INVESTIGATION INTO
GROUNDWATER QUALITY DETERIORATION IN THE OLIFANTS RIVER CATCHMENT ABOVE THE LOSKOP DAM WITH SPECIALISED INVESTIGATION IN THE WITBANK DAM SUB-CATCHMENT

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REGIONAL INVESTIGATION INTO
GROUNDWATER QUALITY DETERIORATION IN THE OLIFANTS RIVER CATCHMENT ABOVE THE LOSKOP DAM WITH SPECIALISED INVESTIGATION IN THE WITBANK DAM SUB-CATCHMENT

The Steering Committee responsible for this project, consisted of the following persons:

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EXECUTIVE SUMMARY

1 INTRODUCTION AND SCOPE OF INVESTIGATIONS

1.1 PARTNERS IN RESEARCH

The Institute for Groundwater Studies at the University of the Orange Free State had entered into a research contract with the Water Research Commission, the Department of Water Affairs and Forestry (DWA&F), the Chamber of Mines of South Africa (excluding Amcoal) and Eskom in 1989, to study the "groundwater quality deterioration in the Olifants Catchment *above the Loskop Dam, with specialised investigations in the Witbank Dam* Sub-catchment". This contract was preceded by a year of negotiations with all interested parties to ensure full co-operation. Most major companies in the Olifants Catchment agreed to co-operate. A condition for co-operation was confidentiality of information on a site-specific basis.

1.2 AIMS

The aims of the investigation were to:

- Quantify the contribution of various activities which may result in a deterioration of the groundwater resources in the catchment above the Loskop Dam, with special emphasis on the Witbank Dam Sub-catchment.
- Predict future salt loads in groundwater, based on projections of probable development in the area. Extrapolate information to other catchments that supply water to the Olifants Catchment.
- Investigate and research improved management and precautionary measures that could be utilised to minimise groundwater quality deterioration.
- Integrate groundwater information with other investigations in the area, including the water management programme of the DWA&F, with the purpose of deriving a catchment management programme at the end of this project.

Since 1990 many investigations have been initiated on specific issues relating to water within the Olifants Catchment. The most important of these investigations were perhaps those which have been initiated by the DWA&F, to establish a surface water quality management programme for the Olifants Catchment. Other studies worthwhile mentioning are the formulation of the Environmental Management Programme Report (EMPR), which has come about during this investigation; research into passive water treatment systems; interception of rainfall thus minimising influx into mined areas; research into liners for waste disposal; research into the use and application of bactericides to limit oxidation of pyrite; promulgation of the Environmental Act, Act 73 of 1989, and the associated licensing of general and hazardous waste sites and many projects initiated by the mines and industries themselves.

Because of the environmental awareness and the significant number of related projects which came about since the original formulation of the present project, the emphasis in this project has shifted to ensure that duplication will not result. The following interpretation is provided to clarify how the emphasis has shifted through the years:

- Quantification of contributions by various activities was to be on the micro- and local scale through monitoring, process studies and modelling at selected sites. Individual mines, industries or municipalities were not to be identified to ensure their continued co-operation.

- Prediction of future salt loads would emphasise the long-term issues tending towards steady state. Prediction will be done through monitoring, process studies, water and salt balances, chemical equilibrium calculations and modelling, thus obtaining a load factor, where possible, for each facility investigated. Prediction on a catchment basis was to be done, where possible, by applying a load factor to existing facilities within a catchment.
- Investigation and research into improved management and precautionary measures were to concentrate on the identification of main pollution sources, formulation of suggestions for improved management strategies, where necessary, implementation of these strategies, where possible, and modelling of the effect of the improved management strategies. The precautionary measures relate to all measures that may be undertaken to minimise an impact on the water resources, as pre- or post-treatment.
- The integration of groundwater information with other investigations in the area was to be done by the DWA&F, incorporating the information that is generated in this investigation, into their catchment management programme.

2 CURRENT WATER ENVIRONMENTS AS THEY RELATE TO THE OLIFANTS CATCHMENT

2.1 GROUNDWATER

Groundwater occurs in the near-surface weathered strata, fractures in the unweathered Karoo sediments and to a lesser extent, in the rock underneath the Karoo sediments (see Section 2.1 for more information). Almost all farmers tap the weathered and Karoo aquifers. Average borehole yield is less than 1 l/sec. Borehole yield is therefore sufficient for domestic use but not for large-scale irrigation. The natural groundwater quality in the weathered aquifer is excellent, because of its short residence time within this zone and also because most of the leachable minerals have been removed through years of leaching. Water quality within the fractured Karoo aquifer is also good, being mainly of the calcium/magnesium bicarbonate type with electrical conductivities in the range of 30 - 70 mS/m.

2.2 SURFACE WATER

Two sources of information were used in this report to demonstrate the degree to which water quality has deteriorated. These sources are the water quality database of the DWA&F, supplemented with macro- and micro constituent analyses for this investigation.

The study shows that a direct link exists between the water quality in many of the streams and development in the catchment. Sulphate levels in many of the streams have risen over the past 20 years from less than 20 mg/l to between 100 - 900 mg/l (Section 2.2).

Manganese levels, often in the range of 1 - 10 mg/l in streams, are telltale signs of acid rock drainage. Other heavy metals, particularly iron and aluminium, are also present in the acidic surface waters around Witbank.

Trace quantities of other constituents, specifically found in the urban, metal industry and agricultural sectors, are present in specific localities.

3 COAL-MINING AND ITS EFFECT ON GROUNDWATER QUALITY AND QUANTITY

3.1 OPENCAST MINING

Of all the activities in the Olifants Catchment, opencast mining has the greatest impact on groundwater quality. The following are the main research achievements and findings of this investigation with regard to opencast mining:

- Methodologies have been developed to quantify the acid-base potential of opencast coal-mines (Appendix A and B).
- These methodologies have been applied at ten opencast mines. Seven of these mines have severe acidic tendencies, two are borderline cases and one colliery is alkaline (Appendix C).
- Heavy metals that may become available for leaching under certain chemical conditions from these mines include iron, aluminium, manganese, copper, zinc and, in rare instances, nickel, cobalt and cadmium. Lead, arsenic, selenium, molybdenum and chromium are usually present only in trace quantities (Appendix C).
- The rate of sulphate generation in backfilled opencast areas, as calculated from this investigation, is between 5 - 10 kg/ha/d.
- On the basis of the present scale of opencast mining in the Witbank Catchment, this amounts to 70 t/d of sulphate. This amount is over and above that presently entering into the Witbank Dam on average from other sources, namely 33 t/d (Wates, Meiring and Barnard, 1993).
- Pits that have been mined out, fill up with water to their decant level within 5 - 10 years after mining has ceased (Section 3.3.1.2).
- Recharge of water to these pits has been calculated by water balance methods and verified by salt balances. The two methods concur that recharge averages 20% of the rainfall, of which 12% is run-off into the mine and 8% is infiltration through the spoil.
- Because of the significant dilution of the pit water by rainfall, saturation levels for salts have not been reached in any of the opencast mines. Sulphate levels are typically between 2000 - 3000 mg/*l* in backfilled opencast areas.
- The mineral most likely to precipitate upon saturation is gypsum. The availability of calcium is, however, restricted by the dolomitic composition of the carbonate minerals within the spoil. By means of chemical equilibrium modelling, it has been demonstrated that gypsum saturation will not be reached in opencast mines, unless the influx of rain water can be reduced to less than 3% of the annual rainfall.
- Once gypsum precipitation occurs, a cycle of chemical instability will be initiated, precipitating most of the calcium, while magnesium and sulphate concentrations could rise in excess of 2 000 mg/l and 10 000 mg/l respectively (Section 3.3.1.5). This situation can be avoided by dilution from rain water. However, if desalination is the preferred treatment option, then dilution through rain water should be minimised.
- Mixing cell modelling has indicated that the inclusion of coal discards into the unsaturated zone in opencast pits creates local acid conditions that will lead to acidification above and below water. Potentially alkaline pits will, under these circumstances, become acid. It is concluded that coal discard disposal constitutes one of the main problems because of the vast volumes of material to be dealt with. Disposal procedures for coal discards, on undisturbed ground, are recommended in the report.

- Three water management options at opencast coal mines stand out (Section 3.3.3.3). These are selective spoil handling, flushing under certain conditions and containment. Each of these options has its pros and cons and environmental impact assessments to select the best management option are recommended for opencast collieries.

3.2 UNDERGROUND MINING

3.2.1 Shallow underground mining *(Section 3.2.1)*

Shallow underground mining has historically been done west and south of Witbank. The following conclusions are drawn:

- Shallow mining has, in many instances, mined into the weathered aquifer, with the result that rain water actively recharges these mines.
- Collapsed structures, connecting the surface with underground workings, have occurred in many areas. Recharge of rain water is enhanced in these areas.
- Recharge has been found to range between 0 - 15% of rainfall events, depending mainly on the intensity of the event. In one instance, 45 000 m^3 of water was recharged to a shallow underground mine as a result of a 157 mm rainfall event.
- Most of the shallow underground mines west of Witbank are currently full of water and are decanting acid water onto surface. A liming plant exists at only one of the decanting mines.
- Other shallow underground mines exist where mining has recently ceased. In due time, these mines will also decant water onto surface, if precautionary measures are not taken.
- Very few precautionary measures exist to control or eliminate pollution from the shallow underground mines. Technologically, it is possible to install liming plants; desalination plants; use sewage sludge, power station fly ash, lime and wetlands for amelioration and prevent mine water from seeping onto surface by actively dewatering the mines. All of these options, except the disposal of fly ash into these mines cannot be viewed as permanent solutions and will require continued management in the future.

3.2.2 Deep underground mining *(Section 3.2.2)*

3.2.2.1 Bord-and-pillar mining

Deep bord-and-pillar mining does not pose a long-term threat to groundwater pollution. It also does not interfere with existing water sources for farmers.

3.2.2.2 High extraction mining

High extraction underground mining results in the collapse of overlying strata, usually with the following effects on the groundwater regime:

- Overlying strata and aquifers adjacent to these areas are dewatered.
- The extent of the dewatering cone is site dependent. As a rule, the effect of dewatering is not noticeable more than 500 m away from current high extraction mining in the Olifants Catchment. This limited extent of the dewatering cone is due to relatively shallow mining (< 50 m), low transmissivity (< 10 m^2) of the rock and recharge by rainfall.
- Water quality deterioration in high extraction areas is a two-phased phenomenon. Firstly, base exchange occurs as groundwater seeps towards areas of high extraction.

This results in sodium bicarbonate/chloride water that enters into high extraction areas. Within the mine, pyrite oxidation introduces sulphate, then releases calcium and magnesium from carbonates in the mine. Water in high extraction areas therefore has any of a range of chemistries, depending on the evolutionary stage of the water.

- All coal mines in South Africa where high extraction underground mining has been done, have had problems to dispose of the water. The quality of this water is inadequate for disposal into public streams. Disposal of this water into evaporation pans leads to further concentration of salt. The only viable disposal option lies in the creation of underground storage space. Several of the mines are currently implementing such underground water storage facilities. This is environmentally the best option since the flooded workings prevent further pyrite oxidation. If alkaline conditions are maintained, future careful planning of such underground storage facilities is necessary to ensure that a stratified water column, with the best water quality at the top, is obtained.

4 POWER GENERATION AND ITS EFFECT ON GROUNDWATER QUALITY AND QUANTITY

Possible sources of groundwater pollution at power stations are mainly that of fly ash disposal, coal stockpiling and dirty water dams. All Eskom's Power Stations in the Olifants Catchment have been equipped with groundwater monitoring facilities. Records at some of the stations date back to 1988. As part of the present investigation, additional monitoring, extensive leaching tests on fly ash, characterisation of in-pit fly ash disposal and solute transport modelling for fly ash dams have been done. The following are the main conclusions:

- The disposal facilities at the power stations are well managed and groundwater pollution is minimal.
- The ash water chemistry is unstable in the presence of air. It readjusts by a drop in pH (from > 12,0 to around 8,0) and precipitation of calcium carbonate.
- The heavy metal leachability from ash at normal pH-levels is insignificant (Section 4.3.2). The base potential of fly ash is usually 2 - 5 times higher than that of spoil in opencast mines. The introduction of fly ash into mining environments can therefore add additional base potential. A prerequisite for fly ash disposal within mining environments is that the system as a whole will remain alkaline.
- Under acid conditions (Section 4.3.2), the heavy metal leachability from fly ash is very high. Tables and graphs are presented in the report to demonstrate the heavy metal availability at pH-levels to below 3,0. It is concluded that great care should be exercised in instances where fly ash is to be introduced into coal mines, to ensure that alkaline systems will be maintained.
- Dry ash disposal is done only at one power station in the Olifants Catchment. A well-managed dry ash dump does not pose any threat to groundwater pollution. It is suggested that the pozzolanic action of the fly ash and water penetration characteristics at dry ash dumps should be evaluated (Section 4.3.2.2).
- Although no pollution could be detected at the modern coal stockyards at some power stations, this does not imply that pollution does not occur. Therefore, it is suggested that leachate monitors should be installed underneath all the coal stockpiles (Section 4.3.3).
- Saline effluents from demineralisation plants at power stations are disposed of in ashing systems. The introduction of sodium into the ash is a concern. In closed

systems, a build-up of sodium results. Research into alternative demineralisation techniques should receive high priority.

- Following the successful demonstration of mass transport modelling at fly ash dams, it is suggested that generic and expert groundwater models are developed for all power stations, to predict future trends (Section 4.4).

5 MUNICIPAL GENERAL WASTE AND ITS EFFECT ON GROUNDWATER QUALITY

The disposal of general waste in the Olifants Catchment is mostly done within environments that are already polluted by mining. It was therefore not possible to identify and separate general waste leachates from other pollution sources in the study area. The conclusion is drawn that although general waste may pose a threat to groundwater pollution, this threat is very localised. Through good planning, the regional impact of leachates from general waste sites will be minimal.

Very few of the general waste sites in the area have been licensed with the DWA&F. Many of the sites will have to be upgraded to comply with requirements.

6 SEWAGE EFFLUENT AND ITS EFFECT ON GROUNDWATER QUALITY

Sewage works do not significantly impact on groundwater quality in the Olifants Catchment. Treated sewage effluent, which is discharged into streams, has ameliorating effects on the acid-rock drainage. This is particularly visible in the area west of Witbank.

Even though the volumes of sewage sludge are insufficient for large-scale application, the possibility of using sewage sludge at an opencast mine to provide the necessary carbon source for sulphate reduction should be investigated.

7 METAL INDUSTRY AND ITS EFFECT ON GROUNDWATER QUALITY

Metal industries, comprising mainly steel, stainless steel, ferrosilicate and vanadium producing plants have little regional impact on groundwater quality. Locally, problems are experienced in the disposal of saline liquids, slimes, slags, phenol and in the case of the stainless steel industry, chromium and nickel.

Groundwater pollution is of a localised but intense nature. Many of these industries are in the process of installing monitoring systems and improving their waste disposal practices. Significant improvements to the waste handling facilities are necessary in the Ferrobank Area.

8 AGRICULTURE AND ITS EFFECT ON GROUNDWATER QUALITY

Agricultural pollution of groundwater is local and isolated. Nitrate (N) and phosphate (PO_4) levels in the groundwater are generally less than 2 mg/*l*.

Groundwater pollution at feed lots is localised and site-specific. Pesticide and herbicide levels are low in surface water and very low in groundwater.

It is recommended that regular monitoring be instigated to ensure that present favourable trends continue.

9 OVERALL ASSESSMENT

An evaluation of the main activities that may lead to groundwater pollution in the Olifants Catchment has been made during this investigation. Regional and local subjective risk ratings for these activities have been compiled. Important conclusions are as follows:

Regional groundwater pollution from point pollution sources in the Eastern Transvaal is not possible because of the layered nature of the aquifers and dynamic shallow throughflow in the upper aquifer.

Pollution that enters into the ground mainly affects the top, weathered aquifer. This pollution moves with the natural groundwater gradient towards streams, where it discharges.

It is therefore only in areas between the sources of pollution and the streams that groundwater pollution is present. This pollution ranges from severe for certain industries to negligible for others, based on the composition of the leachate.

10 RECOMMENDATIONS FOR FURTHER WORK

Many recommendations are made within the report for further work. In this summary, only the main issues are reiterated:

- The acid-base accounting methodologies that have been developed during this investigation should be refined, verified and published as a set of guide-lines, supported by case histories, to be used by the industry, consultants and the DWA&F alike.

- The water management options at current opencast mining operations are very limited. As part of acid-base accounting, the three options, namely selective spoil handling, flushing or containment should be evaluated and recommendations should be made for procedures to decide upon the merit of a specific management option.
- Coal discard disposal poses one of the long-term environmental threats. While certain proposals have been made in this report, based on hydraulic and chemical considerations, these suggestions will have to be evaluated under actual field conditions, followed by the development of a set of guide-lines.
- Control of groundwater pollution is well in hand at the power stations. Qualitative monitoring has been ongoing for many years. It is suggested that the qualitative data should be supported by quantitative modelling, generating generic and expert systems that can be used by the power stations for performance evaluation with respect to groundwater pollution control.
- Many other research issues may be included within this section, such as water treatment, water penetration characteristics, reactions and pathways within systems, amelioration of systems and rehabilitation practices. All of these issues are important in minimising groundwater pollution from waste sites.

11 AIMS VERSUS ACHIEVEMENTS

For the sake of comparing the aims of the investigation with achievements during this research programme, the aims and their interpretation are repeated in this section.

Aim 1: Quantify the contribution of various activities which may result in a deterioration of the groundwater resources in the catchment above the Loskop Dam, with special emphasis on the Witbank Dam Sub-catchment.

Interpretation: 'Quantification' was to be on the micro- and local scale through monitoring, process studies and modelling at selected sites. 'Various activities' refer to activities that could adversely contribute to groundwater deterioration. Within each of these activities, sites were identified and studied on a priority basis, based on the need for information and the severity of the problem. 'Deterioration' is defined as a change in groundwater chemistry that could impact on humans, industries and agriculture. Analyses, where appropriate to the industry, were carried out for selected macro- and micro constituents, some agricultural chemicals, both organic and inorganic, some sewage and some general waste chemical constituents. All extractable groundwater, irrespective of quality, was considered as a 'resource'. The terms 'catchment' and 'sub-catchment' were used in the geographic sense to demarcate the area in which investigations were to be performed. Individual mines, industries or municipalities were not to be identified to ensure their continued co-operation.

Response: A sufficient understanding of processes within the various activities in the Olifants Catchment has been obtained to enable conceptualisation, description, modelling and ranking of all issues that contribute to groundwater pollution. All relevant data have been entered into HydroCom.

Aim 2: Predict future salt loads in groundwater, based on projections of probable development in the area. Extrapolate information to other catchments that supply water to the Olifants Catchment.

Interpretation: Prediction (mainly with the aid of modelling) of future salt loads in groundwater would emphasise the long-term issues tending towards steady state. Verification is a long-term issue that should be done ongoing. Prediction will be done through monitoring, process studies, water and salt balances, chemical equilibrium calculations and modelling, thus obtaining a load factor, where possible, for each facility investigated. Extrapolation on a catchment basis was to be done, where possible, by applying a load factor to existing facilities within a catchment.

Response: Prediction of future salt loads has been done through chemical equilibrium, mixing cell and finite element mass transport modelling. These models are sufficient to describe chemical reactions and pollution plume modelling at any of the waste management facilities in the Olifants Catchment. As part of this modelling exercise, management options at opencast mines and fly ash dams have been evaluated. Superimposing this information onto the average annual run-off for the Witbank Catchment, extrapolation on a catchment basis was accomplished.

Aim 3: Investigate and research improved management and precautionary measures that could be utilised to minimise groundwater quality deterioration.

Interpretation: Investigation and research into improved management and precautionary measures were to concentrate on the identification of main pollution sources, formulation of suggestions for improved management strategies taking cognisance of economic considerations where necessary, implementation of these strategies where possible, and modelling of the effect of the improved management strategies.

Response: Very few improvement management options are available at existing facilities due to cost considerations. Where considered necessary, options that need further investigation have been identified on a site-specific basis.

Aim 4: Integrate groundwater information with other investigations in the area, including the water management programme of the DWA&F, with the purpose of deriving a catchment management programme at the end of this project.

Interpretation: The integration of groundwater information with other investigations in the area was to be done by the DWA&F, incorporating the information generated in this investigation into their catchment management programme.

Response: Approved data and information generated during this investigation are to be offered to the DWA&F for integration within their catchment management programme. This should be done on two levels. Bottom-line values, such as anticipated contribution of sulphate from opencast mining can easily be incorporated by considering entities such as point sources in the catchment management programme. Detailed information, most of which has been presented in summary form in the full report, may have to be re-evaluated, to provide a meaningful contribution for short-term localised predictions.

In view of the processed way in which data have been presented in the final document, it is anticipated that individuals would require more information on specific data sets or procedures used. These individuals are invited to contact the researchers in these respects.

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Executive Summary

INTRODUCTION AND SCOPE OF INVESTIGATION

1.1 PARTNERS IN RESEARCH

The University of the Orange Free State, through its Institute for Groundwater Studies, entered into a research contract with the Water Research Commission, the Department of Water Affairs and Forestry (DWA&F), the Chamber of Mines of South Africa (excluding Amcoal) and Eskom in 1989. This study was entitled "Groundwater quality deterioration in the Olifants Catchment above the Loskop Dam, with specialised investigations in the Witbank Dam Sub-catchment". The contract was preceded by a year of negotiations to ensure co-operation between all interested parties. A condition for co-operation was confidentiality of information on a site-specific basis.

1.2 AIMS

The aims of the investigation were to:

- Quantify the contribution of various activities that may result in a deterioration of the groundwater resources in the catchment above the Loskop Dam, with special emphasis on the Witbank Dam Sub-catchment.
- Predict future salt loads in groundwater, based on projections of probable development in the area. Extrapolate information to other catchments that supply water to the Olifants Catchment.
- Investigate and research improved management and precautionary measures that could be utilised to minimise groundwater quality deterioration.
- Integrate groundwater information with other investigations in the area, including the water management program of the DWA&F, with the purpose of deriving a catchment management programme at the end of this project.

Since 1990, many investigations have been initiated on specific issues relating to water within the Olifants Catchment. The most important of these investigations were perhaps those which have been initiated by the DWA&F, into the establishment of a surface water quality management programme for the Olifants Catchment. Other studies worthwhile mentioning are:

- The formulation of the Environmental Management Program Report (EMPR), which has come about during this investigation.
- Research into passive water treatment systems.
- Interception of rainfall thus minimising influx into mined areas.
- Research into liners for waste disposal, research into the use and application of bactericides to limit oxidation of pyrite; promulgation of the Environmental Act, Act 73 of 1989 and the associated licensing of general and hazardous waste sites.

- Many projects initiated by the mines and industries themselves.

Because of the environmental awareness and the significant number of related projects which came about since the original formulation of the present project, the emphasis in this project has shifted to ensure that duplication will not result. The following interpretation of the aims is provided:

- Quantification of contributions by various activities was to be on the micro- and local scale through monitoring, process studies and modelling at selected sites. Individual mines, industries or municipalities were not to be identified to ensure their continued co-operation.
- Prediction of future salt loads would emphasise the long-term issues tending towards steady state. Prediction will be done through monitoring, process studies, water and salt balances, chemical equilibrium calculations and modelling, thus obtaining a load factor, where possible, for each facility investigated. Prediction on a catchment basis was to be done, where possible, by applying a load factor to existing facilities within a catchment.
- Investigation and research into improved management and precautionary measures were to concentrate on the identification of main pollution sources, formulation of suggestions for improved management strategies where necessary, implementation of these strategies where possible, and modelling of the effect of the improved management strategies. The precautionary measures relate to all measures that may be undertaken to minimise an impact on the water resources, as pre- or post-treatment.
- The integration of groundwater information with other investigations in the area was to be done by the DWA&F, incorporating the information that is generated in this investigation, into their catchment management programme.

1.3 STRUCTURE OF REPORT

For the purpose of this investigation and the report, the main activities within the Olifants Catchment have been grouped into logical units. These are:

- Current water environments as they relate to the Olifants Catchment;
- Coal mining and its effect on water quality and quantity;
- Power generation and its effect on water quality and quantity;
- General waste sites and its effect on water quality;
- Sewage effluent and its effect on water quality;
- Metal industry and its effect on water quality; and
- Agriculture and its effect on water quality.

Apart from these identifiable and somewhat separate units, background information on aspects with which the everyday reader is not necessarily familiar, is included as appendices in the report. Of interest would be:

- Theory of chemical reaction in coal-mines;
- Acid-base experiments and accounting; and
- Suggested water quality monitoring procedures.

Terminologies in the mining industry in particular could lead to confusion. For that reason definitions and descriptors are included in Appendix F.

ENVIRONMENTS AS THEY REGIONALLY RELATE TO WATER IN THE OLIFANTS CATCHMENT

2.1. GEOLOGY AND GEOHYDROLOGY

2.1.1. GEOLOGY

The Karoo Supergroup in the Olifants Catchment comprises the Ecca Group and Dwyka Formation. The total thickness of these sediments ranges from 0 - 100 m.

The Ecca sediments consist predominantly of sandstone, siltstone, shale and coal. Combinations of these rock types are often found in the form of interbedded siltstone, mudstone and coarse-grained sandstone. Typically, coarse-grained sandstones are a characteristic of the sediments in the Witbank Area.

Two coalfields, namely Witbank and Highveld, are distinguished in the area (Figure 2.1). Five coal seams, numbered from bottom to top as No. 1 - 5, are present. Only two of the seams are mineable over most of the area. These are the No. 2 and 4 Seams, which are usually separated by sediments of a total thickness in the order of 20 - 30 m. Seams 1 and 5 are, however, mined locally.

Dolerite intrusions in the form of dykes and sills are present within the Ecca Group. The sills are highly undulating and some might conform to the ring structures described by Burger *et al.* (1981) in the southern Orange Free State. Burger et al. (1981) have discussed their mechanism of emplacement in detail. The sills usually precede the dykes, with the latter being emplaced during a later period of tensional forces within the earth's crust.

The Ecca sediments overlie the Dwyka Group (loosely referred to as the Dwyka tillite). This formation consists of a proper tillite, siltstone and sometimes a thin shale development. The upper portion of the Dwyka sediments may have been reworked, in which case carbonaceous shale and even inclusions of coal may be found.

The Dwyka sediments are underlain by a variety of rock types, such as the Bushveld Complex in the north, Witwatersrand Supergroup in the south, Waterberg Supergroup in the north-west and Transvaal Supergroup to the west.

Tectonically, the Karoo sediments are practically undisturbed. Faults are rare. However, fractures are common in competent rocks such as sandstone and coal.

2.1.2 GEOHYDROLOGY

Three distinct superimposed groundwater systems are present within the Olifants Catchment. They can be classified as the upper weathered Ecca aquifer, the fractured aquifers within the unweathered Ecca sediments and the aquifer below the Ecca sediments.

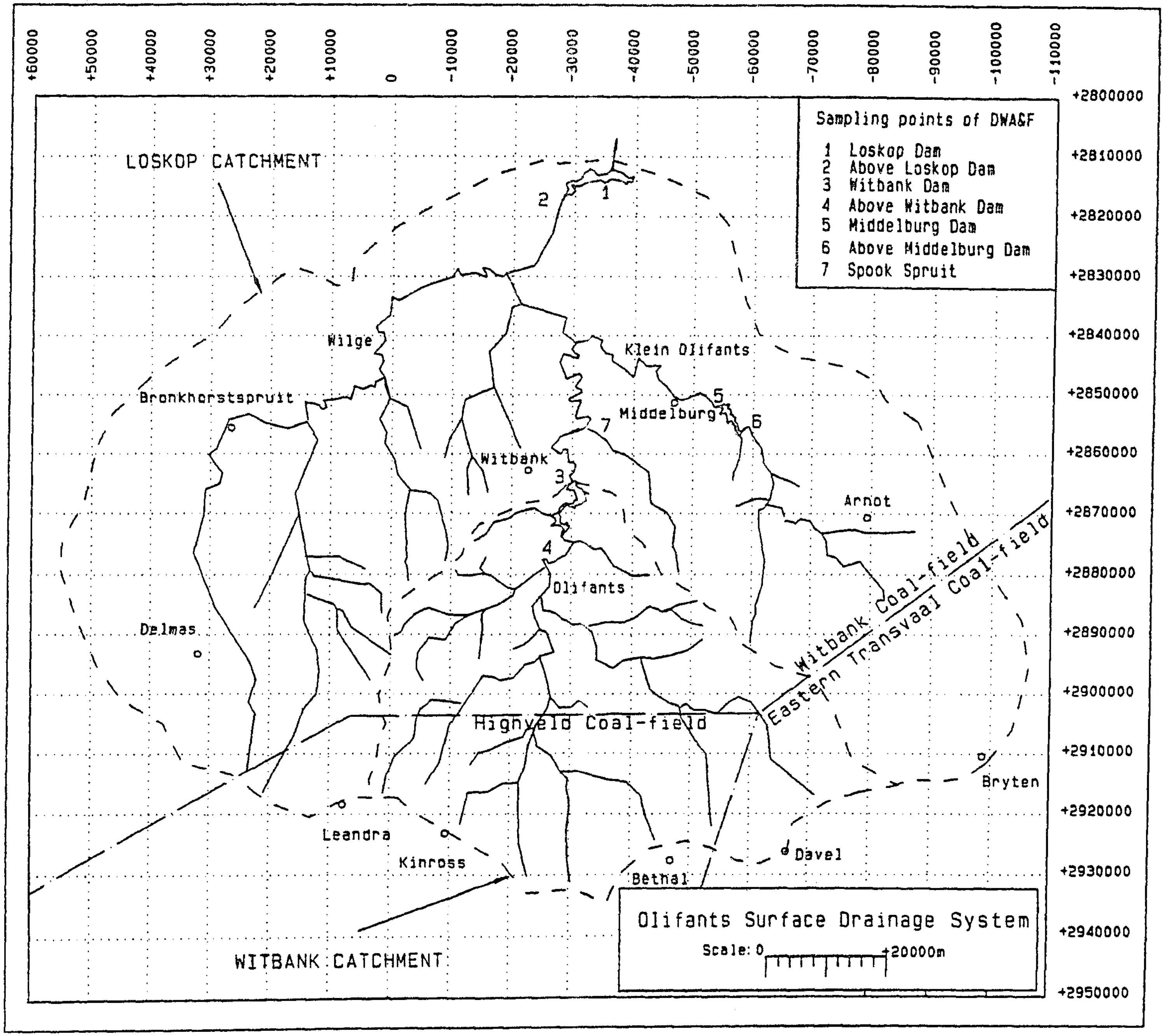


Figure 2.1. Locality map showing the Witbank and Highveld Coalfields, surface draiange system and routine water quality sampling points of the DWA&F

2.1.2.1 The Ecca weathered aquifer

The Ecca sediments are weathered to depths between 5 - 12 metres below surface throughout the Olifants Catchment. The upper aquifer, typically perched, is associated with this weathered zone and water is often found within a few metres below surface. This aquifer is recharged by rainfall. The percentage recharge to this aquifer is estimated to be in the order of 1 - 3% of the annual rainfall, based on work by Kirchner *et al.* (1991) and Bredenkamp (1978) in other parts of the country.

Observed flow within the catchment confirmed isolated occurrences of recharge values as high as 15% of the annual rainfall. It should, however, be realised that within a weathered system such as the Ecca sediments of the Olifants Catchment, highly variable recharge values can be found from one area to the next. This is attributed to the composition of the weathered sediments, which range from coarse-grained sand to fine clays.

Generally speaking, the north-western portion of the coalfields is characterised by coarser grained sandstone and higher recharge values are expected here. It is concluded from the above information that a recharge value in the order of 3% of the annual rainfall is feasible. In terms of the catchment size for the Witbank Dam (3 256 km^2), this amounts to 60 Mm^3 recharge per annum. In comparison to the Witbank Dam, this amounts to about half of the holding capacity of the dam.

Rainfall that infiltrates into the weathered rock soon reaches an impermeable layer of shale underneath the weathered zone. The movement of ground water on top of this shale is lateral and in the direction of the surface slope. This water reappears on surface at fountains where the flow paths are obstructed by a barrier, such as a dolerite dyke, paleo-topographic highs in the bedrock, or where the surface topography cuts into the groundwater level at streams. It is suggested that less than 60% of the water recharged to the weathered zone eventually emanates in streams. The rest of the water is evapotranspirated or drained by some other means.

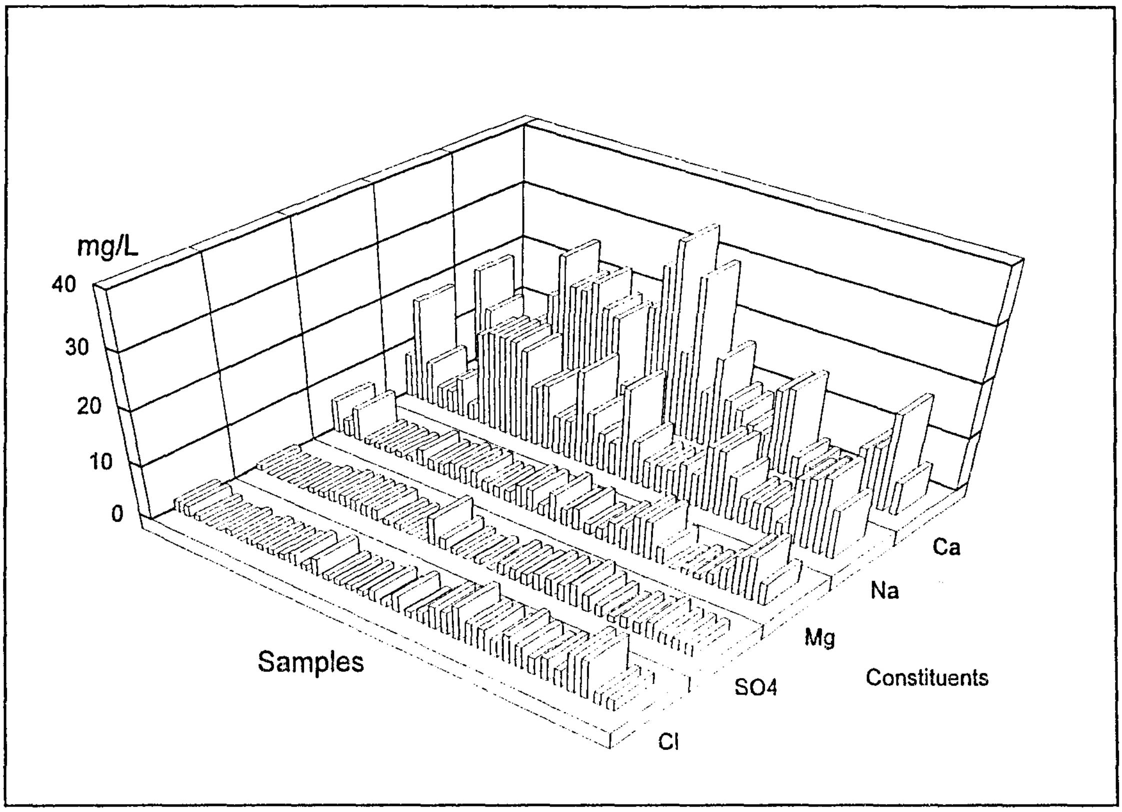
The aquifer within the weathered zone is generally low-yielding (range 100 - 2000 L/hour), because of its insignificant thickness. Few farmers therefore tap this aquifer by borehole. Wells or trenches, dug into this upper aquifer, are often sufficient to secure a constant water supply of excellent quality. Figures 2.2 and 2.3 illustrate the concentration of constituents in a selection of boreholes that tap water from this aquifer.

The excellent quality of this water can be attributed to the many years of dynamic ground-water flow through the weathered sediments. Leachable salts in this zone have been washed from the system long ago and it is only the slow decomposition of clay particles which presently releases some salt into the water.

2.1.2.2 The fractured Ecca aquifers

The pores within the Ecca sediments are too well cementated to allow any significant permeation of water. All ground-water movement is therefore along secondary structures, such as fractures, cracks and joints in the sediments. These structures are better developed in competent rocks such as sandstone, hence the better water-yielding properties of the latter rock type.

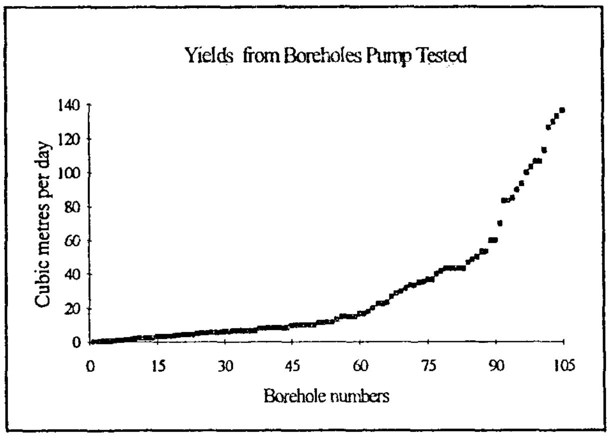
It should, however, be emphasised that not all secondary structures are water-bearing. Many of these structures are constricted because of compressional forces that act within the earth's crust. The chances of intersecting a water-bearing fracture by drilling decrease rapidly with depth. At depths deeper than 30 m, water-bearing fractures with significant yield were observed to be spaced at 100 m or greater. Scientific siting of water-supply boreholes is necessary to intersect these fractures.



Statistics	pH	Chloride	Sulphate	Magnesium	Sodium	Calcium	Elec. Cond.
		mg/l	mg/l	mg/l	mg/l	mg/l	mS/m
Mean	6.23	3.0	1.8	3.1	10.7	12.3	13.0
Median	6.05	2.0	2.0	3.0	8.0	10.0	13.0
Mode	6.00	1.0	2.0	2.0	5.0	6.0	18.0
Standard Deviation	1.23	3.3	1.4	1.9	6.2	8.9	6.2
Minimum	5.02	0.0	0.0	0.0	3.0	1.0	3.7
Maximum	6.98	16.0	6.0	8.0	33.0	35.0	25.0
Number of samples	41	41	41	41	41	41	41

Figure 2.2. Water qualities and statistics within the upper weathered aquifer in the Olifants Catchment.

Very few of the borehole yield within the Olifants Catchment have been quantified on a scientific basis. Statistics from a selection of boreholes which were pump tested by the Institute for Groundwater Studies, are as indicated in Figure 2.3.



Statistics	Yield
Mean yield (m3/d)	30.00
Median yield (m3/d)	11.70
Standard deviation (m3/d)	35.10
Minimum yield (m3/d)	0.30
Maximum yield (m3/d)	136.00
Number of holes tested	105

Figure 2.3. Statistics for the yields of boreholes that were pump tested from within the Olifants Catchment, based on an eight-hour pumping period per day.

The conclusion is drawn that all the tested boreholes have insufficient yields for organised irrigation. This is confirmed by a survey of the catchment, during which no irrigation from this aquifer could be located.

Of all the unweathered sediments in the Ecca, the coal seams often have the highest hydraulic conductivity. Packer testing of the No. 2 Seam and underlying Dwyka tillite has the hydraulic conductivity distribution as indicated in Table 2.1.

Table 2.1. Statistics for results on packer hydraulic conductivity testing of the No. 2 Seam and Dwyka tillite.

Statistics	2 Seam Permeability	Dwyka Permeability
Mean (m/d)	0.1017	0.0034
Median (m/d)	0.0743	0.0024
Standard Deviation (m/d)	0.1295	0.0034
Minimum (m/d)	0.0007	0.0002
Maximum (m/d)	0.5007	0.0148
Number of tests	21	21

From this comparison, it is clear that seepage of water through the No. 2 Seam is possible. Due to its low hydraulic conductivity, the Dwyka tillite forms a hydraulic barrier between the overlying mining activities and the basal floor.

In terms of water quality, the fractured Karoo aquifer always contains higher salt loads than the upper weathered aquifer. This is demonstrated in the selection of water chemistries in Figure 2.4.

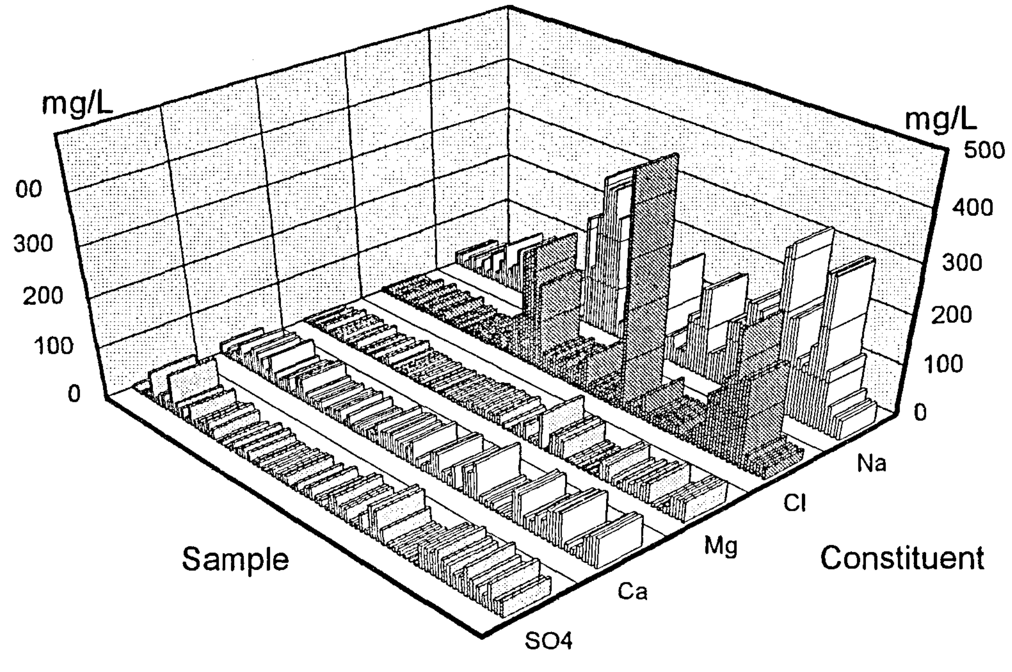


Figure 2.4. Water qualities within the fractured Ecca aquifer in the Olifants Catchment.

Statistics	Sulphate	Calcium	Magnesium	Chloride	Sodium	Elec. Cond.	pH
	mg/l	mg/l	mg/l	mg/l	mg/l	mS/m	
Mean	24	32	15	53	105	64	8.05
Median	20	27	10	22	65	58	8.04
Mode	10	22	8	8	30	59	7.70
Standard Deviation	18	18	13	78	89	34	0.45
Minimum	1	5	1	5	7	15	6.75
Maximum	80	76	69	463	330	145	8.95
Number of Samples	76	76	76	76	76	76	76

Figure 2.4 (Continued). Water qualities within the fractured Ecca aquifer in the Olifants Catchment.

Although the sulphate, magnesium and calcium concentrations in the Ecca fractured aquifer are higher than that in the weathered zone, they are well within expected limits. Their higher concentrations are attributed to the longer contact time between the water and the rock. The occasional high chloride and sodium levels are attributed to boreholes in the vicinity of areas where salts naturally accumulate on surface, such as at pans and some of the fountains.

2.1.2.3 Pre-Karoo aquifers

Drilling in only a few instances has intersected the basement to the Karoo Supergroup. Very few of the farmers, if any, tap water from the aquifer beneath the Dwyka Formation. The reasons for this are:

- The great depth.
- Low-yielding character of the fractures.
- Inferior water quality, with high levels of fluoride, associated with granitic rocks.
- Low recharge characteristics of this aquifer because of the overlying impermeable Dwyka tillite.

No pumping tests have been performed in boreholes that intersect the aquifers below the Karoo sediments.

In the southern portion of the catchment, dewatering of this aquifer has, to some extent, occurred because of the pumping in the Evander Gold-fields. Here, the piezometric pressure in the deep aquifer is generally 10 - 50 m lower than that in the Karoo sediments.

2.2 SURFACE-WATER QUALITY

2.2.1 RAINWATER QUALITY

Detailed studies of the rain water quality have been conducted during the past ten years (Tosen, 1990; Turner, *et al.*, 1990; Held *et al.*, 1991; Held, *et al.*, 1992; Roos, 1992). The conclusion drawn from these investigations is that rain water over the Olifants Catchment contains 2 mg/ℓ sulphate on average. Elevated concentrations of sulphate in rain water are observed over the Witbank Area. Rainfall records for the Witbank Area are presented with the permission of the CCWR in Figure 2.5.

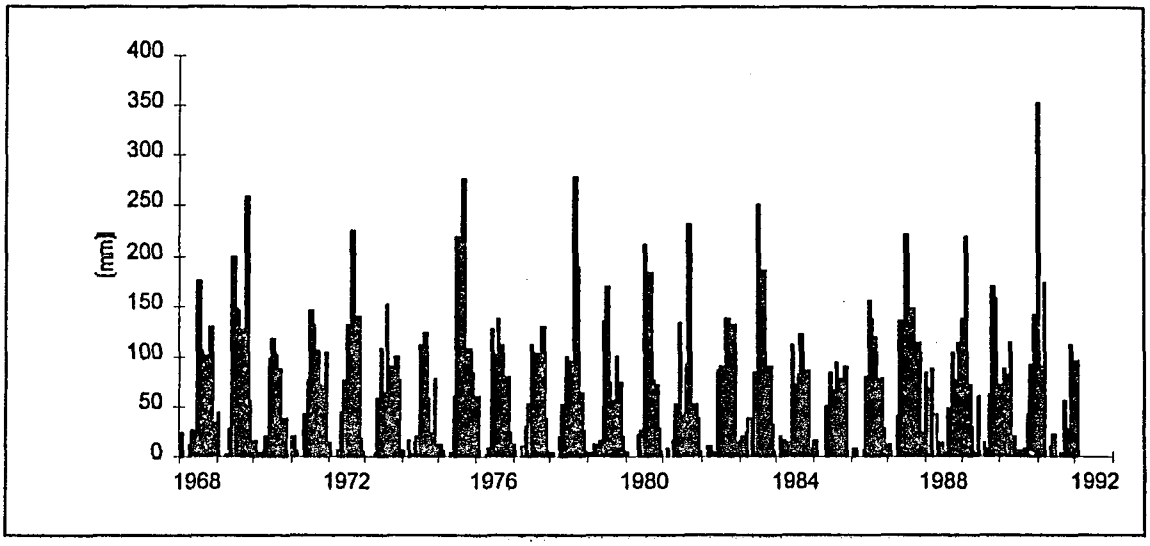


Figure 2.5. Monthly rainfall bars as measured by the Witbank Municipality.

2.2.2 SURFACE RUN-OFF QUALITY

The DWA&F are monitoring surface run-off water quality at many localities within the catchment (Figure 2.1). This information has been kindly made available by the DWA&F. Summaries of water quality, starting with water in the Loskop Dam, are given in Figures 2.6 to 2.15.

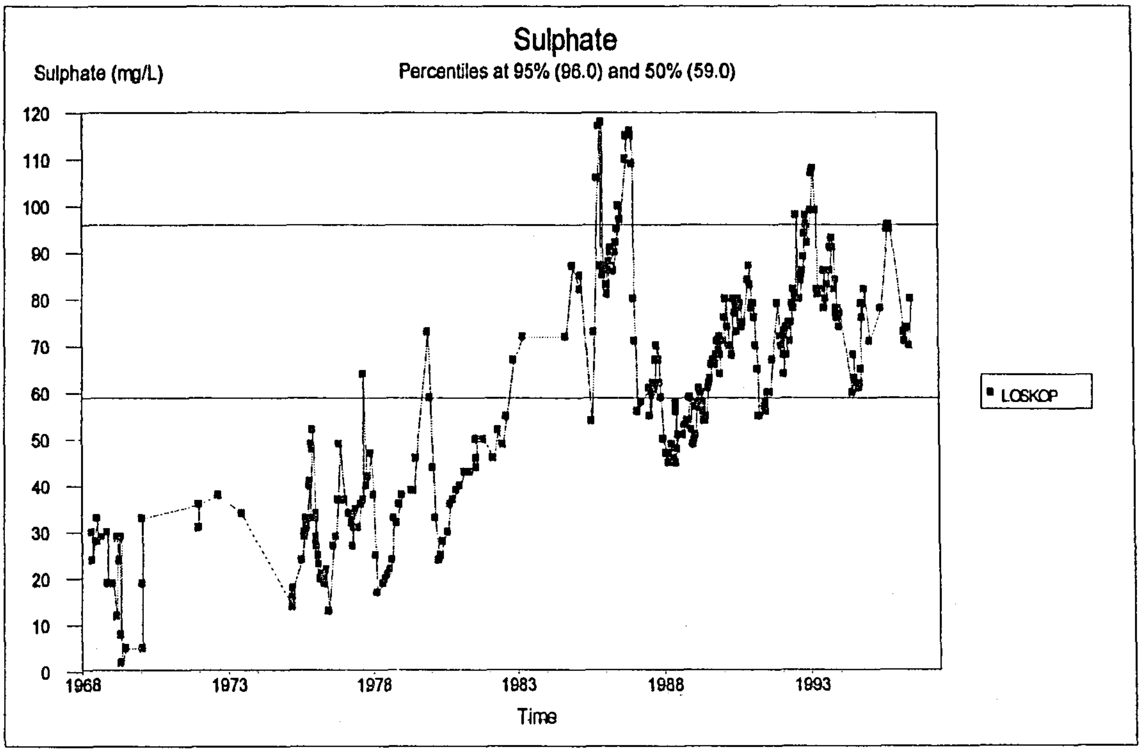


Figure 2.6. Sulphate levels in the Loskop Dam.

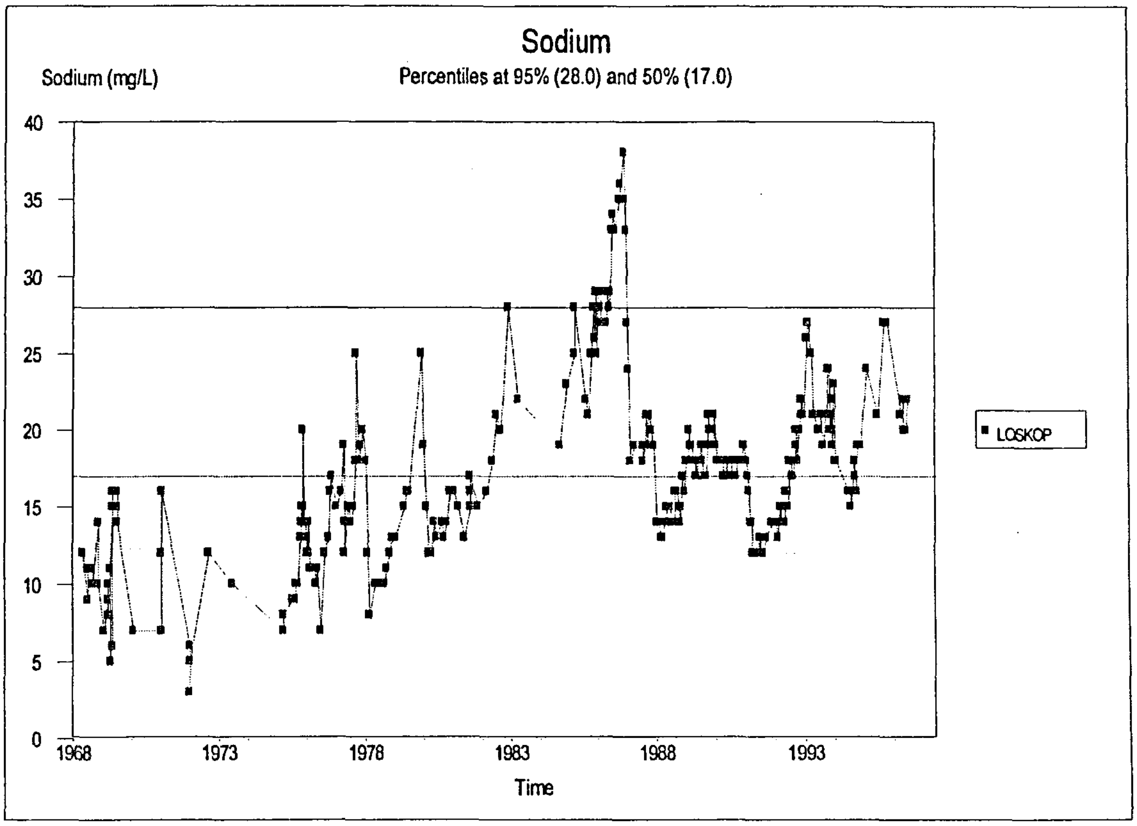


Figure 2.7. Sodium levels in the Loskop Dam.

