of environmental damages related to acid mine drainage will cost about US $ 500 million in Australia and US $ 35 billion in the United States (regulated under the *Superfund* program, see footnote 1) and Canada. The cost figure for South Africa to rehabilitate existing tailings dams and to mitigate damages in the unsaturated and saturated zone is currently unknown. Clean-up costs for contaminated soil material (e.g. soil washing) range from US $ 80-400/ton. This study has shown that at least 5.5 million tons of material would have to be treated in South Africa, if only the polluted topsoil (< 30 cm) underneath the reclaimed sites would have to be considered. Hence, only the topsoil treatment would cost at least US $ 440 million, assuming the lowest treatment cost scenario of US $ 80/ton. Additional rehabilitation measures such as cover systems for present mine-residue deposits, recultivation of reclaimed land or groundwater remediation were not taken into account for this cost scenario. It is obvious that these rehabilitation costs cannot be afforded either by the South African government or by the mining industry. It is also questionable if the predicted cost figures for Australia and North America will ever be spent, in order to rehabilitate such sites. Thus, rehabilitation (including treatment of soils and groundwater) of large-scale polluted sites is uneconomical and this should only be applied at highly contaminated sites or areas characterised by a risk assessment as high risk areas. It is important to realise that the understanding of the short- and long term behaviour of contaminants in the subsurface zone affected by such mining operations, forms an integral part of a risk assessment.

Ten selected reclaimed gold mine tailings dam sites, situated in the Gauteng Province and one in the North-West Province of South Africa, were investigated in this study. All reclaimed sites were analysed in terms of their short-term impact and conservative predictions were also attempted to assess the long-term impact (worst-case scenario). In addition, the contamination source (i.e. tailings dam) was geochemically and mineralogically characterised. Field and laboratory tests were conducted on samples taken from seven reclaimed selected sites within the unsaturated zone and from a shallow groundwater table. The database was supplemented by further soil and groundwater data of the investigated sites, obtained from mining companies, various government departments and associated institutions. Rating and index systems were applied to assess the level of contamination contained in the unsaturated zone underneath reclaimed gold mine tailings dams.

7.2 CONCLUSIONS

In conclusion, this study has shown that contamination occurs in the subsurface below the footprints of former gold-mine tailings deposits. However, based on the findings of this study, it is premature to quantify this impact as the results have to be incorporated into a risk assessment approach for some of the sites. This investigation therefore provides a first step towards a risk assessment and serves as a hazard assessment/identification.

It is important to recognise that slight changes in the pH or Eh conditions (e.g. by changes in land use, climate, hydrology) of soils, affected by acid mine drainage can cause mobilisation of large amounts of contaminants, which are characterised by a geochemical behaviour that is time-delayed and non-linear. Additional field and laboratory testing would be obligatory for an in-depth understanding of the long-term dynamic aspects of these contaminant processes, which pose a serious threat to the vulnerable dolomitic groundwater resources and land development. Salomons & Stigliani (1995) described these processes as "*... precisely the kind of response that catches policymakers, the public, and even scientists by surprise*".

The main findings of this investigation are summarised as follows:

1. Groundwater quality beneath and in close vicinity to the investigated tailings dams is dominated by the Ca-Mg-SO_4 type, indicating acidic seepage, although all sites with relevant groundwater data (sites H, I and K) are underlain by dolomitic rocks. In addition, high TDS (up to 8000 mg/l) values occur mainly as a result of high salt loads (SO_4^{2-} and Cl^-) and are often associated with elevated trace element (e.g. As, Fe, Mn, Ni, Pb and Zn) concentrations, exceeding target water quality standards in some boreholes.
2. The low pH (range between 3-4) in the topsoil enhances leaching of soil nutrients, essential for plant growth and only acid tolerant plant species can survive under such conditions. The acid conditions are a result of acid mine drainage released from the tailings and a $CaCO_3$ deficiency in the soil. When the pH drops around 4.5 in the topsoil, the acidity starts to migrate into the subsoil and indicates a permanent soil degradation.
3. An estimation of seepage quantities and SO_4^{2-} loads for all reclaimed sites (total area of 13 km^2) has shown that an accumulated amount of approximately 1000 Ml seepage is annually discharged into the groundwater system, containing a SO_4^{2-} load between 1000–5000 tons based on conservative recharge and concentration figures.
4. Elevated trace element concentrations in the soils affected by acid mine drainage and the high mobility of phytotoxic elements such as Co and Ni could hinder the establishment of a vegetation cover.
5. Generally, the trace elements are most mobile when the soil pH < 4.5 (prevailing in the upper soil units), and least mobile when a soil pH > 6 (prevailing in the deeper soil units). Cobalt, Ni and Zn are the most mobile trace elements for the selected reclaimed site and exceed soil threshold values. Chromium, Cu, Fe, Pb and U are less mobile compared to the above elements, indicating that a significant portion of those is likely to be contained in the residual fraction (i. e. bound to silicates) of the solid phase.
6. The application of the geochemical load index for the assessment of the long-term impact at seven sites classified three sites as moderately to highly contaminated, three sites as highly contaminated and one site as excessively contaminated.
7. Soil conditions indicating preferential flow (bypass of the soil matrix) were observed in nearly all soil profiles and even sites with relatively impermeable soils showed contamination in the underlying groundwater system.

8. The extractable concentrations of Co, Cu, Ni and Zn found in gold-mine tailings samples exceed threshold concentrations. This confirms that gold mine tailings are a source of trace element contamination for soils and groundwater.
9. Leaching tests and a load calculation have shown that even reclaimed sites (i.e. footprint) continue to release significant salt contents contained in seepage for an extended time period (> 10 years). Incomplete reclamation of tailings would result in tailings material remaining on the surface. Such material provides an additional reservoir for acid generating processes and ongoing contaminant release.
10. The clean-up costs of environmental damages, caused by acid mine drainage from mine waste disposal sites are expected to be excessively high. The most common rehabilitation measure applied in South Africa is liming of acid soils and additional lime is also needed to maintain a certain level of soil quality after the initial treatment. Fertilisers are required to improve growth conditions for plants. However, it can be expected that even after site rehabilitation has occurred, the subsoil and groundwater might remain polluted for an extended period of time.