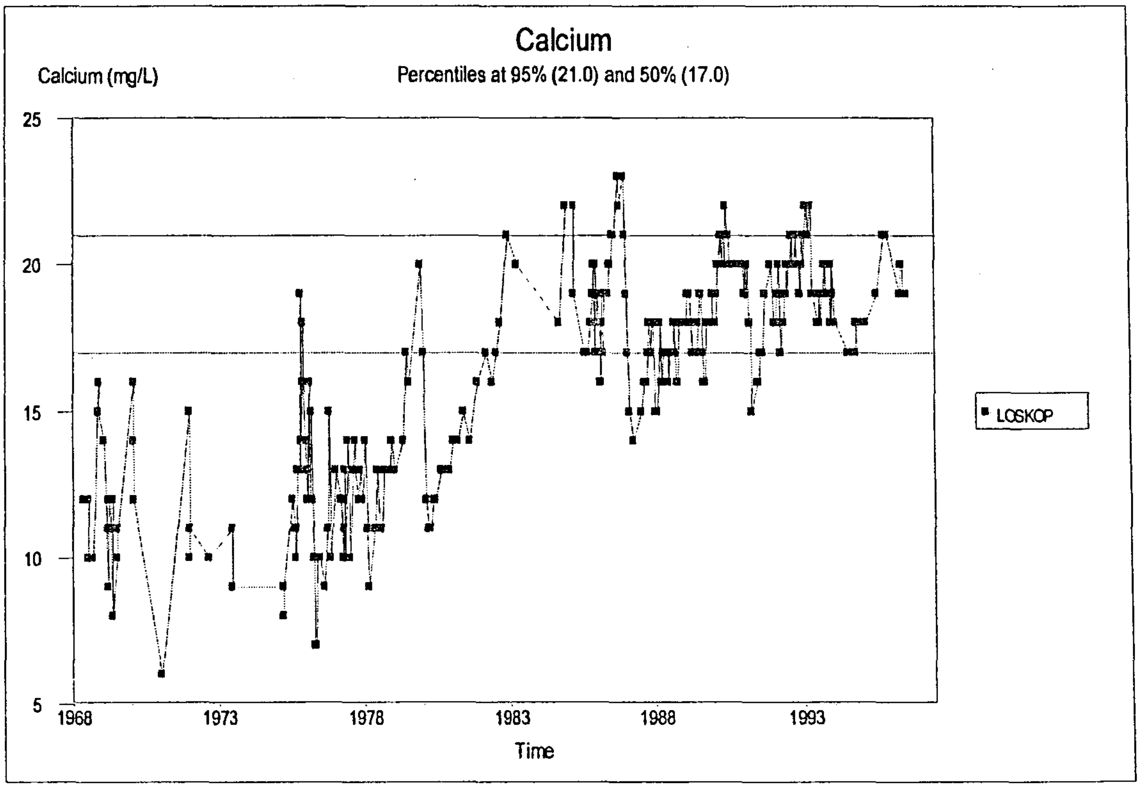


Figure 2.8. Calcium levels in the Loskop Dam.

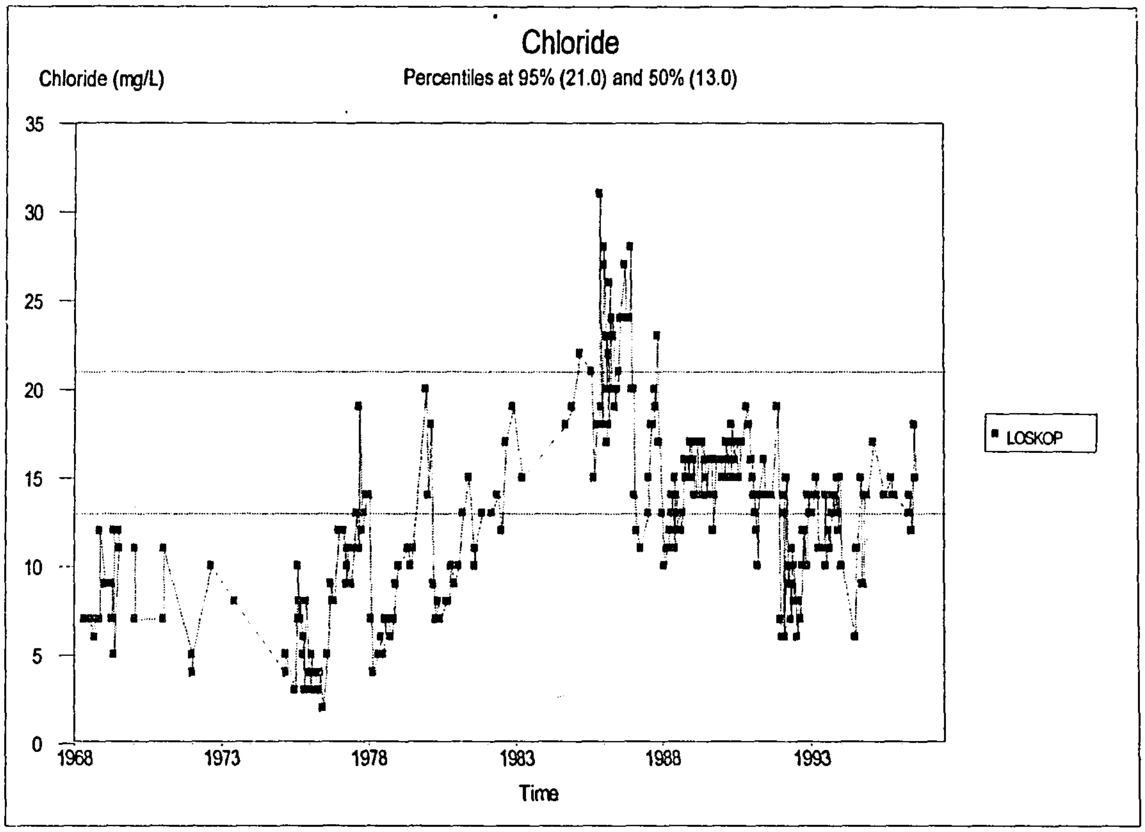


Figure 2.9. Chloride levels in the Loskop Dam.

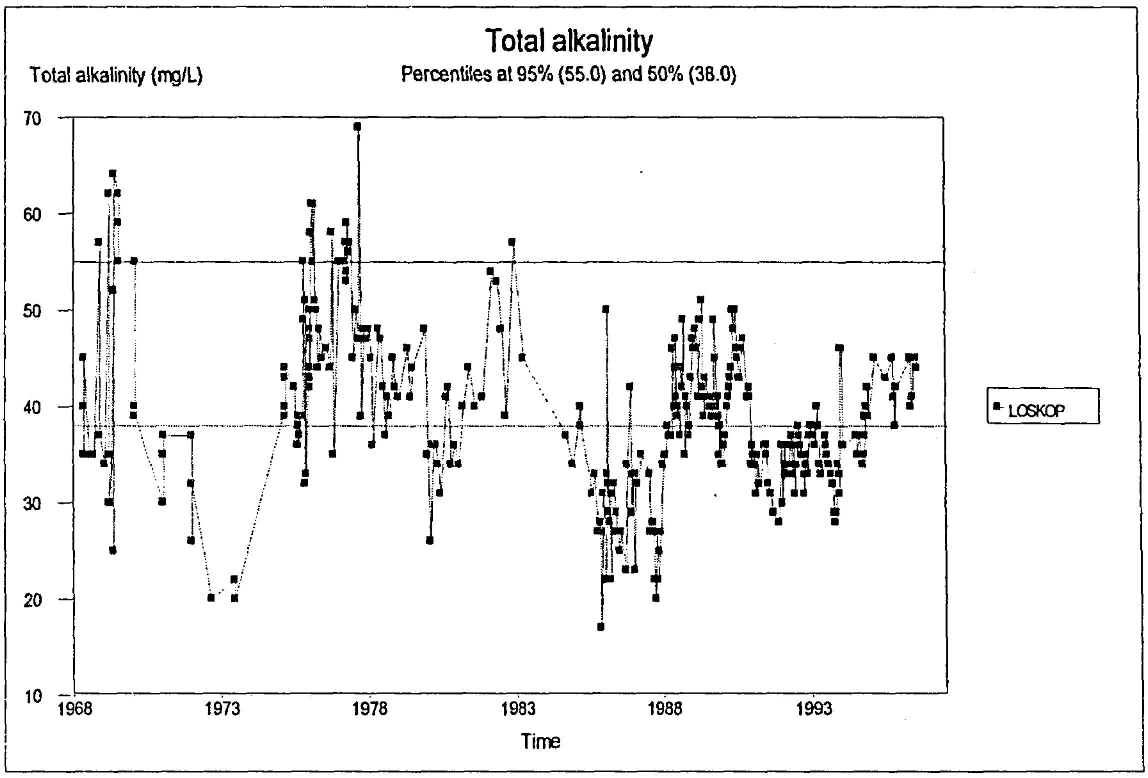


Figure 2.10. Total alkalinity levels in the Loskop Dam.

The general rise in salt concentrations within the water of the Loskop Dam System is evident in the above comparisons. Sulphate concentrations are presently the main concern. The evolution in the original alkaline Loskop Dam water to water dominated by sulphate as the main anion is demonstrated by their inverse relationships as shown in Figures 2.6 and 2.10.

The sulphate concentrations within the Olifants catchment above the Loskop Dam are illustrated in the following figures.

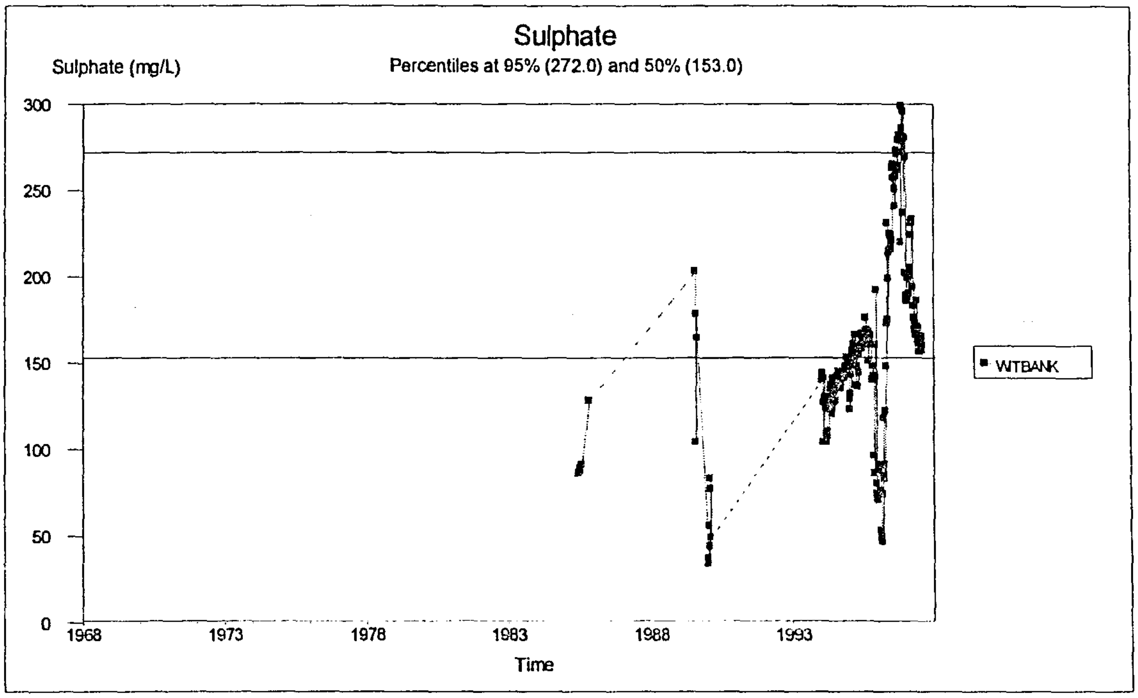


Figure 2.11. Sulphate levels within the Witbank Dam.

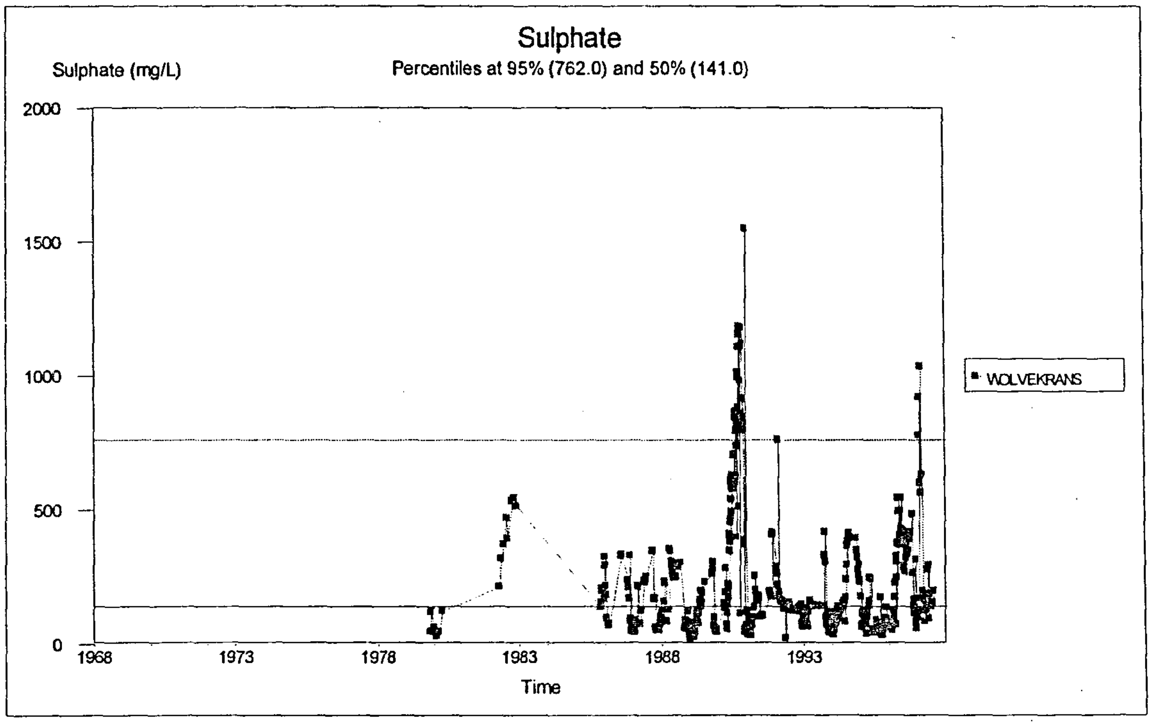


Figure 2.12. Sulphate levels at Wolvekrans above the Witbank Dam

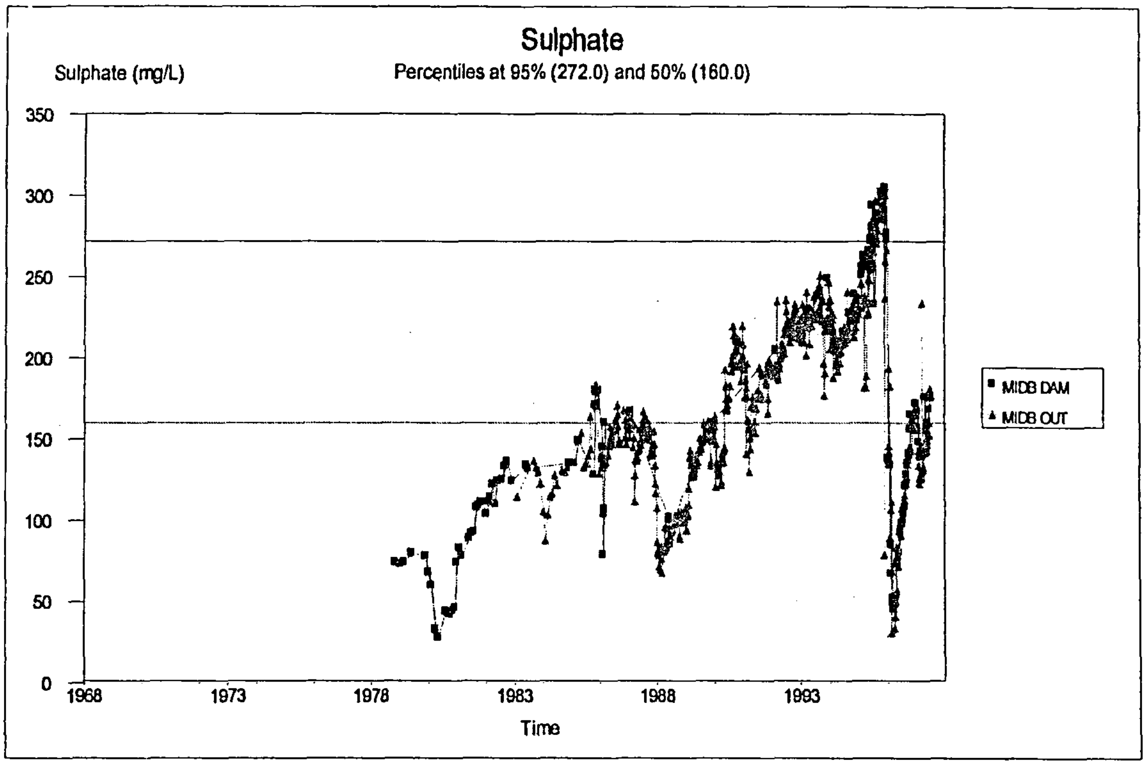


Figure 2.13. Sulphate levels in the Middelburg Dam and at the weir below.

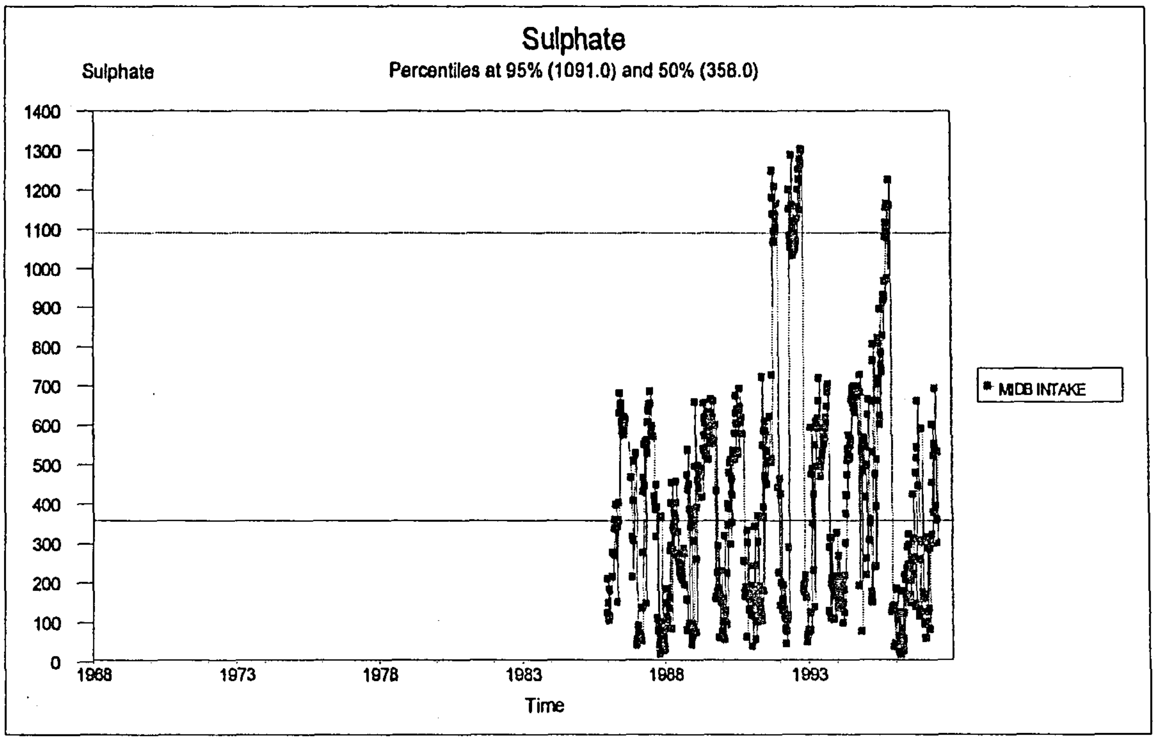


Figure 2.14. Sulphate levels at the weir above Middelburg Dam.

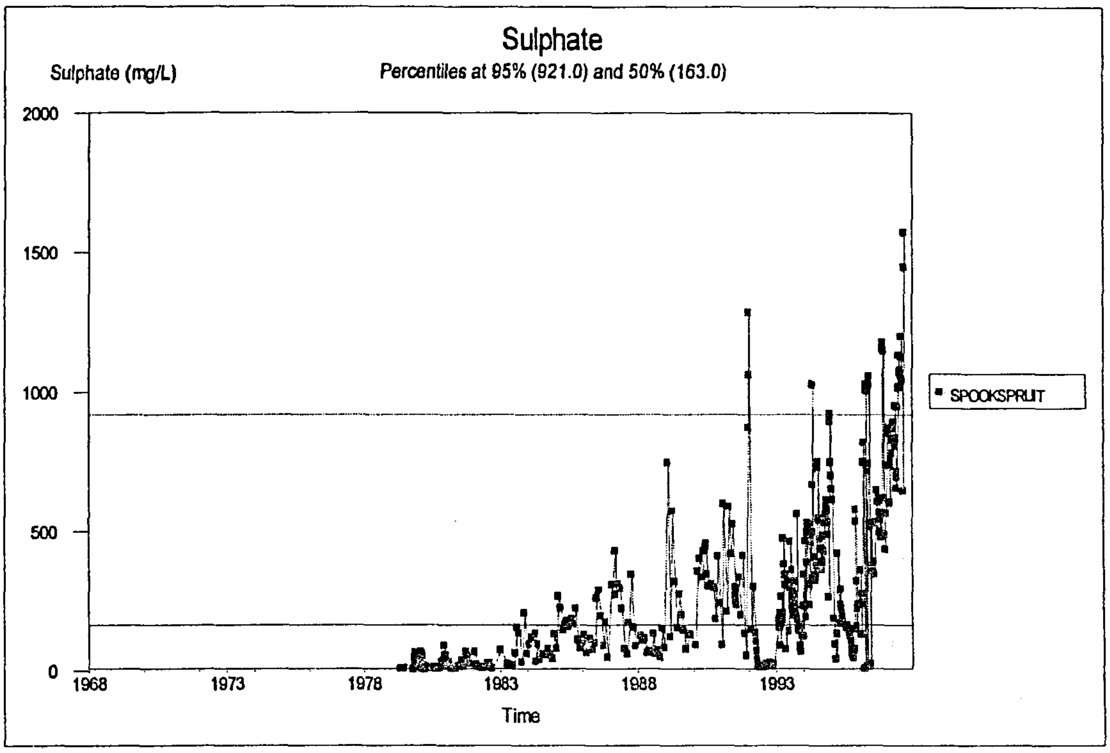


Figure 2.15. Sulphate levels within the Spook Spruit.

When considering that many of the mines have commenced with mining since 1980, it is concluded from the information in the preceding diagrams that a correlation between mining and stream water qualities may exist.

To show the degree to which the sulphate ion dominates the chemistry of the Olifants waters, the surface water chemistry for the major systems has been plotted onto expanded Durov diagrams (Figures 2.16 - 2.20).

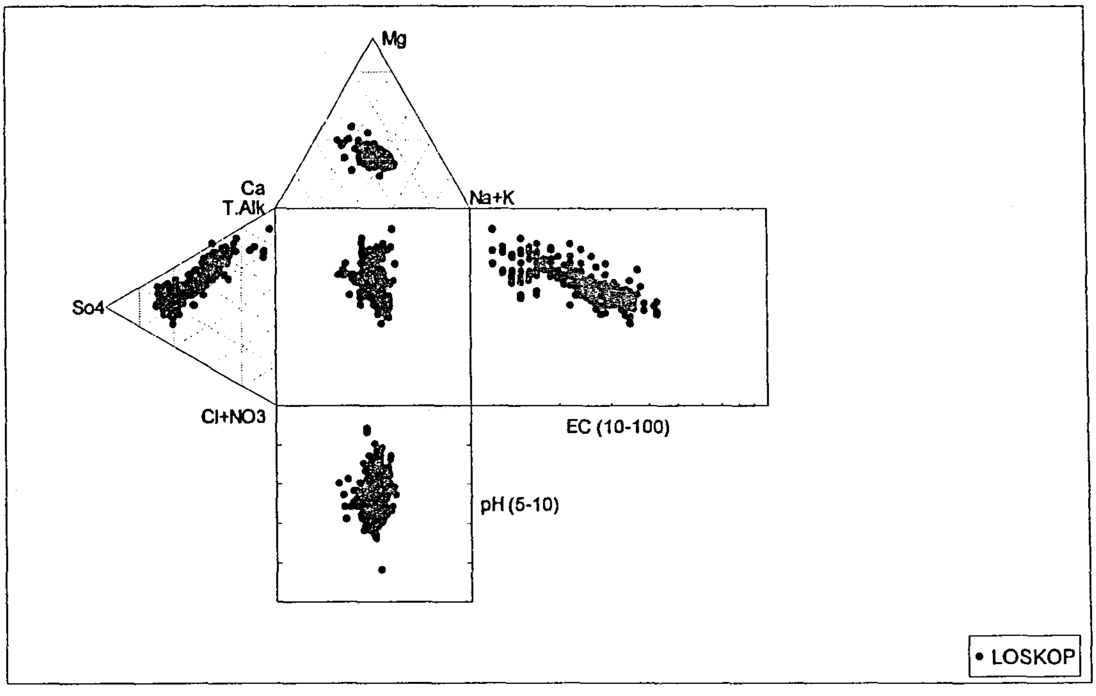


Figure 2.16. Durov plot for the water samples from the Loskop Dam.

The above diagram shows that about 65% of the water samples that have been taken from the Loskop Dam have sulphate as their major anion. The sulphate levels have steadily risen in the Loskop System since 1968. The above plot may therefore be seen as a rough time scale. Samples with high alkalinity ratios are those taken before 1982.

Another interesting aspect highlighted by this diagram, is the chemical evolution of the cation ratios towards a higher sodium percentage.

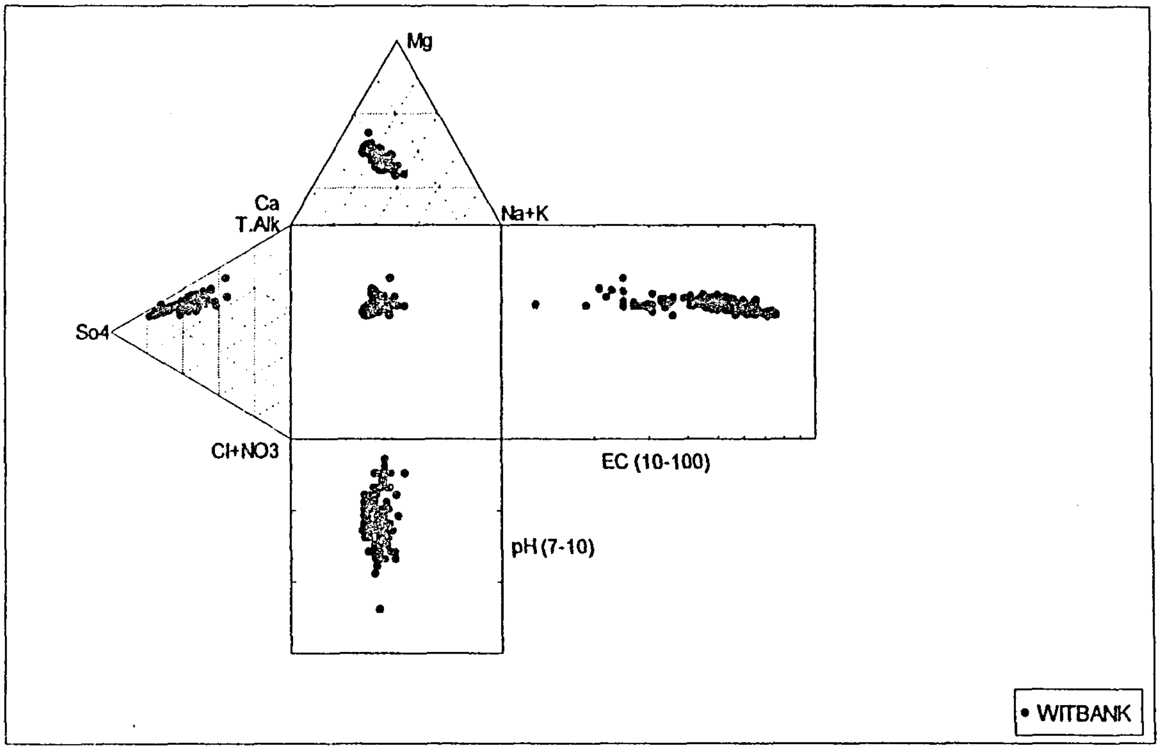


Figure 2.17. Durov plot for the water samples from the Witbank Dam

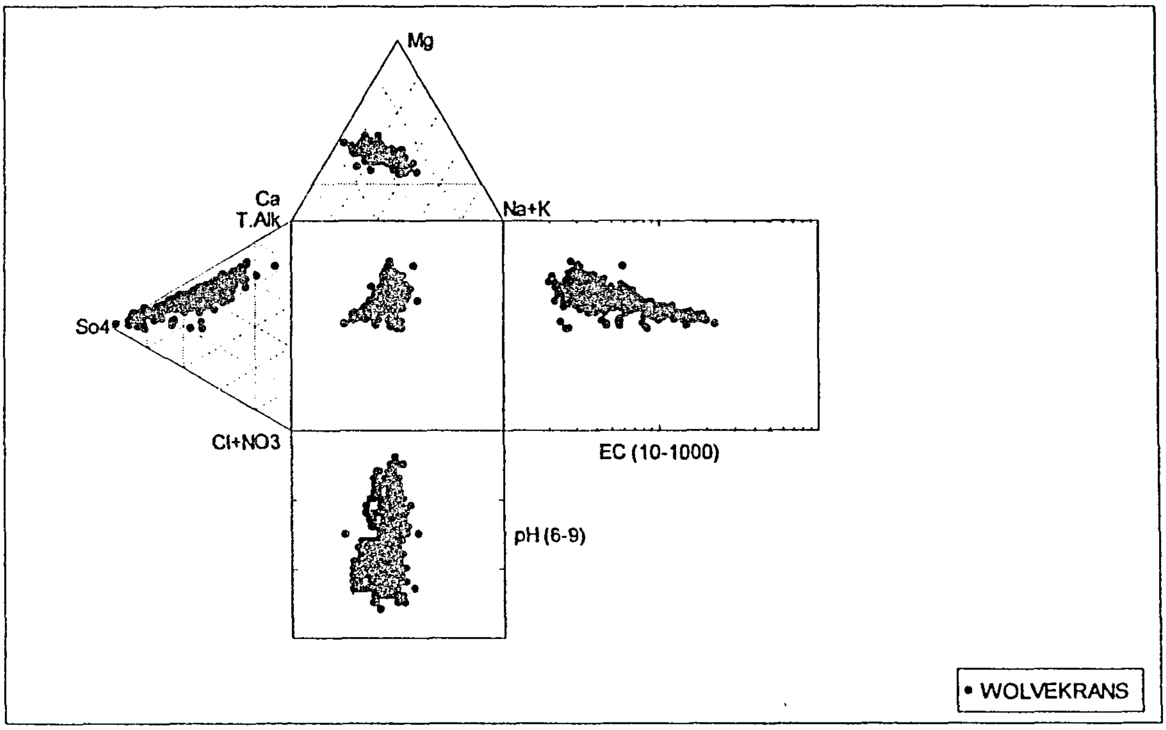


Figure 2.18. Durov plot for the water samples from Wolvekrans

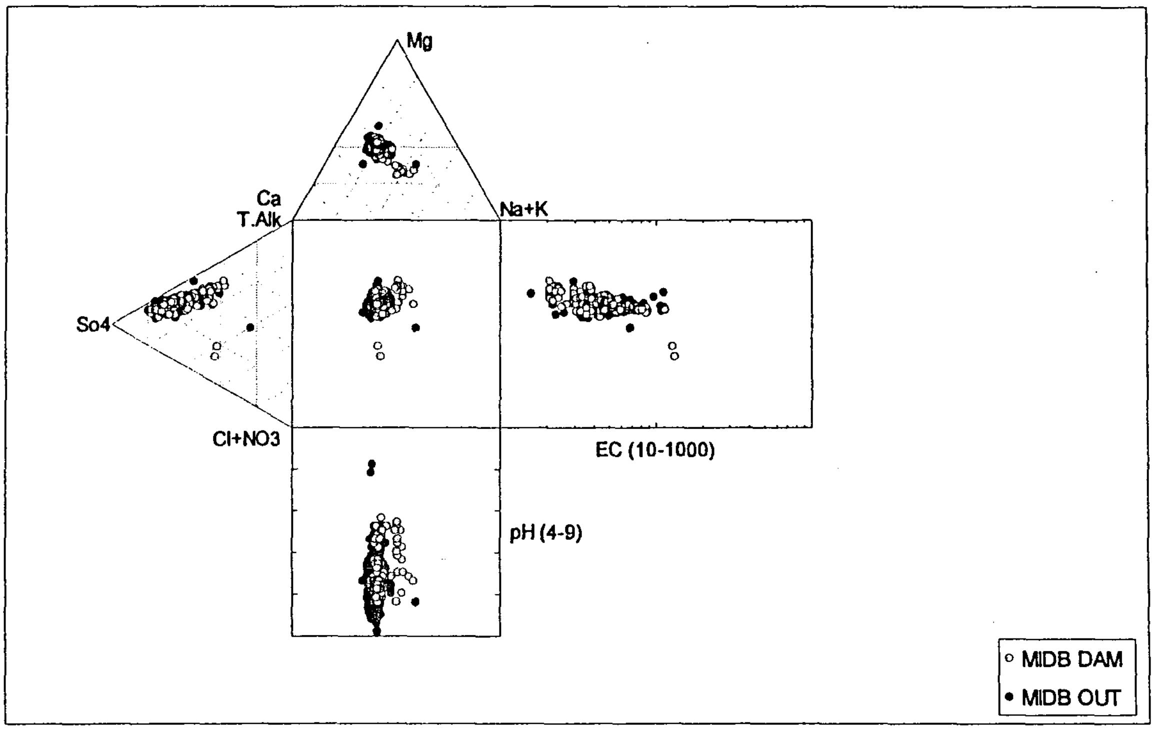


Figure 2.19. Durov plot for the water samples from the Middelburg Dam

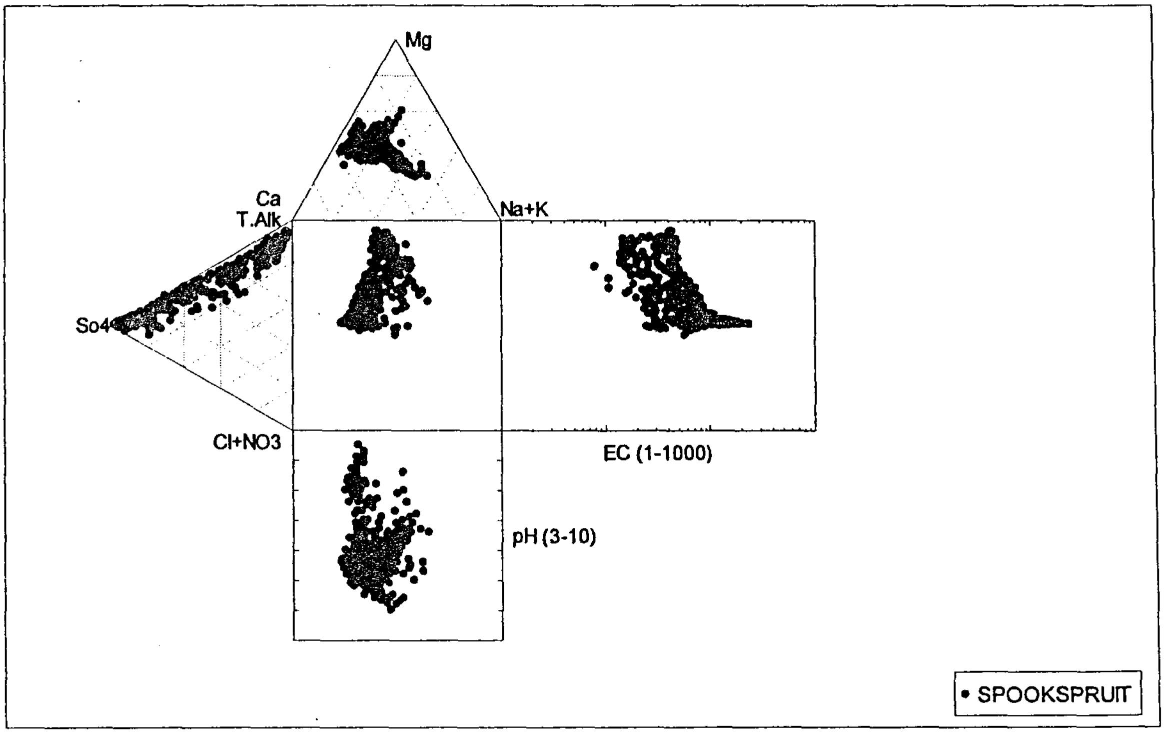


Figure 2.20. Durov plot for surface water samples collected at random from the Spook Spruit Catchment.

It is significant that the anions in the Witbank and Middelburg Dams are almost totally dominated by sulphate. This is due to years of sulphate contamination entering these dams. This is also the case in later years in the Spook Spruit, where their calcium and sulphate content dominate the water chemistries.

The following statistics (Table 2.3) were generated from a survey that included 279 water samples from 80 different localities within the catchments of the Witbank and Middelburg Dams, as part of this investigation:

Table 2.2. Statistics of chemistries for water samples in the catchment above the Witbank and Middelburg Dams, taken during 1990/1991.

Statistic	pH	EC	Ca	Mg	Na	K	Cl	SO4	MAlk	NO3-N
		mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
Mean	7.44	72	75	41	40	7	25	308	96	0.7
Standard Deviation	1.15	83	95	61	37	5	26	491	81	2.0
Minimum	2.75	0	4	1	4	0	2	3	0	0.0
Maximum	8.78	701	498	488	244	35	143	2972	424	21.4
Count	279	279	279	279	279	279	279	279	279	279
Statistic	F	Al	B	Cd	Cu	Fe	Mn	Pb	Sr	Zn
	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
Mean	0.70	0.61	0.02	0.00	0.00	2.55	0.86	0.02	0.67	0.08
Standard Deviation	0.79	1.90	0.07	0.00	0.02	11.46	2.23	0.19	1.07	0.10
Minimum	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01
Maximum	6.60	12.42	0.95	0.06	0.17	148.83	17.48	3.03	8.59	0.82
Count	279	279	279	279	279	279	279	279	279	279

This information supplements the data of the DWA&F, which was discussed above. Differences between the two sets of data are as follows:

- The range of values is wider, since samples were taken closer to sources of pollution.
- The data on heavy metals in the stream waters provide additional input that was not previously available from the DWA&F database. The sporadic high heavy metal content in this water is significant, particularly the manganese. The latter does not precipitate at neutral pH-levels, as is the case with the other heavy metals. Manganese can therefore be used as a tracer to identify acid rock drainage in streams, some distance from the source areas.
- Samples with calcium and sodium enrichment can be seen in the expanded Durov plot (Figure 2.21). The calcium enriched samples are attributed to acid rock drainage, while the sodium enrichment has other origins, such as cation exchange, drainage of water through pans or seepage from power station ashing operations. These aspects will be discussed individually in the relevant chapters.

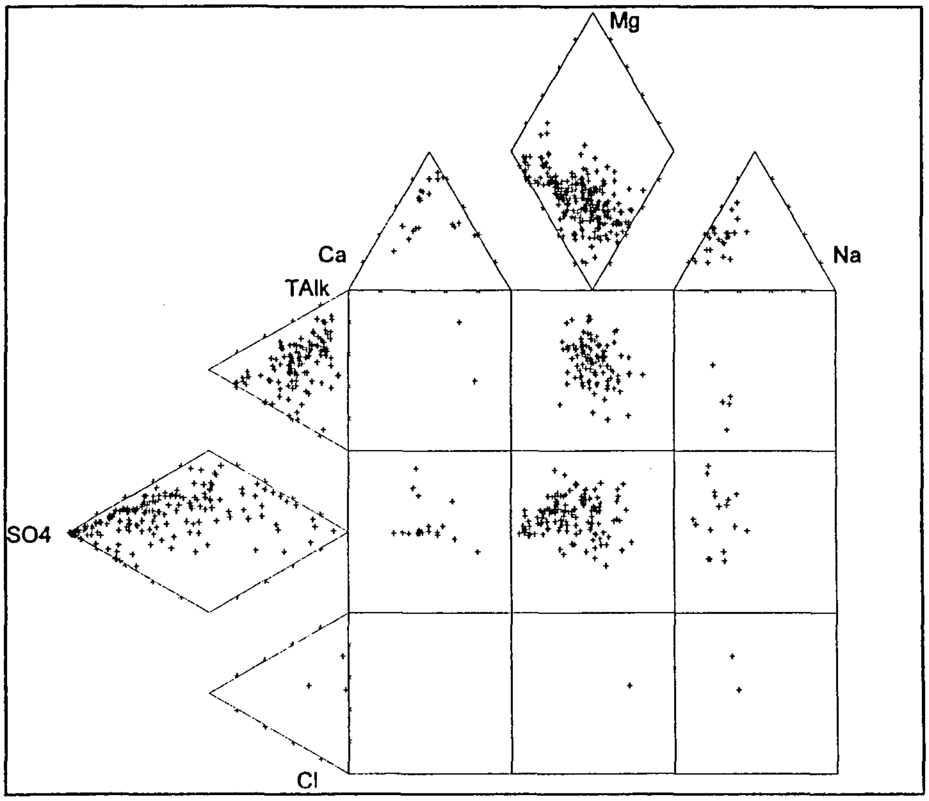


Figure 2.21. Expanded Durov plot for surface water samples collected at random from the Witbank and Middelburg Dam Catchments.

The above observations with regard to surface water quality in the Olifants Catchment are supplemented and confirmed by those of Wates, Meiring and Barnard (1993).

Section 3

COAL MINING AND ITS EFFECT ON GROUNDWATER QUALITY AND QUANTITY

3.1 PREVIOUS RESEARCH

3.1.1 UNDERGROUND MINING

The effects of underground mining on water naturally occurring in the rock have been studied in a number of instances in South Africa. Most of these investigations have related to mine applications for closure, or underground high extraction operations (Hodgson, 1978; Hodgson *et al.*, 1979; Hodgson *et al.*, 1979; Hodgson *et al.*, 1980; Fauconnier and Kersten, 1982; Hodgson, 1985). The monitoring of acid mine drainage by the DWA&F in the Olifants Catchment has been ongoing. Lately, some work has been done in relation to the EMPR requirements. The latter information is, in most instances, confidential.

Some other reports containing general information that could apply to the South African situation are those by Emrich (1969), Keeley (1977), Corbitt (1977), Rae (1978) and Miknis and Lovell (1978). These reports deal with various aspects of short- and long-term pollution of groundwater and discuss a number of means of alleviating the pollution problem.

Chemical degradation of groundwater in underground coal mines is a world-wide problem. Extensive investigations to find a solution have therefore been conducted, especially in the United States of America and the United Kingdom. Most of these investigations pertain to problems associated with acid mine drainage. The reports by Moebs and Krickovic (1970), Knight and Newton (1977), Wahler and Associates (1979) and Sgambat *et al.* (1980) serve as examples of investigations carried out in the United States of America. In the United Kingdom, the problem of water-table rebound has been recognised many years ago. Henton (1976), (1979) and (1981) has indicated that the problem of acid formation has by no means been solved. The ecological effects of the water-table rebound are also discussed. Work by Booth and Spande (1992) has indicated that groundwater levels in the vicinity of longwall mining in the USA tend to recover after a number of years. This is promising in terms of the inhibition of acid generation. Rebound of the water table could also be expected in South Africa, where shale occurs above collapsed longwall panels.

Miorin *et al.* (1977) have investigated the feasibility of flooding an underground coal mine as an amelioration option, so as to restore or partially restore the groundwater table and thus reduce acid formation. Flooding abandoned sections of mines could have two important implications. In the first instance, the quality of surface water is far superior to that of groundwater and flooding could considerably improve the quality of the underground water. Secondly, provided that recharge can be effected within a reasonably short period of time, exclusion of oxygen will restrict acid formation. The investigation by Miorin *et al.* was eventually abandoned, because of the high cost of installing seals to isolate an area in which experiments could be conducted. Conditions in South Africa are far more favourable for such experiments and several studies of this nature are presently in progress.

Information on the cost-effectiveness of dewatering an actively worked underground coal mine as an alternative or supplement to treating acid mine drainage, is contained in a report by Wahler and Associates (1979). Few publications deal with the management of groundwater in areas of increased extraction of coal. Most of the investigations relate only in part to this subject. Miller and Thompson (1974) discuss *inter alia* the use of coal barriers as a means of retarding mine drainage. Knight and Newton (1977) concluded that the detrimental effect on the water quality of some streams might persist for decades after mining has ceased. Shotts *et al.* (1978) investigated ways to select a layout and mining system, and to develop and operate an underground coal mine, while at the same time minimising pollution of the environment.

Additional monitoring programs have recently been initiated in the USA to study the effect of longwall mining on catchment water quality and quantity (Coen, 1992). This also includes a study of the effect of the mine water on fish and the diversity of macro-invertebrates. The study is still ongoing and valuable information is likely to come from this investigation. The need for long-term monitoring is emphasised by this program.

Other references such as Hamilton and Wilson (1977), Manula and Ovili-Eger (1975) and Singh *et al.* (1985) deal with the calculation of groundwater influx into disturbed areas. Schubert (1978) and Balogh and Szepessy (1985) looked at the use of mathematical models as an aid to management in planning efficient systems that will minimise deleterious long-term effects in areas of increased extraction.

3.1.2 OPENCAST MINING

3.1.2.1 Water quality deterioration and hydrochemistry

The change in water quality as a result of opencast coal mining is well documented (Childress and Jones, 1985; Carmon, 1986; Hall, 1986; Grunwald *et al.*, 1988; Cunningham and Jones, 1990). A discussion of this information (much of which is also applicable to underground coal mines) follows:

Dissolution of sodium, potassium and chloride-bearing minerals represents the first interaction between the rock, hydrosphere and lithosphere, following mining activity (Fisher *et al.*, 1985; Williams and Hammond, 1989). The degree of salination during this phase is site specific and normally insignificant in South Africa.

The prolonged oxidation of sulphide-bearing minerals, of which pyrite is the dominant species, is simultaneously set in motion. This generates acidity and releases ferrous ions within the spoil. An inverse correlation exists between the grain size of the pyrite and its reactivity (Barton, 1978; Houghton *et al.*, 1985; McCurry and Rauch, 1986). Finely disseminated framboidal pyrite is the most effective acid-generating source.

Other acid-generating reactions within opencast mines include the precipitation of ferric hydroxide and the accelerated oxidation of pyrite by ferric ions (Stumm and Morgan 1981; Steffen, Robertson and Kirsten 1989). The ferric ion is formed during the oxidation of dissolved iron. Bacteria, such as *Thiobacillus ferrooxidans,* accelerate these reactions (Sobek *et al.*, 1985; Dugan, 1987; Miller and Murray, 1988; Olson, 1991). Total sulphur gives little indication of the final pH of the spoil waters. An extremely low sulphur content may still produce acid drainage, if insufficient neutralising constituents are present (Rauch and Di Pretoro, 1986). The acidification of mine water is referred to as acid rock drainage (Steffen, Robertson and Kirsten, 1989).

Neutralisation of acid water by carbonate minerals has been discussed extensively in literature (Blowes and Jambor, 1990; Glynn, 1991; Liu, 1991; Walter and Norris, 1991). Neutralisation reactions produce metal salts and water (Metcalfe *et al.*,1978). Even though neutralisation may occur, the water quality will still deteriorate as a result of the salinity generated by these processes.

Static tests have been designed to determine the potential of spoil samples to generate net acidity. Such experiments are used in acid-base accounting to determine the net neutralising potential (NNP) of mine samples (Sobek*et al.*, 1978; Ferguson and Erickson, 1988). Steffen, Robertson and Kirsten (1989) and Cravotta*et al.* (1990) described the calculations for NNP. Kinetic testing which involves the construction of leaching columns is used less frequently. This is due to the long periods required to produce significant results.

Computer models have been used to predict the quality of future mine drainage. The reliability of these models depends on their ability to simulate field conditions. Many models are extrapolations of the experimental data acquired from field and kinetic testing. Chemical equilibrium models, such as WATEQ, PHREEQE and MINTEQ (Trusdell and Jones, 1974; Parkhurst *et al.*, 1980; Felmy *et al.*, 1984), deal with the chemistry under equilibrium conditions - a situation seldom found in collieries. Transport models have focused on the numerical solution of partial differential equations concerning the transport of a single reactive solute (Lewis*et al.*, 1986).

Recent models, such as SATRA-CHEM, endeavour to combine the effects of equilibrium chemistry and mass transport (Lewis*et al.*, 1986). Jaynes *et al.* (1984) have developed a model designed specifically for the modelling of spoil-water quality, including the influence of *Thiobacillus ferrooxidans*. Such models still neglect the effects of disequilibrium chemical reactions. Yeh and Tripathi (1989 and 1991) describe their HYDROGEOCHEM model that, according to them, can be adapted to include mixed chemical kinetics coupled with disequilibrium mass transport.

3.1.2.2 Remediation

The Chamber of Mines of South Africa (1981)has developed a set of guide-lines for the rehabilitation of opencast pits. This includes the levelling of the spoil, topsoiling and the establishment of vegetation.

The use of synthetic covers to prevent the oxidation of pyrite, through exclusion of oxygen and water, is expensive (Steffen, Robertson and Kirsten, 1989). A literature survey indicates that spoil heaps and mine dumps in South Africa are typically covered with vegetation and sometimes with soil (Venn*et al.*, 1980; Cartwright, 1983; Ward *et al*, 1984). It should be noted, however, that any form of cover material requires ongoing maintenance. As such, the long-term exclusion of oxygen and water from spoil material is almost impossible to achieve. Water cover or flooding has been demonstrated to be the most effective method of excluding oxygen from in-pit spoil (Steffen, Robertson and Kirsten, 1989).

Various bactericides have been utilised to inhibit the activity of*Thiobacillus ferrooxidans* (Kim *et al.*, 1982; Sobek *et al.*, 1985; Dugan, 1987; Loos *et al.*, 1990; Sanderson and Immelman, 1990; Olem, 1991). Abatement of acid generation has been achieved to various degrees of success. However, the need for the repeated application of the chemicals limits the effectiveness of this method.

The remediation of mine leachate often involves the addition of neutralising agents. Lime and carbonate minerals are commonly utilised in the treatment of acid rock drainage (McDonald and Grandt, 1981; Davison and House, 1988; Kertell*et al.*, 1988; Brady *et al.*, 1990). Sewage and fly ash are other materials which ameliorate acid waters (Barr, 1980; Beaver, 1986; Seaker and Sopper, 1988; Pietz*et al.*, 1989; Bowders and Almes, 1990). Apart from the high costs in applying the neutralising agents to the pit environment, the physical mixing with the spoil presents major problems. Liming of acid leachates in a treatment plant is conducted in a number of instances in South Africa. This precipitates heavy metals from the water, but does not decrease the total salt load. Continued plant treatment of acid water was not generally an economically viable option following mine closure.

Methods such as desalination improve the quality of saline mine water (Stephenson, 1986; Simpson and Buckley, 1988; Sikora *et al.*, 1989). However, Stephenson (1986) indicates that such remediation could be prohibitively expensive. This is confirmed by the work of Forbes and Showers (1988) who found evaporation pans to be more viable. Kempe (1983) and Chapman *et al.* (1988) also mention the use of such facilities for mine leachates. However, many evaporation pans leak, thereby polluting the groundwater.

Du Plessis (1983) proposed the use of lime-treated mine water for irrigation purposes. The coal in the Witbank Area has a low sodium content. Du Plessis (1983) therefore postulated that it is possible to irrigate extensively, without the danger of elevated sodium adsorption ratios. Several such irrigation schemes are presently under way, apparently with promising results.

The development of wetlands for the restoration of the hydrochemistry has received considerable attention in the literature. These environments are successful in removing much of the dissolved heavy metals and raising the pH of leachates (Maree *et al.*, 1987; Donlan, 1989; De Wet *et al.*, 1990; Morea *et al.*, 1990; Walton-day *et al.*, 1991; Machemer and Wildeman, 1992). This is achieved through the activities of heterotrophic metal-utilising bacteria in the aerobic zone (Gordon and Burr, 1988). A further advantage of this ecosystem is the reduction of sulphate to sulphur by bacteria in the anaerobic zone (Maree *et al.*, 1987; Batal *et al.*, 1989; Hedin *et al.*, 1989; Silver, 1989). The sustainability of such a system, under field conditions, is not proven.

3.1.3 CONCLUSION

The conclusion from the above literature survey is that leachate generation within the context of coal mining has been exhaustively studied on a world-wide basis. There is generally a lack of reliable data and information on water aspects of the South African coal mining industry. The present research project therefore focused on the generation and analysis of data to facilitate the meaningful evaluation of the South African situation. Attention was specifically given to the following issues:

- Water quality distribution.
- Processes which contribute to water quality deterioration.
- Quantification of the rate and degree of salination within the Olifants Catchment.
- Long-term chemical behaviour of the system.
- Management options.

3.2 WATER IN UNDERGROUND COLLIERIES

3.2.1 SHALLOW UNDERGROUND COAL MINING

Shallow underground coal mining is defined for the purposes of this study as coal mining that mines into the weathered aquifer. Mining of this kind has been conducted by bord-and-pillar methods in the Witbank Area since the turn of the century. Most of these mines have long since closed down. They have subsequently filled with water to levels that are determined by inflows and seepage losses.

3.2.1.1 Water Quantity

The amount of recharge from rainfall into shallow underground coal mines varies considerably over the Olifants Catchment. The area west of Witbank comprises many shallow mines, where the overlying strata have collapsed to varying degrees. These collapsed structures form conduits for rain water, to infiltrate into the underground workings. In an experiment at one such mine (surface area 191 ha), the amount of recharge for a specific rainfall event could accurately be measured. The experimental conditions were as follows:

- A pump was installed in the mine with the intention to withdraw 55 000 m3 of water from the mine, thus creating underground space that may be recharged by groundwater seepage or by infiltration from rainfall.

- The water level in the mine was monitored for three days prior to pumping, to ensure that no significant water-level fluctuation that could affect the experiment was present.

- On the basis of the floor contours of the mine, volumes of water within the mine were calculated. This information was used to predict the response of the water level within the mine during pumping. The calculated and actual responses of the water level for this experiment are shown in Figure 3.1.

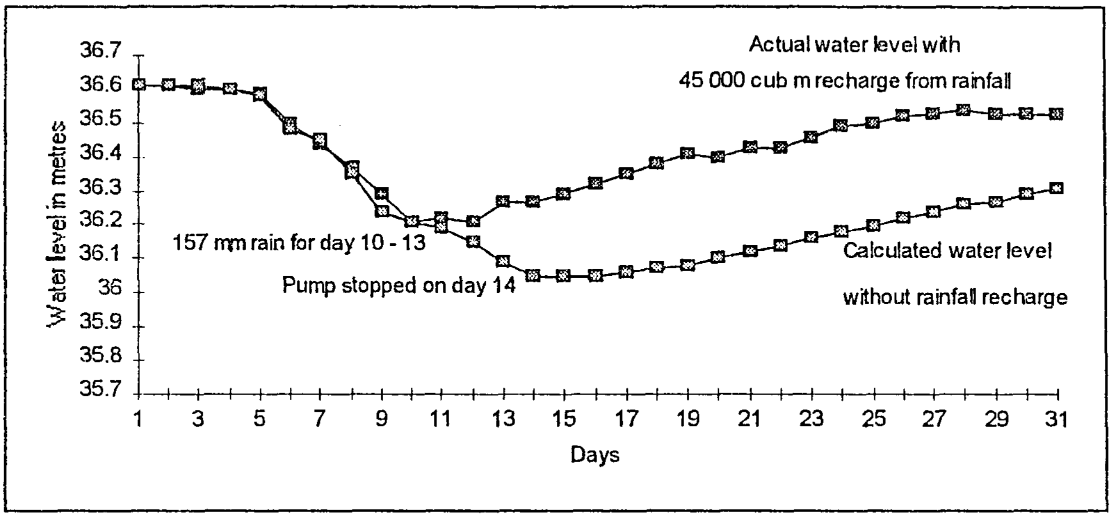


Figure 3.1. Actual and calculated water levels during pumping from an underground mine and recharge from rainfall and seepage.

- The pump was activated on day four, pumping 54 896 m3 of water from the mine up to day fourteen.

- Between days 10 - 13, an abnormally high rainfall event of 157 mm had occurred.

- The effect of rainfall infiltrating into the mine was immediately visible in a rise of the water level in the mine, despite the fact that pumpage continued until a day after the rain had stopped.

- The difference between the observed water level and that predicted from the floor contours of the mine, amounted to 45 000 m^3 at the end of the experiment.

- The percentage recharge, as a function of the rainfall event, amounted to 15% recharge.

- Other underground mines in the area had a three- to fourfold increase in water entering into the underground workings, after this rainfall event. This greater influx was noticeable in the pumping and flow records of the mines, up to more than six months after the rainfall event.

Conclusions:

- The experiment was conducted during an extreme rainfall event, in an area which has actively subsided above a collapsing mine roof. The value of 15% recharge can therefore be taken as the upper limit for shallow underground mines in the area.
- Recharge is immediate in areas of extensive collapse, whereas in undisturbed areas, slow seepage percolates into the underground workings over a matter of months.
- In areas where the overlying strata have not collapsed, experience suggests that recharge from rainfall is likely to be between 0 - 5% of a rainfall event, with 3% recharge on an annual basis. The degree of weathering and composition of the overburden determine the actual percentage of rainfall that reaches the mine. For example, weathered arenaceous overburden, as found in certain areas west of the coalfield, has higher recharge potential.

3.2.1.2 Water quality

Abandoned shallow underground coal mines are only partially filled with water, since water seeps from these mines, through sub-outcrop areas or collapse structures. Known seepages occur in much of the area west and north-west of Witbank. Oxygen and water are therefore freely available to large portions of these shallow mines.

Two examples of mine water quality are included in this document, which illustrate the range of water chemistries that may be expected in the shallow underground coal mines.

Mine 1

Mining ceased at this site more than 25 years ago. This is the same mine that was used for the recharge experiments. Approximately 70% of the mine is filled with water. Numerous collapse structures connect the surface with the underground workings. Active recharge from rainfall and run-off is therefore possible.

A summary of the water chemistries within the mine is presented in Table 3.1. Figure 3.2 provides an indication of actual heavy metal concentrations in the individual water samples. All samples from the mine were taken by pumping water from monitoring boreholes. In every case, the boreholes were pumped until the electrical conductivity of the pumped water had stabilised, before the samples were taken. It has been found during this investigation, that grab samples of mine water that are taken from boreholes which are not purged, do not represent the true mine water quality. This is due to the dilution of the mine water by percolating groundwater from aquifers above.

Table 3.1. Statistics for water chemistries in an acid shallow underground coal mine.

Statistics	pH	EC	Ca	Mg	Na	K	Cl	SO4	MAlk
		mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
Mean	3.85	189	154	59	31	10	24	1475	4
Median	3.20	180	103	43	25	7	12	859	0
Standard Deviation	1.24	150	140	64	30	11	38	1598	11
Minimum	2.41	2	3	0	4	1	2	11	0
Maximum	6.69	501	448	291	137	57	192	7142	61
Number of samples	37	37	37	37	37	37	37	37	37

Statistics	Al	B	Cd	Cu	Fe	Mn	Pb	Sr	Zn
	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
Mean	43.67	0.03	0.04	0.09	129.15	12.24	0.02	0.58	2.89
Median	25.02	0.00	0.00	0.05	32.00	8.43	0.00	0.40	1.66
Standard Deviation	68.02	0.17	0.08	0.18	185.87	12.82	0.07	0.48	3.78
Minimum	0.00	0.00	0.00	0.00	0.11	0.03	0.00	0.01	0.05
Maximum	339.00	1.05	0.37	1.00	829.00	43.30	0.39	1.74	17.21
Number of samples	37	37	37	37	37	37	37	37	37

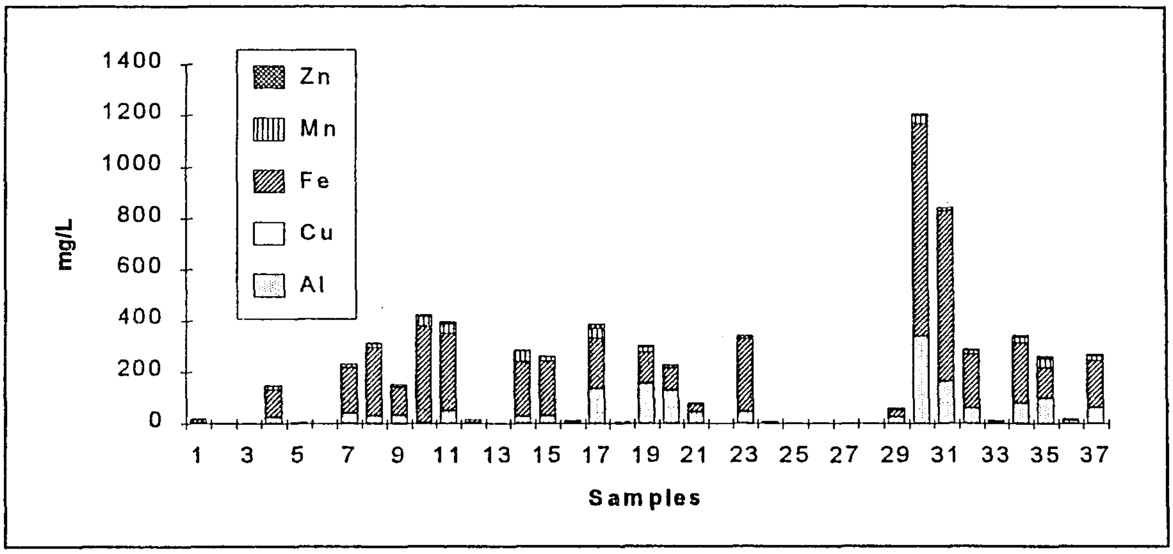


Figure 3.2. Bar chart showing heavy metal concentrations (mg/l) in the mine water.

The following conclusions are drawn from the above information:

- The wide range in values indicates the presence of areas within the mine that are actively recharged by rainfall, as opposed to other areas of very acid, saline and predominantly stagnant water.

- The minimum pH of 2,41 is one of the lowest pH-levels recorded during this investigation. This value lies just above the buffering range of goethite, which probably explains why pH-levels lower than 2,4 are not commonly found.

- The low concentration of calcium and magnesium in these waters indicate that most of the base potential from the flooded portions of the mine has been exhausted. It is likely that the calcium and magnesium concentrations in the waters will decrease with time, as the entire base potential is exhausted.

- Sulphate does not occur at saturation levels within any of the mine waters. This is ascribed to the active recharge and throughflow of water.

- Because of the low pH-levels in these waters, heavy metals occur in high concentrations. Aluminium, iron, manganese and zinc are the common heavy metals. Other tested heavy metals, which are not listed in the above table because of their insignificant concentration in the water, are arsenic, cobalt and selenium.

- Iron is the first heavy metal to precipitate as the water seeps from the mine. This precipitation generates further acidity within the mine effluent. Aluminium, manganese and zinc are carried in solution into streams. Mixing of the mine and stream water results in the precipitation of aluminium. Zinc precipitates are formed as the pH of the combined water continues to rise. Manganese remains in solution, because of its relatively high solubility up to a pH of 8.

- The high heavy metal content and low pH-levels of this water have resulted in periodic fish deaths in the Olifants River.
- Typical water quality in a stream, leading away from such a mine, is illustrated below. Six water chemistries, which were taken progressively downstream away from the mine over a distance of 1,5 km, have been included.

Table 3.2. Acid mine water draining from a shallow underground mine, affecting progressively, from positions No. 1 to No. 6, stream water.

No.	pH	EC	Ca	Mg	Na	Cl	SO4	Al	Co	Fe	Mn	Ni	Zn
		mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
1	3.41	120	60	38	46	66	562	27	0.92	2.7	25.5	0.39	0.74
2	3.44	158	93	59	47	61	923	57	1.19	3.0	32.5	0.71	1.37
3	3.41	174	107	64	43	52	1126	86	1.32	2.6	32.6	0.85	1.65
4	3.20	212	139	83	44	52	1460	110	1.63	5.6	37.5	1.13	2.05
5	3.10	237	160	104	39	43	1677	109	1.82	11.4	38.9	1.10	2.03
6	3.12	219	144	91	40	44	1509	107	1.58	11.8	36.7	0.94	1.95

The following conclusions may be drawn:

- The progressive deterioration of the water quality in a downstream direction is attributed to additional seepage through the weathered sediments, reaching the stream.
- This is one of the few instances in this investigation where significant nickel concentrations have been found in acid mine water. The source of this nickel, i.e. whether it is naturally occurring or whether it has been introduced through disposal into the mine, could not be determined.

Neutralisation of the acid water can be accomplished by lime dosing or some other means. One lime treatment plant presently operates in the area west of Witbank. This facility only treats water draining from the Kromdraai Mine.

In another area, acid seepage in a stream is unintentionally neutralised, by the discharge of treated municipal sewage into the same stream. The input and resultant chemistries for this system are presented below:

Table 3.3. Effect of mixing acid mine water with treated sewage effluent.

Water	pH	EC	Ca	Mg	Na	Cl	SO4	Al	Co	Fe	Mn	Ni	Zn
Source		mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
Sewage eff.	7.60	56	40	16	86	53	150	0	<0.01	0.0	0.2	<0.02	0.03
Mine water	2.85	285	373	130	28	5	2012	7	0.06	50.0	6.4	0.06	0.38
Mixture	6.75	92	66	26	81	51	308	0	<0.01	0.2	0.4	<0.02	0.07

The ratio by volume of sewage water to mine seepage is approximately 12:1. The main advantages of this system are:

- Dilution of the acid water,
- Increase of the pH of the acid water, and
- Precipitation of iron and aluminium.

Further downstream from this confluence, the water again mixes with acid seepage from other underground mines and the composite pH drops below 4.

Mine 2

The second example of shallow underground mine water quality originates from a mine that has been closed down during the past five years. About 40% of the mine is currently flooded. The mine has had a history of acid water, which has been pumped from the workings during mining operations. The present mine water chemistry, which was sampled by purging monitoring holes into the mine, is presented in Table 3.4.

Table 3.4. Current water chemistries in a shallow underground mine.

pH	EC	Ca	Mg	Na	Cl	SO4	Al	Co	Fe	Mn	Ni	Zn
	mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
7.70	70	150	23	6	1	363	<0.1	0.02	<0.06	0.86	<0.02	0.13
7.58	91	177	27	15	3	479	<0.1	0.02	<0.06	0.59	0.04	0.47
7.70	154	308	58	27	6	967	<0.1	0.03	<0.06	1.30	<0.02	0.57
6.78	228	561	107	15	3	1767	<0.1	0.04	21.00	3.20	0.10	0.36
7.69	174	350	78	16	6	907	<0.1	<0.01	<0.06	0.80	<0.02	0.10
7.45	104	189	28	5	2	508	<0.1	0.03	0.28	1.82	0.05	0.28

Several interesting aspects may be highlighted from these chemistries:

- The neutral pH is attributed to the exposure of the mine water to the base potential within the coal seam, as the mine progressively floods. Even though the base potential of the coal is low in comparison to its acid potential, flooding the mine progressively inhibits the acid-generating potential. The base potential within the flooded areas is then available to neutralise, at least on a temporary basis, acid that is produced within the rest of the mine.
- The high calcium content of the carbonates within the coal has enabled gypsum saturation to occur in at least one locality in the mine. This implies that the calcium and sulphate levels will not be able to rise significantly above their present concentrations at this point. The concentrations of all other elements will continue to rise. Magnesium is a particular concern with respect to the long-term water chemistry, because of its availability and high solubility within these environments (see also Figures 3.23 and 3.24 and subsequent discussion).
- The sodium content in this water is low. This mine water, being dominantly calcium sulphate water, may therefore be considered for irrigation on the sandy soils that are generally present in the Witbank Area. However, excessive utilisation of the mine water may lead to further acid generation and in extreme cases, the collapse of the pillars. Such an irrigation scheme can therefore not be implemented without detailed impact studies. Nonetheless, farmers realise the benefits that may be derived from the utilisation of this water and are tapping this resource in ever-increasing numbers. There seems to be no provision within current legislation to prevent or control the groundwater extraction from underground mines. Urgent attention will have to be given to the controlled utilisation of underground mine water by excessive abstraction, before undesirable effects are introduced on a large scale.

3.2.2 DEEP UNDERGROUND COLLIERIES

For the purpose of this document, deep underground collieries are defined as those, which do not mine into (other than for shafts or adits) or interfere with the weathered aquifer. Although the term 'deep' may therefore be a misnomer, it is convenient to use it in the geohydrological sense.

A variety of methods have featured in deep underground coal mining. Until the late seventies, underground coal extraction was predominantly conducted by bord-and-pillar methods. High extraction underground coal mining introduced techniques such as longwall, shortwall and stooping (pillar extraction) to these coalfields. To date, these high extraction methods have only been applied to a limited degree within the Olifants Catchment. The reason for this is the abundance of shallow coal, which can be mined by opencast methods.

3.2.2.1 Water quantity

3.2.2.1.1 Bord-and-Pillar Mining

Water within the deep coal mines in the Olifants Catchment is scarce. Although the coal is intensively fractured, very little water drains from these formations into the workings. Nonetheless, slow seepages are always present on the face in new developments. However, these seepages dry up within a few metres behind the face. This temporary release of groundwater into the mine is the result of pressure release withinthe coal aquifer. The hydraulic resistance within the coal is such that dewatering within the seam usually only extends a metre or two into the unmined areas. An average hydraulic conductivity for the No. 2 Coal seam of 0,1 m/d has been suggested in Section 2 of this document.

So-called 'wet' areas are intersected from time to time in underground bord-and-pillar mining. These areas are associated with tectonically more active zones, where the overlying or underlying rocks contribute a continuous supply of water into the mine. In some instances, these wet areas can be predicted on geological or structural grounds. Very few instances occur where development was stopped because of the amount of water flowing into the mine.

Apart from the water sources described above, the collieries can be regarded as dry. Large portions of the collieries are completely devoid of water seepage where the vertical percolation of groundwater is inhibited. This is attributed to the horizontal stratification of the Karoo Sediments, particularly the shale and Dwyka tillite which are situated above and below the coal seams respectively. In many areas, even sandstone does not permit water percolation, because of its massive and well-cemented nature.

Prediction of water influx into bord-and-pillar mining operations is difficult and often inaccurate. This is mainly due to the irregular occurrence of fissures in the mines. Water is allowed to accumulate in abandoned underground workings at all underground mines. It is therefore impossible to obtain a true reflection of the amounts of groundwater that flow into these mines, even if the pumpage was metered. Very few of the mines actually have operating flow measurement devices. As a guide-line, 0,5 - 1,0 Mℓ/d can be expected in the first two to four years of bord-and-pillar mining, eventually escalating to values between 2 to 4 Mℓ/d for the larger established underground mines. Values as high as 15 Mℓ/d have been recorded in exceptional cases. However, these values probably include water that has accumulated within the mine and which is removed for mining purposes.

A significant conclusion, however, is the fact that even though several megalitres of groundwater flow into each of the deeper underground mines on a daily basis, no instance could be found where the overlying aquifers were totally drained because of bord-and-pillar mining. This is significant in the sense that generally this form of mining does not deplete the farmers' groundwater resources.

3.2.2.1.2 High extraction underground mining

Longwall mining has been the predominant underground high extraction mining method in the Olifants Catchment to date. Shortwall mining experiments have been conducted, while pillar extraction has been performed on a small scale. In other areas of South Africa, high extraction underground coal mining is performed on a larger scale. For the purpose of this document, information on the effect of high extraction mining on groundwater will be drawn from all high extraction mining in the Highveld Area and not just from the Olifants Catchment.

Longwall mining

Longwall panels are 200 m wide on average. Panel lengths range from 1 000 - 3 000 metres. The overlying strata collapse in a circular fashion and major fractures extend all the way to surface. Borehole geophysical logs conducted in collapsed areas during this investigation indicate that the bottom 2 - 5 metres of the roof above longwall panels have extensively collapsed onto the floor of the panel. Porosity values for the collapsed material indicate the presence of 30% void space within the coal seam horizon. It has been observed during this investigation that total subsidence of the surface above longwall panels tends to be in the order of 50% of the mining height. The remainder of the void space, i.e. 20%, is therefore distributed within the zone between the extensively collapsed panel roof and the surface. Actual relationships could be much more complex than this and dependent on site-specific factors.

Influx of water occurs during the first collapse. The amount of water derived from the overlying strata is a function of time and the transmissivity of the adjacent strata. Typical ranges for influx values are demonstrated in Table 3.5.

Table 3.5. Range of influx volumes at longwall panels.

Days after first collapse	Influx ranges measured (m3/d)
1	4000 - 13000
4	800 - 4000
10	600 - 3500
40	500 - 3200
100	450 - 3000

The following interpretation can be made:

- The initial influx of water into longwall panels is comparatively high. This is due to the dewatering of strata immediately overlying the longwall panel.
- Influx of water into longwall panels continues indefinitely, while water is drained at the bottom. This is due to recharge from rainfall.
- Although the dewatering effect may be noticeable up to 1,5 km from longwall panels in exceptional cases, it is generally less than this maximum figure. The system is actually much more complicated than drawing a single interference line. Several aquifers overlie each other and each aquifer has its own dewatering characteristics. Situations therefore arise where the dewatering effect extends more than a kilometre from a longwall panel in one aquifer, while the other is barely affected. It is clearly a combination of factors that controls the distribution of the dewatering cones around longwall panels. The most important include the aquifer depth, recharge to the aquifer, aquifer transmissivity and associated boundary conditions.
- The inflow rates that are described in Table 3.5, represent the values for the first panels within specific areas. As subsequent panels are developed adjacent to the first panel, the influx rates increase due to drainage into the larger interconnected areas. A 10% increase in the influx for both panels has been observed where the dewatering cone around the first panel has actually stabilised before the mining of the second panel. This figure is, however, a gross generalisation and each instance should be evaluated individually.
- The inflow rates in the above table represent estimates at seven different mines. The true influx values could not be accurately established in any of these mines, because of

depression storage within the collapsed panels. The true influx values are therefore expected to be slightly higher than those reported above.

Shortwall Mining

Shortwall mining has only been conducted on a limited scale in South Africa. However, this mining method is likely to become more popular in the future, because of the greater mobility of shortwall equipment over that being used in longwall panels.

Shortwall mining also removes all support to the overlying strata, though over much smaller widths than in the case of longwall mining. In theory, the immediate roof of the coal seam will still collapse. Because of the small span of shortwall mining, the fractures into the overlying strata may not necessarily extend into the weathered aquifer. If this is the case, the long-term flux of water into the mine will be reduced significantly (probably by 40 - 60%). No information is available to quantify this impact more precisely.

Stooping

Stooping involves the extraction of coal pillars, after bord-and-pillar mining has been completed. Stooping has only been performed on a limited scale in the Olifants Catchment. It also results in limited collapse and possible drainage of the overlying strata. The quantity of water that flows into a stooped area is, amongst other things, a function of the horizontal extent of operations.

3.2.2.1.3 Modelling of groundwater influx into high extraction areas

Influx into underground high extraction areas has been modelled by Hodgson (1979), Hodgson *et al.* (1979), Hodgson *et al.* (1979), Hodgson *et al.* (1980) and Hodgson (1985). In all instances, finite element techniques were used to solve the governing flow equations.

A comparison of the predicted long-term influx rates and those actually recorded through measurement of the dewatering pumping is presented in Table 3.6:

Table 3.6. Predicted and actual influx at longwall panels.

	Predicted long-term influx	Actual long-term influx
	m3/d	m3/d
Mine 1	423	450
Mine 2	2000	2010
Mine 3	450	300
Mine 4	400	350

The high accuracy of these predictions is attributable to extensive field investigations at each of the mines.

Several shortcomings in the finite element technique had to be overcome during these simulations. These were:

- Simulation of the steep water-level gradients next to the fractured columns above the longwall panels.
- Flow of water from multiple aquifers into the fractured columns.
- Calculation of the influx through iterative procedures.
- Accommodating recharge from rainfall.

Steep water-level gradients

Because of the dewatering effect that longwall mining has on the aquifers adjacent to it, a significant drawdown of the groundwater levels occurs towards the fractured column. This can only be accommodated in a finite element model if the size of the elements decreases accordingly towards the fractured column. In all the simulations, element sizes were decreased logarithmically towards the fractured column. Elements were sized towards the fractured column on the following metre scale: 1000, 400, 100, 40, 10, 4, 1, 0,4 and 0,1. It was found that elements, as small as 0,1 m, were required to allow 90% accuracy in the finite element approximation of the analytical solution.

Multiple aquifers

At the stage that the modelling was done, 3-D capabilities were non-existing, because of the limited capacity of computers. To overcome this problem, the response of each overlying aquifer was calculated individually. Even today, this is still considered to be the best approach.

Influx calculation

The mathematics behind the calculation of water influx into high extraction areas has been detailed by Hodgson *et al.* (1985). While influx rates for homogeneous and isotropic systems can be calculated according to this work, influxes in the actual field situation differ markedly from these ideal situations. Numerical simulations of influxes are therefore required. This is accomplished through the finite element method. Care should be taken that sufficient detail in the finite element network design is provided for in the area immediately surrounding a collapsed panel. Experience during these simulations has shown that at least 35 nodes are required along the length of a panel.

Recharge by rainfall

Recharge from rainfall and run-off introduced the greatest uncertainty into the simulations. Rainfall was incorporated into the models by assuming that a certain percentage of the rainwater is recharged to the groundwater. This assumed percentage was readjusted after the first collapse during refinement of the models.

In summary, it can be stated that the numeric simulation of groundwater influx into underground areas of high extraction is cumbersome and complicated. The models sited above have indicated that a high degree of accuracy can be achieved on condition that accurate field data are available, that the problem is well conceptualised and that there is an understanding of the constraints of the finite element method.

3.2.2.2 Water Quality

Water qualities within deep underground mines can be related to the mining method as described below. Other factors that also contribute to the water quality are:

- The quality of the source of the water,
- The rate at which this water enters into the mine,
- The rate at which the mine is flooded,
- The acid-base relationship on the coal seam horizon, and
- The period of retention within the mine.

3.2.2.2.1 Water quality in bord-and-pillar mining

Bord-and-pillar mining causes minimum rupture of the aquifer. Water seeps into the mine as a result of pressure release during mining and also as free flow in larger fractures. Two water qualities can be identified in this respect. The first is interstitial water, which is in equilibrium with its surroundings. This water usually contains sodium and chloride as its dominant ions. The salt load contribution from this water is very small, because of the small quantities of water released in this manner. Most of the mine water is derived from fissure flow. This dynamic system does not allow the concentration of salts within the water, with the result that the latter has salinities equivalent to that of the deep aquifer, as discussed in Section 2.

Stagnant pools of water in the mine deteriorate in quality. This could be due to pyrite oxidation and the resultant neutralisation of the acidity by carbonates within the coal seam. A linear relationship between sulphate and calcium concentrations is observed during this stage. When water has depleted the base potential of the coal seam in a specific puddle, the sulphate concentration continues to increase, without the introduction of additional calcium. At that stage, the buffering capacity of the calcium carbonate in the coal seam has been depleted and the pH will drop to the next level of buffering, i.e. that of siderite, which lies between 5 - 6 pH. A plot of calcium versus sulphate concentrations for mine water will therefore indicate areas where the acidity has exhausted the calcium carbonate base potential.

Statistics on the chemistry of water samples from a deep bord-and-pillar coal mine are included below:

Table 3.7. Water quality in a deep underground bord-and-pillar colliery

Statistics	pH	EC	Ca	Mg	Na	M. Alk.	Cl	SO4
		mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
Mean	8.06	164	186	89	120	140	13	910
Median	8.09	136	126	60	104	145	10	647
Standard Deviation	0.25	109	152	84	75	46	7	773
Minimum	7.28	49	30	14	35	66	6	127
Maximum	8.55	435	482	354	379	178	31	2780
Number of samples	20	20	20	20	20	20	20	20

Conclusions are:

- The elevated sulphate concentrations in the samples indicate that the coal seam has a high acid-generating potential. However, the alkaline pH-level of the water samples suggests that the coal seam also has sufficient base potential to neutralise the generated acid at this point in time.
- A plot of total calcium and magnesium against sulphate concentrations indicates a linear relationship (Figure 3.3). This confirms that the dolomitic base potential within the coal seam exceeds the acid that has been generated to date at all the localities within the mine.

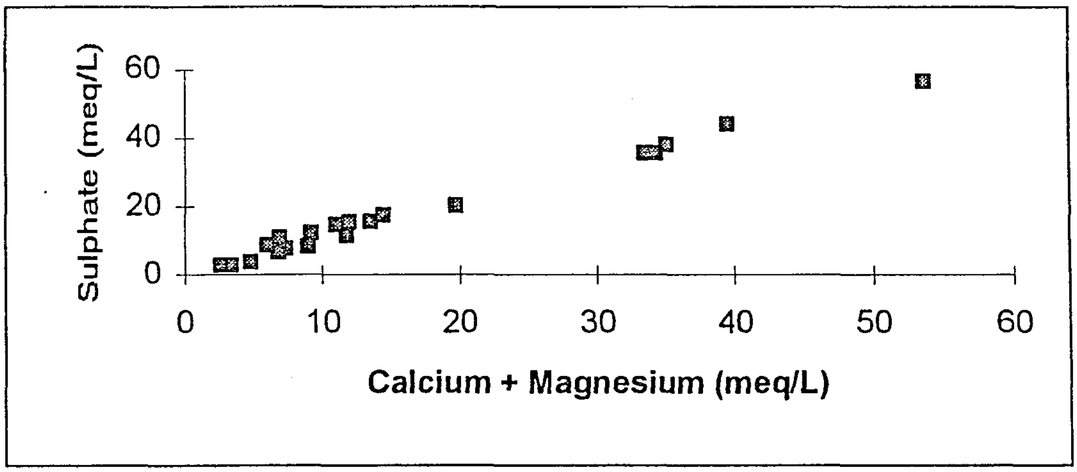


Figure 3.3. Plot of total calcium and magnesium versus sulphate concentrations in water from a deep underground coal mine.

- The average ratio of calcium plus magnesium, versus sulphate is 0,92:1. This suggests that other means of buffering the acid, though slight, are also present. This could be in the form of carbon dioxide from the air, or natural alkalinity of the water that flows into the mine. The relationships between alkalinity and sulphate versus pH are demonstrated in Figure 3. 4.

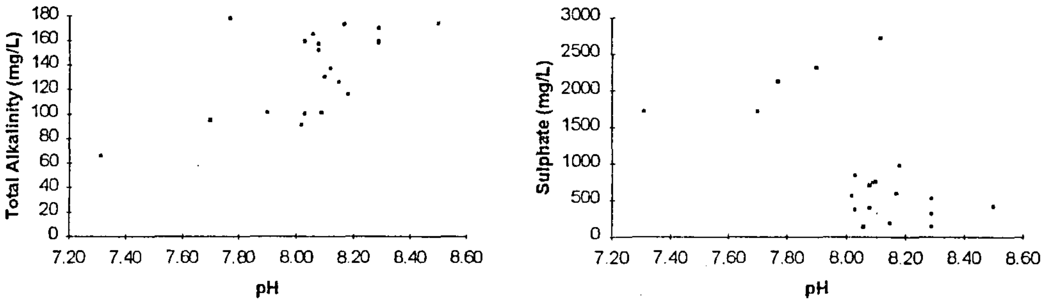


Figure 3.4. Relationship between pH versus total alkalinity and sulphate in water from a deep underground coal mine.

The above example could be regarded as typical of waters in many of the operating bord-and-pillar collieries. The significant variations in the water quality are a function of the dynamics of the water within the mine. Areas actively recharged or where water is removed as soon as it flows into the mine, have better water qualities than those where the mine water stagnates. The worst water quality is usually found within the small puddles that are isolated from the remainder of the groundwater regime. Here an ample supply of oxygen and bacterial activity accelerates the acid-generating reactions, which soon depletes the base potential of the coal locally.

At least one underground colliery exists where active underground recharge from a stream is practised during summer months. This water is used to supplement domestic and mine water requirements during low-flow conditions in the stream. The quality of the mine water in this compartment remains similar to that recharged into the mine, because of the dynamic nature of the system.

3.2.2.2.2 Water quality in high extraction areas

Water quality in high extraction underground mining (Figure 3.5) differs significantly from that in bord-and-pillar areas. To explain this discrepancy, the following environments are identified:

- Water, which drains towards and into the collapsed columns above high extraction areas, undergoes base exchange. Calcium and magnesium are adsorbed onto clays within the sediments, in exchange for sodium. An increase in the sodium percentage in the water is therefore present in these areas. X-ray diffraction analyses of the clays in the Olifants Catchment indicate that kaolinite is the dominant mineral. With an average cation exchange capacity of 10 meq/100 g of clay, it can be demonstrated that the exchange capacity of the clay within a dewatering radius of 1 km around a high extraction area, is sufficient to bind all the calcium of the groundwater flowing into a fractured column for 40 years. Apart from kaolinite, other minerals such as chlorite also occur in the rock, which will add to the base exchange capacity of the system.
- Interstitial release of sodium from the shale occurs as the water falls down the fractured columns.
- The degree to which cation exchange has affected the chemistry of the water, is a function of the distance which the water has travelled and the composition of the rock through which it has moved. The resultant water that ends up on the coal seam horizon

ranges from water almost totally devoid of calcium and magnesium, to water, which may be slightly enriched in sodium.

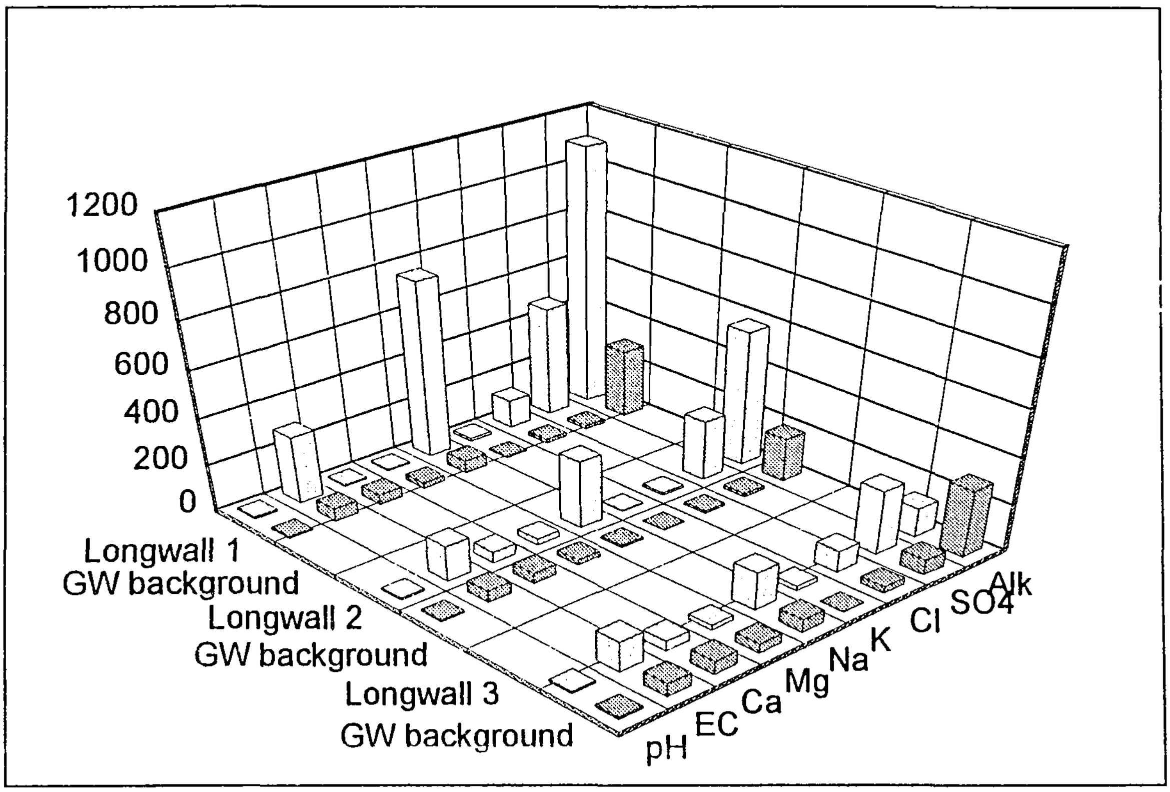


Figure 3.5. Typical background chemistries for groundwater before longwall mining, compared to water chemistries of water from longwall panels.

- The conclusion is drawn that the initial character of water in high extraction areas is that of high sodium water. Disposal of this water on surface is undesirable because of the sodium content.
- Water is left to accumulate in the worked out areas, following high extraction mining. Here a gradual deterioration of the water quality occurs as pyrite is oxidised in the fractured column and on the coal seam horizon. An increase in sulphate, calcium and magnesium in the water is the result.
- The panel waters currently have alkaline tendencies, despite the high acid potential of the seams as reflected by the elevated sulphate concentrations. This is due to the availability of carbonates within the crushed rock in the panels.

3.2.3 WATER MANAGEMENT STRATEGIES IN UNDERGROUND COLLIERIES

Many management strategies have been devised to cope with underground water over the years. These techniques ultimately relate to the environmental protection of water sources outside the mines. A selection of the many available measures and scenarios were identified for further discussion in this document. They are as follows:

- Seepage of acid water from shallow mines.
- Handling of water within longwall panels.

- Disposal of underground mine water in evaporation areas.
- Disposal of underground mine water into old workings.
- Disposal of coal slurry, power station fly ash and sewage in old workings.
- Flooding the underground workings from external water sources.

Seepage of acid water from shallow mines

A single liming plant exists in the whole Olifants Catchment. Other seepages from underground mines are at best partially controlled by earth dams or not controlled at all.

Liming acid mine drainage raises the pH to above 6, thus precipitating all the heavy metals, with the exception of manganese. However, this pH is insufficient for the precipitation of magnesium for instance, which is the only major cation that can be removed by raising the pH further. Furthermore, the calcium content in this water is insufficient to precipitate gypsum. The resultant effluent therefore has a higher salt concentration than the original water.

Dams that are designed to contain acid seepage from underground mines are not effective. All those studied leak into the weathered aquifer. This water then moves down gradient and emanates into the nearest stream. The little water that is contained within the dams concentrates through evaporation, thus aggravating the salinity problem.

Uncontrolled acid seepages are plentiful from defunct mines in the Witbank Area. These have been quantified by the study of Wates, Meiring and Barnard (1993). The severity of the problem lies in the acidity of the water and the resultant heavy metal content. There is no economic solution to this problem at present.

Handling of water within longwall panels

Handling of water within longwall panels is normally more of a nuisance than a problem. Volumes that enter into the workings are small enough to be handled by moderate sized equipment. In areas where the slope of the coal seam is away from the direction of mining, no problems are usually encountered. This is ascribed to the fact that the coalface advances faster than the build-up of water in the collapsed area.

Mines have therefore been advised that, from a safety and water handling point of view, the first panel in any new area should proceed up-dip for at least the first 400 metres. This allows the creation of enough emergency water storage space behind the face. In the event that the inrush is abnormally high, sufficient time would be available to introduce an emergency dewatering scheme through boreholes from surface. Such drastic precautions have been unnecessary within the Olifants Catchment to date. Water is usually left to accumulate in collapsed areas behind the face.

At most mines, the coal pillars between longwall panels are also mined or are crushed under the weight of the overlying strata. The whole of the longwall area may therefore be regarded as an underground lake, full of crushed rock. The porosity of this mix has been quantified by dewatering one such a lake and a value of 30% void space has been obtained. Because of elevation variations within the coal seam, not all of the longwall areas fill with water. The level to which it fills, is regulated by the overspill level which connects the longwall area to development or, as in the case of some of the mines, the brick walls which have been built to keep the longwall water from interfering with the development.

Disposal of underground mine water in evaporation areas

Two practices exist for the disposal of underground mine water in evaporation areas. The one is into natural pans and the other into constructed evaporation areas.

Many natural pans occur in the Olifants Catchment. These pans are usually located on topographic highs, such as the ridges separating one catchment from another. Most of these pans are so-called fresh water pans. This suggests a downward flux of water and salts into the underlying strata. The latter is to be expected, in view of the high topographic location of these features. Water that is disposed of in these pans is therefore certain to permeate into the underlying weathered aquifer.

Although constructed evaporation areas are limited in the Olifants Catchment, the use of these facilities is expanding. Seepage from these areas into the weathered aquifer occurs. The success of these areas is based on the unplanned seepage losses to the groundwater, which prevents the build-up of salts that would otherwise occur. The build-up of salts to very high levels in these systems is not observed in any of the present evaporation systems.

Disposal of underground mine water into old workings

Disposal of underground mine water into old workings is presently accepted as the preferred manner of effectively controlling and containing mine effluent. Several mines have, or are in the process of setting aside mined-out underground areas, which will be used for the disposal of mine water from other areas. The practical restrictions that apply to these systems are:

- In terms of available space, the natural groundwater influx into these areas should be small in comparison to the amount disposed of.
- That these areas can naturally or artificially be separated from the rest of the underground workings. Natural barriers may occur in the form of a roll in the coal floor, separating portions of the mine from one another, or mine excavations that are not interconnected. Artificial seals may be installed in roadways. These are expensive and have to withstand significant hydraulic pressures. Of the two, natural seals are preferred.
- In the installation of artificial seals, the natural hydraulic conductivity of the coal has to be considered. Detailed geological and mechanical investigations are required to establish the integrity of such a system. Typical hydraulic conductivities, with an average of 0,1 m/d, for the No. 2 coal seam have been quoted in Section 3. It can be demonstrated on the basis of these values, that a seepage of 0,4 $m^3/d/m^2$ is possible through coal under the following conditions:

Coal hydraulic conductivity	0,1 m/d
Thickness of the coal pillar separating the flooded compartment from development within the mine	25m
Water level above flooded compartment floor	100m

The secret behind the successful flooding of compartments within close proximity to any development, is therefore:

- To perform initial feasibility studies,
- To initiate flooding, and
- To regulate the level of water within the flooded compartment according to the anticipated seepage tolerance within the development.

Dolerite sills of significant thickness (20 - 40 m) cut through the coal seams in certain areas. These sills are predominantly impermeable and artificial water seals can be installed within these rocks with great success.

Disposal of coal slurry, power station fly ash and sewage in old workings

Coal slurry, if deposited within water, cannot oxidise under neutral pH-conditions. The subaqueous deposition of coal may therefore be applied as a successful management option at all deep mines where the slurry can be flooded totally.

Power station fly ash has been introduced at one deep colliery to support the roof of mined-out areas. The power station fly ash has a high base potential. This material, together with the alkaline water that has also been introduced into the mine, improves the chemistry of underground mine water. Gypsum and brucite saturation and precipitation are observed. This precipitation lowers the salt content of the mine effluent by 40%.

Sewage effluent disposal into underground workings has occurred at one mine within the Olifants Catchment. Thirty per cent of the compartment was flooded with this water. The level of water within the mine was regulated by seepage into two adjacent compartments to the west and north, an underground fire on the south side and a seepage level into the weathered aquifer to the east. Results of water chemical analyses from this mine indicate negligible amounts of nitrate and ammonia in water seeping from the system. A well-controlled sewage disposal system into an underground mine will help to keep the salt concentrations low. The reason for this is twofold. The most obvious effect is the dilution of the mine water. Also of importance, is the introduction of organic material, which is utilised during bacterial (*Desulfovibrio*) reduction of sulphate within the mine water. Anaerobic, neutral conditions are thus created and oxidation of pyrite is retarded.

Flooding the underground workings from other external water sources

Flooding underground workings from other external water sources has been considered in a number of instances. Two sources have been utilised in the past. These are surface water or industrial effluent.

The introduction of surface water into an underground mine to eliminate long-term pyrite oxidation or for controlled re-use during low-flow periods, is an acceptable practice. Significant surface streams cross most underground workings. It should be theoretically possible to divert surface run-off into most deep underground mines and flood them in a matter of years.

3.3 WATER IN OPENCAST COLLIERIES

3.3.1 THE OPENCAST MINE ENVIRONMENT

3.3.1.1 Mining methods and significance

Pre-stripping of soil in accordance with guide-lines set out by the Chamber of Mines (1981) is a standard procedure at all large opencast collieries and at the majority of the smaller ventures.

The overburden above the coal is removed by dragline at the large pits and by truck and shovel at smaller operations. Truck and shovel methods allow selective handling of spoil, since material may be transported, dumped and even compacted in areas specifically allocated for spoil of specific compositions. To date, selective spoil handling has not been practised to any significant degree within the Olifants Catchment. In the case of dragline mining, the stratigraphic column is inverted during mining operations.Dragline operations can, at most, do limited selective spoil handling by dumping at different angles with respect to the operating cut.

Mining depths normally range from 0 - 40 m below surface. Only one of the collieries has reached a depth of 80 m below surface in a staged dragline operation.

The large opencast collieries mine between approximately 7 Mt and 14 Mt per annum as run of mines. During this operation, an average of 50 Mt of overburden is moved and replaced per large colliery per annum. This amounts to approximately 400 Mt spoil per annum for opencast mining within the Olifants Catchment.

Many opencast collieries produce coal for power generation. In most of these instances, the entire coal product is delivered to the power station. Eight power stations presently exist within the Olifants Catchment, of which two have been mothballed on a temporary basis. One of the power station collieries produces coal for the power station as well as for the export market. Export collieries may use heavy medium separation for the beneficiation of coal. The waste from coal beneficiation is called discard and may consist of carbonaceous shale and similar products, usually with a high pyrite content. These may be mixed with breaker discards and are disposed of at the mines, mostly on surface.

3.3.1.2 Water sources in coal opencast mines

3.3.1.2.1 Water volumes

The water within opencast coal mining is derived from various sources. Table 3.8 lists a breakdown of these sources as a function of the average annual rainfall or the total ingress of water into a pit:

Table 3.8. Water recharge characteristics for opencast mining.

Sources which contribute water	Water sources into opencast pits	Suggested average values
Rain onto ramps and voids	20 - 100% of rainfall	70% of rainfall
Rain onto unrehabilitated spoils (run-off and seepage)	30 - 80% of rainfall	60% of rainfall
Rain onto levelled spoils (run-off)	3 - 7% of rainfall	5% of rainfall
Rain onto levelled spoils (seepage)	15 - 30% of rainfall	20% of rainfall
Rain onto rehabilitated spoils (run-off)	5 - 15% of rainfall	10% of rainfall
Rain onto rehabilitated spoils (seepage)	5 - 10% of rainfall	8% of rainfall
Surface run-off from pit surroundings into pits	5 - 15% of total pit water	6% of total pit water
Groundwater seepage	2 - 15% of total pit water	10% of total pit water

These statistics on recharge values have been taken from observations at nine opencast collieries within the Olifants Catchment. These values require further clarification:

- Not all rainfall that precipitates within ramps and voids contributes to the overall water balance. Much of this water is accumulated in local depressions, from where it evaporates. Many of the ramps and voids are likely to remain dry, even after a pit has closed down and has filled with water to the decanting level. It can be concluded that the percentage recharge through ramps and voids is a function of the slope of the floor of the pit and the degree to which these structures are filled with water. With the average evaporation potential from water exceeding the annual rainfall by 700 - 800 mm in the Olifants Catchment, ramps and voids with standing water should therefore have a beneficial effect on the water balance.
- Unrehabilitated spoil heaps constitute a significant percentage of the disturbed areas within South African coal mines. A survey of opencast collieries in the Olifants Catchment indicates that rehabilitation lags 2 - 10 cuts behind the operating cut per dragline operation. These spoil heaps have a high rainfall recharge potential. The lack of erosion scars on the spoil heads suggests that rain water penetrate into spoil without much obstruction. Optimally, from the point of view of water control, rehabilitation should follow within 2 cuts of the operating cut.

- Spoils are usually levelled by dozing equipment. This often results in compaction of the upper portion of the spoil, particularly where a high percentage of shale is present. Sandstones and siltstones are usually not crushed during dozing and permeation of water through this material is still possible, though usually at a reduced rate. Run-off from levelled spoils easily erodes scars into the spoil. Erosion channels permit uncontrolled water recharge to the pit. With time, levelled spoils become less permeable, because of decomposition of the argillaceous material and silting up of channels. It can be concluded that the vertical hydraulic conductivity of levelled spoils is highly variable and dependent on the amount of compaction, surface slopes, spoil composition and age.
- Levelled spoils are usually covered by soil. The soil is vegetated with different seed mixtures varying from one company to another. Mixtures are designed to establish a quick growth, allowing the sword to develop over a number of years. The evapotranspiration properties of the soil and vegetation differ between mines and seasons. Areas exist where grass growth exceeds one metre in height. At some of the collieries, channelling of surface run-off has, in places, eroded through the topsoil, thus exposing the underlying spoil. Infiltration in these areas is envisaged to be variable, ranging from high to low.
- Surface run-off from the surrounding areas is usually diverted away from the pits by cut-off trenches. Despite these precautionary measures, some overland run-off still enters into the pits. Through good management, the contribution of water from this source should be small compared to the total influx of pit water.
- Groundwater seepage into opencast coal mines, as a percentage of the total water in the pit, is small. As indicated in the discussion on geohydrology and borehole yields, the Ecca rocks yield very little water. Groundwater influx is mainly from the upper weathered aquifer which, in turn, is recharged by rainfall. Groundwater seeps into the pits at the bottom of the weathered aquifer or from weathered zones next to the odd dolerite dyke, from sporadic fractures in the unweathered Ecca sandstones lower down and from the coal seams themselves. These seepages are negligible in comparison to the scale of mining and other water sources within the pit.

3.3.1.2.2 Dewatering of surrounding aquifers

The dewatering effect of opencast mining on the adjacent strata, as a result of groundwater influx into the pits, usually extends less than 40 m ahead of mining.This is substantiated by the necessity to perform pre-spitting, to drain water from blasting holes ahead of the highwalls in many of the pits.

3.3.1.2.3 Hydraulic characteristics of the spoil

3.3.1.2.3.1 Porosity

The porosity of the Ecca sediments, in their natural state, ranges from 5 - 12% (average 9%, sample size 30). Although the pores in the rock may be saturated with water, it does not drain under the influence of gravity alone, due to retention of interstitial water by capillary forces. The drainable or effective porosity of the Ecca sediments in their unweathered state is therefore negligible. Fractures that dissect the sediments are the main water release mechanism. The amount of water stored within fractures is a small percentage of the total rock volume. Pumping tests performed on such structures indicate that the percentage of the rock mass that contains drainable water is 0,01% on average.

As rock is removed and replaced during mining, an increase in total volume from the natural state to the uncompacted state, results. The increase in volume is a function of the state of weathering of the rock and its composition. Sandstones generally have a larger bulking factor than shale, because of its larger average rock size.

Bulking factors of 22 - 30% (average 26%) of the original rock volume are commonly used in the coal industry for the projection of final post-mining spoil surfaces.

Since the rock matrix has practically no effective porosity, the bulking factor may be used as a measure of the total porosity of spoil. Measurements and calculations performed at a number of flooded pits indicate that the drainable porosity is much lower than the total porosity of the spoil. This is attributed to the retention of water within the fine spoil material, which usually has a high clay content. Effective porosities in the order of 5 - 10% of the total spoil volume, have been measured. This has been established in the field by draining spoils. This implies that when pits fill up, 260 litres of water will be absorbed for every cubic metre of spoil, while only 80 litres (on average) will be released, should the water level drop.

3.3.1.2.3.2 Hydraulic conductivity of the spoil

As part of this investigation, only two pumping tests were performed in boreholes that were drilled into spoil. The pumping rate has been 6,5*l*/sec (30 000 *l*/h) in both instances. Water flow into the boreholes was so great during the tests that no measurable dewatering cone developed in the pumping holes. The conclusion is drawn that for the site in question, the basal portion of spoil is highly permeable. The high hydraulic conductivity of the basal portion of the spoil can be attributed to the way in which the dragline dumps spoil, so that large boulders roll to the bottom of the pit. Dozer does also not rework the bottom portion of the spoil after dumping and no compaction of this material therefore occurs. Radiometric logs in spoil boreholes have also confirmed the presence of large cavities that exist within the spoil.

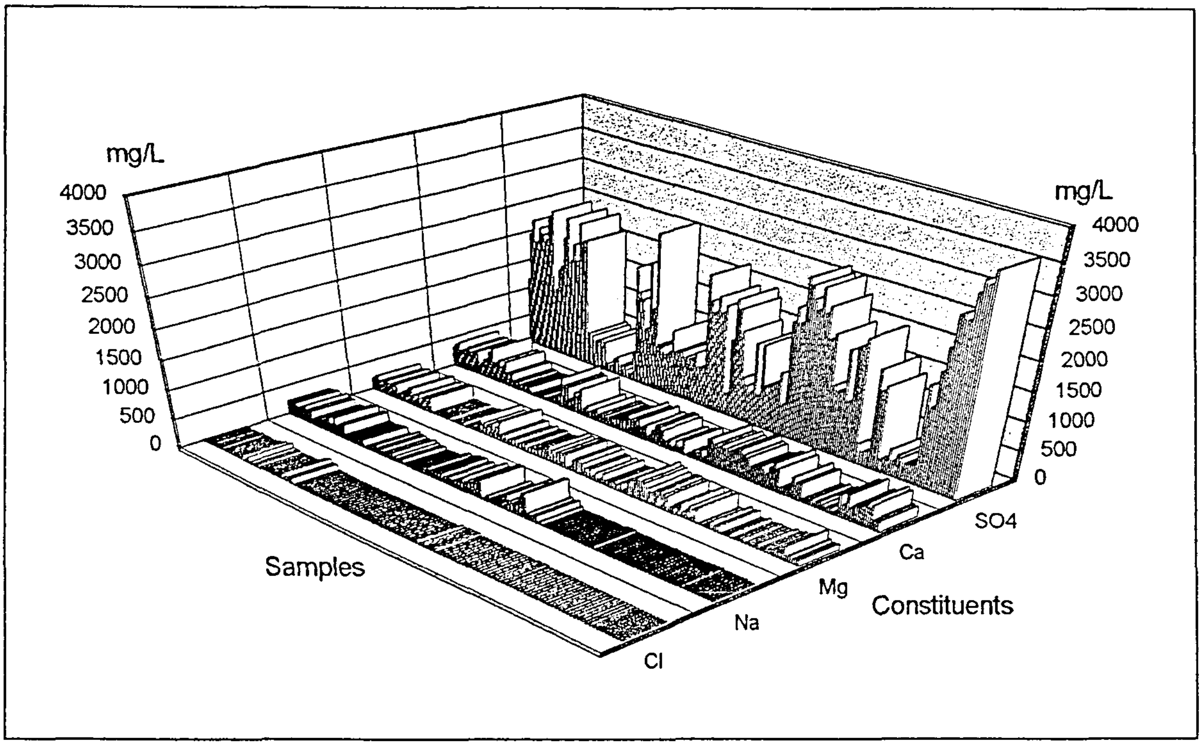
The high hydraulic conductivity within the spoil results in horizontal water levels within the pits. This has been confirmed through measurements in all the pits monitored during this study.

3.3.1.3 Spoil-water quality

Water quality within spoil varies greatly between localities. The variations of spoil-water quality are demonstrated in Figure 3.6.

The following characteristics of the spoil water should be emphasised:

- Although natural groundwater occasionally contains high chloride levels, elevated levels of chloride are seldom found within the opencast pits. High concentrations of this element are associated with natural evaporation pans, which are incorporated into the spoil material by mining activity.
- The source of sodium in opencast mines is twofold. A slight increase in the sodium content has been observed in monitoring holes adjacent to some of the pits. This is ascribed to cation exchange as groundwater drains towards the mine excavations. However, the dominant source of sodium is undoubtedly the spoil itself. All naturally occurring sodium salts are water-soluble. Water that seeps through the spoil will immediately dissolve all the available sodium from the outer surface areas of the spoil fragments. As the spoil subsequently decomposes over many years, a steady supply of sodium will become available. Leaching tests on the spoil indicated that the sodium dissolution potential is 0,04 to 0,11 mg/g.
- Elevated sodium concentrations within the sediments are found at some of the collieries. The Highveld Coalfield sediments generally have a higher sodium content than those of the Witbank Coalfield.

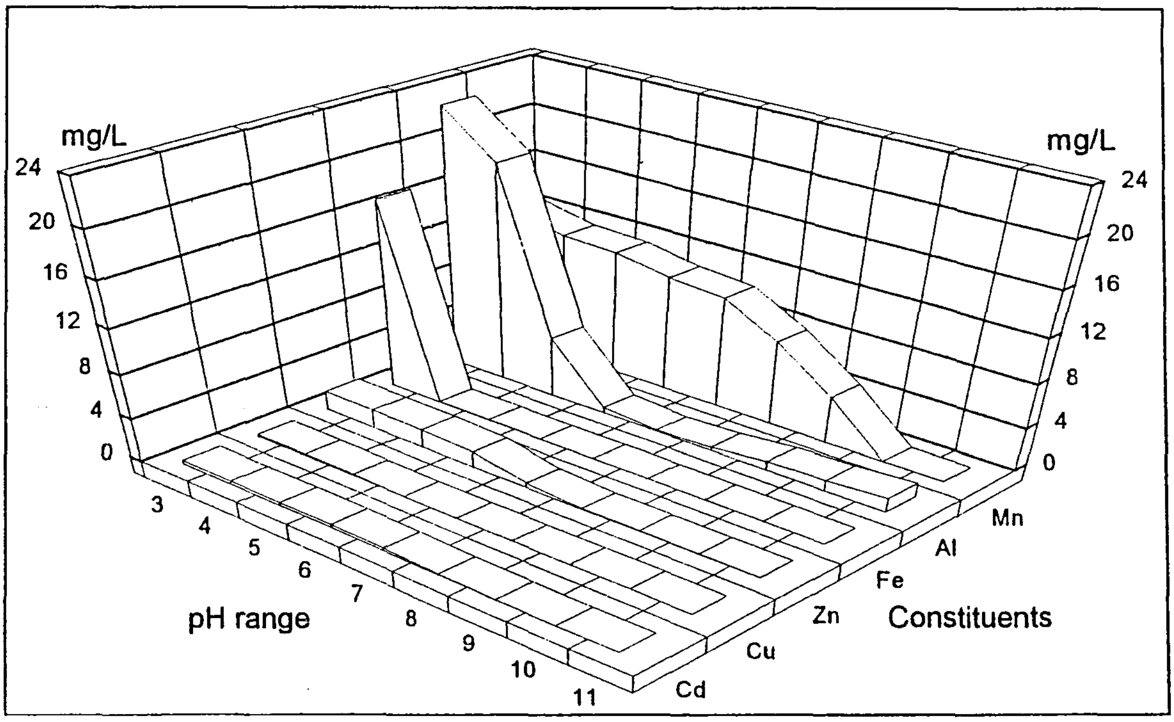


Statistics	Cl	Na	Mg	Ca	SO4	Fe	Al	Mn	Zn	Cu	pH	EC
	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l		mS/m
Mean	19	109	160	261	1465	84	2	8	0.40	0.02	7.16	247
Median	11	84	142	249	1426	1	0	3	0.08	0.01	7.60	233
Mode	12	19	20	188	1990	1	0	0	0.07	0.00	7.61	346
Standard Deviation	24	95	95	136	1073	313	9	12	1.12	0.08	1.49	130
Minimum	0	3	19	33	108	0	0	0	0	0	2.6	50
Maximum	147	435	387	570	3500	2596	70	52	6.1	0.71	8.72	700
Number of samples	126	126	126	126	126	126	126	126	126	126	126	126

Figure 3.6. Spoil-water quality in boreholes drilled into spoil.

- Calcium and magnesium are mostly derived from the neutralisation reactions between acidic leachates and the calcium and magnesium carbonates within the sediments. The ratio of calcium:magnesium is 1,63:1 in the spoil water, which is almost identical to the ratio of calcium:magnesium in dolomite [$CaMg(CO_3)_2$], which is 1,66:1. This confirms that the carbonates within the sediments are dolomitic in nature, and not calcitic [$CaCO_3$], as was generally believed. This has far-reaching implications in terms of solubility limits of gypsum in the spoil water, which in turn dictates sulphate concentrations in the water. These aspects will be discussed later.
- A wide range of sulphate concentrations is present within the spoil waters in well-established pits. Sulphate concentrations are high in areas of water stagnation (range 2 000 to 3 000 mg/l). The sulphate levels are much lower in areas where the dilution of spoil water occurs as a result of recharge. The sulphate concentration level at which the spoil water becomes saturated with respect to gypsum, is dependent on the concentrations of the various cations. At calcium levels of 600 mg/l, the maximum level for sulphate solubility is 1 400 mg/l. At an average level of 263 mg/l calcium, sulphate concentrations may theoretically rise to 8 000 mg/l. The range of calcium values listed in Figure 3.7 indicates that none of the samples have reached gypsum saturation limits.

It is concluded that if dilution from rainfall is excluded from the pits, the concentration of sulphates and other elements will increase.



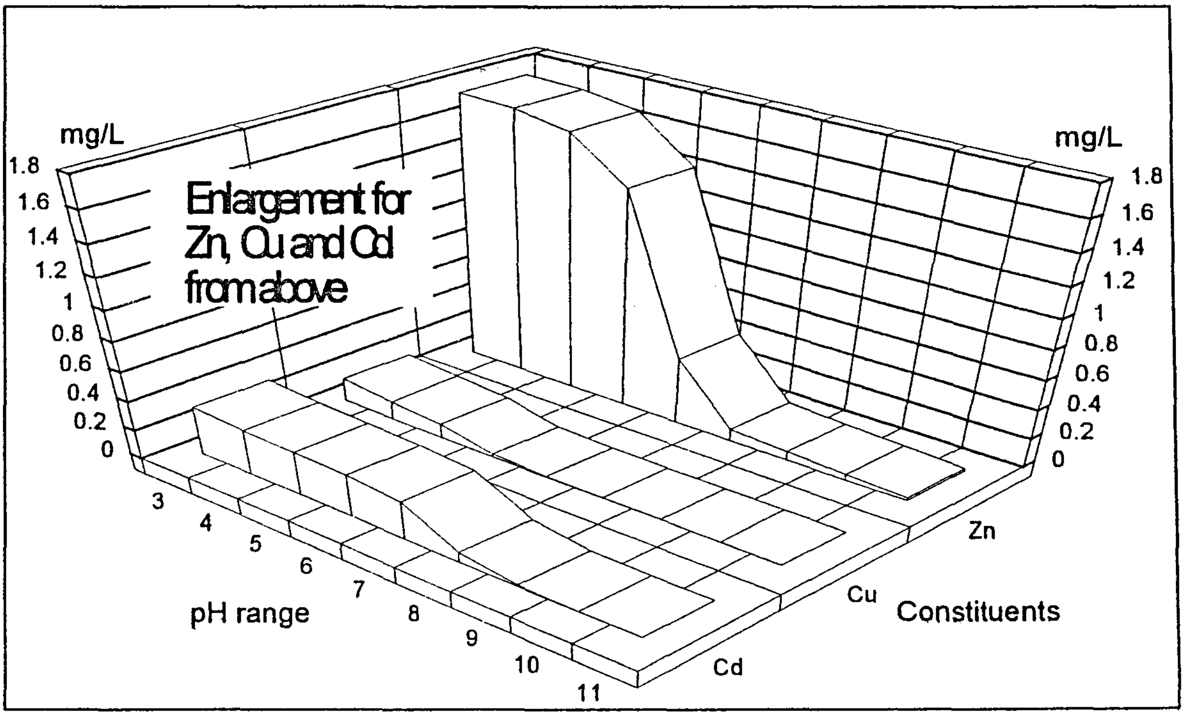
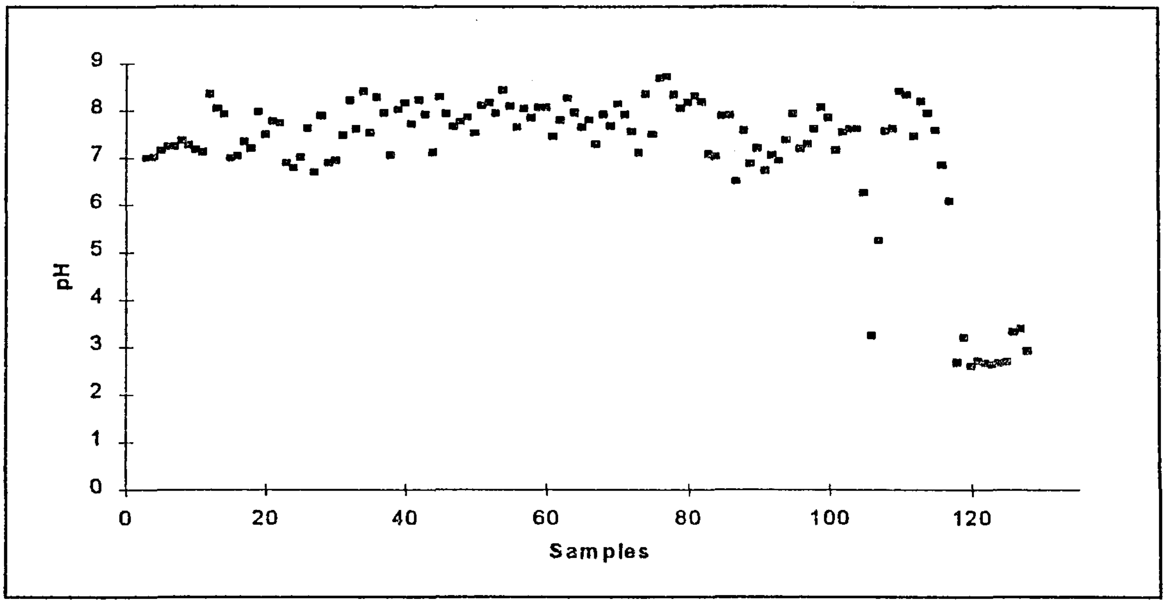


Figure 3.7. Heavy element concentration drop in a laboratory experiment, starting with acid spoil water at a pH of 3, titrating upwards to a pH of 11 with NaOH.

- The only common heavy metal to be found in neutral spoil waters is manganese. The solubility of manganese only drops off at pH-levels from above 8. Other heavy metals commonly found in acid mine water are aluminium, cadmium, copper, iron and zinc. Solubility graphs for these elements versus different pH-levels, as determined by titration, are included for the reader's reference below. These graphs illustrate the pH-levels at which the various heavy metals precipitate from spoil water. Collieries where acid treatment is done through the addition of lime, usually raise the pH of the mine water to a level between 6 - 7. This is sufficient to remove all heavy metals, with the exception of manganese and, to a certain extent, cadmium.

- The pH of the spoil waters in the Olifants Catchment is neutral, with the exception of one colliery. This is illustrated in Figure 3.8.



Statistics	pH
Mean	7.16
Median	7.60
Mode	7.61
Standard Deviation	1.49
Minimum	2.60
Maximum	8.72
Number of samples	126

Figure 3.8. pH-values as measured in spoil monitoring holes.

Water seepage out of the spoil is often characterised by an acidic pH. This is ascribed to the acidity which is produced through the oxidation of dissolved iron. Vertical stratification of water quality has been observed in the majority of the monitoring boreholes. Figure 3.9 illustrates this phenomenon by means of electrical conductivity graphs, which have been measured in three boreholes.

Two significant features are evident in these electrical conductivity profiles.

- The conductivity at the water surface is higher than one metre below the surface in all of the profiles. This is attributed to the reaction of oxygen and carbon dioxide with the water. A reaction, which can be expected at the water/air interface, would be the oxidation of ferrous to ferric iron. This produces a drop in the pH of the water, which is, in turn, neutralised by the base potential of the rock. The result is an increase in the salinity at the top 0,6 metres of the water column, which is reflected by the higher electrical conductivity. The higher electrical conductivity only occurs within the borehole itself.

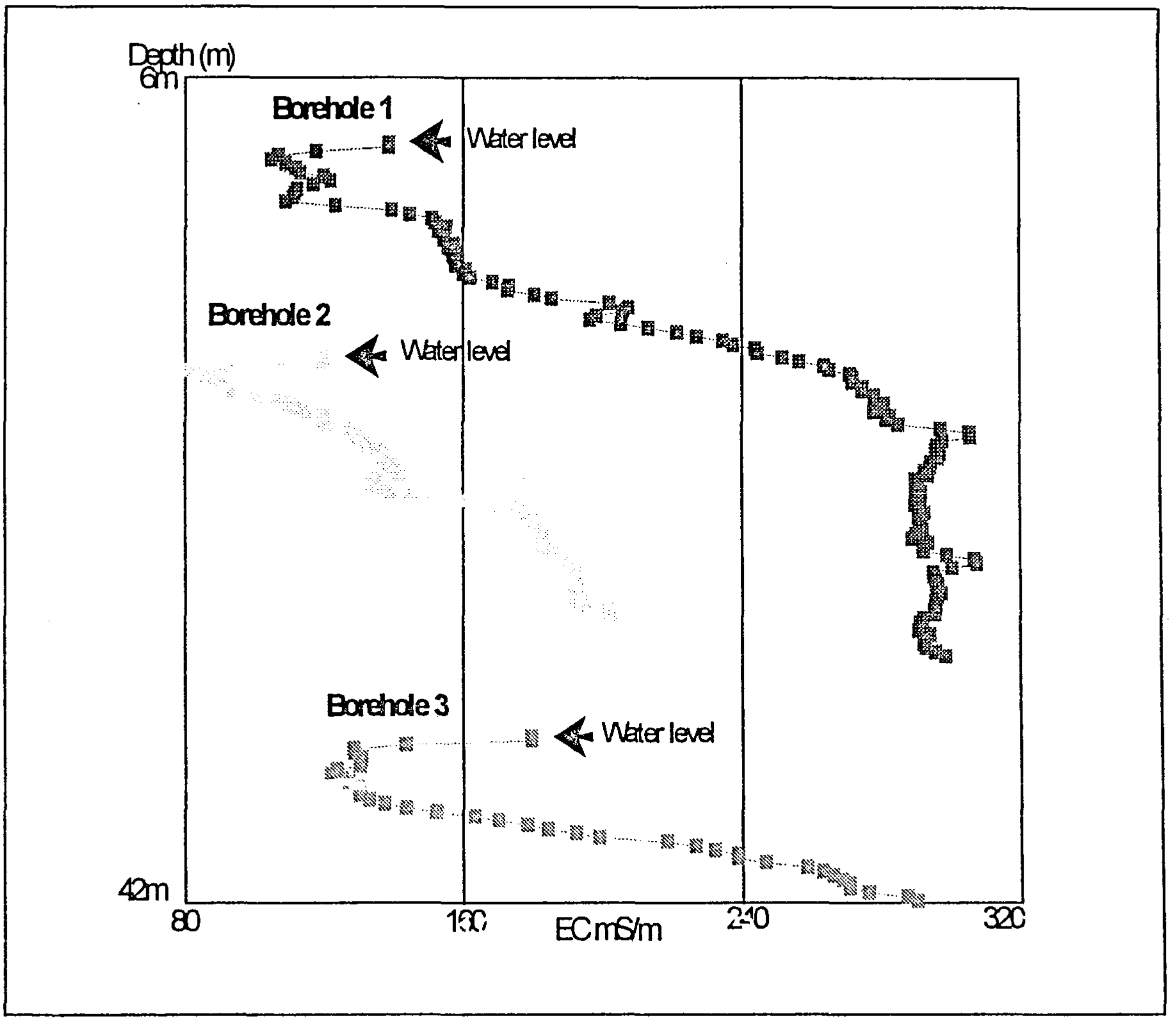


Figure 3.9. Examples of electrical conductivity ranges of water in a selection of boreholes drilled into spoil.

- The marked increase in the electrical conductivity with depth below the water level is well illustrated in Figure 3.9. These profiles are typical of semi-stagnated spoil waters. The better quality near the water/air interface is the result of recharge from rainfall, or in the case of borehole 2, recharge from a river which crosses the spoil.

3.3.2 Acid-Base Potential of Mines

3.3.2.1 Acid-base determinations

3.3.2.1.1 Sampling methods

The ideal sampling procedure for the determination of the acid-base potential of rock is described in Appendix B. The majority of samples taken during this investigation comply with these requirements. Grab samples have been taken where diamond drilled cores were not available. In the latter instance, the thickness of the strata could not be inferred. Without this information, the mine NNP cannot be calculated.

In specific instances, the current mine acid-base accounting is restricted to one borehole. This information is clearly insufficient for the determination of the mine NNP, particularly where the lithologies have a limited lateral extent such as channel sandstones.

3.3.2.1.2 Regional interpretation of the acid-base potential

Statistics on the NNP for the samples analysed during this investigation are presented in Figures 3.10 and 3.11 as whisker plots. The key for the lithological codes that are used in the diagrams is on the right:

LITHOLOGY	CODE
Shale	SHLE
Residual soil	SOIL
Interbedded shale/siltstone	SLSN
Interbedded sandstone/shale	SDSL
Sandstone	SNDS
WEATHERING	
Weathered	WT
Fresh	FR

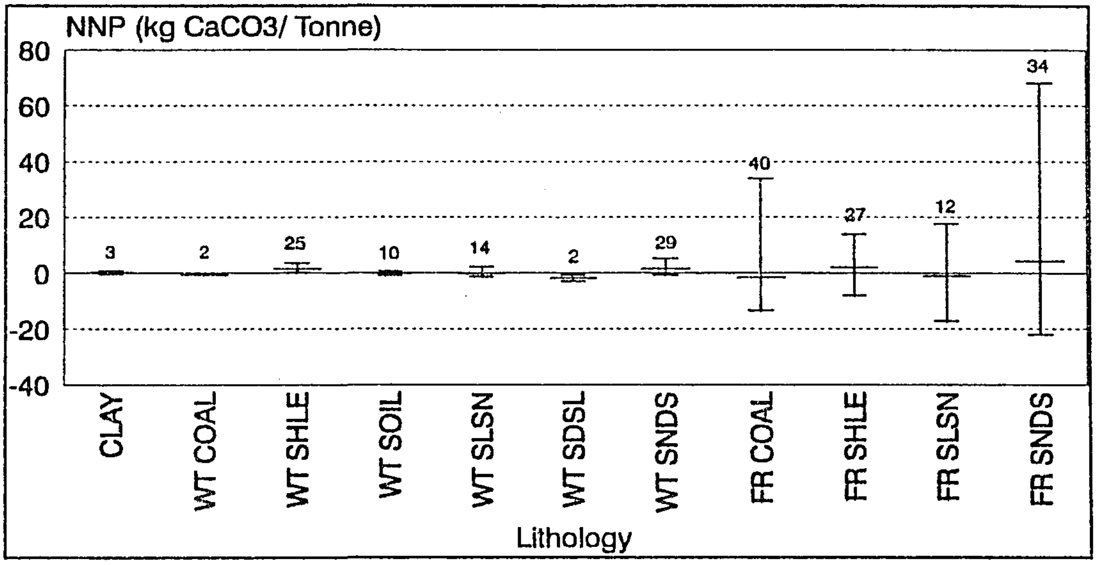


Figure 3.10. Whisker plots (10 percentile, 90 percentile and mean) for NNP in spoil under open conditions. (Note: WT denotes weathered strata, FR denotes fresh strata).

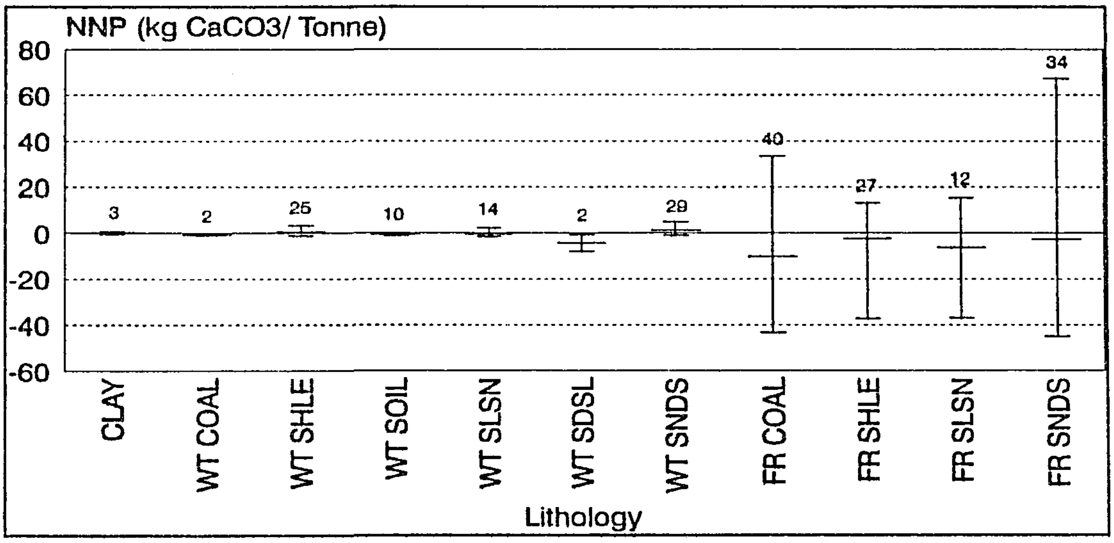


Figure 3.11. Whisker plots (10 percentile, 90 percentile and mean) for NNP in spoil under closed conditions. (Note: WT denotes weathered strata, FR denotes fresh strata).

The plotted values have not been compensated for strata thickness. Weathered and unweathered lithologies have been differentiated. Within each of these groups, the sedimentary units are arranged in order of increasing grain-size. Plots for each lithological unit show the 10 percentile, 90 percentile and average values. The number of observations within each category is shown above the respective plots.

The following conclusions are drawn from this diagram:

- The weathered strata are predominantly characterised by neutral NNP's. This is ascribed to the fact that most of their acid and base components have been leached through weathering within the vadose zone. It suggests the significant movement of oxygen and water through the sediments, above the groundwater level.
- The average NNP's for the unweathered lithologies for closed carbonate systems are acidic. This is in contrast to an open carbonate system where the sediments generally tend towards alkaline NNP's.
- The high base-potential of some of the sandstone samples is significant. This aspect may be incorporated into the amelioration plan for a mine.

The whisker plots for cadmium and aluminium (Figures 3.12 and 3.13) show interesting relationships.

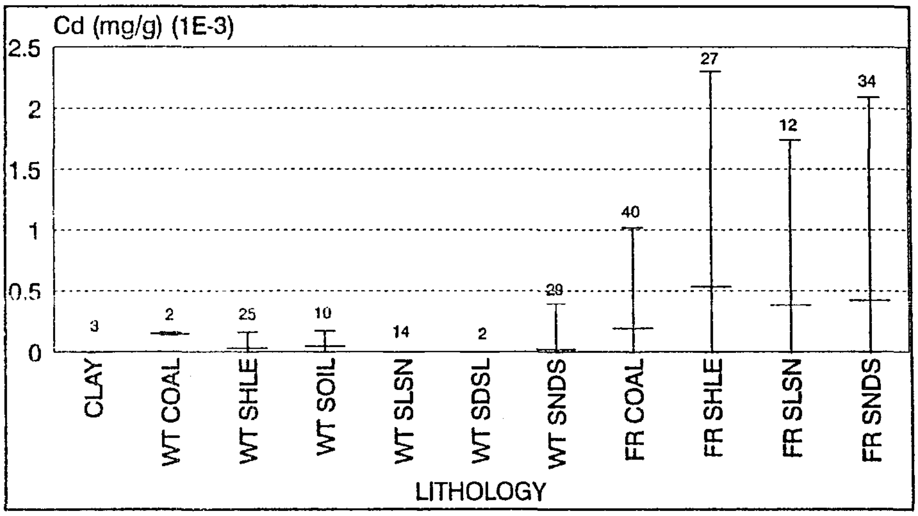


Figure 3.12. Whisker plots (10 percentile, 90 percentile and mean) for ranges of cadmium concentrations in spoil. (Note: WT denotes weathered strata, FR denotes fresh strata).

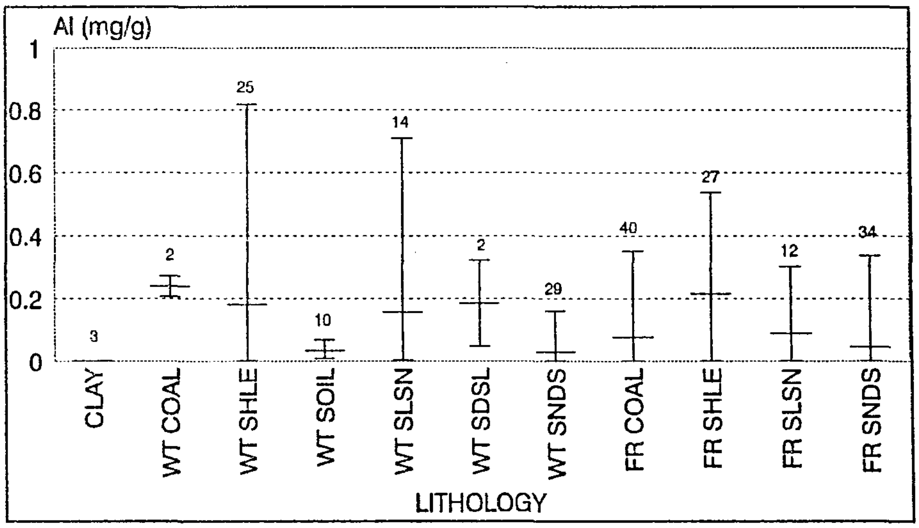


Figure 3.13. Whisker plots (10 percentile, 90 percentile and mean) for ranges of aluminium concentrations in spoil. (Note: WT denotes weathered strata, FR denotes fresh strata).

Cadmium occurs within the sediments as a sulphide ore. It is singled out in this example because of its potential toxicity as a cumulative poison (Kempster and Smith, 1985). The cadmium relationships in Figure 3.12 follow similar patterns to that of the NNP distribution. Leaching has removed this element from the weathered material, whereas the fresh sediments in opencast mines have a high potential to release cadmium. This trend also applies to other heavy metal-bearing sulphides. The actual release will depend on the local NNP relationship.

Aluminium occurs as oxides, hydroxides or silicates within the sediments. Since aluminium is practically insoluble at neutral pH-levels, a drop in the pH of the water is required to mobilise aluminium from the sediments. This drop in the pH is the result of the oxidation of sulphides within the sediments. Usually therefore, sediments characterised by an alkaline NNP, reflect low aluminium concentrations, even though much more aluminium is available. The graphs for aluminium (Figure 3.13) stress the importance of maintaining neutral pH-conditions, thus immobilising heavy metals.

3.3.2.1.3 Interpretation of the acid-base potential on a mine basis

Ten mines have been investigated with respect to their acid-base potential. The results of some of these investigations are presented in Appendix C.

Although the use of powdered samples is not representative of spoil fragments, this procedure is currently used internationally. Two options are available for evaluating whether a mine is potentially acid or not.

The first option is to assume that the spoil material approximates a closed system. Although this represents the maximum acid potential in terms of calcium carbonate consumption, it is probably a realistic approximation in view of the long-term projected behaviour of the spoil environment.

A second alternative involves the concept of a threshold value for the NNP of an open system. Fergusson and Erickson (1988) found that mines produce alkaline water if the NNP is >30 kg $CaCO_3$ per ton of spoil. On the other hand, 59% of the mines with a NNP of 7 - 30 kg $CaCO_3$ per ton of spoil produced alkaline drainage. Only 11% of the mines with an NNP below 7 kg $CaCO_3$ per ton of spoil produced alkaline drainage. The threshold value of 30 kg $CaCO_3$ per ton of spoil is confirmed by several other researchers (Cravotta *et al.*, 1990).

The reason for such a high threshold value may be attributed to a number of factors:

- The spoil material approximates a closed system that increases the carbonate consumption.
- Some of the base potential of the spoil becomes coated with mineral precipitates, following the neutralisation reactions.
- A portion of the carbonate material is leached out of the spoil by percolating waters prior to the development of significant acid generation.

Although such a threshold serves as a provisional guide-line, it is still necessary to confirm this value for South African conditions. It is recommended that further work be done in this regard.

Only three of the mines which have been investigated, are alkaline according to a closed system approach (Appendix C). These are mines 3, 4 and 5. Although the thickness of the strata for mine 4 is not available, it is evident that this mine is alkaline.

Using the concept of the threshold value, the three alkaline mines can be categorised as follows:

- Mine 4 (NNP >30 kg $CaCO_3$ per ton) falls into the category, where mines always produce alkaline drainage.

- Mines 3 and 5 (NNP = 7 - 30 kg $CaCO_3$ per ton) fall into the category, where 59% of the mines produce alkaline drainage.

Further sampling and research are required to determine whether mines 3 and 5 may still produce acidic drainage, despite the alkaline NNP determined by laboratory experiments.

It has been speculated in previous work that the depositional environment of coal deposits dictates the nature of the drainage which emanates from the seam horizon. Carrucio and Geidel (1970) have observed that acidic mine drainage is primarily associated with coals deposited in marine brackish waters. Although drainage from freshwater coals is also characterised by high sulphate concentrations, they are normally in the neutral pH-range.

The Number 1 and 2 Seams in the study area are normally associated with glaciofluvial sandstones, whereas the 3, 4, and 5 Seams are associated with fluvio-deltaic deposits (Cairncross *et al.*, 1990). Seams 3, 4, and 5 should therefore theoretically have a greater tendency towards the generation of acidic drainage. Further work is necessary in this regard as some observations contradict the generalisation.However, this relationship is generally true for the study area. Analyses from the Number 3 and 4 Seams indicate that their NNP is negative. Coal from the Number 2 Seams reflects a positive NNP. There are exceptions, particularly in Mine 1, where the Number 4 Seam is characterised by an alkaline NNP. It should be noted, however, that three of the four coal samples have a high acid potential. This suggests that secondary mineralisation may have contributed neutralising components to produce the alkaline NNP obtained for these coal seams.

In Mine 3, host rocks confirm the acidic or alkaline nature of the coal seams. The interlaminated sandstone and shale associated with the Number 4 Seam are more acidic than that of the same lithology associated with the Number 2 Seam.

Mine 5 has a more complex relationship. In this mine, the lithologies which are located between the Number 3 and 4 Seams are the most acidic. However, the strata, which lie directly above the Number 4 Seam and immediately below the Number 3 Seam, are the most alkaline strata of the entire sequence. Furthermore, the Number 2 Seam is bounded by acidic strata.

It can be seen from the above information, that the broad concept for coal drainage quality of Carrucio and Geidel (1970) is generally valid for the area. These relationships may also apply to the strata that bound the coal seams in certain instances. However, it should be noted that there are exceptions. These deviations may be functions of the localised groundwater flow patterns and secondary mineralisation. Each mine should therefore be studied individually to confirm the nature of relationships between the various coal seams and the NNP's of the surrounding strata.

It may therefore be concluded that the NNP for the lithologies of the area are a function of the depositional environment and secondary mineralisation. This latter parameter may be correlated with the hydraulic conductivity of the strata and the local groundwater flow directions.

Further development and sampling is required to confirm whether the above exceptions (see also Appendix C) which were noted during these experiments are consistent throughout the respective mines, or whether these phenomena are localised occurrences.

3.3.2.2 Total acid-base potential for a specific mine

Laboratory results only provide a framework for spoil and water management. Acid leachate may still develop in an alkaline mine as a result of the following factors:

- Preferential pathways through the spoil material.
- Placement of coal discards.
- The distribution of pyrite and neutralising components within the spoil fragments.

It is therefore necessary that experimental results be related to actual field conditions. This can be done through computer modelling. Although models of this kind can be applied in the generic sense, a typical mine example follows to demonstrate the application of the model to a real life situation.

3.3.2.2.1 Water quality simulation for an opencast mine

The following assumptions apply to this simulation:

- The opencast pit is closed.
- The rehabilitated area comprises 1670 ha.
- All existing river systems have been diverted around the pit.
- Run-off into the pit has been reduced to a minimum and is not considered as input into this example.
- An open water area has formed on the northern perimeter of the pit, where insufficient spoil was available to raise the spoil topography above the level of water in the pit. Water may decant from this open water area into the stream.
- All spoil waters drain through the spoil towards the open water area.
- Coal discards have been placed in the southern portion of the pit.

The cross-sections (Figures 3.14 and 3.15) indicate the distribution of the flooded spoil, the pit floor morphology and the topography of the rehabilitated areas.

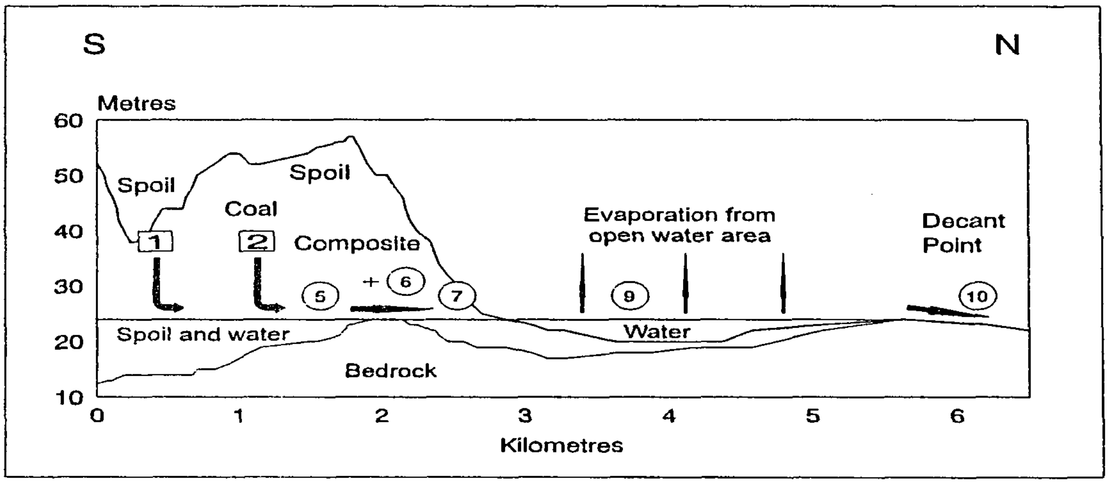


Figure 3.14. S-N section through the conceptual mine, showing the various solution cells.

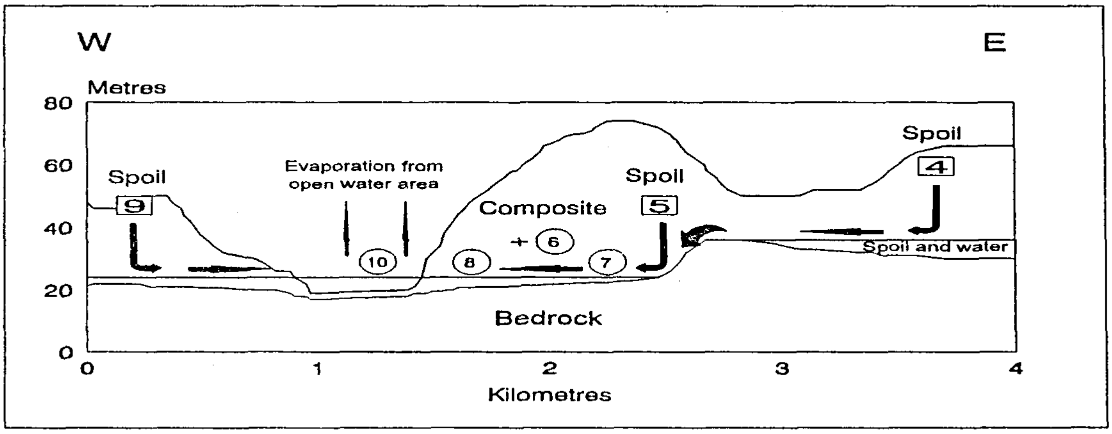


Figure 3.15. W-E section through the conceptual mine, showing the various solution cells.

3.3.2.2.1.1 *Acid-base characteristics of the spoil*

In the example to be modelled, coal discards have been disposed of within the pit. The coal discards, having a high acid-generating potential, are included into the model as a separate cell.

The NNP of the spoil has been determined through laboratory investigation and is set at 15,5 $CaCO_3$ kg/t for an open system and at 11,0 $CaCO_3$ kg/t for a closed system. The coal discard material is set at NNP's of -94 and -188 $CaCO_3$ kg/t for open and closed systems respectively.

3.3.2.2.1.2 *Sources of water and water balances*

The water balance for simulation is derived as follows:

- The recharge from rainfall has been determined through field investigations and set at 8,5% of the average annual rainfall (727 mm). Models such as ACRU(Schulze, 1989), MOUSE (Steenhuis and Parlange, 1990) or the Richards' Equation (Richards, 1931) may also be used to predict recharge on a daily basis.
- Groundwater influx has been quantified from measurements in the highwall areas of the pit. A value of 1 000 m^3/d is used for this model. Models such as FESIM (HydroSolutions Inc., 1991), AQUAMOD (Van Tonder, 1992) or MODFLOW (McDonald and Harbaugh, 1983) may also be used to simulate groundwater inflow.
- Nett evaporation is approximated at 0,7 m per annum. A daily evaporation model such as Thornthwaite (Thornthwaite, 1948) could also have been used if predictions on a shorter time scale were necessary.
- Recharge, due to run-off, is negligible for the purposes of this model, since surface contouring leads run-off off the pit area. This source of water may, however, be simulated and included into the model.

3.3.2.2.1.3 *Volumes of flooded spoil*

The volume of spoil that will be flooded for the conceptual model is 32,8 million cubic metres. The total volume of the spoil in the pit is 380 million cubic metres. For an NNP of 11 kg $CaCO_3$, for a closed carbonate system, this may be recalculated as an excess base potential of 7,5 million tons $CaCO_3$ for the total mine. Theoretically, this excess base potential has the capacity to neutralise 40 million tons of coal discards, if placed evenly within the spoil. According to Fergusson and Erickson (1988), the probability that this mine will produce alkaline drainage is 0,59. Using their criteria, it is uncertain whether the mine will remain alkaline.

3.3.2.2.1.4 *Areas prone to acidification*

Deposition of acid-generating material, such as coal discards, underneath water, is accepted practice since this prevents the oxidation of sulphides by oxygen. This is, however, only effective where the disposal takes place within an alkaline environment. The mechanism for progressive deterioration of coal discards underneath water is demonstrated in Figure 3.16.

Acidic water from overlying spoils comes into contact with pyrite in the flooded zone (Figure 3.16B). Ferric ions within the acid water react with the pyrite:

$$FeS_2 + 14Fe^{3+} + 8H_2O = 15Fe^{2+} + 2SO_4^{2-} + 16H^+$$

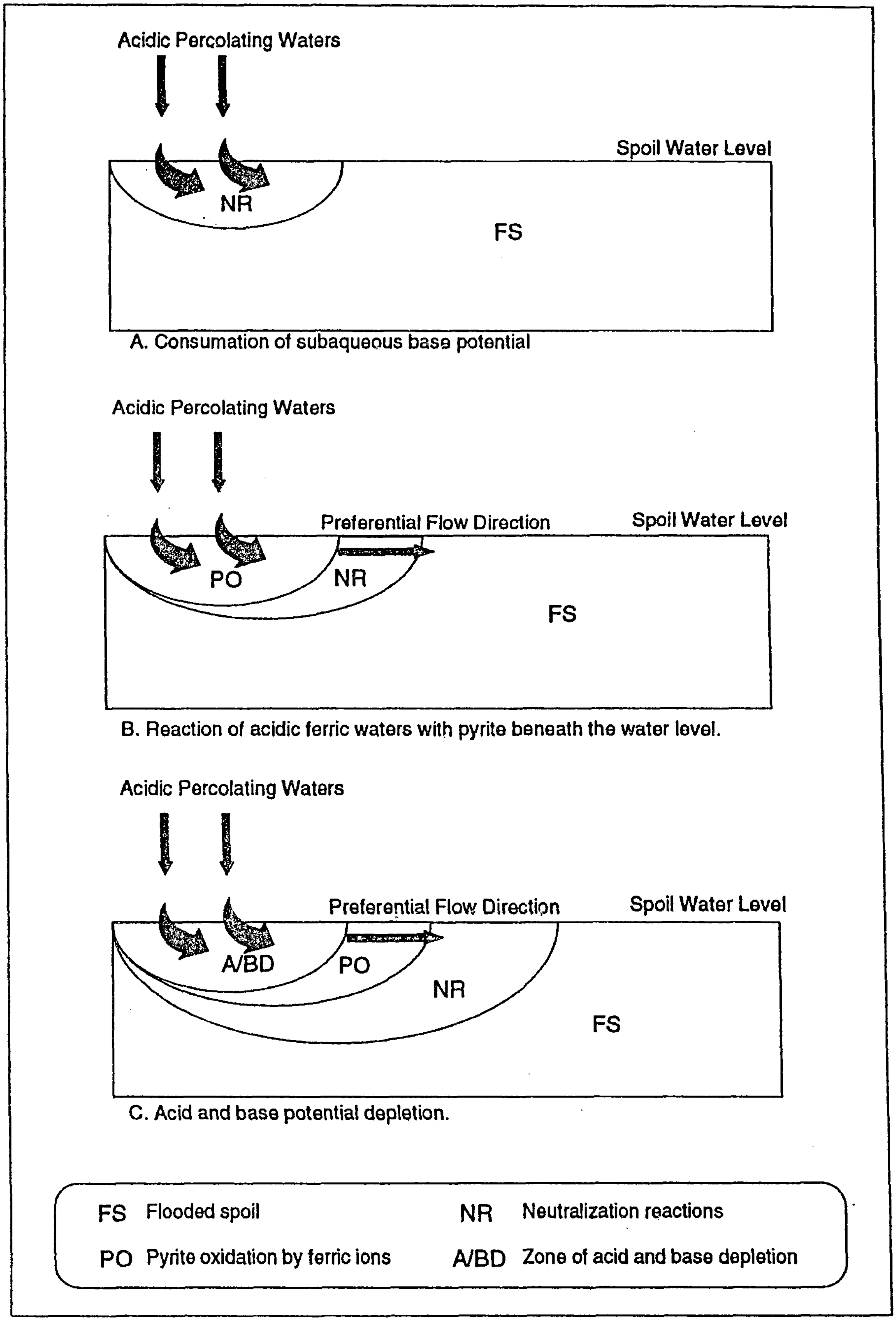


Figure 3.16. Schematic representation of the diffusion of acidic leachates through flooded spoils.

This rapid reaction generates acidity and ferrous ions. The acid and base potential are exhausted within the flooded spoil with time (Figure 3.16C). As the water emanates from the spoil (Figure 3.15), the ferrous iron reacts with dissolved oxygen in the pan to produce ferric ions (Barton 1978):

$$14Fe^{2+} + 3.5O_2 + 14H^+ = 14Fe^{3+} + 7H_2O$$

Subaqueous deposition of coal discards is therefore only effective where the material is not subjected to ferric-rich acidic waters.

The area that will first be prone to acidification is cell 2, containing the coal discards.

3.3.2.2.1.5 *Initial water chemistries and mixing cells within the spoil*

The initial water chemistries within cells were measured in the field through monitoring boreholes that have been drilled into the spoil and confirmed by calculations, using the PHREEQE model. Stratification within the spoil water has also not been accounted for in this example, since this would complicate the example beyond the scope of this demonstration. Average chemistries were therefore used for areas where significant stratification is present.

Cells containing calculated water chemistries, verified by field sampling, are 1, 2, 3, 4 and 8 (Figures 3.14 - 3.15). The chemistries within the remainder of the cells are calculated by mixing water from the various initial solutions. Flow paths between cells and their associated percentage of the total volume of water are shown in Figures 3.17 - 3.18. Each cell has a percentage that indicates the relative proportion of this source of water within the mine. The open water area is set at 100%, since this represents the maximum water volume within the pit at any given time. Two simulations were run with different CO_2 pressures within the spoil. The two scenarios represent partially closed systems, which tend towards open (pCO_2 = -2,0) and closed (pCO_2 = -0,6) systems.

(pCO_2 = -2,0)   (pCO_2 = -0,6)

3.3.2.2.1.6 *Geochemical modelling*

There are three evolutionary stages in geochemical modelling (Plummer, 1992):

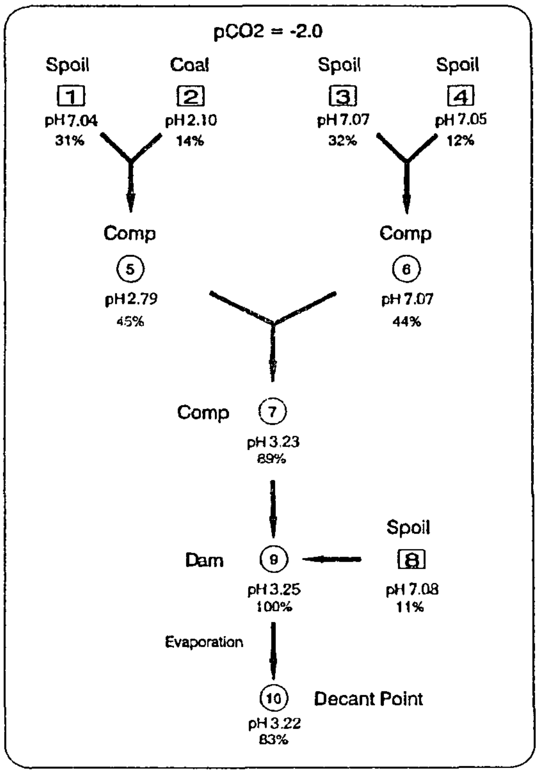
- Aqueous speciation modelling solves speciation and saturation for individual solutions.
- Forward and inverse modelling allows for water-rock interaction.
- Reaction transport modelling couples the geochemical models with flow transport.

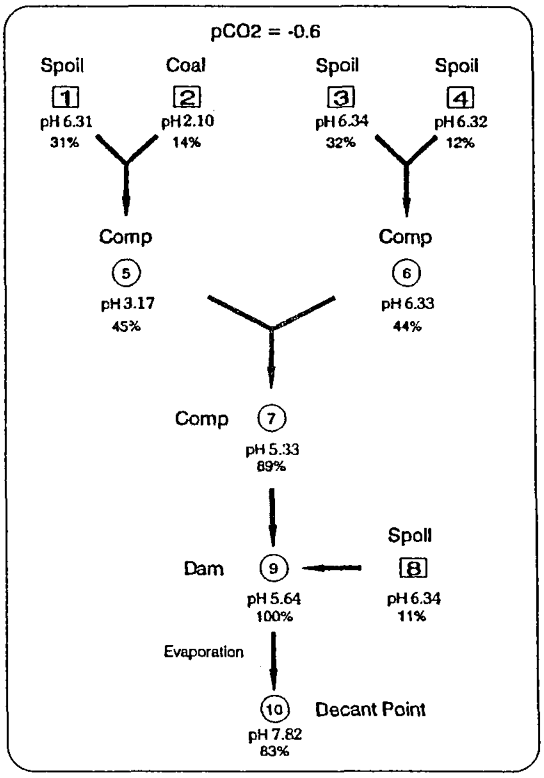
Aqueous speciation modelling, with programs such as WATEQ (Trusdell and Jones, 1974) and MINTEQ (Felmy *et al.*, 1984) or their later derivatives, is too simplistic for the current study. Reaction transport models such as SATRACHEM (Lewis *et al.*, 1986) and HYDROGEOCHEM (Yeh and Tripathi, 1989) are the most advanced forms of modelling presently available for this purpose. However, the hydraulic properties of the aquifer must be known before such modelling is possible. It is physically impossible to measure these parameters on a representative basis within the South African spoils and this form of modelling is therefore too advanced for the current modelling exercise.

The abilities of forward modelling with the aid of PHREEQE (Parkhurst*et al.*, 1980) is well-documented (Wright and Turner, 1987; Sanford and Konikow, 1989; Macalady *et al.*1990; Longmire *et al.*, 1990; Roy *et al.*, 1991). This model is suitable for the modelling of long-term chemistries under steady-state equilibrium conditions and has been selected for this modelling exercise.

INITIAL ELEMENTS mg/l	Cell 1	Cell 2	Cell 3	Cell 4	Cell 8
	SPOIL	COAL	SPOIL	SPOIL	SPOIL
Ca	557	0	438	517	401
Mg	340	0	266	313	243
Na	19	0	15	18	191
Fe	0	747	0	0	0
Cl	30	0	23	27	294
SO4	2563	2570	1980	2360	1800
pH	7.04	2.10	7.07	7.05	7.08
TOT ALK	119	0	125	121	127
MINERALS mg/l					
PYRITE	1605	1605	1237	1474	1124
Fe(OH)3	-1430	0	-1101	-1313	-1001
DOLOMITE	2576	0	2016	2377	1845
GYPSUM	-12	0	0	0	0
NOTE					
pCO2=-2.0					
Spoil indicates spoil drainage.					
Coal indicates discard drainage.					
Neg. mineral phase indicates precipitation.					
Pos. mineral phase indicates minerals reacted.					

INITIAL ELEMENTS mg/l	CELL 1	CELL 2	CELL 3	CELL 4	CELL 8
	SPOIL	COAL	SPOIL	SPOIL	SPOIL
Ca	620	0	531	607	495
Mg	394	0	322	368	300
Na	19	0	15	18	191
Fe	0	747	0	0	0
Cl	30	0	23	27	294
SO4	2499	2570	1980	2360	1800
pH	6.31	2.10	6.34	6.32	6.34
TOT ALK	567	0	588	574	595
MINERALS mg/l					
PYRITE	1605	1605	1237	1474	1124
Fe(OH)3	-1430	0	-1101	-1313	-1001
DOLOMITE	2989	0	2442	2794	2276
GYPSUM	-128	0	0	0	0
CALCITE	0	0	0	0	0
NOTE					
pCO2=-0.6					
Spoil indicates spoil drainage.					
Coal indicates discard drainage.					
Neg. mineral phase indicates precipitation.					
Pos. mineral phase indicates minerals reacted.					





Figures 3.17 (left) and 3.18 (right) Input chemistries of open (left) and closed (right) spoil systems with flow paths and pH's.

3.3.2.2.1.7 Flow paths and resultant chemistries

CO_2 pressure = -2,0

At a pCO_2 pressure of -2.0, the final water (cell 10) is acidic (Table 3.9). This is due to the impact of the discards (cell 2) on the entire system. The alkalinity of the water in the cells is incapable of neutralising acidity in the water.

Table 3.9. Initial and calculated chemistries for spoil waters for an open system.

INITIAL AND CALCULATED CHEMISTRIES										
ELEMENTS	Cell	Cell	Cell	Cell	Cell	Cell	Cell	Cell	Cell	Cell
mg/l	1	2	3	4	5	6	7	8	9	10
	SPOIL	COAL	SPOIL	SPOIL	COMP	COMP	COMP	SPOIL	DAM	DECANT
Ca	557	0	438	517	379	459	419	401	417	503
Mg	340	0	266	313	231	279	255	243	253	305
Na	19	0	15	18	13	16	14	191	34	41
Fe	0	747	0	0	239	0	110	0	92	111
Cl	30	0	23	27	20	24	22	294	52	63
SO4	2563	2570	1980	2360	2565	2083	2324	1800	2266	2730
pH	7.04	2.10	7.07	7.05	2.79	7.07	3.23	7.08	3.25	3.22
TOT ALK	119	0	125	121	0	124	0	127	0	0
MINERALS										
mg/l										
PYRITE	1605	1605	1237	1474	0	0	0	1124	0	0
Fe(OH)3	-1430	0	-1101	-1313	0	0	-18	-1001	-12	0
DOLOMITE	2576	0	2016	2377	0	0	0	1845	0	0
GYPSUM	-12	0	0	0	0	0	0	0	0	0
NOTE										
pCO2=-2.0										
Spoil indicates spoil drainage.										
coal indicates discard drainage.										
Dam indicates chemistry prior to evaporation.										
Comp indicates composite chemistry from mixing cells.										
Decant indicates decant point chemistry after evaporation.										
Neg. mineral phase indicates precipitation.										
Pos. mineral phase indicates minerals reacted.										

An important aspect of the discard environment is the elevated iron concentrations within the water (Table 3.9). These ions are in the ferric state and remain in solution as a result of the low solution pH.

Most of the $CaCO_3$ and $Fe(OH)_3$ precipitation occurs within the cells. Gypsum precipitation only occurs once the effect of evaporation from the open water area (dam) has raised the concentrations of the elements sufficiently for this mineral to become oversaturated.

pCO_2 pressure = -0,6

The second model was simulated for a spoil pCO_2 pressure of -0.6. This effectively tends towards a closed system and is reflected by a higher base consumption (Table 3.10).

Although the initial cells have a lower pH than the previous simulation, they have a higher alkalinity than the initial chemistries of the first model. This is explained as follows:

Table 3.10. Initial and calculated chemistries for spoil waters for a closed system.

INITIAL AND CALCULATED CHEMISTRIES										
ELEMENTS	CELL	CELL	CELL	CELL	CELL	CELL	CELL	CELL	CELL	CELL
	1	2	3	4	5	6	7	8	9	10
mg/l	SPOIL	COAL	SPOIL	SPOIL	COMP	COMP	COMP	SPOIL	DAM	DECANT
Ca	620	0	531	607	422	551	486	495	487	541
Mg	394	0	322	368	268	334	301	300	301	363
Na	19	0	15	18	13	16	14	191	34	41
Fe	0	747	0	0	164	0	0	0	0	0
Cl	30	0	23	27	20	24	22	294	52	63
SO4	2499	2570	1980	2360	2522	2083	2302	1800	2247	2707
pH	6.31	2.10	6.34	6.32	3.17	6.33	5.33	6.34	5.64	7.82
TOT ALK	567	0	588	574	0	584	57	595	116	24
MINERALS										
mg/l										
PYRITE	1605	1605	1237	1474	0	0	0	1124	0	0
Fe(OH)3	-1430	0	-1101	-1313	-144	0	-157	-1001	0	0
DOLOMITE	2989	0	2442	2794	0	0	0	2276	0	0
GYPSUM	-128	0	0	0	0	0	0	0	0	-1
CALCITE	0	0	0	0	0	0	0	0	0	-116
NOTE										
pCO2=-0.6										
Spoil indicates spoil drainage.										
coal indicates discard drainage.										
Dam indicates chemistry prior to evaporation.										
Comp indicates composite chemistry from mixing cells.										
Decant indicates decant point chemistry after evaporation.										
Neg. mineral phase indicates precipitation.										
Pos. mineral phase indicates precipitation.										

- CO_2 dissolves in water by the reaction: $CO_2 + H_2O = H_2CO_3^*$
- The amount of $H_2CO_3^*$ that is formed is a function of the pCO_2 pressure within the spoil.
- The $H_2CO_3^*$ dissociates to form bicarbonate, which causes the alkalinity of the water. However, acidity is also produced by the following reaction: $H_2CO_3^* = H^+ + HCO_3^-$

The alkalinity in the waters of the second model is sufficient to neutralise acidity (Table 3.10). It will be noted that the pH rises between cell 9 and cell 10 for this model. This is a result of the equilibration of the water with the atmosphere. Carbon dioxide is exsolved and the acidity produced by the $H_2CO_3^*$ is consequently reduced.

The locality of the mineral precipitation is similar to that of the previous model.

As an extension to the model, the effect of introducing 3 Mℓ/d of ash water into the coal cell has been simulated to study the effect of this management option on the system. Schematically, the cross-section and input are modified as illustrated in Figures 3.19 and 3.20:

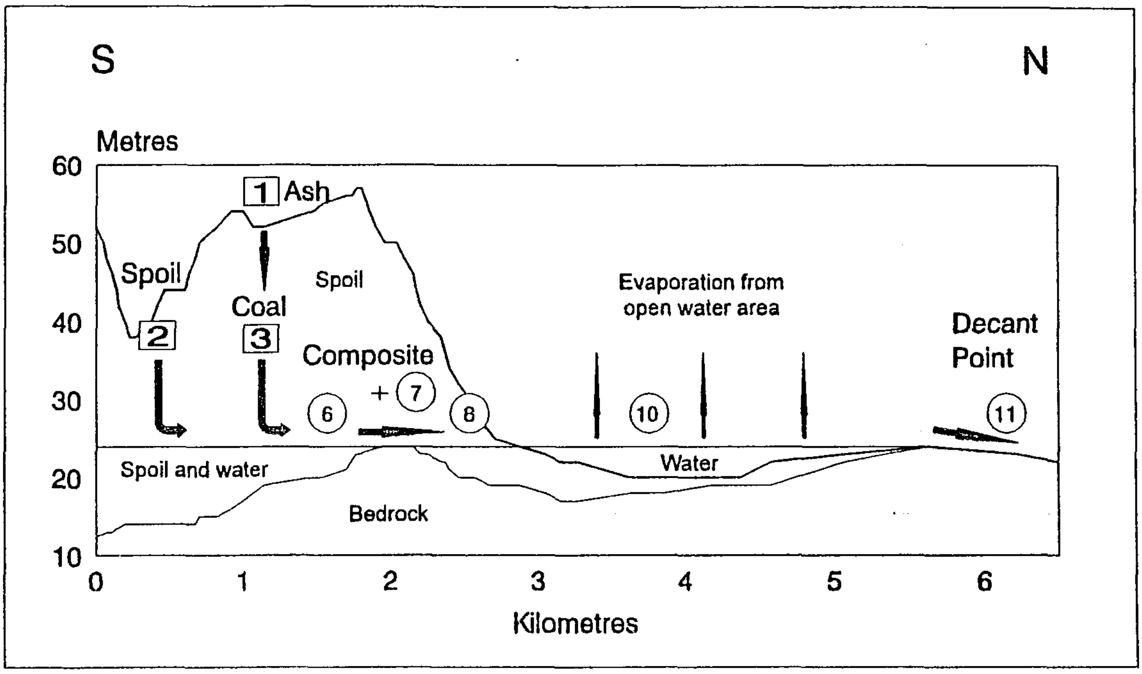


Figure 3.19. S-N section through conceptual mine, showing the various solution cells for the addition of ash water.

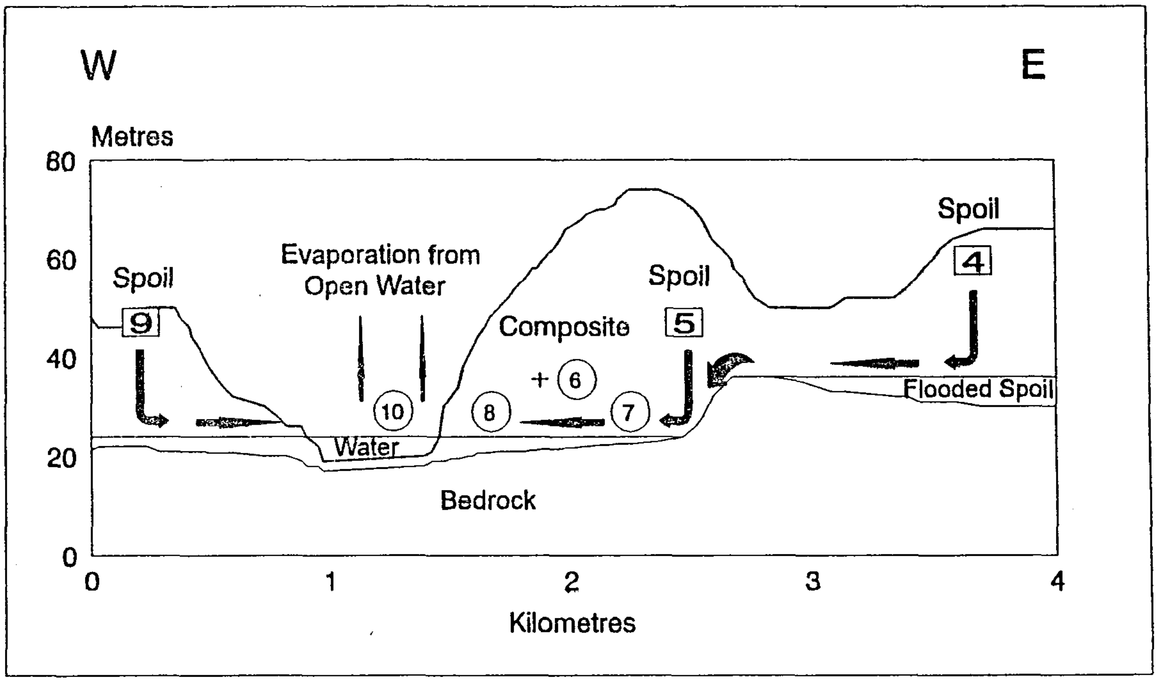


Figure 3.20. W-E section through the conceptual mine, showing the various solution cells for the addition of ash water.

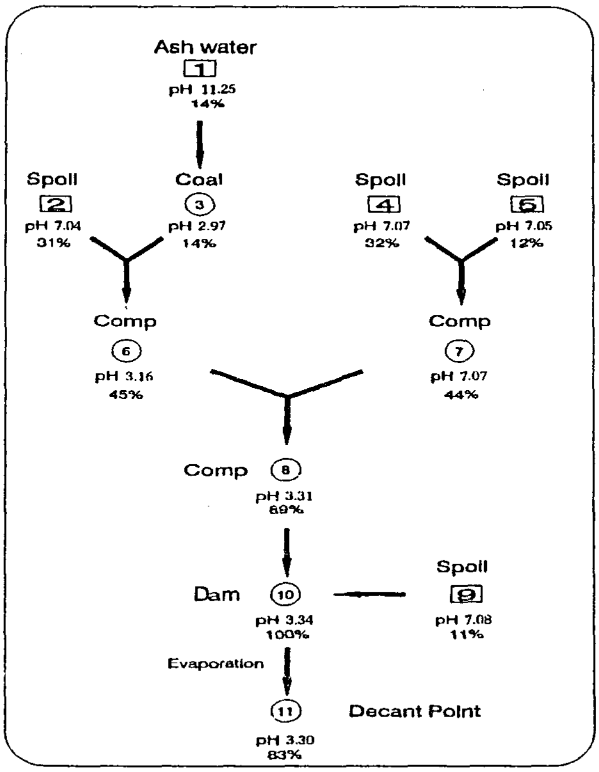


Figure 3.21. Flow diagram and pH-levels for ash water recharge calculations.

The conclusion is drawn that the alkalinity of the ash water is insufficient to neutralise the acid potential of the discards in the spoil as is demonstrated in Table 3.11.

Table 3.11. Input and calculated chemistries for the ash water recharge experiment.

INITIAL AND CALCULATED CHEMISTRIES										
ELEMENTS	CELL	CELL	CELL	CELL	CELL	CELL	CELL	CELL	CELL	CELL
mg/l	1	2	3	4	5	6	7	8	9	10
	ASH	SPOIL	COAL	SPOIL	SPOIL	COMP	COMP	COMP	SPOIL	DAM
Ca	581	557	525	438	517	536	459	498	401	487
Mg	0	340	0	266	313	231	279	255	243	253
Na	150	19	150	15	18	61	16	38	191	55
Fe	0	0	729	0	0	192	0	68	0	55
Cl	15	30	15	23	27	25	24	25	294	54
SO_4	1002	2563	3437	1980	2360	2816	2083	2450	1800	2378
pH	11.25	7.04	2.97	7.07	7.05	3.16	7.07	3.31	7.08	3.34
TOT ALK	712	119	0	125	121	0	124	0	127	0
MINERALS										
mg/l										
PYRITE	0	1605	1605	1243	1474	0	0	0	1124	0
$Fe(OH)_3$	0	-1430	-35	-1101	-1313	-79	0	-53	-1001	-11
DOLOMITE	0	2576	0	2016	2377	0	0	0	1845	0
GYPSUM	0	-12	-243	0	0	-47	0	0	0	0
NOTE										
$pCO_2=-2.0$										
Ash indicates ash drainage above the discard cell.										
Spoil indicates spoil drainage.										
coal indicates discard drainage.										
Dam indicates chemistry prior to evaporation.										
Comp indicates composite chemistry from mixing cells.										
Decant indicates decant point chemistry after evaporation.										
-Mineral phase indicates precipitation.										
+Mineral phase indicates minerals reacted.										

In a further simulation, a dilution option has been considered, by recharging 3 Mℓ/d of clean run-off water into cell 1, which will dilute acid generation from the coal cell. The resultant chemistries are listed in Table 3.12.

Table 3.12. Input and calculated chemistries for the recharge experiment.

	INITIAL AND CALCULATED CHEMISTRIES									
ELEMENTS mg/l	**CELL 1**	**CELL 2**	**CELL 3**	**CELL 4**	**CELL 5**	CELL 6	**CELL 7**	**CELL 8**	**CELL 9**	**CELL 10**
	SPOIL	COAL	SPOIL	SPOIL	COMP	COMP	COMP	SPOIL	DAM	DECANT
Ca	222	0	438	517	151	459	305	401	316	380
Mg	135	0	266	313	92	279	185	243	191	231
Na	19	0	15	18	13	16	14	191	34	41
Fe	0	269	0	0	69	0	10	0	5	6
Cl	30	0	23	27	20	24	22	294	52	63
SO4	925	925	1980	2360	925	2083	1504	1800	1536	1851
pH	7.17	2.43	7.07	7.05	3.21	7.07	3.59	7.08	3.75	3.70
TOT ALK	145	0	125	121	0	124	0	127	0	0
MINERALS										
mg/l										
PYRITE	578	578	1237	1474	0	0	0	1124	0	0
Fe(OH)3	-515	0	-1101	-1313	-32	0	-47	-1001	-8	0
DOLOMITE	1022	0	2016	2377	0	0	0	1845	0	0
GYPSUM	1022	0	0	0	0	0	0	0	0	0
CALCITE	0	0	0	0	0	0	0	0	0	0

The conclusion is drawn that the beneficial effect of the dilution is minimal, due to the low pH of the water. As another alternative, clean water was introduced into cell 10, where the spoil water decants into the stream. Similar results were obtained, with the final water composition as follows:

Constituent	**mg/l**	**Constituent**	**mg/l**
Ca	276	Cl	35
Mg	168	SO4	1502
Na	22	pH	3

Although the dilution will not significantly change the final pH of the outflow water, it will reduce the salt concentrations.

3.3.2.3 Conclusion

The prediction of long-term water quality within coal opencast mines is complicated because of the many variables that need to be considered. The current models are based on advanced geochemical and mathematical principles that need to be understood to enable the selection of an appropriate model. More research is necessary to ascertain the confidence with which these advanced models can be applied to simulate opencast water chemistries.

3.3.3 WATER MANAGEMENT STRATEGIES IN OPENCAST MINES

Many options for the management of water quality within opencast coal mines have been considered in the past by the mining community. Very few practical solutions have generally been available to date.

The potential strategies to deal with mine-related pollution may be categorised as follows:

- Prevention of acid generation by forestalling the pyrite oxidation at source.
- Management of contaminated spoil water.

The concepts behind these options for water management are discussed below.

3.3.3.1 Preventative measures

3.3.3.1.1 Rehabilitation of spoils

According to Steffen, Robertson and Kirsten (1989), the aims of spoil rehabilitation for water quality control are twofold:

- To control the rate of acid generation.
- To limit the migration of the acid generation products.

The good rehabilitation of spoil heaps retards the ingress of oxygen and water into the subsurface. Rehabilitation effectiveness should be evaluated in terms of the combined benefits of limiting both these reactants but must fit into the overall objective of returning the land to beneficial use. Spoil rehabilitation techniques include levelling, topsoiling and revegetation. Vegetation prevents sheet-wash erosion of the soil cover. Gully erosion may be minimised through contouring methods.

Effective natural covers rely on the soil moisture content to inhibit the ingress of oxygen into the spoil. A triple layer rehabilitation plan could be considered. This constitutes a layer of clay, overlain by a coarse breaker layer and finally a growth medium at surface. It is crucial that the clay layer is prevented from desiccating and cracking. The purpose of such a cover is also to prevent spontaneous combustion of the spoil and discards (Watson, 1983).

Factors that influence the economy and efficiency of covers under field conditions are:

- slumping and settling of the underlying spoil material,
- gully erosion,
- available material to act as a growth medium on top of the clay layer,
- geological availability of clay, and
- cost benefit and practical feasibility of selective spoil placement.

The clay layer may be substituted by compacting the upper layer of the levelled spoil. The latter operation will only be successful if the spoil contains a high percentage of clay material. In view of the large scale of opencast mining operations within the study area, it is impossible to rehabilitate the spoils evenly. Localised ingress of oxygen and water into the spoil may therefore still occur and lead to the development of "hot spots" or acidic micro-environments.

Water ingress from rainfall

Water ingress through the topsoil may be quantified by measuring the field capacity and saturated hydraulic conductivity. It can be demonstrated that rainfall events less than 10 mm, do not contribute significantly to recharge through the soil cover. Thereafter, a range of 0 - 15% of the rainfall event applies, depending on the amount and duration of the precipitation. The assumptions for this model are:

- No cracks are present in the soil.
- No pounding occurs on the rehabilitation.
- Field capacity for silty loam is 28%.

- Wilting point for this soil is 14%.
- Effective root depth is 30 cm.
- Soil thickness is 30 cm.

Based on the daily rainfall history of Witbank for the past 64 years, an average recharge of 5,5% of the annual rainfall has been simulated for such conditions. The fact that significant volumes of water enter the spoil in the form of rainfall is confirmed by the stratification of the spoil-water quality, as observed in many of the monitoring boreholes. To reduce the recharge further, the soil thickness could be increased or the triple layer rehabilitation concept could be introduced.

Water ingress from run-off

Water ingress from run-off presently constitutes a large proportion of the recharge which enters the pit. Many internal drainage areas exist on top of the rehabilitated spoils. Run-off from these areas constitutes 5 - 15% (average 10%) of the annual rainfall. The localities of these internal drainage areas are mostly related to past mining history. They constitute final voids and ramps, some of which have been insloped. Low-lying areas such as pans that have been mined through and a drop in the general surface due to the removal of several metres of coal are also factors that contribute to internal drained areas. The percentage internal drainage areas range from 10 - 100% of the total pit area for different mines in the Olifants Catchment.

Most of the run-off from these internal drainage areas accumulates in the pits. This results in the dilution of the spoil water and reduces the saturation level of the potential precipitates.

Elimination of internal drainage areas at existing pits is mostly impractical due to the size of these depressions. Insufficient material is available to fill them. In some instances trenching, thus connecting these internal drainage areas with external drainage systems is possible. A cost benefit analysis is necessary to evaluate each situation with respect to the Catchment as a whole.

At new opencast operations, every effort should be made to rehabilitate the spoils in such a way that surface run-off will drain off the spoils. This can be accomplished by modelling the post-mining surface before the onset of mining. Provision should be made for sufficient diversion trenches and fill material to effectively drain water off the spoils.

Water ingress from groundwater influx

Groundwater entering into the pit constitutes only 10 - 20% of the total water make. Seepage into the pits occurs through the pit perimeter and pit floors. This water will continue to enter the opencast mines indefinitely, since the recovery water levels within pits are lower than the natural state before mining.

Oxygen ingress

Good rehabilitation should reduce the ingress of oxygen into the spoils during barometric variations. Detailed studies by Jaynes *et al.* (1984) indicated that the percentage of oxygen within rehabilitated spoils varies greatly from one locality to the next. In one instance, the oxygen was almost depleted within 4,3 m below surface. However, at five other localities in the mine, no significant depletion of oxygen was found at depths of 12 m. This variation is ascribed to preferential diffusion paths of air within the spoil, during barometric fluctuations. In well-rehabilitated areas, a minimum of air diffuses into the spoil. However, significant volumes of air will penetrate the spoil where gully erosion has occurred, for instance. This venturi effect has led to the spontaneous combustion of the carbon which remained within the spoil at some of the collieries in South Africa. More work is required to determine the pattern and extent of oxygen ingress in spoils.

Carbon dioxide movement

Jaynes *et al.* (1984) have also performed detailed studies on the distribution of carbon dioxide within spoils. These researchers postulate that the significant quantities of carbon dioxide within the spoil are derived from bacteria or neutralisation reactions between acidic leachates and base minerals. Carbon dioxide production during neutralisation reactions for South African conditions is calculated at 4,4 kg/ha/d CO_2. This gas is either dissolved in water or exsolved into the spoil atmosphere. It is anticipated that much of the carbon dioxide dissolves into the accumulated water within the spoil because of the long diffusion pathways to the spoil surface. Good rehabilitation therefore promotes the retention of carbon dioxide within the system.

The absorption of carbon dioxide creates additional acidity within the spoil water. This increases the amount of base potential that is required for the neutralisation of the spoil leachates. Although this may lead to further acidification, the alkalinity of the water is also increased through the simultaneous formation of the bicarbonate ion.

3.3.3.1.2 Selective spoil handling

Spoil handling by the dragline results in an inversion of the stratigraphic sequence. The limited spoil handling ability of the dragline may, however, be utilised by spoiling the weathered and fresh material at different angles with respect to the operating cut.

The truck and shovel method employed by the smaller collieries is ideally suited to selective spoil handling as demonstrated in surface mines at Gillette, Wisconsin. Selective spoil handling has not yet been attempted on any significant scale in South Africa.

It has been demonstrated in this report that weathered strata are inert with respect to acid-base potential. The weathered strata are generally about 10 m thick, within a range from 1 m in the vicinity of streams to 20 m in localised occurrences. Weathered material should preferably be used to cover the more reactive spoils, such as fresh sandstone, shale and coal. Jaynes *et al.* (1984) have demonstrated, with the aid of the program POLS, that the oxidation of pyrite is inhibited if the sulphide-bearing layers are placed 5 - 10 m beneath the spoil surface. This practice increases the oxygen diffusion pathways from the atmosphere.

The concept of selective spoil handling involves the intelligent management of the various acidic or alkaline materials. The following examples are possible management options:

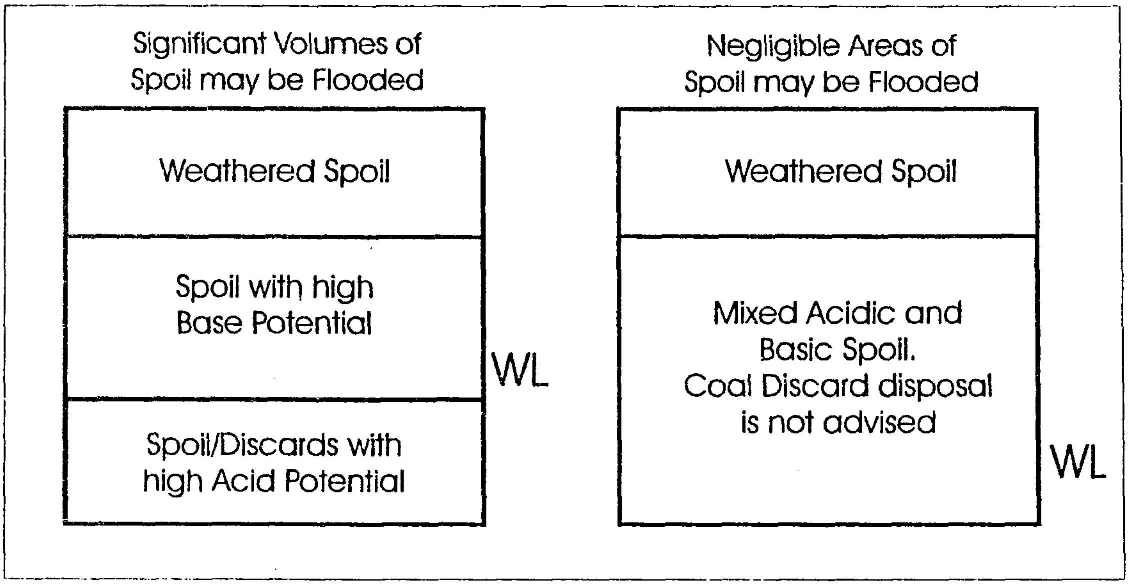


Figure 3.22. Management options for the selective handling of spoil.

In the first instance, it is predicted that water will accumulate within the pit after mine closure. The projected water level is ascertained by field investigations and acidic spoils are placed below the flooding elevation. Their acid-generating potential is thus eliminated unless a source of ferric ions is available. The spoil with a high base potential is placed above the acidic spoils to act as a source for alkaline water. Although this arrangement will eliminate acidic conditions within the spoil water, sulphides within the basic spoils will still oxidise and introduce sulphate into the system. Sulphate levels within the spoil water will therefore still be high.

In the second instance, it is predicted that insufficient water will accumulate within the pit to flood the acidic spoils. In this case, the acidic and alkaline spoils should be mixed to ensure *in situ* neutralisation of acid leachates. This will maintain a neutral pH within the spoil leachates as long as sufficient base potential is present within the system.

In both scenarios, the weathered material is placed above the reactive zone to retard the ingress of water and oxygen.

There are many advantages associated with selective spoil handling. These are:

- Acidic conditions in the spoil water may be eliminated.
- Heavy metals are precipitated within the spoils.
- The net salt load of the system is reduced.
- The weathered material at the top is a better growth medium than the unweathered spoils.
- The weathered material inhibits the pyrite oxidation by reducing the influx of oxygen to the reactive surface.

Selective spoil handling is an effective long-term management strategy. This aspect therefore warrants further research to investigate its practical feasibility in terms of mine engineering and operational procedures.

3.3.3.1.3 Introduction of buffering agents

The addition of acid-consuming components increases the salt load in the environment. However, saline alkaline leachates are generally more favoured in the environment than acidic waters. Buffering agents which may be introduced into spoils are varied.

3.3.3.1.3.1 Introduction of lime

The nett neutralising potentials for ten mines have been determined during this investigation. This potential has been expressed in terms of kg $CaCO_3$ per ton of rock. Negative values suggest that lime dosage will be required to keep the spoils neutral.

Based on the closed system concept, the following summary of lime requirements is provided:

Table 3.13. Estimates for the lime requirements of selected mines within the study area.

	Mine NNP (kg CaCO3/t	**Cost R/ha for 20m spoil depth**
Mine 1	-1.5	8880
Mine 2	-0.7	4128
Mine 3	11.0	No application necessary
Mine 5	25.0	No application necessary
Mine 7	-3.8	22800
Mine 9	-2.6	15600

The above costs have been calculated on the 1995 price for lime and do not include handling costs.

Where lime is added to the spoil surface, the base potential is transported into the spoil via the dissolution of the material by percolating rain water. However, there is a limit to the solubility of mineral phases within rain water. The resultant weakly buffered alkaline water may be insufficient to neutralise all the acidity of the spoil, depending on the rate of acid production.

Lime addition is most effective where the neutralising components are thoroughly mixed within the spoil. This places the acid-consuming materials in close proximity to the acid-generating sites, thereby inhibiting the development of acidic environments. This counteracts bacterial oxidation of the pyrite.

3.3.3.1.3.2 *Introduction of power station fly ash*

Power station fly ash is considered as a salable commodity by the power generating industry in South Africa.

Tests performed on power station fly ash from the eight power stations in the Olifants Catchment, provided the following base potentials:

Table 3.14. Power station fly ash NNP-determinations for eight power stations situated in the Olifants Catchment.

POWER STATION	CaCO3
	kg/t
1	54
2	55
3	71
4	81
5	103
6	132
7	159
8	171

The capacity of the fly ash to neutralise acid mine drainage is only about 5% - 16% of that of lime. A further disadvantage of fly ash addition is that the material contains significant concentrations of heavy metals. If the fly ash dosage is insufficient for complete neutralisation of the system, the resultant acidity will be accompanied by the additional release of heavy metals. An excess dosage of fly ash will therefore be required to compensate for the higher risk associated with the use of this ameliorant.

Fly ash is currently used for the filling of ramps and final voids in two opencast pits. The spoils in this area are characterised by an alkaline NNP and the addition of fly ash will further increase the alkalinity of the system. Heavy metals will therefore be retained within the spoil and ash environments. More information is available in Section 4.3.2, where aspects of ash disposal from power stations are discussed.

It may be concluded that the use of power station fly ash is only suitable for mines which are slightly acidic or alkaline in nature. It is not recommended that this material is used in very acidic mines, due to the high heavy metal content of the ash.

Investigations into the use of burnt discard and burnt spoil, for neutralisation purposes, have also been performed. The conclusion was drawn that the discard and spoil do not burn fully and that salts are readily available from partially burnt material. Water that comes into contact with this material is high in salinity.

3.3.3.1.3.3 *Introduction of sewage*

Extensive research has been conducted in the United States on the application of sewage sludge to mine spoils (Kardos *et al.*, 1979; Sopper and Kerr, 1981). An improvement of the spoil pH from 2,7 to 4,4 - 5,2 is evident in pilot studies where sludge applications have been tested (Joost *et al.*, 1987). Van der Merwe *et al.* (1983) described the beneficial effect of ferric iron in the removal of phosphates from a sewage system. Ferric-rich acid mine water may therefore be amalgamated with sewage to improve the overall quality of both effluents. The introduction of organic carbon will also assist in the bacterial reduction of sulphate in the mine water (Berner, 1964).

Before sewage is introduced into spoils, the following aspects require attention:

- Nitrification of the sewage is inhibited by acidic solutions. Furthermore, the nitrification process may generate additional acidity in the solution. Van der Merwe *et al.* (1983) describe the following reaction:

  $NH_4^+ + 2O_2 = NO_3^- + 2H^+ + H_2O$

  This suggests that 7,1 mg alkalinity (as $CaCO_3$) is destroyed for each milligram ammonia (as N) which is oxidised.

- The characteristics of the mine water in opencast collieries are too erratic to allow the prediction of the optimum mixing proportions.
- The alkalinity of the spoil water may be insufficient for the neutralisation of the combined acidity of the spoil and the nitrification reactions.
- The sewage generated within a specific area may be inadequate for the amelioration of the vast quantities of mine water. Significant quantities of sewage are only available at Witbank and Middelburg.
- Specific mine waters may lack the necessary iron concentrations required for the effective removal of phosphates from the resultant solution.

According to the reactions of Maree *et al.* (1986), 64 g of carbon is required for the reduction of 96 g of sulphate. A total of 6,7 t/d of carbon will therefore be required to reduce the anticipated average sulphate load of 10 t/d per mine. It is expected that the quantities of sewage sludge available would not have a meaningful impact on a catchment scale.

The conclusion is drawn that the introduction of sewage sludge can be considered onto opencast mines west of Witbank, an area dominated by its high acid-generating potential. Further research will be necessary.

3.3.3.1.4 *The application of bactericides*

Extensive research has been conducted on micro-organism control within coal environments. A combination of sodium lauryl sulphate (SLS) and sodium benzoate is an effective inhibitor of *Thiobacillus ferrooxidans* (Dugan, 1985) on laboratory scale. This is confirmed by the work of Loos *et al.* (1990) for South African coal dumps.

SLS is used as an ingredient in toothpaste while sodium benzoate is an accepted food preservative (Budavari *et al.*, 1989). These bactericides are therefore relatively non-toxic and should not pose an environmental threat on the short term. The health effects of long-term exposure to bactericides do not appear to have been investigated to date.

Although the impact of the chemicals is immediate, the presence of bacteria may be noted two to five weeks after treatment (Dugan, 1985). In the case of opencast mining, leaching of the inhibitors from the spoil will reduce the effectiveness of these chemicals. Further work by Dugan (1987) indicates that low concentrations of the bactericides (25 mg/ℓ) actually stimulate the acidification process.

Pellets containing bactericides were successfully applied to spoil waters in the United States (Sobek *et al.*, 1985). The water quality improved between 82% - 95% with respect to acidity, iron, aluminium and sulphate concentrations. The duration of release of the bactericides from such pellets is three to five years (Sobek*et al.*, 1985).

A pilot study was conducted for South African coal dumps (Loos*et al.*, 1990). Effective inhibition of the bacteria only occurred once the adsorption sites for the bactericides within the coal material were saturated. These researchers estimate that the quantity of SLS and sodium benzoate required to achieve the saturation is in the order of 600 mg/kg and 500 mg/kg respectively.

Attaining such saturation for the top 0,5 m of the spoil would cost R29 000/ha for SLS at 1990 prices (Loos *et al.*, 1990). The application of sodium benzoate would cost approximately R13 000/ha at 1990 prices. This translates to R100 000 000 for a mine of 2 000 ha for a once off application. This is clearly not economically viable for opencast mines.

3.3.3.1.5 Containment

The purpose of containment may be twofold:

- To flood as much of the spoils as possible, thus eliminating oxidation of sulphides, and/or
- To contain and evaporate the spoil water.

Although the philosophy behind containment is sound, there are several practical limitations. These are discussed in the following paragraphs.

The final water level in any pit is regulated by its decant level. The decant level is the lowest topographic level where water from the pit will eventually overflow onto surface. All pits that are not subjected to artificial water extraction will decant in future. The rate of water accumulation within existing abandoned pits is usually such that the decant level is reached within 5 - 10 years after pit closure.

The majority of opencast mines lie adjacent to streams. Mining usually ranges from 0 - 40 m in depth. The regional topography is such that within 1 - 2 km from the stream, the coal is too deep to be mined by opencast methods. Because of the close proximity of opencast mines and their decant positions to streams, the greater portion of the spoils cannot be flooded. Exceptions may occur at small pits.

The decant level may be artificially raised by building levees next to the streams. In instances where opencast mining has disrupted existing streams, dams may be built within the catchment immediately below the mines, thus flooding as many pits as possible. However, it is envisaged that flooding of significant proportions (>30%) of the larger opencast pits is not possible. The possibilities of increased flooding are therefore limited and this management option offers a long-term solution for a few of the smaller pits.

The second reason for considering containment of spoil water is where evaporation is a potential management strategy. The net evaporation rate from free standing water surfaces for the Witbank Area is 700 mm per annum. The average recharge to a well-rehabilitated opencast pit of 1 000 ha, with 40% internal drainage, is 2,5 megalitres per day. To evaporate this amount of water, an area of 130 ha (i.e. 13% of the pit area) will be required. Ideally, an evaporation area should be located within the pit itself, to avoid the cost of pumping water from the mine into the evaporation area. Systems that require pumping are not self-sustaining and consequently cannot be considered as long-term solutions. The design of in-pit evaporation systems as an integral part of the final rehabilitated topography is a promising option that requires further investigation in terms of practical mining/operational feasibility for new and existing mines. However, the creation of sufficient evaporation areas at most of the existing mines, is not possible.

Containment, whether within the spoil or in evaporation areas, will lead to further salination of the water. Experiments and calculations have been done to predict the chemical evolution of spoil water under these conditions. The results are presented in Figures 3.23 and 3.24. In these figures, the experimental results are designated as 'Lab' values and calculated values, using chemical equilibrium models as such in these diagrams.

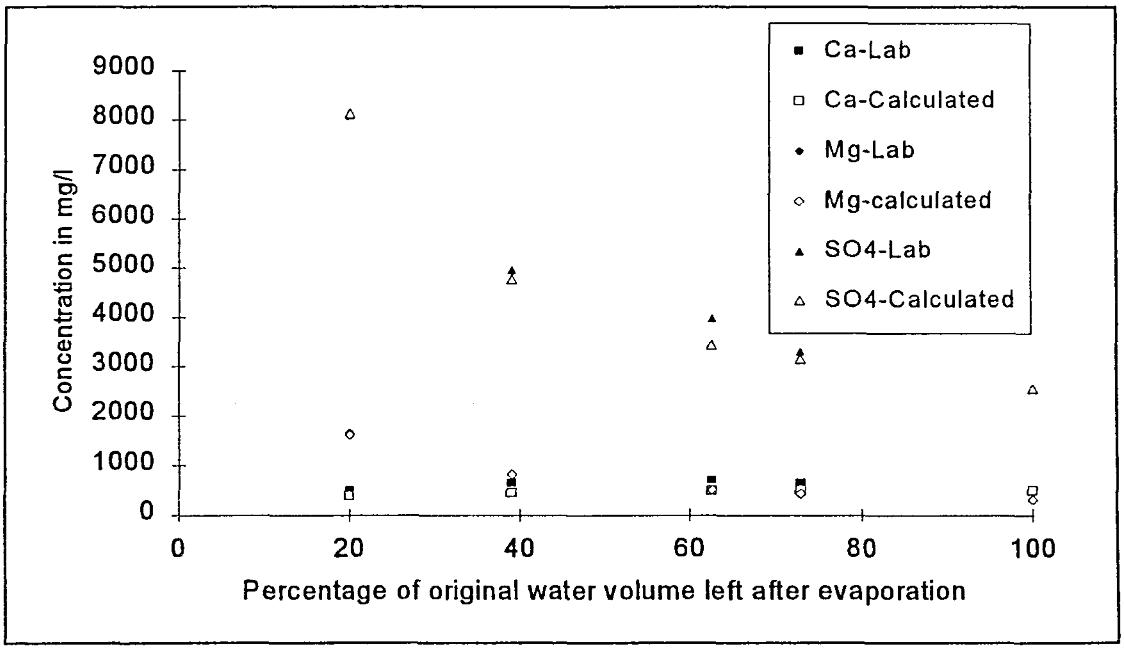


Figure 3.23. Concentrations of constituents in alkaline (pH 8,03) spoil water during evaporation experiments and modelling.

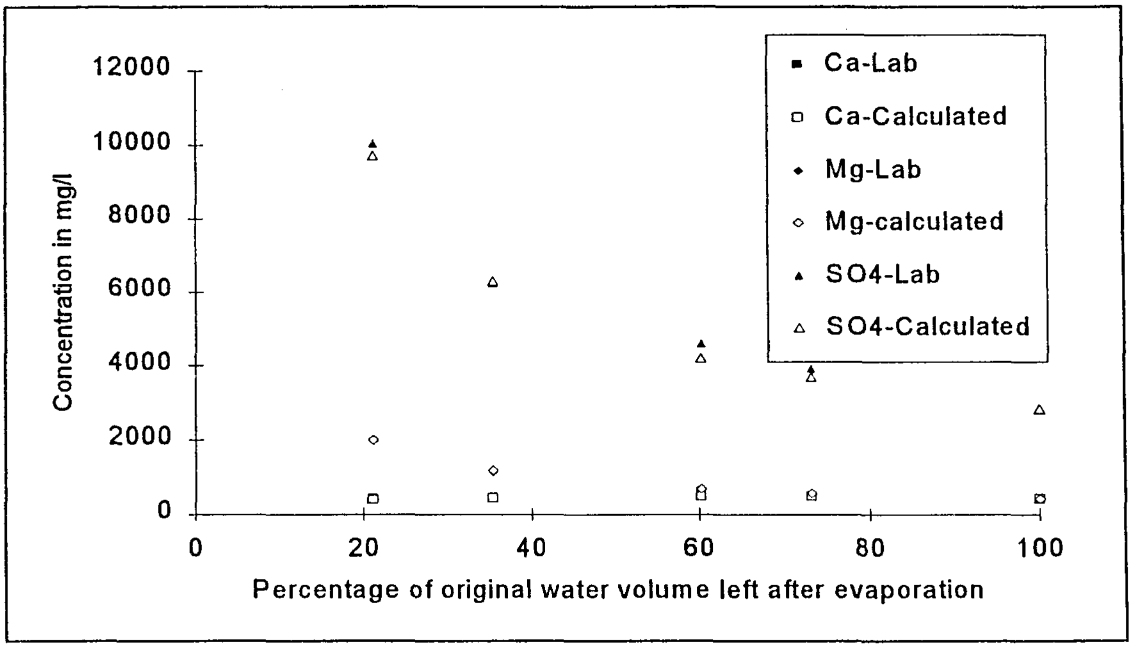


Figure 3.24. Concentrations of constituents in acid (pH 3,38) spoil water during evaporation experiments and modelling.

The following conclusions are drawn from these experiments:

- The close correlation between the experimental and calculated values was expected, since the chemical equilibrium model PHREEQE, that has been used, is known to accurately simulate field conditions.
- The precipitation of calcium after 40% of the water has been evaporated, proves:

- That the spoil water has not reached gypsum saturation at current levels of 2560 mg/l SO4 and Ca of 500 mg/l.
- Gypsum saturation is only reached in the vicinity of 3900 mg/l SO4 and Ca of 700 mg/l in the case of the alkaline spoil water.
- Precipitation of gypsum has negligible influence on sulphate concentrations. Sulphate concentrations continue to increase to more than 10 000 mg/l.
- Magnesium saturation is not reached, even at 80% concentration.
- In the case of water containment within the spoil, similar concentrations, as in the case of evaporation, will occur. Arguments behind this statement are as follows:
- The base potential in the spoil occurs as dolomitic lime, containing a 1:1 ratio of Ca:Mg as meq.
- Upon reaction of acid with the base minerals, calcium and magnesium are released in equal proportions.
- Upon reaching saturation, calcium will precipitate as gypsum, in the same way as in the evaporation experiments.
- Magnesium and sulphate concentrations will continue to increase along similar lines as in the evaporation experiments. This is unfavourable, because water that contains high magnesium and sulphate concentrations is toxic (Kempster and Smith, 1985)

The conclusion is drawn that containment is a possible option for the management of spoil water. However, containment appears to be more applicable to new opencast mining, where total containment can be planned. At existing collieries, it would be practically impossible to reduce the inflow of water to such quantities that containment is achieved.

3.3.3.1.6 Flushing

Flushing implies the introduction of clean water into opencast pits or portions of opencast pits, to regulate the water quality within the spoil, thus ensuring that the water quality does not deteriorate beyond that which can be used locally for irrigation or other purposes.

Natural flushing has been ongoing for up to 15 years at operational and closed collieries. Examples of the positive effect that flushing has on the in-pit water quality have been included in several sections in this document. It therefore suffices to summarise the evidence and refer the reader to these sections:

- The conductivity graphs in Figure 3.9 illustrate two important flushing principles:
- The stratification in water quality in the spoil is the direct result of water being recharged from surface, adding clean water on top of polluted pit water.
- The much better quality of the water in electrical conductivity profile 2 is due to the additional flushing effect of a river that crosses the spoil at that particular mine.
- The fact that none of the pit waters in the Olifants Catchment have reached gypsum saturation, is evidence of the effect which flushing by rain water has. Based on the current average concentration of 2 700 mg/l SO_4 in semi-stagnant pit waters, it can be demonstrated that rain water to the extent of 1,9 m^3/d/ha on average is currently recharged. This amounts to 12% of the annual rainfall.

In closed pits, flushing can be optimised by for instance designing the rehabilitation to introduce run-off into a pit. Water may for instance be introduced into pits through ramps and voids that have not totally been filled in. Such recharge points already exist at many collieries. Sulphate concentrations in the vicinity of these recharge points range from 50 - 600 mg/l, as opposed to 2 500 - 3 000 mg/l sulphate in more stagnant in-pit waters.

Although flushing will reduce the concentration of salts in the pit water, sulphate concentrations may still be above acceptable limits for discharge into streams or dams. Irrigation with this water is an alternative use. Several farmers are presently using spoil water for irrigation within the Olifants Catchment.

Flushing systems at opencast mines will have to be designed for and properly managed by:

- Connecting low-lying areas within pits by trenches to the decant point,
- Contouring rehabilitated surfaces to introduce run-off into selected areas of the pits, and/or
- Diverting natural stream flow into the pits.

This management style considers sulphate concentration to be more important than sulphate load. This is in direct contrast with the philosophy of keeping "clean water clean" or "containment of spoil water". It is clear that the long-term behaviour of a system needs to be understood before a decision on any of the options can be made.

In conclusion, it may be indicated that for the current opencast mines, some degree of flushing will always be part and parcel of their water management strategy. Large internal surface drainage areas, infiltration of rain water and influx of groundwater exist at these mines. At new opencast mines, environmental impact studies must be done to decide on either containment or flushing. The designs of the two systems are different and in most instances, flushing systems cannot afterwards be converted into containment systems.

3.3.3.2 A comparison of groundwater amelioration techniques

A review of the long-term effectiveness of the various groundwater amelioration options at opencast mines, is presented in the following tables:

Table 3.15. Summary of amelioration techniques associated with rehabilitation at opencast mining.

AMELIORATION OPTION	LONG-TERM EFFECTIVENESS
REHABILITATION	
Contouring of the rehabilitated surface	**Pros:** This effectively diverts surface water, thus minimising infiltration through the spoil.
Natural covers	**Pros:** This is an effective option to retard water and oxygen ingress into the spoil. **Cons:** Covers erode and crack, thus allowing free movement of air and water. Current covers are mostly inadequate to prevent ingress of water and moisture.
Selective spoil handling	**Pros:** This is a very effective option to retard oxidation and eliminate a portion of the acid potential. **Cons:** Modifications to accommodate this at existing mines will be very costly. At new mines, this may be implemented.

Table 3.16. Summary of amelioration techniques associated with the introduction of buffering agents at opencast mining.

AMELIORATION OPTION	LONG-TERM EFFECTIVENESS
BUFFERING AGENTS	
Lime	**Pros**: This is an efficient and proven neutralisation agent. **Cons:** High cost. More research is required into the feasibility of using lime, as opposed to carbonate minerals.
Caustic soda	**Cons:** This additive should not be used due to the danger of elevated sodium levels.
Power Station Fly Ash	**Pros:** Although this additive is not as efficient as lime, it is often readily available. **Cons:** This additive should not be used in mines with a nett acid potential, since acidic leachates will mobilise heavy metals from the ash.
Sewage sludge	**Pros:** Sewage sludge has been used overseas with success. **Cons:** Sewage in the Olifants Catchment is insufficient as a carbon source for the reduction of sulphate.
Bactericides	**Cons:** The application of bactericides is very expensive. It is not recommended because repeated application is required.

Table 3.17. Summary of other amelioration techniques at opencast mining.

AMELIORATION OPTION	LONG-TERM EFFECTIVENESS
OTHER OPTIONS	
Containment	**Cons:** Total containment cannot be achieved in current opencast mines. Containment will lead to an increase in salinity. This water will be unusable because of the high sulphate and magnesium concentrations.
Flushing	**Cons:** Excess water from the pits will have to be utilised locally. This requires further research.

3.3.3.3 Groundwater management strategies for opencast mines

It is possible to summarise the groundwater management strategies for opencast mines in three diagrams (Figures 3.25 - 3.27).

The first diagram provides a step by step summary of the investigations required for a proactive water management strategy at opencast mining. This is followed by Figure 3.26 with the policy for mine effluent management. Lastly, the implications of selective spoil handling are outlined (Figure 3.27).

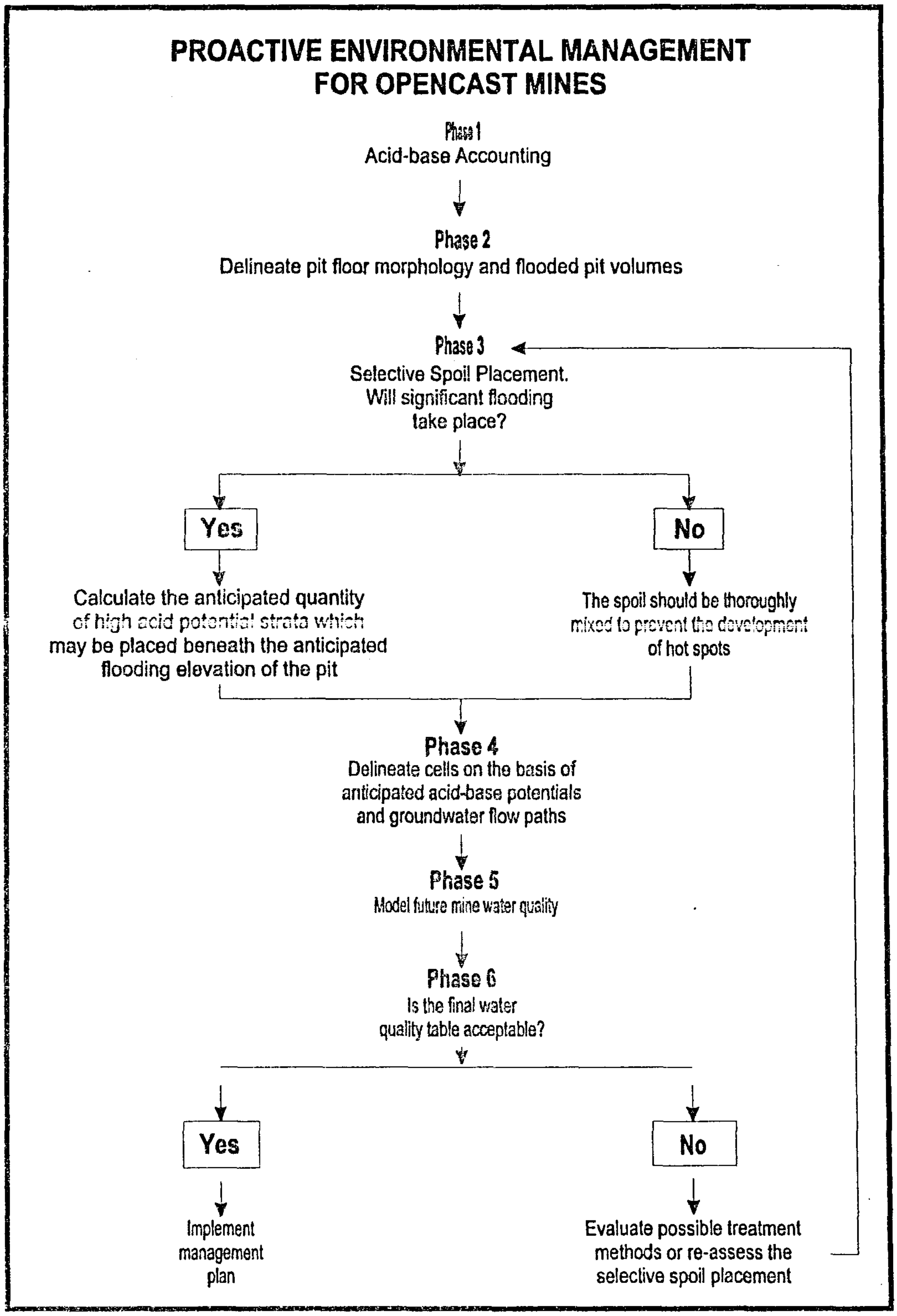


Figure 3.25. Flow chart for proactive environmental management at opencast mines.

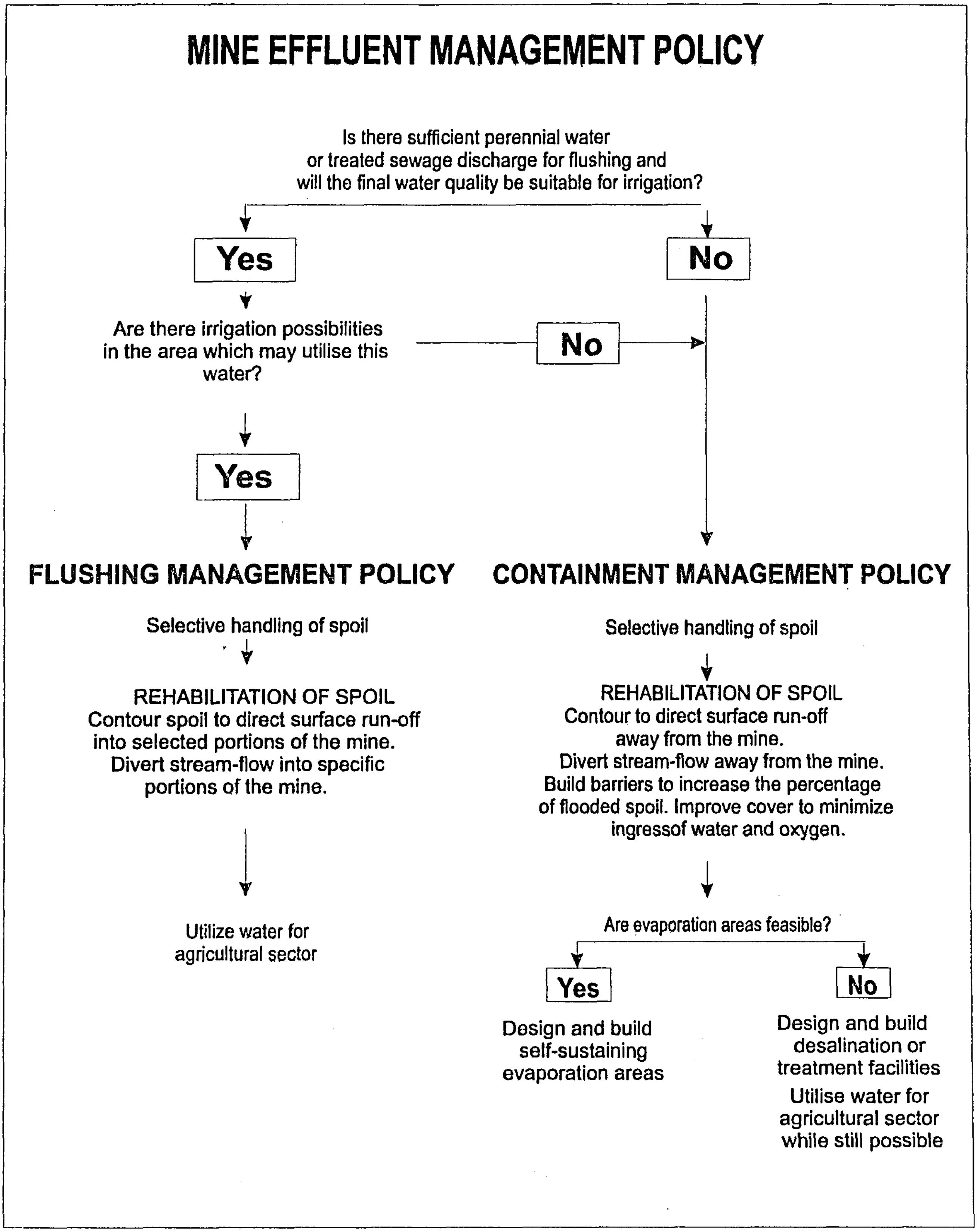


Figure 3.26. Flow chart for mine effluent management policy.

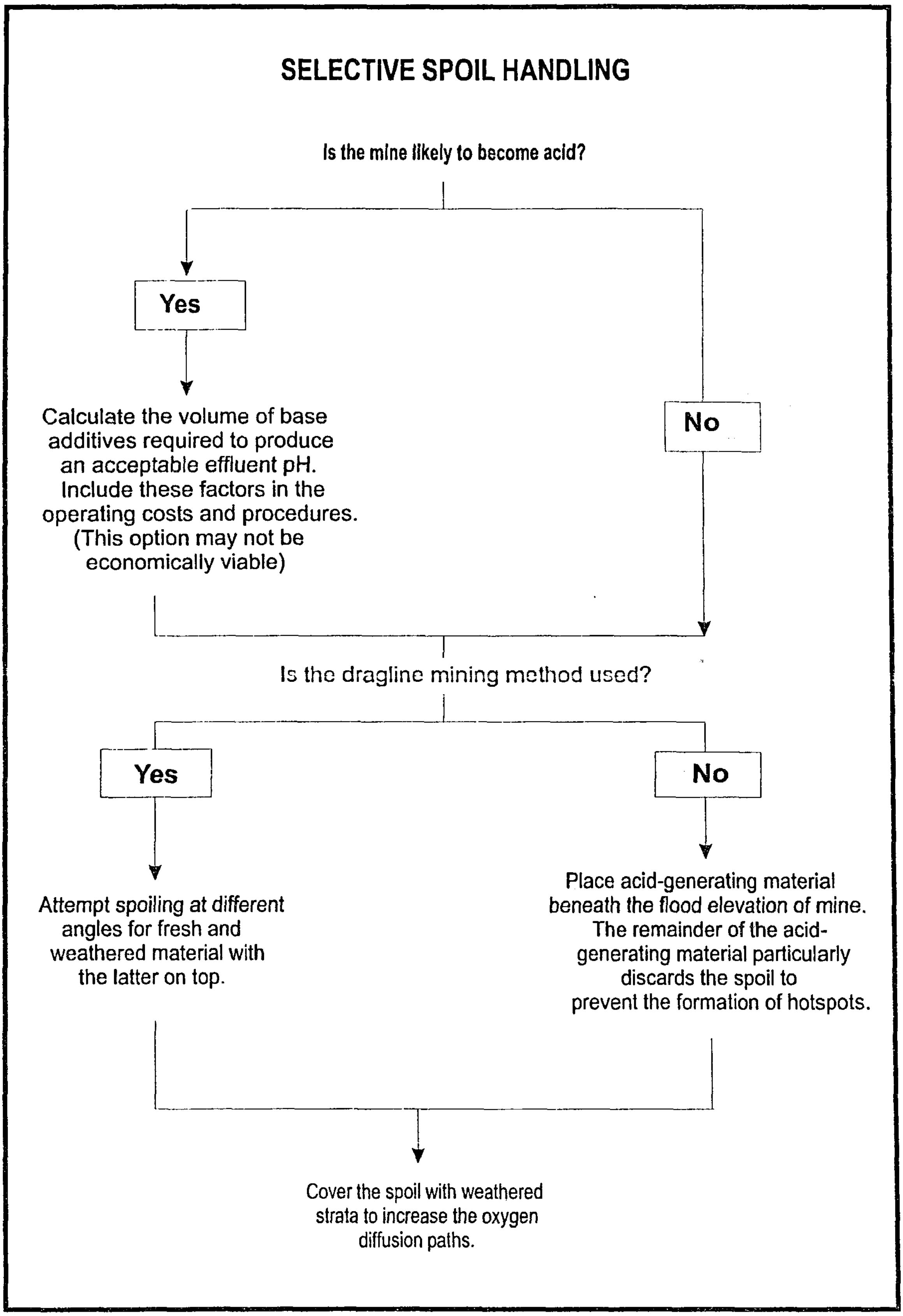


Figure 3.27. Flow chart for selective spoil handling.

These somewhat simplistic representations should be considered as broad guidelines of the major issues at stake. Within each of these issues are endless arrays of secondary issues that need to be considered as part of the greater environmental impact assessment.

3.3.3.4 Conclusions

Many conclusions have been derived within the main body of this section of the report. The conclusions that follow are therefore of a general sense and the reader should refer to the relevant section of the report for more information:

- Opencast mines have significant potential to pollute groundwater in the mines themselves.
- Current opencast mines are likely to decant water onto surface when they close. This will only occur if the void/pond overtops and follows surface contours to a stream. Seepage through undisturbed/natural aquifers in juxtaposition with the pit spoil will also carry affected groundwater down gradient to emerge at decant points away from the pit.
- The local groundwater impact will therefore be transmitted overland into streams and dams.
- It is suggested that very little can be done cost-effectively to minimise surface water pollution from existing opencast mining.

3.4 OTHER ACTIVITIES THAT MAY POLLUTE GROUNDWATER

3.4.1 COAL DISCARD DISPOSAL

3.4.1.1 Introduction

Coal discard has historically been disposed of in surface dumps at the mines. Numerous dumps of this kind, large and small, exist in the Olifants Catchment. The environmental impact of these dumps is known, since their impact is reflected in several ways, namely:

- Most of the old dumps have been burning for many years. It is only during the past 15 years that some of the mining companies are controlling the burning dumps, mostly by compacting outer shells of coal discard and rock around the burning dumps.
- Ingress of water into the old dumps is almost unhindered. In one infiltration experiment, water could not be piped fast enough into the infiltration pond, to flood the bottom of the pond. During rainfall events, very little run-off is observed from these dumps. At burning dumps, the cloud of steam that emanates from the dump is a tell-tale sign that water penetrates into the dumps.
- Acid seepage from all of the older dumps is obvious in the valleys below the dumps. Low pH-levels, high sulphate, high heavy metals and brown staining of the soil and vegetation are present.
- Groundwater pollution has apparently not been monitored extensively in the past. Sites monitored during this investigation, revealed groundwater pH-levels are below 2,5 and sulphate concentrations in excess of 6 000 mg/*l*.

Coal discard is derived from coal beneficiation processes. Coal beneficiation is performed at all coal export collieries and at many collieries that supply the local market. The only collieries that do not have coal beneficiation plants are those that supply coal solely for power generation.

Coal beneficiation usually produces between 20 - 30% coal discard that has to be disposed of or stockpiled locally. In terms of tonnage, estimations range from 25 - 40 Mt discard per annum. This amounts to about 1 Mt of sulphur per annum.

The coal discard is usually stockpiled on undisturbed surface, or within the opencast pit environment. Where possible, topsoil stripping is performed. Coal discards at modern dumps are usually compacted to some degree, thus reducing the possibility of spontaneous combustion of the coal discards. The quality of compaction ranges from very poor to excellent. At many collieries, the coal discard is merely dumped from trucks. At the other end of the scale, are a few collieries where coal discards are dumped through belly spreaders and compacted by heavy roller. This allows better compaction and therefore reduces water ingress and groundwater pollution.

The amount of seepage from a coal discard facility that enters into the ground is a site-specific issue. It is controlled by:

- The degree of compaction of the coal discard.
- The shape of the discard dump, i.e. the ease with which water can run off from the dump.
- The nature of soil cover and vegetation on the rehabilitated portion of the dump.
- The nature of the floor of the dump.
- The hydraulic conductivity of the weathered aquifer below the dump.

To quantify the leachate contribution from specific dumps as part of this investigation served no purpose. Of greater significance in terms of this study, would be the evaluation of coal discard disposal practices as such, and to indicate how these may be improved on. In this respect, the following guidelines may be suggested, in terms of the overall experience with the mining industry in the Olifants Catchment.

In-pit coal discard disposal is the worst possible method because the coal discards raise the acid-generating potential of the pit, thus often turning a potentially alkaline pit into an acid environment. Many such examples exist in the Olifants Catchment.

- Disposal of coal discard in dumps on top of the spoil is not acceptable. Acid from the dump percolates vertically into the spoil, depleting the base potential of the spoil directly underneath these dumps. The acidity then spreads along the floor of the pit. Henceforth, oxidation of pyrite underneath the water in the pit is possible and further acidification results.
- Disposal of coal discards underneath water in a pit should only be considered if sufficient evidence can be given that the water within the pit will never become acid.
- At pits where a sufficient base potential is present, coal discard may be spread across the whole of the pit and mixed in with the spoil during levelling operations. A careful check will have to be kept of the system, to ensure that the base potential of the pit is not exhausted.

3.4.1.2 Suggestions for improved coal discard disposal

Based on observations at many mines where coal discard disposal is done, the following suggestions are made:

Coal discard onto virgin ground

Placement of coal discard onto virgin ground is the accepted current practice. The following is a list of precautionary measures to be taken:

- Strip topsoil.

- All coal discard dumps on virgin ground overlie weathered and permeable aquifers. The depth of weathering is such (5 - 15 m) that interception trenches cannot be dug deep enough. Pollution of the underlying aquifer is therefore inevitable. The design of these discard dumps should therefore be towards minimising the impact on the aquifer.
- As a first measure, discard dumps should be sited on continuous slopes, next to streams, just outside the 1:200 year flood line. This will ensure a minimum of dispersion of the pollution plume. The pollution plume will surface at the stream, because of groundwater flow from the opposite side of the stream, forcing the plume to surface.
- As indicated in the modelling exercise of the pollution plume for fly ash (Section 4.4), the pollution plume from the discard dump can be intercepted by an efficiently designed agricultural drain within the stream itself.
- Compaction of discard dumps is essential to eliminate spontaneous combustion and minimise rain water ingress. Belly spreaders followed by heavy roller is the only true way in which sufficient compaction can be achieved.
- The dumps should be shaped to maximise run-off and minimise erosion by correctly sloping and vegetating the sides. Whaleback dumps or saucer-shaped dumps with outlets are preferred because they allow run-off. Saucer-shaped dumps without outlets should not be allowed, because they accumulate water , which slowly infiltrates into the dumps, thus gathering salt. Stepped-terrace dumps could have erosion problems along steep slopes and water may gather on the flat areas. Seepage of water from these dumps, along the steep slopes, may also occur. This will destroy vegetation at seepage points.
- Co-disposal of slimes and coal discard is performed at several collieries. This may be done by creating a slimes dam within the coal discard or by depositing alternating layers of coal discard and slimes. Van Rooyen (1992) has discussed the advantages of the latter system. He indicated that a dense, practically impermeable composite has been achieved in field experiments. Both systems have excess water, some of which will seep into the underlying aquifer. This seepage may be intercepted in the same way as suggested above. The seepage from a slimes/coal discard disposal site is usually neutral in pH, because of the active circulation of slimes water between the washing plant where it is limed and the slimes dam.
- In instances where coal discard dumps overlie underground workings, particularly where the workings are relatively deep, consideration may be given to the drilling of drainage holes from surface into the mine. The idea is that the holes will create dewatering cones that will intercept acid seepage along the weathered aquifer. These holes will have to be spaced close enough to each other to ensure that all lateral flow is intercepted. In some instances, this may mean a spacing of the holes as close as 20 m apart. Even though this will require a significant number of drainage holes, this option may still be cheaper than other interception systems. Of importance is also the consideration of whether or not the underlying mine can accept the acid water and what the long-term effect of this will be.

Clean and dirty water management systems are in place at certain discard dumps. These facilities should be improved upon to minimise pollution.

Disposal of coal discards in underground mines

Another last possible solution to the problem of discard disposal is one that has not been implemented to any large degree. This entails disposal of coal discards in environmentally safe underground mines. At many of the mines, coal slimes have been disposed of in the underground workings. This is considered environmentally safe, because the mines will be flooded afterwards. To dispose of coal discards in underground workings presents a handling problem and this may only be feasible if the discards can be pulverised on surface, prior to underground disposal. This aspect will have to be researched further, before any meaningful recommendation can be made.

Conclusion

All in all, coal discard disposal is probably the second biggest environmental problem in the Olifants Catchment, after opencast mining. The only true solution to this problem is not having discards in the first place. Steps are presently under way at specific mines to make discards available to nearby power stations. This is certainly one solution. It will imply higher concentrations of SO_X in the power station stacks though, because coal discards typically contain 3% sulphur. Reworking of dumps is also occurring.

Section 4

POWER GENERATION AND ITS IMPACT ON GROUNDWATER QUALITY AND QUANTITY

4.1 PREVIOUS WORK

In South Africa, all of the Eskom power stations have been equipped with groundwater monitoring systems (Hodgson, 1987; Hodgson, 1987; Hodgson, 1988; Hodgson, 1989; Hodgson, 1989; Hodgson, 1989; Hodgson, 1990; Hodgson, 1990; Hodgson, 1991; Hodgson, 1991; Hodgson and Van Niekerk, 1991; Hodgson, 1991). Many other related studies have been completed by consultants and by Eskom, for the siting, construction and operation of various waste and stockpiling facilities at power stations. Eskom has unconditionally volunteered this information for the present investigation. This provided a significant database on which this investigation could continue to build.

Overseas experience mainly deals with possible groundwater contamination through fly ash disposal. It should be realised though, that much of the overseas experience lies in the co-disposal of flue gas desulphurization wastes with fly ash (Wahlstrom *et al.*, 1990; Beaver *et al.*, 1987; Beaver, 1986; Hupe and Golden, 1983) and that this experience does not reflect the South African situation. The latter causes a drop in the base potential of the ash and could lead to severe leaching of heavy metals from the ash.

Sulphur trioxide in low concentrations (20 ppm) is injected upstream of electrostatic precipitators at Hendrina, Kriel and Matla Power Stations. This improves the resistivity of fly ash and hence enhances the efficiency of particulate removal in the stacks. The net result of SO_3 injection is the formation of small amounts of calcium sulphate on the fly ash particles which results in some addition of calcium sulphate to the ash water (J.B. Conlin, personal communication, 1994).

The overseas experience generally agrees that dry ash disposal has less potential to pollute groundwater than wet ash disposal (Theis *et al.*, 1989; Simsiman *et al.*, 1987; Ripp and Villaume, 1985; Rogers and Kean, 1980). Wet ash disposal systems, having excess water, have been recorded to pollute groundwater by introducing mainly sodium and sulphate into the environment (Hodgson, 1987). Heavy metal mobilisation is not a problem, because of neutral or high pH-levels in the ash environments.

The disposal of power station fly ash in coal opencast pits has been addressed extensively by Libicki (1978, 1983 and 1985). He has concluded that for the systems studied by him, co-disposal of fly ash and opencast spoil can be done safely. He goes on to discuss various disposal options, such as in-pit below water, in-pit above water and on top of opencast spoils. In South Africa, in-pit and on-pit disposal are currently being done at Kriel and Matla Power Stations. These systems have been researched extensively as part of this investigation.

In terms of modelling of pollution from ash dams, two approaches are present in literature. Source term release modelling was done by Serne and Treat (1987), while Whelan (1988) emphasises the need for mass transport modelling to describe the propagation of pollution plumes. Although modelling of pollution has been ongoing in South Africa since 1976, the necessity for modelling of pollution plumes at power stations has not received much attention.

Benson and Parsley (1984) have addressed the problem of efficient groundwater monitoring. The purpose of their Groundwater Monitor Plan is to determine the hydrogeological setting of each facility, provide long-term monitoring of discharge zones and develop a three-dimensional description of any contaminant plume within the zone of discharge. They furthermore emphasise the point that correct chemical and physical conceptualisation of the system are essential prerequisites for successful modelling.

Apart from the above-sited work, numerous other publications relate to environmental issues of fly ash disposal and the co-disposal of fly ash with other wastes.

In terms of other storage and waste disposal facilities, such as coal stockyards and domestic and solid waste, previous work on these items will be discussed under the relevant chapters later in this document.

4.2 THE POWER STATION ENVIRONMENT AS IT RELATES TO WATER QUALITY

4.2.1 INTRODUCTION

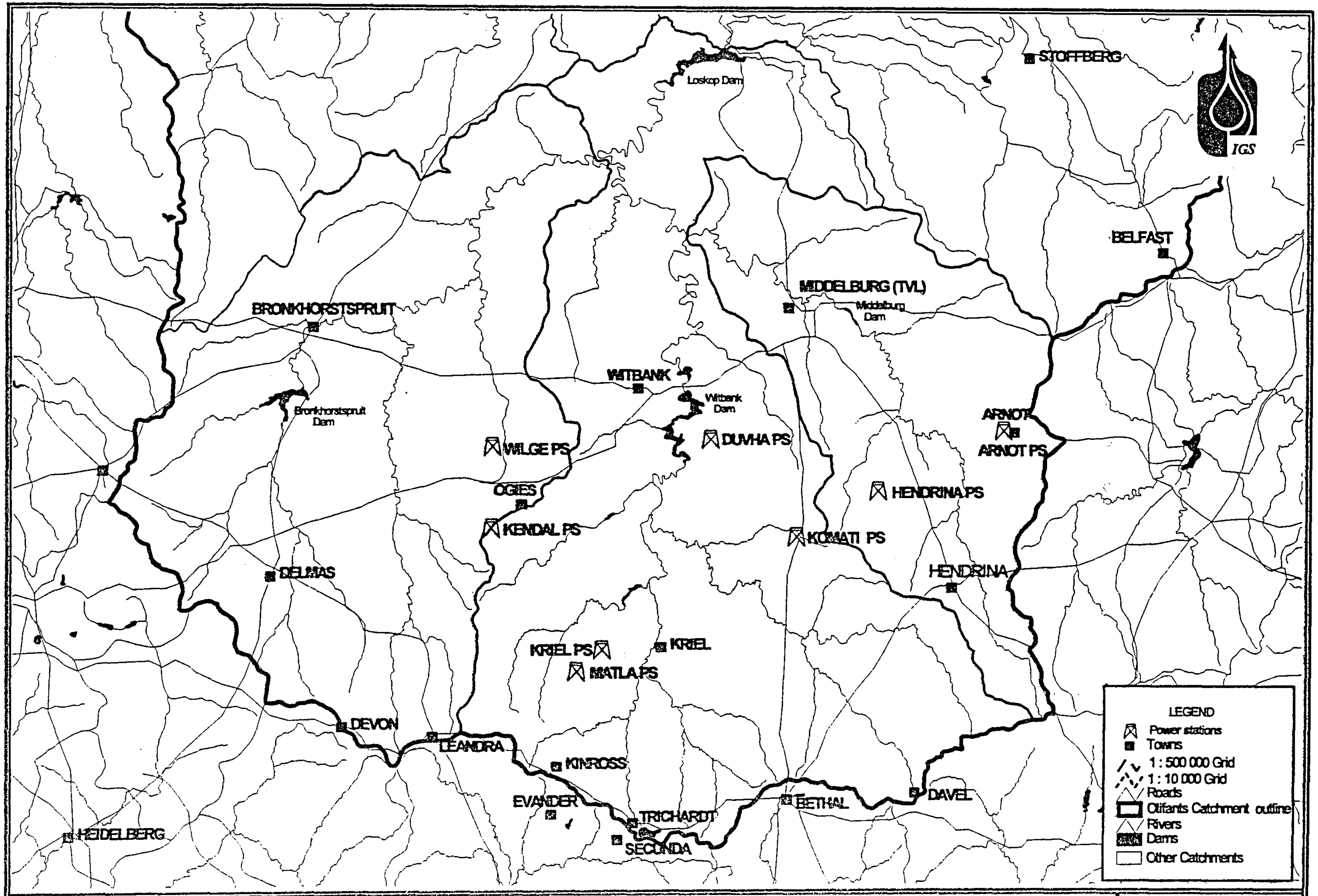
Eight Eskom power stations are located within the larger Olifants Catchment in the Eastern Transvaal. Their localities are shown in Figure 4.1.

One of these power stations, namely Komati, has been mothballed during the course of this investigation. In this time, Wilge Power Station has been scaled down to a training facility and it runs at a very low capacity. Arnot Power Station is also not running at full power. Of the remainder of the power stations, Hendrina is the oldest, having been in production since the mid-seventies. Kriel Power Station came on line in the late seventies. Matla, Duvha and Kendal are the latest flagships of Eskom in the Eastern Transvaal.

In size, the power stations have significantly increased through the years and present-day stations usually consist of six generation units, each with a capacity to generate 600 Megawatt. They typically burn coal at a rate of 25000 tons per day and produce in the order of 7 000 tons of ash per day. Raw water intake is on average 120 Ml/d except for Kendal Power Station, where the intake is only 12 Ml/day for the dry cooled design.

The scale of operations is huge compared to many of the overseas facilities. The potential for power stations to pollute the environment is, for this reason, real. Through the years though, the Eskom power stations have developed methodologies to minimise the impact on the environment.

Associated with power generation, are six distinct activities that could result in a deterioration of groundwater quality. These are: ash disposal, coal stockpiling, dirty water dams, domestic and other solid waste, sewage and surface run-off from the power station area in general. Other items that could contribute to groundwater pollution, such as oil, grease, pesticides, fertiliser and medical waste will not be dealt with separately, because of their small potential impact. Eskom is well aware of the requirements for the disposal of wastes such as medical, oils and other potential toxic substances and these are managed separately.



INSTITUTE FOR GROUNDWATER STUDIES

Figure 4.1. Locality plan of the power stations in the Olifants catchment

DESIGN : A4 SCALE 1: 620 000
PROJECTION : TRANSVERSE MERCATOR
CENTRAL MERIDIAN : 29 DEGREES

4.2.2 FLY ASH DISPOSAL

Coal that is burnt for power generation within the Olifants Catchment generates between 25 - 30 per cent ash. Between 80 - 90% of this ash exists in the form of fly ash with a very fine particle size. The larger power stations bum on average about 10 million tons of coal per annum. This leads to the problem that between 2,5 - 3,0 million tons of fly ash has to be disposed of annually, at each of the larger power stations.

Wet ash disposal has been the preferred disposal methodology in the past. It is only at Kendal Power Station, which is the most recent station to come onto line, where dry ash disposal is currently being done. In the case of wet ash disposal, ash is generally being pumped from the power station to the ash dams in ash-to-water ratios of 1:5 to 1:10 by volume. The excess water on top of the ash dams is decanted through a penstock arrangement, draining water into ash water return dams. From here, water is returned to the power station to pump more ash.

In the case of dry ash disposal, ash is partially wetted at the power station before being transported by conveyor belt to the ash disposal dump. Ideally, the ash on the conveyor belt contains about 15% moisture. This arrangement prevents ash from blowing off the conveyor belt, or in the area where it is being disposed of. Disposal occurs by merely tipping the ash at the end of the conveyor belt. No compaction of the ash, other than under its own weight and under the weight of machinery being used on top of the ash dump, therefore occurs. In addition to the moisture added to the ash within the power station, a watering gun is available in the area where the ash is being tipped to prevent the ash from drying out and creating a dust problem.

Rehabilitation of the sides of ash dams is possible, though usually not done during active ash disposal at an ash dam. Exceptions do, however, occur and rehabilitation is for instance being performed on the sides of the ash dam at Duvha Power Station. Rehabilitation usually consists of the placing of topsoil on slopes of the ash dams, and planting grass and trees. Dry ash dumps lend themselves towards rehabilitation from a very early stage. It is possible to rehabilitate the total dump as it progresses and Eskom has achieved significant success in this respect. The advantages of immediate rehabilitation are many:

- Dust suppression problems are minimised.
- Infiltration of rain water is minimised.
- Rehabilitation costs are part of the operational costs.
- The rehabilitated portions of the dump are aesthetically pleasing to the eye.

Fly ash dams and dumps usually cover in excess of 200 ha at the larger power stations. Duvha Power Station currently has the largest planned wet ashing area, in the range of 460 ha. The dry ash dump at Kendal Power Station is planned to eventually extend over 550 ha. In height, the ash dams and dumps may be as high as 40 m in some areas.

4.2.3 COAL STOCKYARDS

Each of the power stations has a strategic coal stockyard. These coal stockyards are usually located within the security fence that surrounds the power stations. In size, coal stockyards vary significantly, depending on the coal requirement in the power station. Generally, coal stockyards hold sufficient coal to supply the power station for at least two months, in case of an emergency. At larger power stations, coal stockyards typically contain about 2 million tons of coal.

Coal stockyards usually comprise two distinct areas. Most of a coal stockyard (about 80%) is used for storage of compacted coal. The rest of the area comprises a live coal stockpile from where coal is used on a day to day basis. Coal within the live stockpile is not compacted.

The floor area of a coal stockpile is usually a well-prepared flat surface. Soil, clay and lime may have been used to prepare the flat surface. Generally, the flat surface is designed for ease of operation and coal recovery and not specifically to prevent groundwater pollution by incorporating clay into the liner.

A trench to prevent surface run-off and fine coal material from being washed onto neighbouring areas often surrounds coal stockyards.

Coal stockpiles are usually dome-shaped, which encourages rainfall to run off from the stockpile. At the larger coal stockpiles though, it is not always possible to create continuous slopes for run-off and water may pond in isolated areas on top of the coal stockpiles. Until recently, some of the smaller coal stockpiles were saucer-shaped on top. They were also provided with benches along the sides of the stockpiles. This design was intended to minimise erosion of the sides of the stockpiles during heavy rainfall events. Although this objective was achieved to some extent, this resulted in significant pounding of water on top of the coal stockpiles. Much of this water seeped into the coal stockpile, eventually to emanate as contaminated water either at the floor of the stockpile or as seepage within the groundwater underneath the coal stockpiles. This practice of shaping and contouring coal stockpiles to prevent erosion has now mostly been abandoned for environmental reasons.

The compacted coal stockpiles may from time to time be recovered for a variety of reasons.

4.2.4 DIRTY WATER DAMS

All power stations have wastewater streams that do not conform to the general or special industrial effluent standards. These streams are usually disposed of temporarily or permanently into dirty water dams. Two types of dirty water dams usually exist at the power stations, namely man-made dams or natural pans. The man-made dams have usually been constructed within nearby streams. Extensive engineering is, as a rule, necessary to prevent dirty water in these dams from polluting other systems. An example of a dirty water dam arrangement is presented in Figure 4.2.

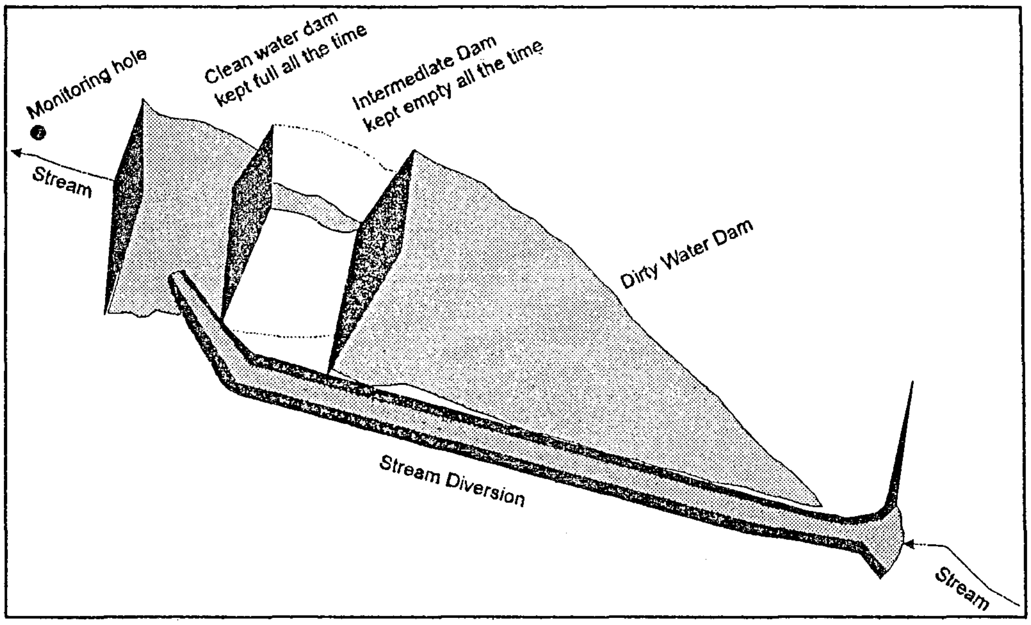


Figure 4.2. Typical arrangement of dirty and clean water dams at power stations.

Two precautions that are generally taken are:

- The natural clean surface run-off from the catchment above the dirty water dam is diverted around the dirty water dam.
- A series of dams are constructed as part of the dirty water holding facility. Typically, three dams would be constructed, immediately below one another within the stream. Usually, the upper dam contains dirty effluent. The intermediate dam is mostly kept dry and acts as a buffer between the upper dirty water dam and the bottom clean water dam. The clean water dam usually receives water from the clean water stream diversion.

Pans are also used at several of the power stations for intermediate storage of dirty water.

Depending on the quality and characteristics of the water within the dirty water dams, the water may be used to wet ash during ash disposal, or for conditioning specific waters within the power station itself.

Certain high salinity effluents from the power station such as the demineralisation effluents may directly be disposed of within the ashing system. The larger power stations typically use about 0,5 ton sodium hydroxide and a similar amount of sulphuric acid daily for a regeneration of resin in the power station demineralisation plant. These plants receive raw water from various sources such as the Komati Sub-system, Usutu-Vaal System and Witbank Dam.

Flue gas desulphurization is not done at power plants within the Olifants Catchment.

4.2.5 DOMESTIC AND OTHER SOLID WASTE

Very small amounts of domestic and other solid waste are being generated at each of the power stations. In view of legislation that has been passed in 1989, regarding licensing of solid waste disposal sites, Eskom has upgraded their domestic waste disposal facilities to comply with existing legislation. Currently, all of the domestic waste disposal sites are fenced in. Disposal is done in an organised way. Applications have been made to the Department of Water Affairs and Forestry for licensing of the domestic waste disposal facilities.

Small amounts of hazardous waste have, in some instances, been disposed of at the power stations. A licensed asbestos disposal site exists at Komati Power Station. Small amounts of hospital and laboratory wastes may also have been disposed of in the past. There is a general awareness at the power stations of possible consequences of disposing hazardous material within domestic waste dumps. Eskom is currently looking into the feasibility of centralised storage for specific hazardous wastes to enable future destruction.

4.2.6 SEWAGE

Each of the power stations has their own sewage treatment facilities. These facilities are managed in accordance to specifications and requirements by the DWA&F. Larger power stations discharge about 1 Mℓ of treated sewage effluent daily.

4.2.7 SURFACE RUN-OFF

Surface run-off from areas surrounding the power stations themselves, is generally not a pollution hazard. Waste disposal facilities are usually provided with adequate mechanisms whereby surface run-off from the waste disposal facility itself is contained. In certain areas around the disposal sites, particularly where large amounts of water are handled, crystallisation of salts at surface may occur. White or brown crystals that appear on surface during dry periods usually note these areas. The crystals usually consist of sodium sulphate, which is easily dissolved and mobilised during follow-up rainfall events. Another example of contamination of surface run-off can be found particularly at the older power stations, where coal discards material may have been used to stabilise roads. In these instances, evidence of acid rock drainage, followed by low pH-values and precipitation of iron, has been noticed.

At modern power stations, coal discard is generally not used for road stabilisation. There is a general awareness of the severe consequences which shallow seepage and acid rock drainage can hold for environmental pollution. It is therefore unlikely that similar problems will surface at the newer power stations.

4.2.8 AIR POLLUTION

The contribution of stack emission to air pollution and thus water pollution has been studied exhaustively by the CSIR and Eskom (Turner *et al.*, 1990). Apart from monitoring groundwater quality, no direct work relating to the effect that stack emissions have on groundwater quality, has been done during this investigation. The impact of air pollution on groundwater will be discussed in the section to follow.

4.3 IMPACT OF POWER GENERATION ON GROUNDWATER

4.3.1 INTRODUCTION

The main impact that power generation may have on groundwater quality and quantity is from temporary or permanent disposal of waste on surface. As indicated in Chapter 4.2, this waste may exist in a variety of forms.

Groundwater monitoring systems have been installed at all Eskom's power stations within the Olifants Catchment. These monitoring facilities consist mainly of boreholes that have been drilled in strategic positions to observe possible movement of pollution plumes away from waste management facilities. Eskom also collects surface water samples on a routine basis. Much of the information is entered into HydroCom databases at the power stations.

4.3.2 FLY ASH DISPOSAL

4.3.2.1 Wet ash disposal

4.3.2.1.1 Ash dams

Typical ash water chemistries within wet ash disposal systems for each of the power stations are presented in Figure 4.3.

The ranges of chemistries for the ash water are not necessarily true reflections of only the ash water chemistry, since ash water is often used within the power station for other purposes or other waters are mixed with the ash water, before it reaches the ash dams.

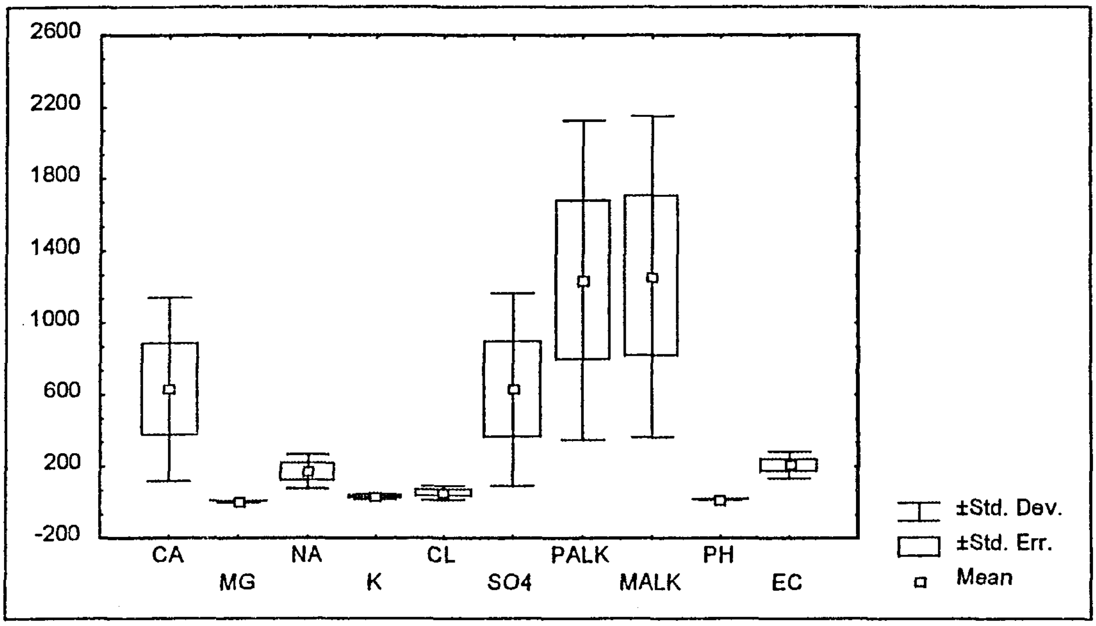


Figure 4.3. Box and whisker plot of various constituents (mg/l where applicable) in fly ash water from power stations in the Olifants Catchment.

The main characteristics of these waters are:

- High pH, usually above 12,0.
- High calcium content (Figure 4.4), often reaching saturation. Repeated leaching of calcium from the ash is possible, because of the significant amounts of calcium oxide present within the ash.

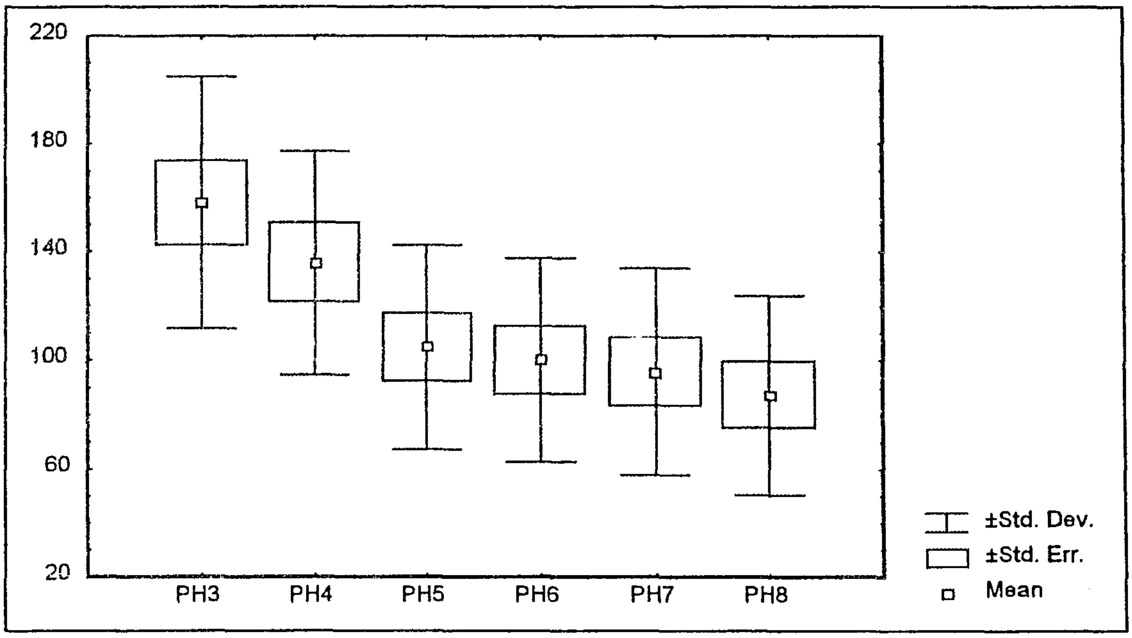


Figure 4.4. Potential of power station fly ash (kg $CaCO_3$/t of ash) to counteract acidity, at different pH-levels.

- The absence of magnesium in the ash water is due to the very limited solubility of magnesium at pH-levels above 11.

- Ash water generally contains low fluoride levels, partly because of low fluoride in the coal and partly because of the limited solubility of fluoride in the presence of high calcium concentrations in water.
- While most of the heavy metals are insoluble at pH-levels above 11,0, aluminium has a significant solubility at this high pH-level. It has been found, however, that aluminium concentrations are generally lower than expected. This deviation is discussed later in this section of the report.
- Sodium is generally not a constituent that is present in significant quantities within ash water. Sodium is, however, added to the ash water during the disposal of demineralisation effluent. In the actual field situation, variable amounts of sodium may therefore be encountered within the ash water, depending on the exact circumstances under which the sample was taken.
- The main source of sulphate in fly ash water is from the demineralisation effluent. Concentrations are typically in the range of 200 - 1000 SQ.

The above information should therefore not be regarded as conclusive and fully representative of ash water characteristics for all existing power stations. Differences do occur from one power station to the next.

The ash water chemistries as described above are unstable in the presence of carbon dioxide. As carbon dioxide from the air enters into the water at the ash dams, calcium carbonate precipitates. This results in a drop of the pH. Sodium, which has an almost unlimited solubility, remains in solution in the ash water. Gradually, through the years, sodium concentrations in the ash water may rise. At some of the older power stations, sodium levels in excess of 300 mg/l have been recorded.

Some of the ash water from wet ash disposal facilities seeps into underlying aquifers. At certain power stations such as Kriel, wet ashing facilities are underlain by clay material that limits vertical permeation of ash water into the underlying aquifer. A typical hydraulic conductivity range for the underlying clays at Kriel Power Station is presented in Figure 4.5. This is certainly lower than that of the ash, as is also demonstrated in this diagram. The wet fringe along the bottom of the ash dam at Kriel Power Station (Figure 4.6) confirms the results of this comparison.

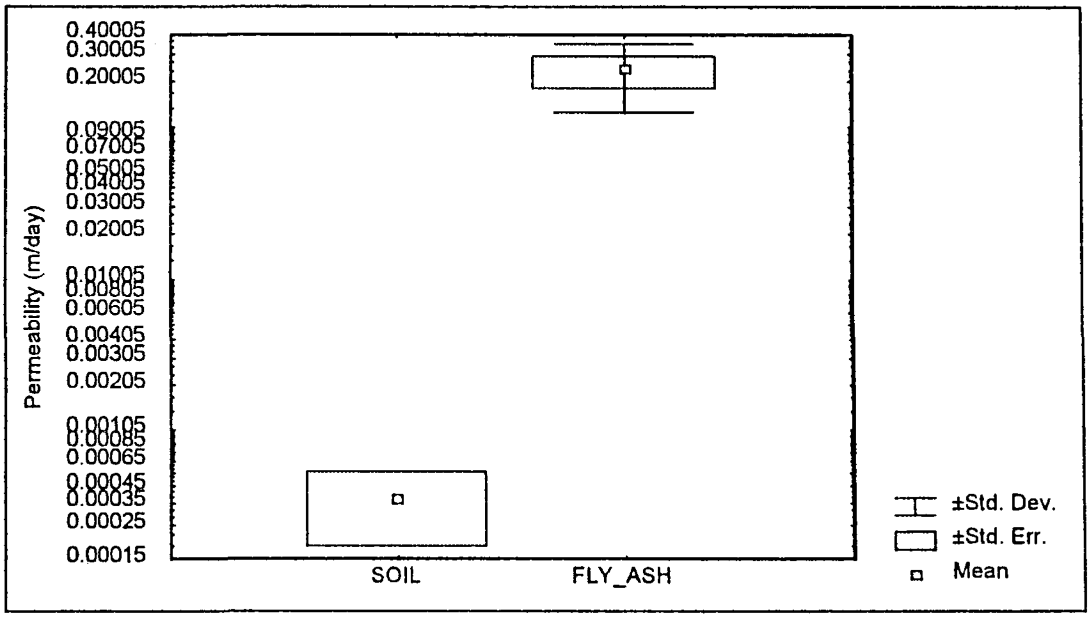


Figure 4.5. Hydraulic conductivity ranges for soil and fly ash at Kriel Power Station.

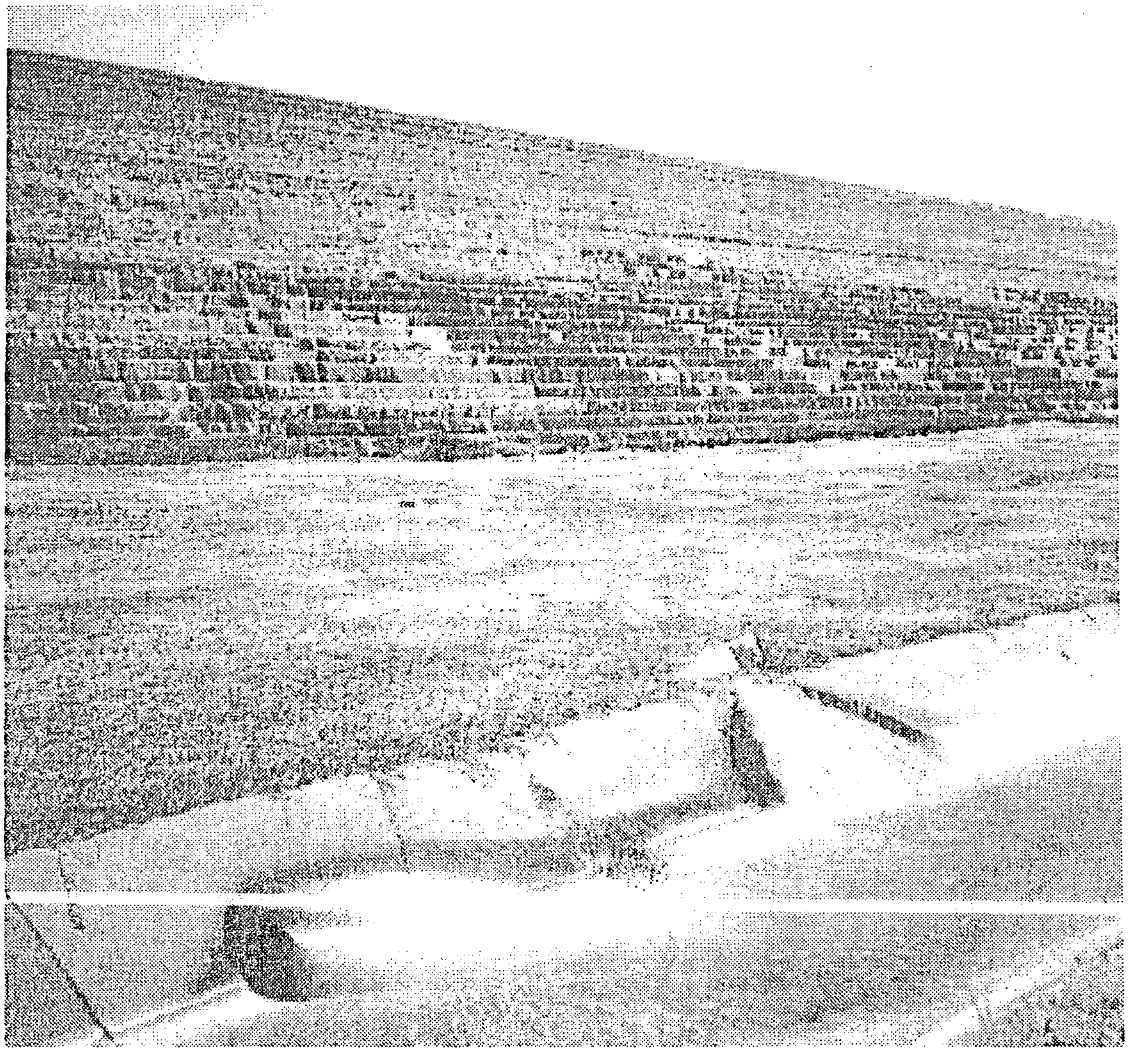


Figure 4.6. Photograph of the old fly ash dam at Kriel Power Station, showing the wet fringe around the bottom portion of the dam and cut-off drain with penstocks decanting water.

In the photograph (Figure 4.6), the darker bottom portion of the ash dam represents seepage of ash water from the ash dam, because of the insufficient hydraulic conductivity of the underlying clays to transmit water fast enough into the underlying strata. The trench to intercept overland flow from the ash dams and pipes that decant water through penstocks on top of the ash dam can also be seen in this photograph.

The strata underneath all wet ash dams have some hydraulic conductivity. Elevated water levels within monitoring boreholes (Figure 4.7) at all ash disposal facilities leave no doubt that significant amounts of ash water permeate from the ashing facilities into the underlying groundwater.

Once the ash water has percolated through the 1 - 3 m of soil, it is free to permeate deeper down into the weathered Karoo sediments. The weathered strata have significantly higher permeabilities, generally to a depth of about 10 m (Figure 4.8). As can be seen from this presentation, the unweathered Karoo sediments deeper down have very low permeabilities. Seepage from the ash dams should therefore affect only the weathered portion of the Karoo sediments. The weathered horizon extends laterally over the whole of the Eastern Transvaal Highveld, at thicknesses ranging from less than 5 to 20 m, with an average of 10 m. Lateral flow of ash water within the weathered zones is therefore possible.

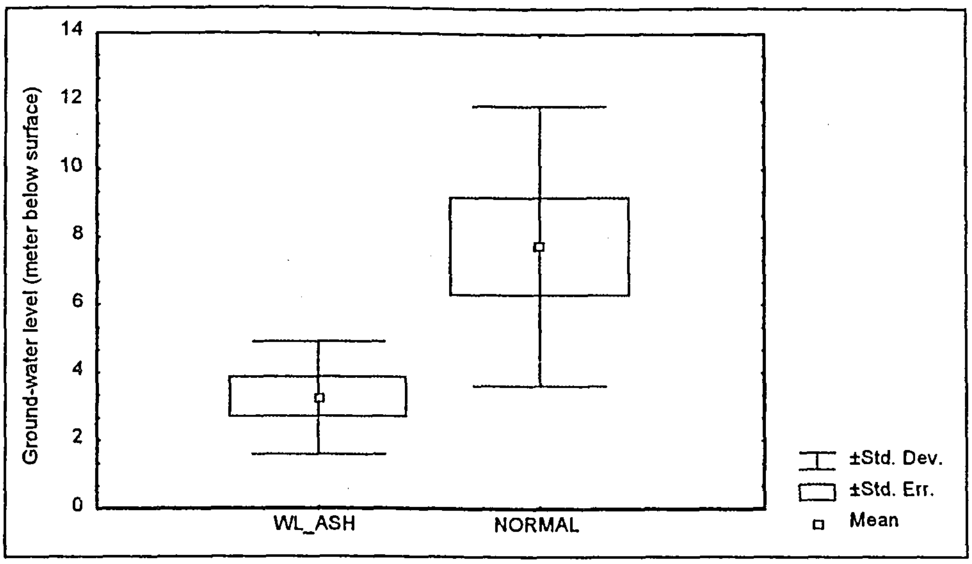


Figure 4.7 Range of groundwater levels within close proximity of wet ash dams in comparison to the normal range of groundwater level depths, away from the ashing facilities.

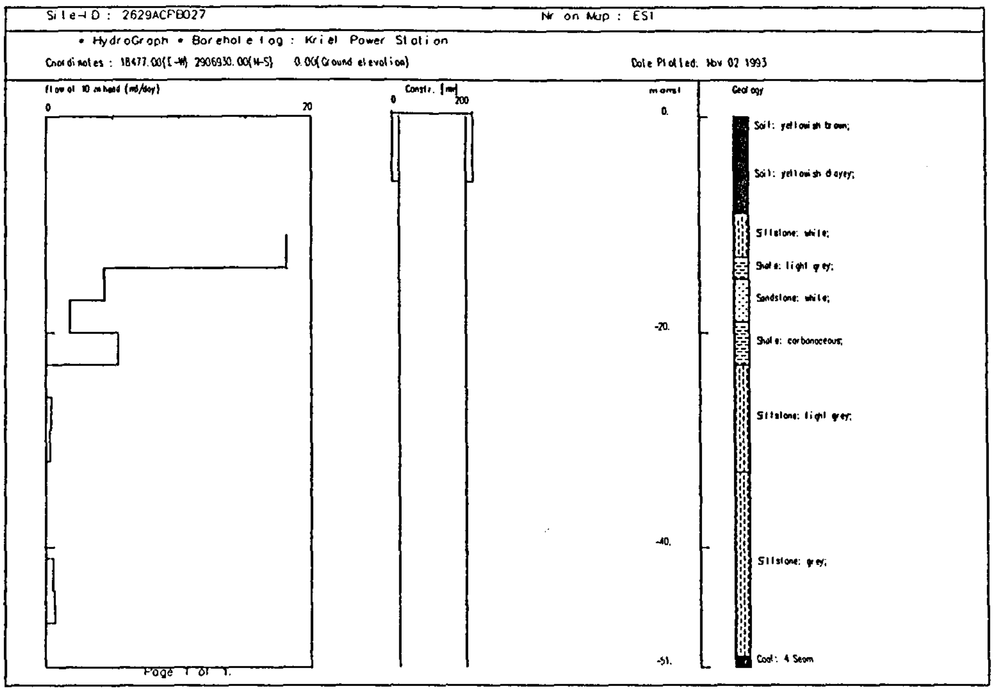


Figure 4.8 An example of hydraulic conductivity distribution of the lithological profile underneath the Kriel ash dam.

Even though this seepage into the underlying strata occurs, very few of the ash water's components are carried into the underlying aquifer. This is due to the unstable chemistry of the ash water. Examples of ash water and groundwater chemistries within boreholes in close proximity to ash dams are presented in Figure 4.9.

The following conclusions are drawn from this comparison:

- The unstable components are filtered out to a very significant degree from the ash water, before it reaches the aquifer.
- This filtering action is probably a combination of the unstable chemistry of the ash water, as well as adsorption or complexation of specific ions onto clay material underneath the ash dams.

To determine which of these mechanisms are the main contributing factors, two further experiments have been conducted. These constituted XRF-analysis of the clays, and sulphate adsorption tests by the clays.

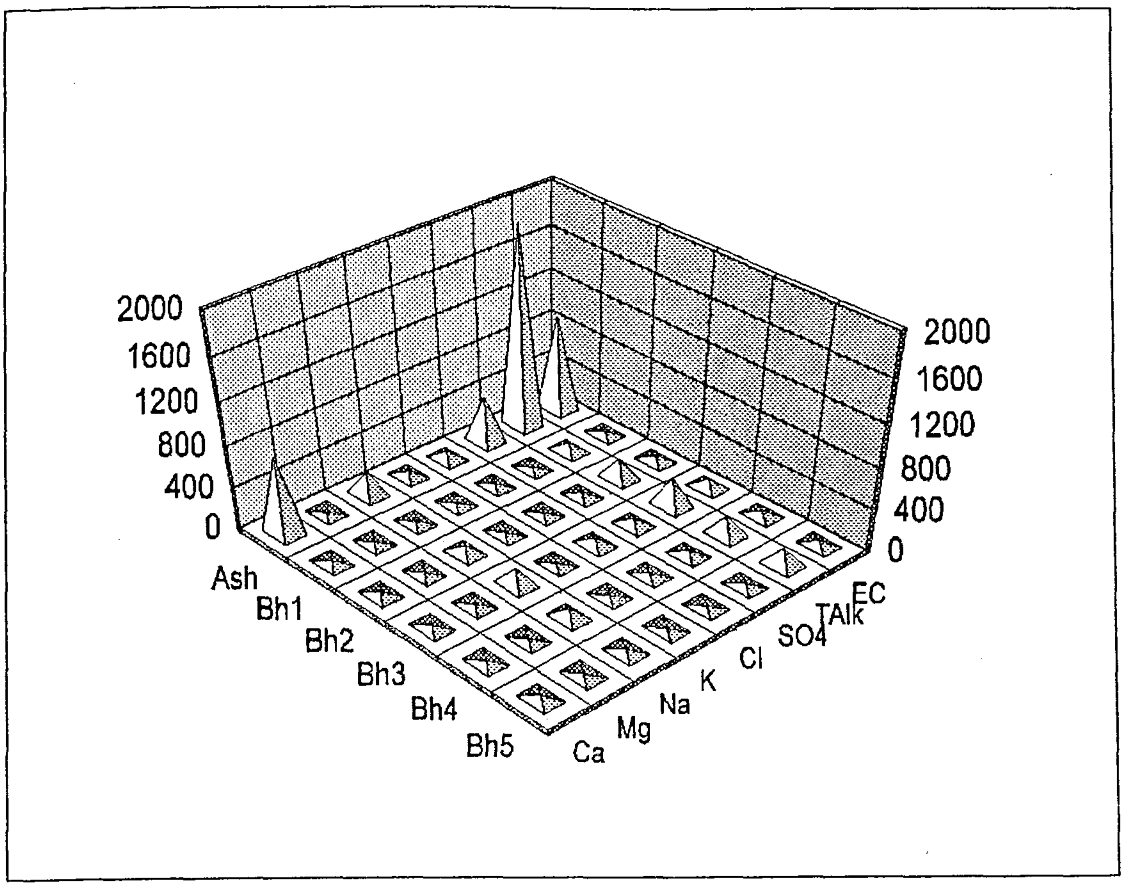


Figure 4.9. Examples of ash water chemistry, in comparison to groundwater chemistries within boreholes in close proximity to the Kriel ash dam, which has been in operation for 15 years at the time of sampling (units in mg/l).

The XRF-analyses indicated that kaolinite is the main clay mineral. The cation exchange capacity (CEC) of kaolinite is the lowest of all common clay minerals, with an average exchange capacity of 100 meq per kg of kaolinite (Fitzpatrick, 1980). The exchange reaction may be written as follows (Fergusson 1982):

$\text{Kaolinite-Na} + H_2O = \text{Kaolinite-H} + OH^- + Na^+$

Iron and aluminium, on the other hand, tend to produce acidic solutions when they are desorped from clay surfaces:

$\text{Kaolinite-Al} + 5H_2O = \text{Kaolinite-}H_3 + Al(OH)_4^- + H_3O^+$

From the above information, it can be demonstrated that the CEC of clays underneath fly ash dams will soon be masked because of the high calcium content within the ash water. Based on the following assumptions, the total duration for the CEC to be depleted, can be calculated:

- Surface area of the ash dam: 263,8 ha; Clay thickness underneath the ash dam: 6 m; % Kaolinite within the clay: 33%; Average hydraulic conductivity of the clay: 10^{-7} cm/sec.; CEC for kaolinite: 100 meq/kg; Calcium concentration in ash water: 1000 mg/l; Gradient is unity.

On the basis of these assumptions, it can be demonstrated that the clay underneath the ash dam has the capacity to adsorb a maximum of 31,6 tons of calcium and that it would take 32,4 days for this amount of calcium to move with the ash water through the clay. CEC is therefore not considered as the major long-term mechanism, which after 15 years at Kriel Power Station, still filters all the calcium from the ash water.

To study the sulphate adsorption from the ash water, six soils have been collected from the area, classified and subjected to laboratory testing. The results of these investigations are presented in Table 4.1 below.

Table 4.1/ Results of sulphate adsorption tests on soils from the Olifants Catchment.

Soil Type	pH (H20)	pH (KCl)	H20 Soluble SO4 mg/l	SO4 adsorption (H2SO4) mg/l	SO4 adsorption (CaSO4) mg/l
Clovelly A	5.5	4.4	19.0	70.8	31.7
Clovelly B	5.4	4.6	53.3	99.6	45.6
Milkwood A	6.1	5.0	9.2	66.5	9.2
Hutton A	6.0	4.9	28.5	80.7	27.8
Hutton B	6.2	5.3	19.1	95.2	45.0
Hutton A	4.7	4.0	35.2	75.0	17.2
Hutton B	4.6	4.0	20.3	90.7	38.5
Hutton A	4.9	4.0	16.9	82.6	25.2
Hutton B	4.9	4.2	33.6	86.9	29.9
Avalon A	5.2	4.2	36.0	79.0	22.6
Avalon B1	4.6	4.1	206.5	43.2	-19.2
Avalon B2	5.7	4.8	56.0	94.7	28.8

The following methodology applies:

- 10 g soil samples (<2 mm) were used.
- 25 ml solution was added for pH-determinations.
- pH was measured after stirring for 5 seconds and again stirring for 55 seconds.
- Soluble sulphate content was determined by adding 30 ml de-ionised water to 10 g soil sample (<2 mm). Stirred well. Stirred again after 12 hours. Filtrate was analysed on IC.
- Adsorbed sulphate was determined by adding 30 ml solution (100 mg/l SO_4 as H_2SO_4 or $CaSO_4$) to 10 g soil (<2 mm). Stirred well. Stirred again after 12 hours. Filtrate was analysed on IC.

To emphasise the contrast between sulphate adsorption for acid and alkaline solutions, Figure 4.10 has been constructed from the information in Table 4.1.

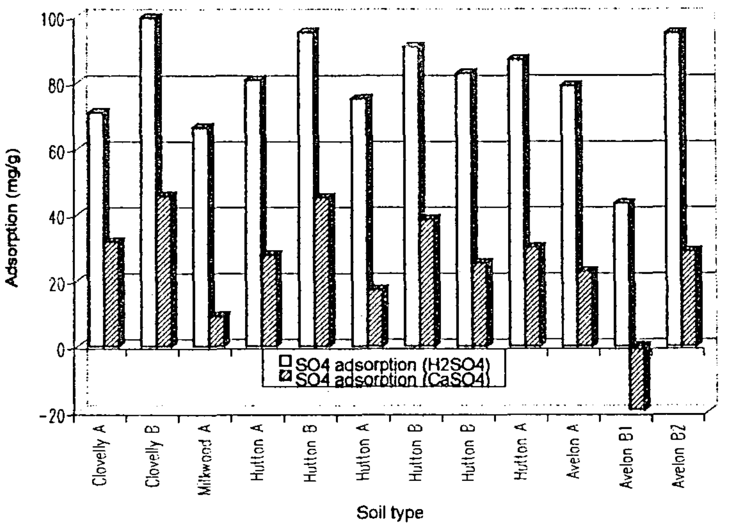


Figure 4.10. Adsorption quantities of sulphate for different soils.

The following conclusions are drawn from the experiment:

- The adsorption capacity for the soils under acid conditions greatly exceeds that of the soils under alkaline conditions.
- Even though the soils in the Olifants Catchment are naturally acid, this acidity will be depleted almost instantaneously as the alkaline ash water is disposed of on the soil, because of the high alkalinity of the ash water.
- The specific sulphate adsorption capacity for a soil will greatly depend on the composition of the water. The above information can therefore not be used to infer quantitatively the capacity of soils at the different power stations. Further work will be necessary to accurately quantify the sulphate adsorption capacity of the soils at each of the power stations, if this information is required. It is felt though, that sufficient information is at hand to conclude that the sulphate absorption capacity of the soils under alkaline conditions is small in comparison to the amount of sulphate within the ash waters. It is concluded that other mechanisms must be researched to account for the removal of sulphate from the ash water, before or as it moves into the aquifer.

Clearly, the potential of the soils to remove calcium and sulphate from the ash water is limited in comparison to that contained within ash waters. Another mechanism must therefore be sought to explain the seemingly endless capacity of the system to remove calcium and sulphate from the ash water. Another approach to explain this phenomenon lies within the chemistry of the ash water itself. Ash water typically contains calcium, sulphate and aluminium in high concentrations. It can be demonstrated through chemical equilibrium modelling that a drop in pH, from 11,25 to 7,0, does not have the potential to precipitate significant amounts of gypsum from the ash water (Figure 4.11). The calcium would rather precipitate as calcium carbonate, as carbon dioxide from the air enters into the ash water. The latter is then probably responsible for the precipitation of a large proportion of the calcium in the ash water. This accounts for the pozzolanic layer that forms on the outside of the ash dams.

Apart from calcium and sulphate, the ash water could contain significant amounts of aluminium. Relationships between aluminium, calcium and sulphate concentrations have been obtained through laboratory solubility experiments and the results have been plotted in Figures 4.12 - 4.14. These demonstrate that up to 4 mg/l aluminium may be contained in ash water at high water to ash ratios but, as the water to ash ratio decreases, such as within the ash dam, both the aluminium and sulphate concentrations drop. Possible forms of aluminium precipitation include gibbsite ($Al(OH)_3$) and ettringite ($6CaO.Al_2O_3.3SO_3.33H_2O$). This precipitation is likely to be within the top layer of the ash dam, where the ash water seeps into the ash. As the ash dam rises with time, ettringite should continue to precipitate with the top portion of the dam and thus eventually occur throughout an ash dam.

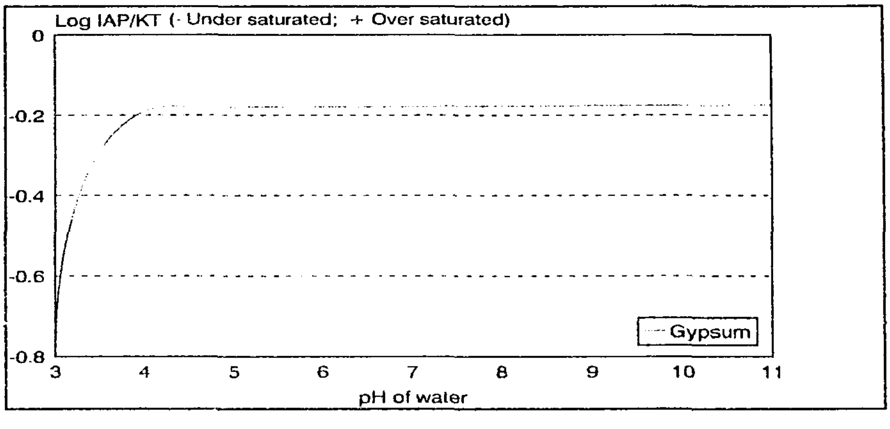


Figure 4.11. Gypsum saturation curve for different pH-levels, Ca of 200 mg/l and SO_4 of 2000 mg/l.

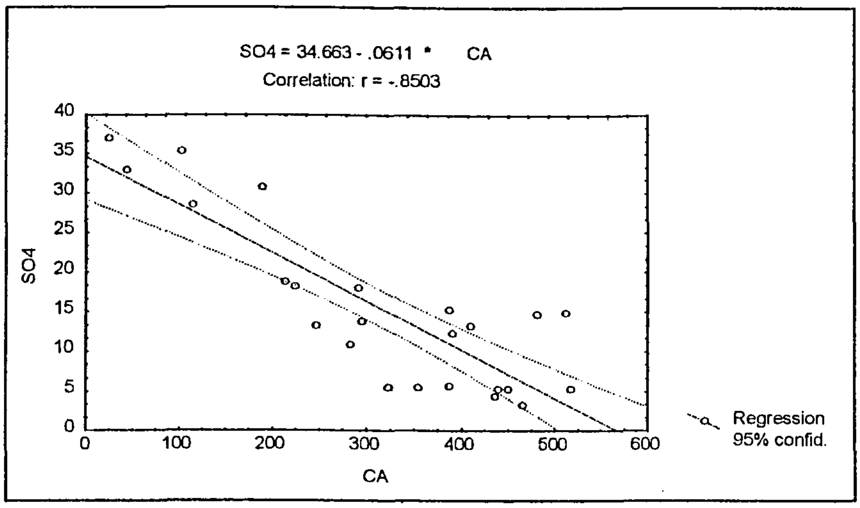


Figure 4.12. Calcium and sulphate concentrations (mg/l) in laboratory experiments, using different fly ash:water ratios.

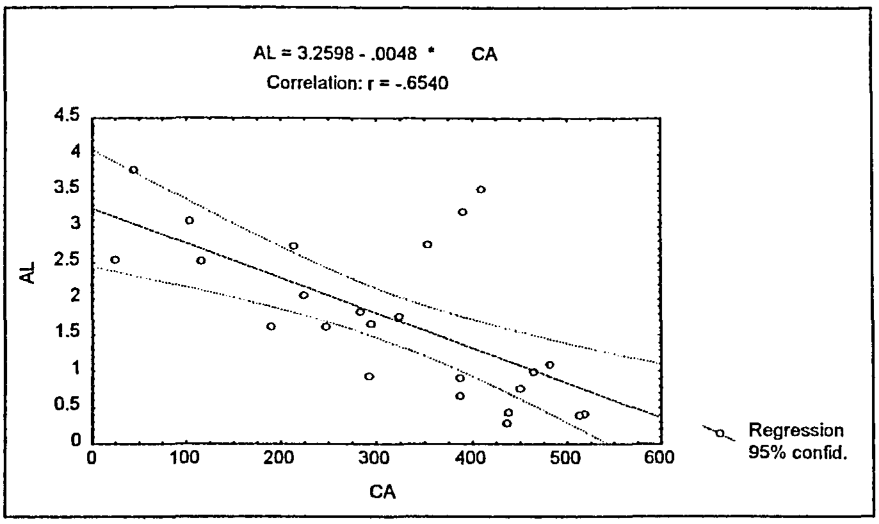


Figure 4.13. Calcium and aluminium concentrations (mg/l) in laboratory experiments, using different fly ash:water ratios.

Table 4.2. A comparison of the ash water chemistry with the chemistry of the water seeping through the bottom portion of the ash dam, at Kriel Power Station.

	Ca	Mg	Na	K	Cl	SO4	TAlk	EC mS/m	pH
	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mS/m	
Ash water	750	0	228	52	74	445	1920	930	11.45
Seepage Water 1	5	0	250	165	32	80	690	250	11.15
Seepage Water 2	3	0	196	99	32	62	464	225	11.05

Eskom is currently conducting extensive laboratory experimentation in the precipitation of sulphate as ettringite, from ash water. In view of the success which has to date been achieved with these experiments (J.B. Conlin, personal communication), it is suggested that sulphate removal through ettringite precipitation is possibly the main mechanism of sulphate precipitation in the current ash dams. The possible precipitation of these constituents within the ash itself is substantiated by the information in Table 4.2, for Kriel Power Station. The conclusion is therefore drawn that sulphate addition to the ash water is an essential component of ettringite precipitation within the ash dams.

At the older power stations, a tendency exists for sodium to migrate from the ash dams into the adjacent areas. The sodium is mostly derived from sodium hydroxide that is being used to regenerate resin in the demineralisation plant. At all power stations, the demineralisation effluent is disposed of in the ash dams.

Sodium is almost endlessly soluble in water, with the result that a build-up of sodium in semi-closed systems, such as the ash dams, is inevitable. In the Olifants Catchment, it is presently only Wilge and Komati Power Stations where noticeable sodium concentration has occurred. As the ash water seeps away from the ash dams, capillary action draws water to the surface and white crystals of sodium sulphate crystallise on surface. Upon rainfall events, the sodium sulphate is again dissolved and mobilised with overland run-off. In all, sodium pollution from ash dams is probably the only real concern for groundwater and surface water pollution at Eskom's wet ash dams. Komati Power Station is currently in the process of installing drains to prevent the rising of water to surface and crystallisation of sodium sulphate in the area between the ash dam and the stream.

During this investigation, the Komati Power Station has been mothballed. The opportunity therefore existed to observe the impact which ashing has on the groundwater regime, under two scenarios. Two borehole profiles, indicating the geology, borehole construction details, depth of water intersections, natural gamma radiation from the rock and electrical conductivity profiles of the water in the boreholes, are given in Figure 4.14. Borehole B2 is located within 50 m and B6 within 100 m below the ash dams. It is clear from the electrical conductivity profiles that a significant improvement in the groundwater quality has resulted from 1989 till 1993. Ash disposal has ceased during 1991.

The decrease in electrical conductivity in borehole 2 is attributed to a drop in the sulphate, sodium, magnesium and calcium levels. In borehole 6, the drop in sulphate concentration is even more pronounced, with a corresponding drop in all the major cations. A comparison of the actual chemistries for these waters is presented in Figure 4.15.

The conclusion is drawn that the potential for groundwater pollution from wet ashing operations decreases dramatically when ash disposal is stopped. The reasons for this decrease are:

- No further addition of salts, particularly from the demineralisation plant.

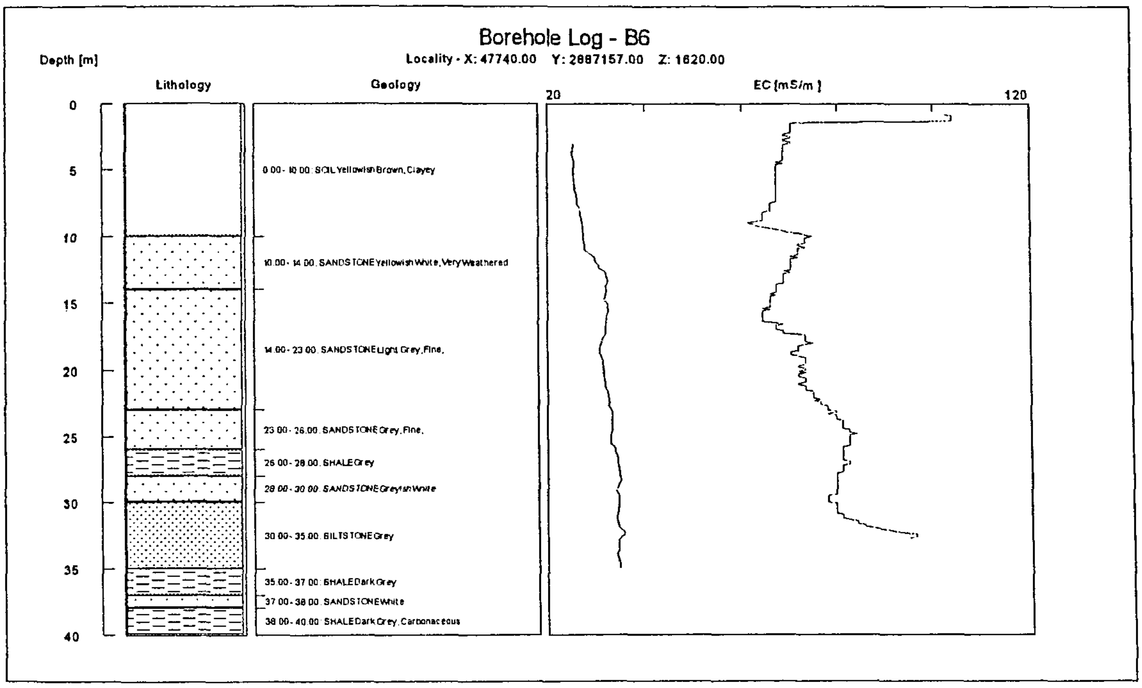


Figure 4.14.EC in boreholes during (right) and after (left) ash disposal at Komati Power Station

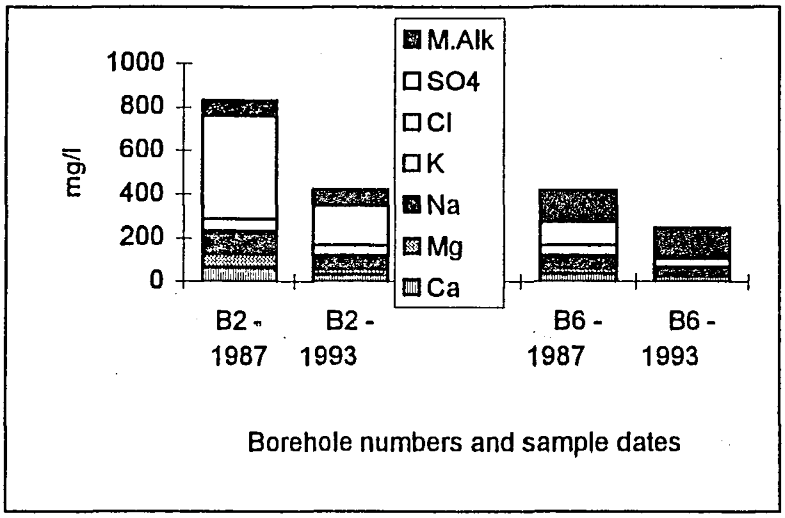


Figure 4.15. A comparison of water qualities in two monitoring boreholes at Komati Power Station, during and after ashing.

- A pozzolanic layer forms on top of the ash dams, thus minimising infiltration from rainfall.
- The unstable chemistry within the ash dam, precipitating elements within the dam itself, as suggested above.

4.3.2.1.2 In-pit ash disposal

Investigations into the possible effect which in-pit ash disposal may have on mine water and the environment have been conducted overseas as early as 1976 (Dougherty and Holzen). The use of fly ash for surface treatment of acidic opencast spoils was studied and found to be feasible and suitable for producing a soil cover capable of sustaining vegetation. In these early years, several studies have also been performed with respect to in-pit ash disposal in Poland (Libicki 1978, 1983 and 1985). His early studies mostly described results of monitoring in-pit ash disposal, while the publication in 1985 discusses the consequences of placing the ash at different horizons within the pit. He also discusses the planning of monitoring boreholes and control performances. These papers have formed a valuable foundation for the present investigation.

In-pit ash disposal is presently being done at only two power stations, namely Kriel and Matla. Kriel Power Station disposes of fly ash into some of the ramps and voids of Pit 1, previously mined by Kriel Colliery. Matla Power Station disposes of fly ash into and onto Pit 3N, also previously mined by Kriel Colliery. In both instances, have the power stations taken over the ownership of the pits and thus the responsibility of water management and final rehabilitation of the pits.

The disposal of fly ash into the opencast coal pits has been preceded by extensive evaluations of possible environmental impacts. In terms of water qualities and quantities, the following investigations have been performed:

- Compatibility of ash water with spoil water.
- Compatibility of ash chemistry with the long-term pit chemistry.
- Water and salt balances for present and future systems.

Compatibility of ash water with spoil water

The spoil water chemistry of the two pits has been studied since 1983. Figure 4.16 demonstrates the concentration of sulphate in the spoil water, before ash disposal commenced. The level of approximately 2 500 mg/ℓ sulphate is considered the maximum sulphate levels that would have been reached in the pit water, because of a dilution effect from rainfall, which is being recharged to the pit. More comprehensive pit water and ash water chemistries are presented in Table 4.3.

Because of the neutral pH of the spoil water, heavy metals are not present in significant quantities and are therefore not listed.

The above chemistries have been entered into a chemical equilibrium model (PHREEQE) and composite chemistries for various ratios of ash and ramp water have been calculated. The saturation values for these mixtures, with respect to gypsum, are presented in Table 4.4. From this, it was concluded that gypsum precipitation would not occur, irrespective of the ratios in which these waters are mixed. The calculation of the resultant chemistry is therefore mainly a mixing cell calculation and not of the chemical equilibrium type. The conclusion was drawn that no detrimental effects would result by mixing ash and spoil water.

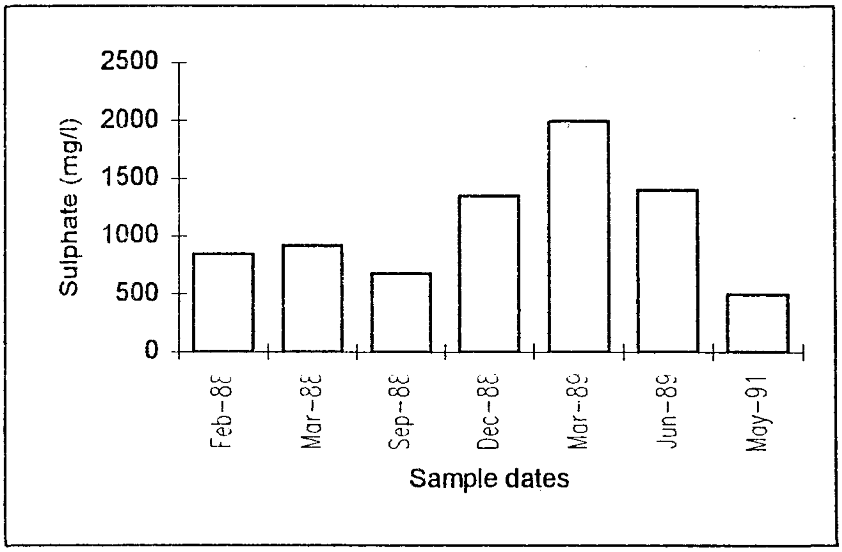


Figure 4.16 Sulphate levels in pit water samples before and after in-pit ashing commenced in 1990.

Table 4.3. Pit and ash water chemistry before in-pit fly ash disposal.

Sample Origin	pH	EC	Ca	Mg	Na	K	Cl	SO4	TAlk
		mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
Pit Water	7.5	337	452	233	214	17	32	1990	580
Ash Water	11.5	932	750	0	228	52	74	445	1920

Table 4.4. Calculated chemistries for different ratios of pit and ash waters, under non-equilibrium conditions with air.

% PIT WATER	pH	GYPSUM SATURATION	Ca mg/l	SO4 mg/l
100	7.5	-0.141	454	1997
90	8.7	-0.157	483	1842
80	9.1	-0.184	513	1687
70	9.4	-0.217	543	1532
60	9.6	-0.256	573	1377
50	9.8	-0.302	603	1222
40	10.1	-0.357	633	1067
30	10.4	-0.423	663	912
20	10.7	-0.502	693	757
10	11.1	-0.594	723	602
0	11.5	-0.708	753	447

Compatibility of ash chemistry with the long-term pit chemistry

The long-term chemistry of the in-pit water will be a function of two reactions, namely oxidation of pyrite that occurs within the spoil above the pit water level, and the neutralising potential of the spoil combined with that of the fly ash.

The acid-base potential of the spoil has been studied in detail. This information indicates that the spoil in general has a net neutralising potential (NNP) of 58 kg $CaCO_3$ per ton of rock. This amounts to 30 times the acid potential within the spoil. Twardowska (1990) has suggested through studies in Poland, that if the base potential exceeds the acid potential by a factor of 2,5, then the system is unlikely to become acid. Fergusson and Erickson (1988) found that mines produce alkaline water if the NNP is >30 kg $CaCO_3$/t of spoil. The threshold value of 30 kg $CaCO_3$/t of spoil is confirmed by several other researchers (Cravotta *et al.*, 1990). The fly ash from Kriel Power Station has a base potential of 103 kg $CaCO_3$ per ton of ash, at pH 7.

The following conclusions are derived from this information:

- The spoil in the pit has an inherent resistance to acidification and it is unlikely that the pit water will turn acid, because of pyrite oxidation.
- The ash has an even higher base potential than the spoil, thus further reducing the risk of in-pit water acidification.
- It is therefore safe to dispose of power station fly ash into Pit 1, on condition that the necessary precautions are taken that ash water does not decant from the pit into public streams.

Through similar experimentation and calculation, it has been demonstrated that Pit 3N, receiving ash with a base potential of 154 kg $CaCO_3$/t from the Matla Power Station, will also not acidify with time.

Water and salt balances for present and future systems

The establishment of an accurate water balance is an essential component of long-term chemistries at in-pit ash disposal sites. Such a water balance for Pit 1 is presented in Table 4.5.

The following conclusions can be drawn and difficulties that may arise during in-pit ashing, are pointed out:

- During ashing in the ramp, significant volumes of water will be replaced by the ash. Provision must be made to handle this water and use or dispose of it elsewhere.

Table 4.5. Current and projected water and salt balances for Pit 1.

Water Source	Annual Factor (m)	Current Areas (ha)	Current influx (Ml/d)	Areas after ashing (ha)
Rainfall recharge on open water	0.70	20	0.38	4
Rainfall recharge on ramp slopes	0.35	32	0.03	0
Rainfall recharge on rehabilitation	0.06	220	0.36	320
Rainfall recharge on spoil heaps	0.17	48	0.22	0
Internal drainage areas in pit	0.08	288	0.63	30
Run-off into pit from surroundings	0.06	20	0.03	20
Seepage from new ash dam			0.45	
Seepage from old ash dam			0.18	
Groundwater seepage into pit			0.10	

- By filling in a ramp with ash, pit run-off that previously ended up in the pit can be diverted away from the pit. This will reduce the amount of water ending up in the pit. Final rehabilitation of the pit is therefore easier.
- It will be impossible to eliminate all recharge to the pit, because of natural groundwater throughflow and recharge through the rehabilitated spoils and ash. Provision must therefore be made to either accept the extra water into public systems or contain the water within the pit by creating an evaporation facility within the pit itself. An evaporation scheme may be either through direct evaporation from standing water or through indirect evaporation by planting trees in areas with shallow spoil water levels. Evaporation will lead to concentration of salts within the system, which is undesirable in the long run.

Not all spoils in other opencast collieries have the same high net base potential, as in the case of Kriel Colliery. Tests have shown that a large percentage of the collieries has negative net neutralising potentials, i.e. their acid-generating potential exceeds their base potential. At these collieries, it would be extremely risky to dispose of power station fly ash into the pits, because the total system may become acid with time, and mobilise heavy metals. Detailed investigations will be necessary in the event if fly ash disposal is considered within a potentially acid environment.

Extensive leaching tests have been performed as part of this investigation. Ash samples have been obtained from all eight power stations within the Olifants Catchment and subjected to leaching tests at pH-levels ranging from 8 down to 3. The results of some of these tests are presented in Appendix D. Information on the potential for the ash samples to counteract acidity, at different pH-levels, is presented in Table 4.6 and Figure 4.17.

These potentials of the ash samples have been determined by adding excess acid and titrating upwards to the desired pH. Upon reaching the desired pH, the sample was covered and allowed to stand for 24 hours. Thereafter, the pH was again measured and corrected by additional titration. This procedure is repeated until stable pH-values at the different levels were obtained.

In Table 4.6 and Figure 4.17, the results have been arranged for power stations from east to west in the Olifants Catchment. The conclusion is drawn that ash from power stations in the east generally has lower base potential than those in the central and western parts of the coal-field. This is, however, a gross generalisation and should not be taken as representative for the total coalfield. Kriel and Matla Collieries are in any case part of the Highveld Coalfield, while the other power stations obtain their coal from the Witbank Coalfield.

Table 4.6. Potentials (kg $CaCO_3$/t) for power station fly ash samples to counteract acidity at different pH-levels.

pH	Arnot	Hendrina North	Hendrina South	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3	104	98	114	189	137	175	221	174	214
4	91	88	100	175	117	155	179		182
5	65	65	75	135	85	114	167	91	146
6	62	61	68	131	78	109	160	87	144
7	58	59	60	130	75	104	154	79	143
8	54	53	55	124	61	93	139	69	136

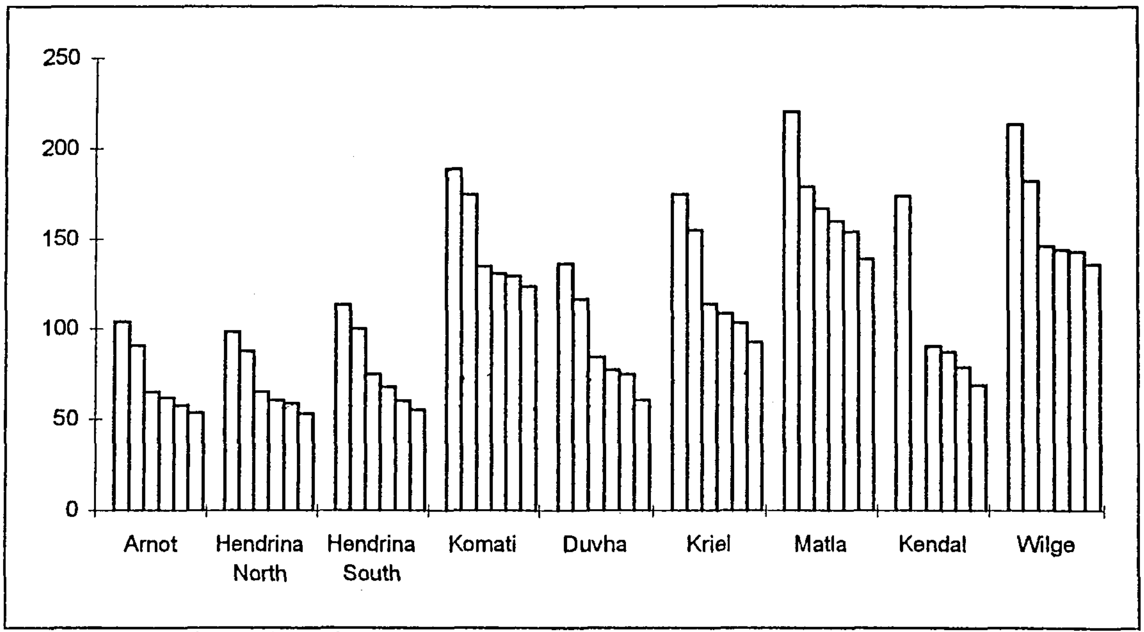


Figure 4.17. Potentials of power station fly ash (kg $CaCO_3$/t of ash) to counteract acidity at pH-levels of 3 to 8.

Appendix D lists and graphs concentrations of different elements that can be leached from the fly ash at different pH-levels. While this kind of presentation is monotonous and repetitive, some of the more interesting aspects may be highlighted in the form of box and whisker plots. In these diagrams (Figures 4.18 - 4.22), the means for all the ash waters at different pH-levels, standard errors of the means at a 95% confidence level and standard deviations have been plotted for a range of elements that may be leached from the ash. The interesting issues are the increase of concentrations at lower pH-levels and also in the range of values. It is clear that aluminium is the main heavy metal which could be mobilised from the ash.

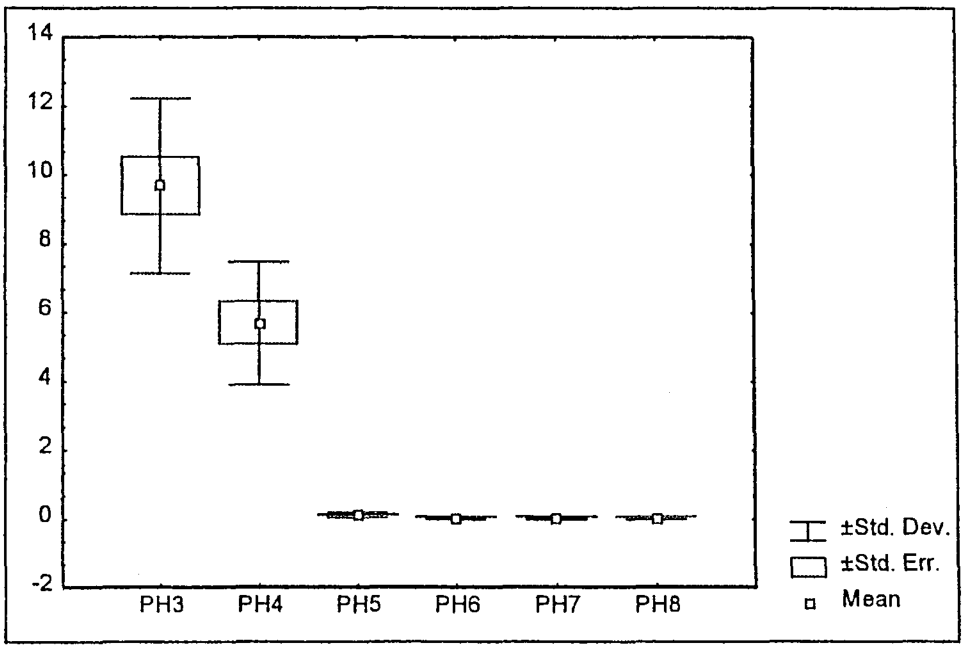


Figure 4.18. Box and whisker plot of aluminium concentrations (kg/t of ash) in leachates from ash samples in the Olifants Catchment, at different pH-levels.

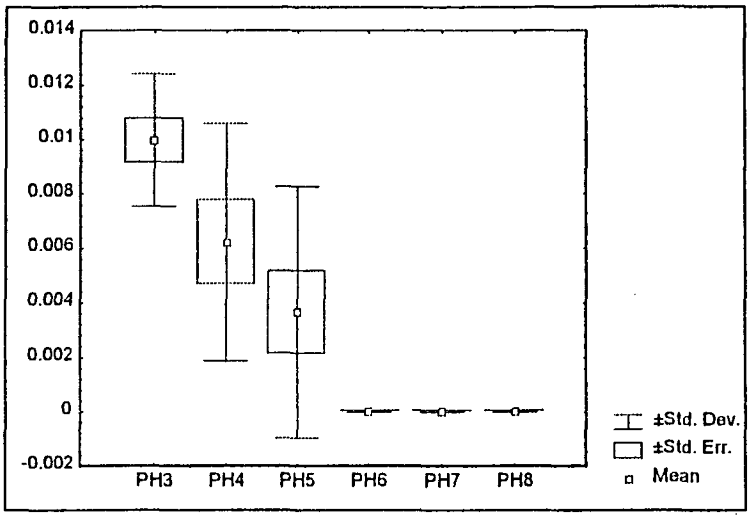


Figure 4.19. Box and whisker plot of copper concentrations (kg/t of ash) in leachates from ash samples in the Olifants Catchment, at different pH-levels.

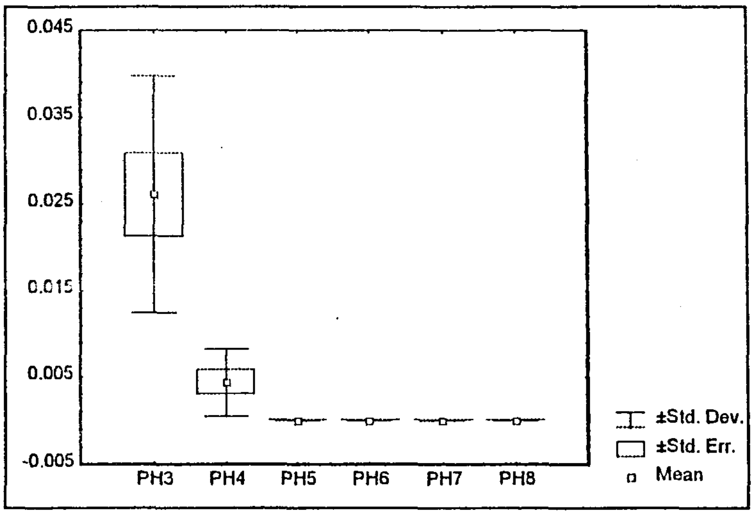


Figure 4.20. Box and whisker plot of chromium concentrations (kg/t of ash) in leachates from ash samples in the Olifants Catchment, at different pH-levels.

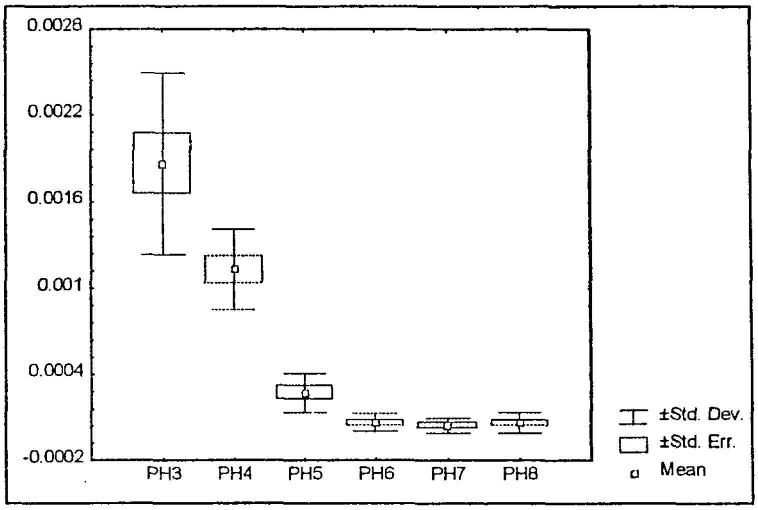


Figure 4.21. Box and whisker plot of beryllium concentrations (kg/t of ash) in leachates from ash samples in the Olifants Catchment, at different pH-levels.

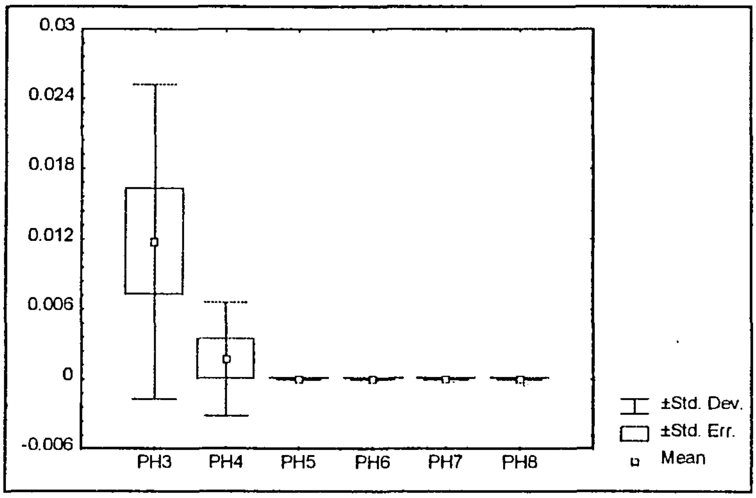


Figure 4.22. Box and whisker plot of arsenic concentrations (kg/t of ash) in leachates from ash samples in the Olifants Catchment, at different pH-levels.

Another interesting comparison is the total quantities of elements that are released from the ash during acidification, per power station. Figures 4.23 to 4.31 display these results.

The following conclusions are drawn:

- Matla and Kriel Power Stations have elevated boron concentrations in their ash. Water samples from monitoring boreholes at these power stations do not have similarly elevated boron values. Boron is usually only mobilised under acid environments.
- A very close correlation exists between the calcium and magnesium contents of the ash and the base potential for the fly ash samples. The conclusion is drawn from the almost 100% correlation, that calcium and magnesium account for almost all the base potential at neutral pH-levels. Base potential determinations in fly ash samples involve lengthy titrations. Calcium and magnesium determinations can, on the other hand, easily be done. The latter information may therefore be used as a first approximation of the base potential of the ash, at neutral pH-level.
- The other elements shown in the remainder of the box and whisker plots do not exhibit specific trends that are worthwhile pointing out. It suffices to state that a significant number of trace elements exist which may be leached from the fly ash under severe acid conditions. It is unlikely that any of the existing ash dams will ever be subjected to such severe acid conditions. The ranges for the elements should therefore be taken as an indication of the maximum availability of these elements from the ash and not as a potential environmental threat.

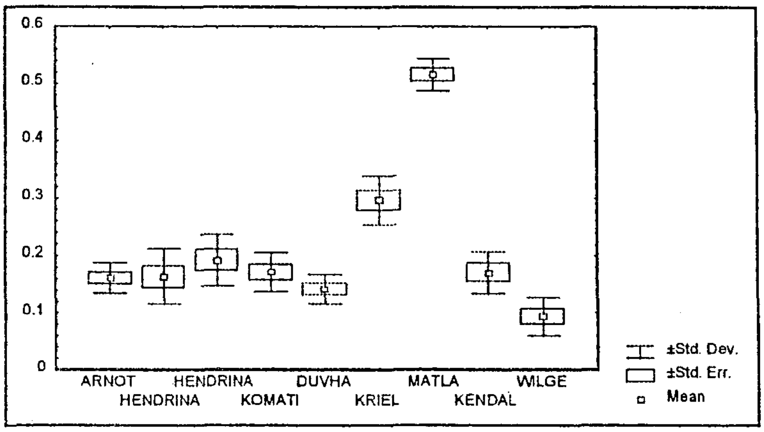


Figure 4.23. Box and whisker plot of boron concentrations (kg/t of ash) in leachates from ash samples from power stations in the Olifants Catchment.

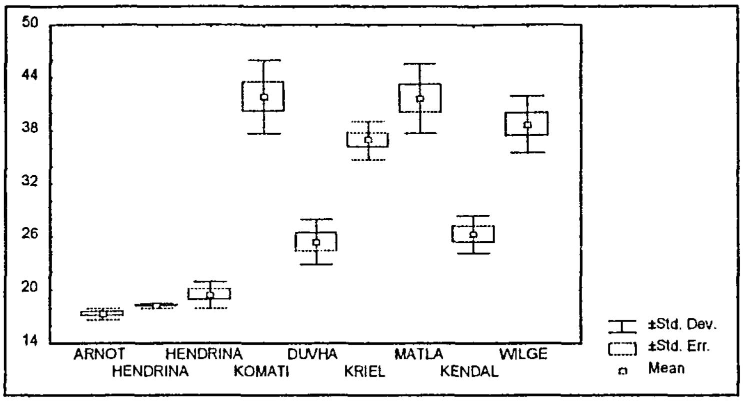


Figure 4.24. Box and whisker plot of calcium concentrations (kg/t of ash) in leachates from ash samples from power stations in the Olifants Catchment.

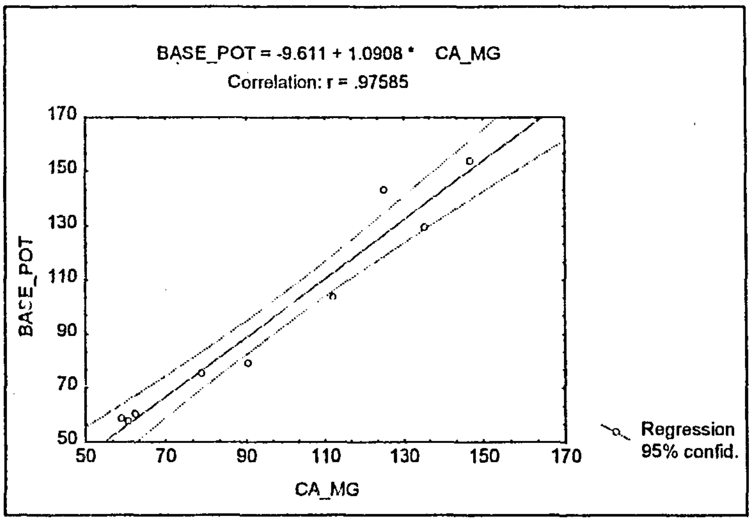


Figure 4.25. Plot of calcium plus magnesium (as kg/t $CaCO_3$) from analytical results versus the base potential (as kg/t $CaCO_3$) of the ash samples at pH 7.

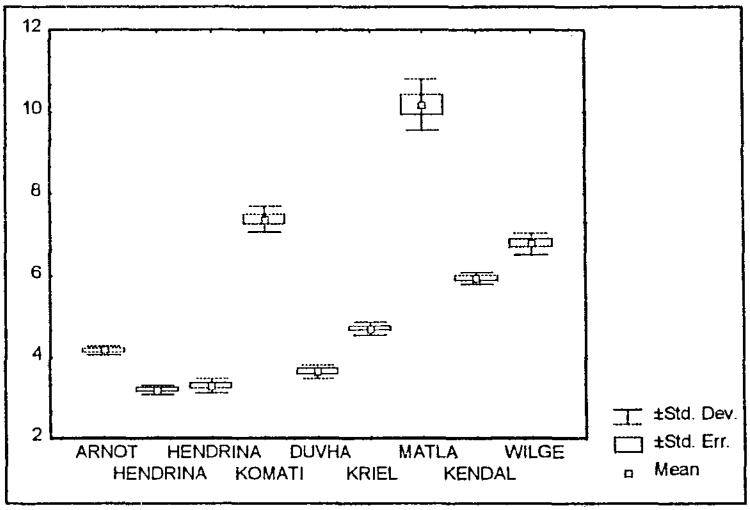


Figure 4.26. Box and whisker plot of magnesium concentrations (kg/t of ash) in leachates from ash samples from power stations in the Olifants Catchment.

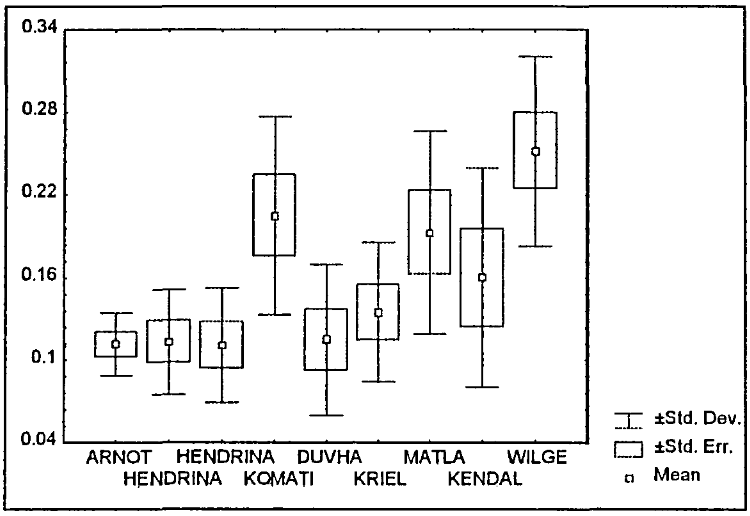


Figure 4.27. Box and whisker plot of manganese concentrations (kg/t of ash) in leachates from ash samples from power stations in the Olifants Catchment.

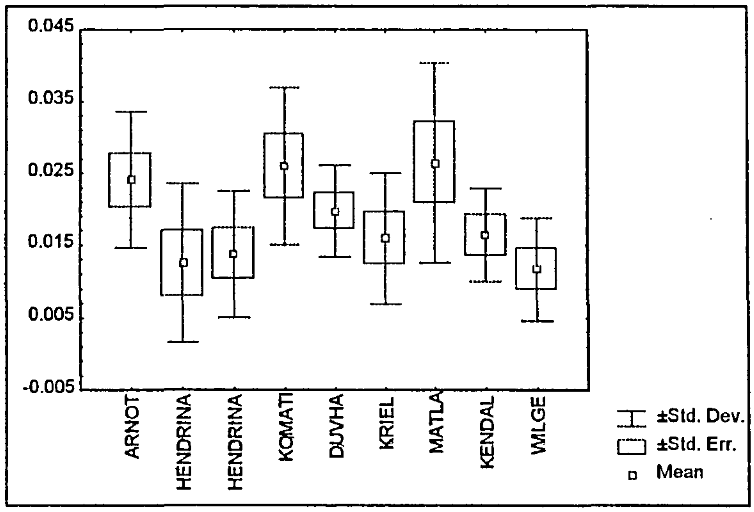


Figure 4.28. Box and whisker plot of zinc concentrations in leachates (kg/t of ash) from ash samples from power stations in the Olifants Catchment.

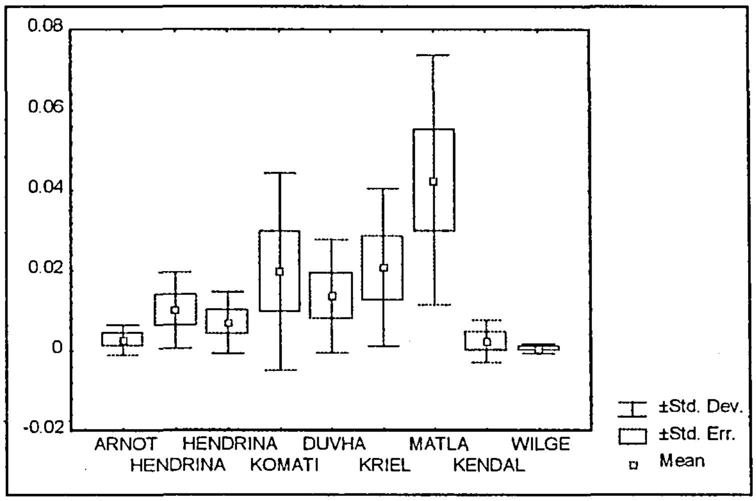


Figure 4.29. Box and whisker plot of vanadium concentrations (kg/t of ash) in leachates from ash samples from power stations in the Olifants Catchment.

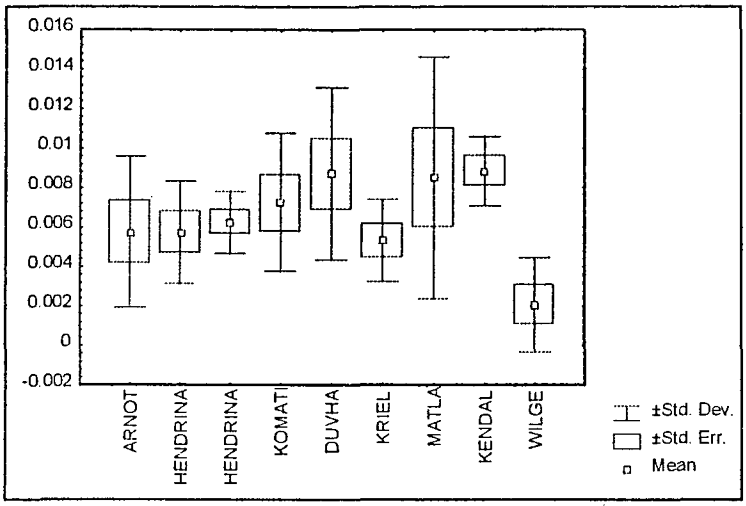


Figure 4.30. Box and whisker plot of molybdenum concentrations (kg/t of ash) in leachates from ash samples from power stations in the Olifants Catchment.

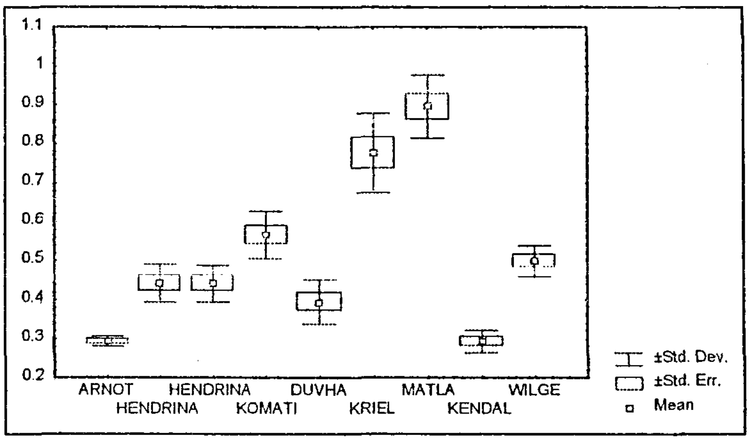


Figure 4.31. Box and whisker plot of strontium concentrations (kg/t of ash) in leachates from ash samples from power stations in the Olifants Catchment.

4.3.2.2 Dry ash disposal

Dry ash disposal is currently only being done at Kendal Power Station within the Olifants Catchment. Other power stations in South Africa where dry ash disposal is being done, are Lethabo, Tutuka and Majuba.

The ash is wetted by adding about 15% water within the power station. This amount is considerably lower than the field capacity of the ash, which is in the order of 30% moisture, depending on the bulk density of the ash. The moist ash is then transported by conveyor belt to the ash dump. Here, it is dumped off the end of the conveyor belt in a single or double stacking operation. No mechanical compaction, other than traffic moving over the ash dump, is done.

The ash is again wetted by means of a pivot irrigation system as it is dumped off the conveyor belt. This reduces the problem of dust. Under severe windy conditions, dust does, however, still blow from the exposed ash.

A soil cover is placed on the ash as soon as possible. This cover is usually in the order of 10 - 30 cm thick. The soil cover prevents ash from blowing off the dump and serves as a growth medium for grass. Vegetation establishes well under these conditions.

Most ash dumps have continuous surface gradients, which allow rainfall to run off the dumps. Small areas do, however, exist where pounding occurs after heavy rainfall events.

An investigation has been carried out into the penetration of rainwater into the ash. These experiments consisted of two approaches, namely:

- Measurement of moisture variation in the top 2 m of the ash by means of neutron probing.
- Hydraulic conductivity measurement of the ash by means of ring infiltrometer testing.

Neutron moisture probing

Two typical neutron moisture profiles are presented in Figure 4.32. These profiles were obtained from monitoring holes on a dry ash dump. The profiles were measured during a variety of conditions, ranging from dry to very wet (75 mm rainfall in two days).

The following field conditions apply:

- The ash is covered with 30 cm of sandy topsoil.
- Deeper down, the ash is homogeneous. Variations in the moisture profiles are therefore attributable to variations in moisture content.
- Hole 1 is located on top of the ash dump. Grass had been established at this position with a maximum root zone of 30 cm.
- Hole 2 is located on the side of the dump, about one-third from the top. It receives recharge from rainfall and surface run-off, as well as downslope seepage through the sandy soil. Grass has been established on the slope.
- The ash has been disposed of six months before experimentation commenced.
- Experimentation continued for one season.

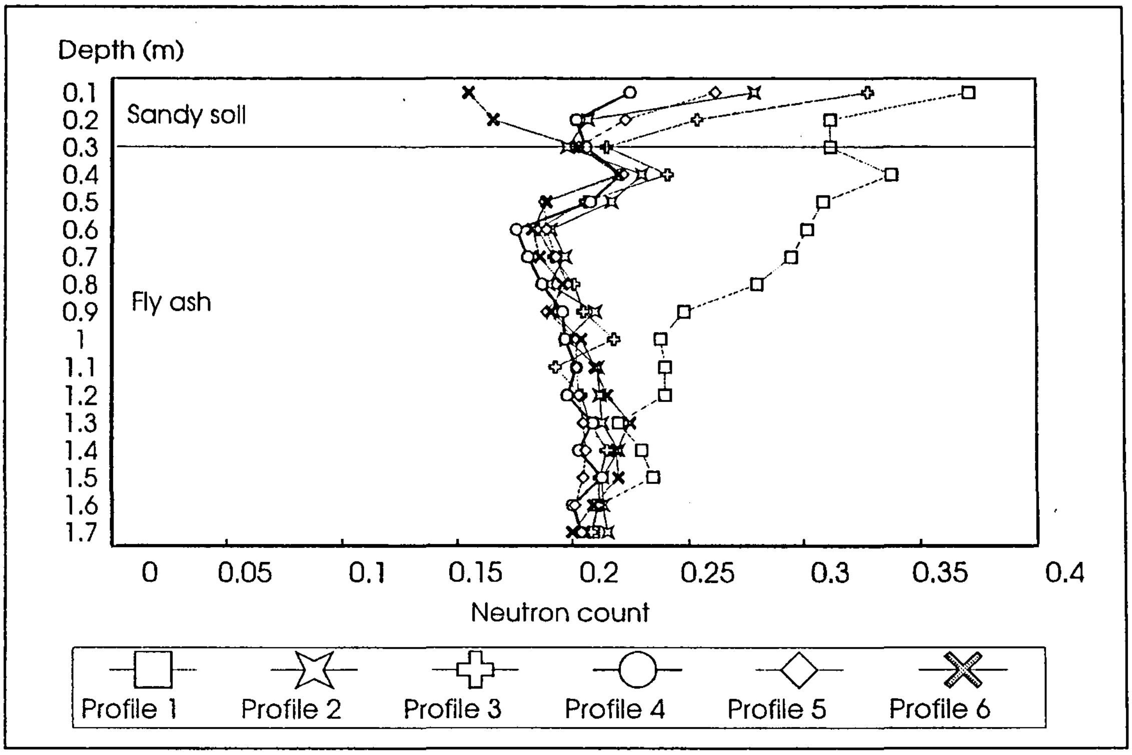


Figure 4.32(a). Examples of neutron soil moisture profiles in Monitoring Borehole 1 on a dry ash dump.

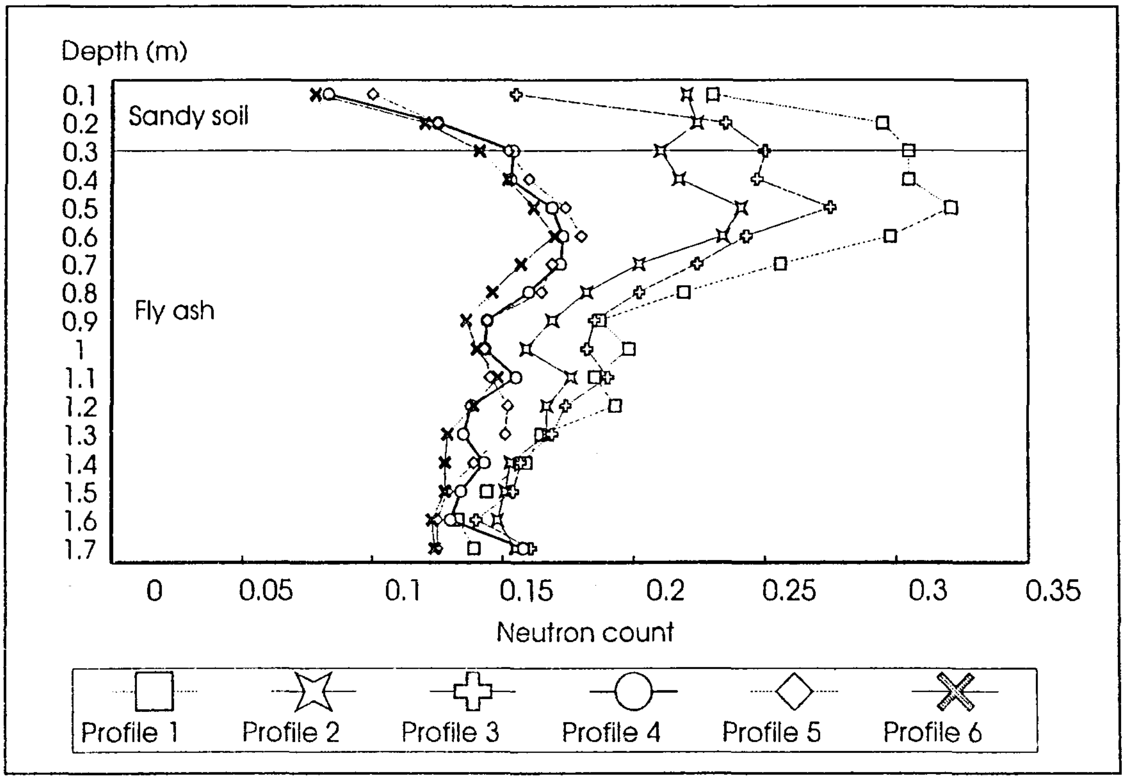


Figure 4.32(b). Examples of neutron soil moisture profiles in Monitoring Borehole 2 on a dry ash dump.

The following conclusions are drawn:

- Normal rainfall events of up to 25 mm, followed by a dry period, contribute only to variation in moisture content to a depth of 30 cm at Hole 1. The profile in Hole 1 that shows significantly higher moisture content was obtained after a rainfall event of 75 mm. In this instance, penetration of the rainwater, to a depth of 1,4 m, can be seen where this profile joins with the other profiles.
- At Hole 2, moisture penetration from normal rainfall events is greater because of the downslope seepage on the contact between the sandy soil and the ash. The effect of the high rainfall event can again be seen, but at least two other events also show significant recharge. During these two events, rainfall of 30 mm and 25 mm was respectively recorded. Also significant, is that a small percentage of the rainwater penetrated to depths beyond 2 m, which was the limit of measurement. The low moisture content close to surface is attributed to the evaporation from the very dense grass cover in the area. This suggests that the sandy soil is inadequate as a water reservoir for the grass. The water holding properties of the sandy soil may be improved by mixing it with the ash.
- It is concluded that negligible amounts of rainfall will penetrate into a well-rehabilitated dry ash dump. The pollution potential of dry ash dumps, if located above the groundwater level, is therefore very small.

Infiltrometer tests

Ring infiltrometer tests were done on the ash dump to determine the rate of water seepage under saturated conditions. The results of this experiment are presented in Table 4.7.

Table 4.7. Results of ring infiltrometer tests in relation to the ash dump.

Position and description	*Equilibrium infiltration rate (l/h/m2)*
At Hole 1, on 10 cm of soil, on top of ash	7.9
At Hole 1, below 10 cm of sand, directly on the ash	1.9
On top of newly dumped ash	25.5

The following conclusions are drawn:

- The sandy soil on top of the ash is significantly more permeable than the ash. This will lead to water decanting along the sandy soil off the top of the ash dump. This has been confirmed by the neutron measurements at Hole 2.
- The freshly disposed of and uncompacted ash is significantly more permeable than the ash that underlies the soil in rehabilitated areas. This is partially the result of compaction of the ash during spreading of the soil. However, a far greater impact is that of the pozzolanic action within the top portion of the ash. At dry ash dams, carbon dioxide moves into the ash with the rain water. The carbon dioxide reacts with the calcium oxide in the ash and lime ($CaCO_3$) precipitates. It has been established during this investigation that the pozzolanic layer at a dry ash dump is typically up to 500 mm thick. As the crystallisation of lime continues, the top portion of the ash becomes less and less permeable.

A stage should therefore be reached where the hydraulic conductivity of the pozzolanic layer has been reduced to such an extent, that rainwater can no longer effectively penetrate into the ash. The pozzolanic potential of the ash differs between power stations. From the base potential and other chemical analyses, it can be stated that power stations such as Arnot and Hendrina have the lowest pozzolanic potentials, while Kendal has an intermediate pozzolanic potential with respect to the other power stations in the Olifants Catchment. It is unclear at this stage what base potential would be sufficient to create a pozzolanic layer that is totally impermeable to water. It is fully realised that other factors also play a role and the suggestion is therefore made that Eskom should monitor the establishment of the pozzolanic layer at Kendal and other dry ash disposal power stations. This information will be valuable in future decision-making processes, where fly ash may be used to cover other wastes in order to minimise pollution from these sites.

The ability of pozzolanic ash to successfully act as a sealant, has also been demonstrated in many other research projects, such as that by Edil, Sandstrom and Berthouex (1992) in Wisconsin, in which they state that ash permeabilities are reduced to less than 10^7 with time.

4.3.3 COAL STOCKYARDS

Coal stockyards, by virtue of the fact that the coal contains pyrite, have a far greater pollution potential than fly ash. The potential for coal stockyards to generate acid is, however, greatly reduced through compaction of the coal. The coal is, in the first instance, being compacted to prevent spontaneous combustion. This reduces throughflow of oxygen. A request was made to Eskom to install seepage collectors at their coal stockyards, but they did not consider this feasible, in view of the fact that all the strategic coal stockyards have already been established. The installation of seepage collectors would have involved digging up of portions of the coal stockyards, to install seepage collectors underneath the coal. For the same reason, gas samplers could not be installed.

Monitoring at coal stockyards was therefore limited to boreholes in the areas adjacent to the coal stockyards.

At the newer power stations, the coal stockyards are well-constructed, separate entities that can be monitored without interference from other wastes. At older power stations, such as Komati, the coal stockyard lies in the vicinity of a dirty water dam and coal loading operations at the mine. Examples of monitoring results for some of the power stations are presented in Table 4.8.

Table 4.8. Groundwater quality in the vicinity of coal stockyards at the newer power stations in the Olifants Catchment.

pH	EC	Ca	Mg	Na	K	Cl	SO4	M.Alk	NO3-N	F
	mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
6.77	8	7	4	7	3	6	3	39	0	0.4
6.64	8	6	4	6	3	4	3	36	0	0.4
7.26	17	15	7	12	3	4	3	83	0	0.2
7.44	18	16	7	12	3	3	4	90	0	0.2
6.43	8	4	3	8	4	5	3	32	0	0.2

The following conclusions are drawn:

- The monitoring results indicate that coal stockpiling is not an environmental hazard at the newer power stations.
- Sulphate concentrations are still low in monitoring boreholes even after more than 15 years of coal stockpiling.
- Even though the coal stockpiles have some hydraulic conductivity at the top, as determined by ring infiltrometer tests, the net ingress of water into the coal is apparently minimal. It is expected that the evaporation potential of water from within the coal is significant.
- In view of the present general lack of groundwater pollution and, to a certain degree, also a lack of understanding the hydraulic and chemical variables that operate within a coal stockpile environment, it is suggested that Eskom initiates a research programme to quantify these unknowns. Field experiments would include leachate collectors and gas samplers.

4.3.4 DIRTY WATER DAMS

Dirty water dams or pans exist at all the power stations. At three of the power stations, pans are being used to hold dirty water that is recycled. Monitoring holes have been drilled in the downstream direction from the pans and the results are presented in Table 4.9.

Table 4.9. Typical groundwater qualities in monitoring boreholes adjacent to ash water pans.

pH	EC	Ca	Mg	Na	K	Cl	SO4	M.Alk	F
	mS/m	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
6.87	25	18	8	14	3	2	2	107	0

The conclusion is drawn that pollution from pans with internal drainage systems is minimal and that it does not impact regionally on water quality. In these systems, care should, however, be exercised that groundwater gradients should always be towards the pans.

At some power stations, dirty water dams have been constructed in nearby streambeds. A typical arrangement is shown in Figure 4.33. The results from monitoring holes are presented in Table 4.10.

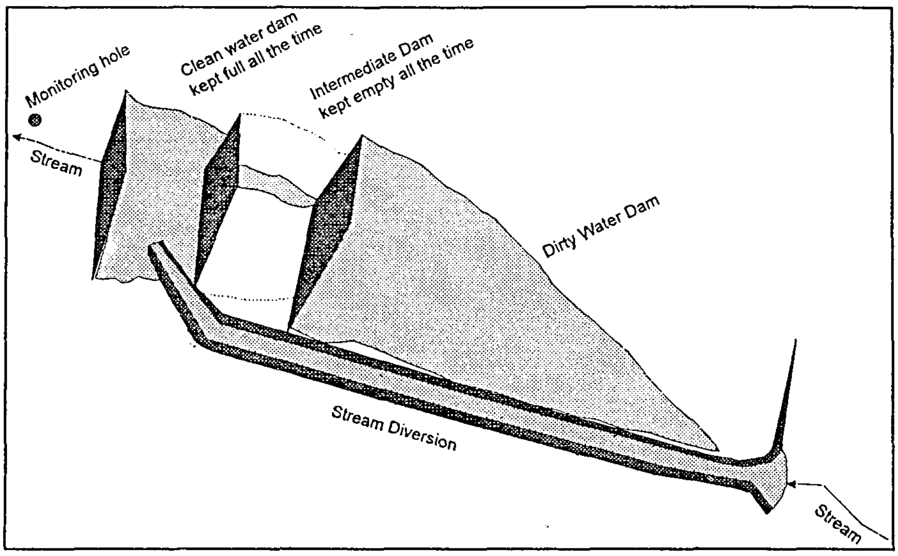


Figure 4.33. Arrangement of dirty and clean water dams at Kendal Power Station.

Table 4.10. Observed electrical conductivity values in a monitoring borehole below the dirty water dam arrangement at Kendal Power Station.

Date	Electrical Conductivity
	(mS/m)
Jun-87	26
May-88	18
Oct-88	20
Aug-90	20
Nov-90	20
Jan-91	18
Apr-91	21
May-91	21
Mar-92	22
Jun-92	21
Aug-92	22

This arrangement works well. Water that seeps from the dirty water dam is intercepted in the intermediate dam and pumped back into the dirty water dam. The clean water dam serves as a last front, from where a positive hydraulic gradient forces water back into the intermediate dam. As long as the capacity of the dirty water dam is sufficient and the water level in the clean water dam is above that in the intermediate dam, no pollution of groundwater resources is possible beyond the clean water dam.

Brines could be used to wet ash as it leaves the power station, thus spreading the brine evenly throughout the dump. Brines usually contain high concentrations of sodium. Compatibility studies of sodium with pozzolanic capabilities of the ash will have to be made before addition of the brine is made. It is essential that the pozzolanic nature of the ash be preserved, to ensure efficient capping of the ash dump. This will minimise water throughflow and erosion of the ash by wind and water.

Encapsulation of brines is another possibility for disposal. Eskom is currently looking into the viability of encapsulation as a method to safeguard brines.

Another possible way of disposing of the brines is into isolated areas in coal mines. Initial studies have indicated that this could be a viable proposition under the right circumstances. Since each instance will have to be evaluated on its own merit, no further details will be provided here.

4.3.5 OTHER SOLID WASTE

All domestic waste disposal sites of Eskom have been fenced in, equipped with groundwater monitoring facilities and applications have been made to license the sites according to requirements by the Department of Water Affairs and Forestry.

Results of monitoring within boreholes are presented in Figure 4.34.

The following conclusions are drawn:

- Domestic waste disposal facilities are often located within the immediate vicinity of other wastes, such as fly ash. This is an advantage, because alkaline seepage from the ash dam will precipitate heavy metals that may derive from the domestic waste disposal sites.

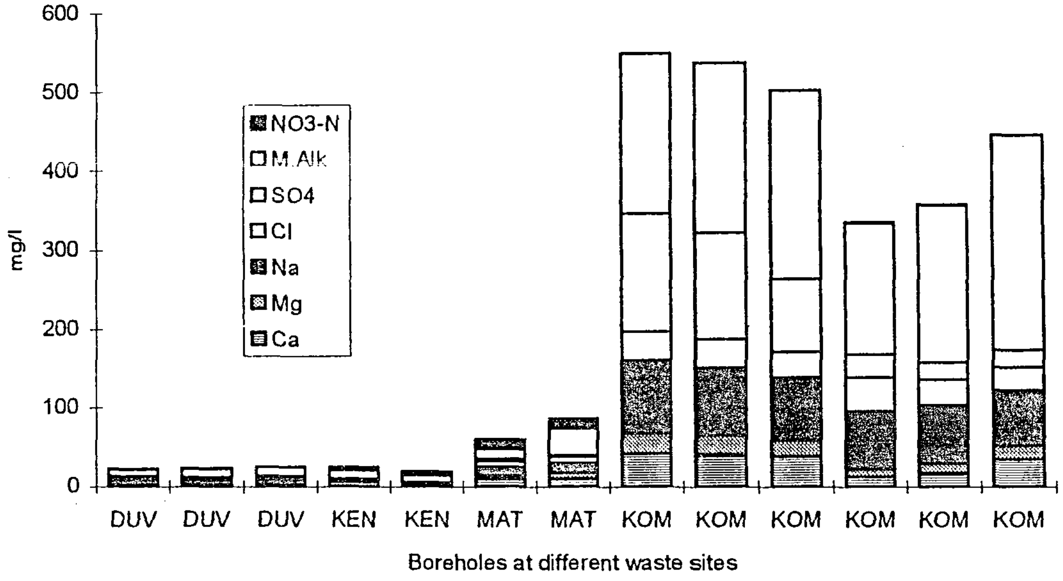


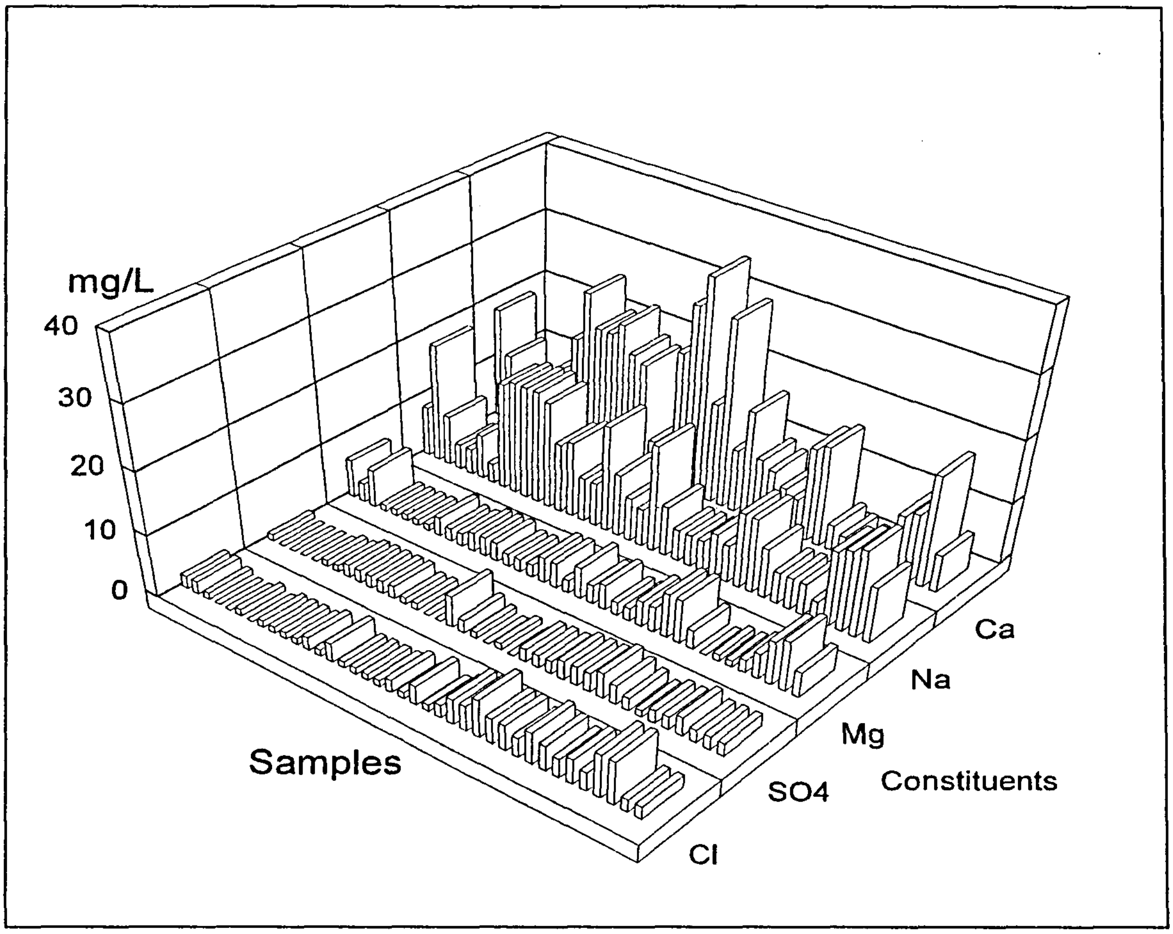
Figure 4.34. Examples of groundwater qualities in the monitoring boreholes at domestic waste disposal facilities at various power stations. Nitrate values range from 1 - 10 mg/l.

- To date, no significant pollution from the domestic waste sites has been detected. The pollution at the Komati Power Station (KOM samples above) is derived from the ash disposal.
- Most of the sites are managed well, though improvements at little extra cost are possible, for example:
- Concentration of waste on as small a site as possible, using the cell concept.
- Sorting of the waste, separating paper, glass and garden refuse.
- Effective covering of the waste to minimise oxygen ingress, thus creating anaerobic conditions within the waste and eliminating acidification.

- Coal discards are not placed within the domestic waste dump as had occurred at one power station because of the high acid-generating potential of the coal.

4.3.6 AIR POLLUTION

Data on sulphate concentrations in the upper weathered aquifer (Figure 4.35), suggest that there is no noticeable concentration of sulphur from stack emissions in the groundwater.



Statistics	Chloride	Sulphate	Magnesium	Sodium	Calcium	Elec. Cond.
	mg/l	mg/l	mg/l	mg/l	mg/l	mS/m
Mean	3	2	3	11	12	13
Median	2	2	3	8	10	13
Mode	1	2	2	5	6	18
Standard Deviation	3	1	2	6	9	6
Minimum	0.0	0.0	0.0	3.0	1.0	3.7
Maximum	16.0	6.000	8.00	33	35	25
Number of samples	41.0	41.000	41.00	41	41	41

Figure 4.35. Information on water qualities within the upper weathered aquifer in the Olifants Catchment.

This conclusion is supported in Table 4.1 and Figure 4.10, where it is demonstrated that although soils in the Olifants Catchment have significant amounts of sulphate adsorbed, the total sulphate adsorption capacity, especially under acid conditions, is far from being depleted. As a matter of fact, sulphate desorption is likely to occur at the low sulphate concentrations (< 3 mg/ℓ) in rainwater.

4.4 MODELLING OF GROUNDWATER POLLUTION AT FLY ASH DAMS

4.4.1 INTRODUCTION

The term 'modelling' has different connotations to different people and it is important that the purpose and types of models to be used, be spelt out at the onset.

4.4.1.1 Types of models

Modelling required by the power stations may be done on five different levels:

The Conceptual Level:

The conceptual level is the starting block, during which the following aspects are considered:

- Identification of the sources that can contribute to possible groundwater pollution.
- A study of the interaction of the various components within each system and between systems.
- Decisions as to the relative importance of the individual components in terms of their possible overall contribution.
- Decisions in terms of the types of model that should be used for the simulation.
- The estimation of parameter values that have little potential to contribute to the overall situations, thus incorporating them empirically into the model.

The Chemical Level:

The chemical reactions within and between leachates and the potential to precipitate, mobilise, transform and retard movement of salts have to be studied. Detailed field investigations are an integral part of this phase. Investigations into management practices at each of the waste facilities are investigated, since different management practices lead to different leachate compositions and loads.

The Numerical Level:

Numerical (deterministic) models are used for the modelling of pollution plumes in groundwater. These models are based on the solution of the mass transport equation. Prediction of the rate of pollution migration, away from a site, is the result. The predicted concentration of pollutants along flow lines becomes available from these simulations. Contours may be used to depict the rate of leachate movement and concentrations. Breakthrough curves for specific points are obtained.

The Generic Level:

Within a power station environment, no single modelling technique can satisfy all the requirements. The results obtained from the different models will have to be consolidated into a generic model for the whole of the groundwater environment at a power station. The results from this exercise are in the form of a table or flow chart, as well as a contour plan, showing salt loads and concentrations to be derived from the different sources.

The Expert Level:

An expert system is a knowledge-based information system. In terms of the power station environment, expert systems can be generated for specific situations. It should be pointed out though, that most of the situations to be modelled at the power stations are dynamic and that expert systems that are devised around these systems can only operate within known constraints. For instance, if a new ash dam is to be established, site selection cannot be accomplished solely on the expert system in existence for the old ash dam, because of site-specific information that is valid only for each of the sites. The HydroCom database at each of the power stations should ideally be updated and linked to the expert system.

4.4.1.2 History of numerical modelling

Since modelling of this kind is involved, it was decided to concentrate on steps one to three in the present document, i.e. the conceptual, hydrochemical and numeric levels of modelling for an ash disposal facility. However, before actually performing the modelling, additional background information on previous work and modelling methodologies with respect to mass transport modelling is presented.

Groundwater modelling, using computer programs, commenced in 1968. During this year, Prickett and Lonnquist as well as Pinder and Bredehoeft, introduced their first finite difference flow models. Between the years 1969 to 1973, Bredehoeft and Pinder set the basis for groundwater modelling by means of the finite element method. They produced models that could be used to predict groundwater flow as well as mass transport. In later years, particularly within the USGS (Pollock, 1989), the trend was again back to the finite difference techniques for solving the flow and transport equations, because of the greater ease with which solutions could be obtained. This limitation of the finite element technique has since been overcome and better accuracies are presently achieved with the finite element solution technique, for the same amount of work.

Modelling of pollution migration from waste management facilities at power stations is complex. Benson and Pasley (1984) have stressed the importance of correctly conceptualising the situation to be modelled, amongst others. They suggest separate physical and chemical conceptualisation, whereafter the two models can be combined to form a refined conceptual model, which describes the three-dimensional site conditions with the required level of accuracy.

Serne and Treat (1987) discussed different types of models, such as diffusion, desorption, solubility, binary combinations of these three mechanisms and strictly empirical leach models. They stress the importance of source-term modelling in performance assessments, prior and during waste disposal.

De Groot et al. (1989) has stressed a proper understanding of the leaching characteristics of the system to be modelled. They developed a three-part standard test, namely: a column leaching test, a serial batch leaching test and a test for the determination of maximum leachability. Their work has shown that the results of the leaching tests closely confirm those found in the field.

Many researchers to obtain a better understanding of chemical reactions that occur within waste facilities have in the past used chemical equilibrium models. The computer code MINEQL has been used as a precipitation/complexation model to predict the solution chemistry of leachates from various extractions of flue gas desulphurization sludge and fly ash with fair to good accuracy by Tiskanen and Fitzpatrick (1984). One of their conclusions was that model discrepancies seem, in most cases, to involve thermodynamic data deficiencies for ash mineral chemistry or the neglection of adsorption phenomena.

Another example of groundwater pollution modelling as it relates to power station fly ash disposal, is that of Whelan *et al.* (1989). In their extensive report on groundwater assessment modelling with respect to the Resource Conservation and Recovery Act, they discuss the development of a two-level modelling approach. Level I consists of preliminary screening with minimal data requirements and Level II of detailed mathematical analysis with extensive data requirements. Their idea was to develop a methodology for the Electric Power Research Institute (EPRI) that will provide the electric utility industry with a means of quantifying health risks posed by the disposal of utility wastes (e.g. fly ash and bottom ash).

At the Institute for Groundwater Studies, groundwater flow modelling has been done since 1973 and mass transport modelling since 1977. Preference has been given within the Institute to finite element models, because of the ease with which complex geometric shapes are accommodated. Extensions of the finite element models have been developed through the years. An example of such an extension is the Bayes model developed by Van Tonder (1990) and Van Zandwyk*et al.* (1992), which provides an estimation of the water-level distribution in the area to be modelled.

The above-mentioned models are suited for modelling pollution movement from waste management facilities, such as fly ash disposal, leachates from coal stockyards and domestic waste disposal. Typical questions that can be answered by running these models are:

- The possibility of pollution migration into the underlying groundwater,
- The rate of pollution movement through the aquifer,
- Predicting areas where such pollution will emanate on surface, including the concentrations of salts within this water
- Advising the waste disposal manager on the best means for interception of pollution plumes.

To demonstrate the degree to which groundwater models can assist in effective waste disposal facility placing and waste management, several typical model runs have been made.

4.4.2 DESCRIPTION OF MODELLING EXERCISE

4.4.2.1 Physical geometry

Several model runs with different geometries have been made during this investigation. The first few runs relate to hypothetical problems, which have been simulated to demonstrate specific characteristics of groundwater pollution migration as it relates to aquifers within the Olifants Catchment.

The layout for the first conceptual site is depicted in Figure 4.36. Two sources of pollution in the form of fly ash dams with a total surface area of 194 ha exist. A stream passes within 300 m of the fly ash dams, draining to the north. No overland flow of water from the ash disposal facilities into the stream is possible. Surficial clays underneath the ash dams restrict seepage of ash water into the groundwater system. Lateral movement of pollution within the aquifer is possible along the weathered sediments and fractures within the rock.

This configuration is typical for the Eastern Transvaal Highveld. The results obtained from this modelling exercise may therefore be indicative of what may happen at many of the power stations within the Olifants Catchment. Specific geometries for each of the power stations will have to be considered, however, before site specific predictions can be made.

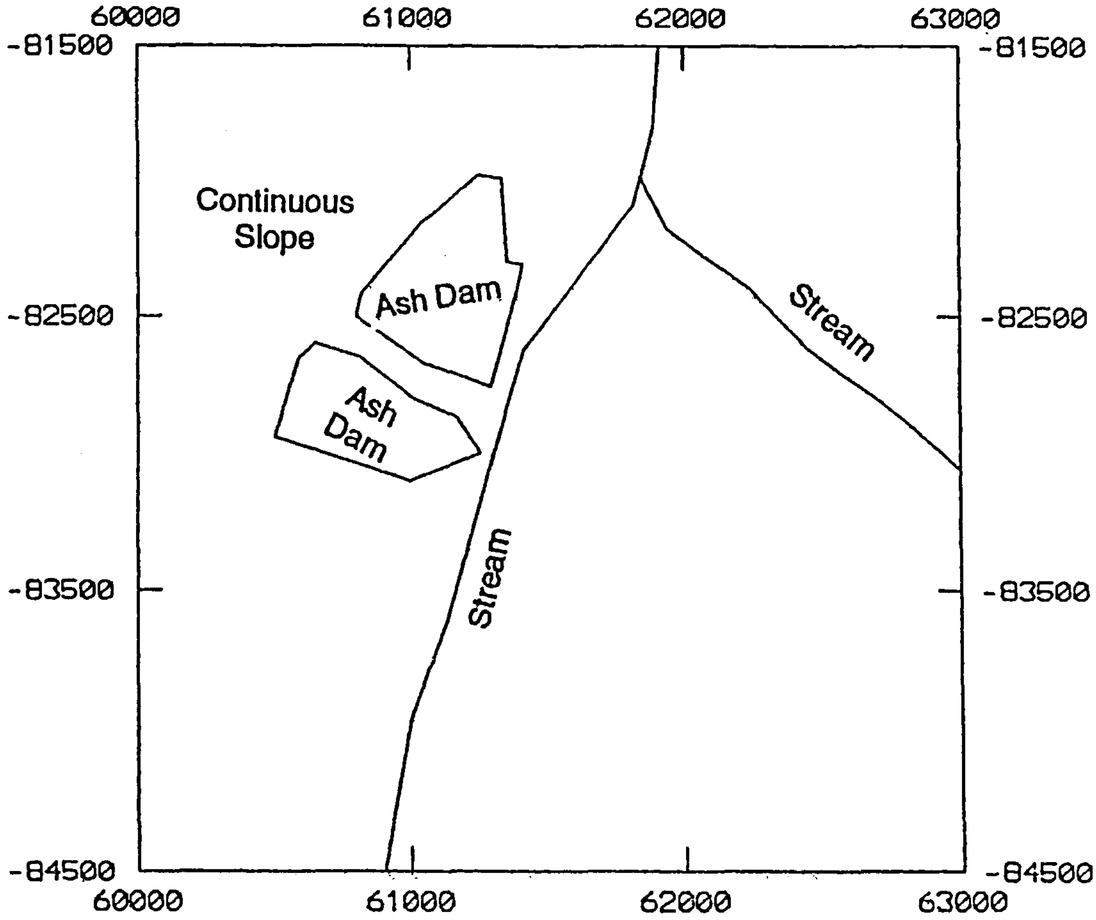


Figure 4.36. Site layout for Model I.

4.4.2.2 Governing equations for modelling

Two sets of equations govern pollution movement through an aquifer as described above. The first equation is the groundwater flow equation, which may be presented in a three-dimensional form as follows:

$$T(\partial^2h/\partial x^2) + T(\partial^2h/\partial y^2) + T(\partial^2h/\partial z^2) = S\partial h/\partial t - Q$$

where

S = Storage Coefficient

$\partial h/\partial t$ = Change in hydraulic head with time (m/d)

T = Transmissivity (m^2/d)

$(\partial^2h/\partial x^2)$, $(\partial^2h/\partial y^2)$, $(\partial^2h/\partial z^2)$ = Flux directions

Q = Abstraction/Recharge (m^3/d)

This equation is stated in this form so that the interrelationship between the individual variables is apparent. This approach differs from that used in literature, where the mathematical concepts behind the numerical solution of the flow and mass transport equations are emphasized. The above equation describes three-dimensional flow of groundwater through the substrata, calculating the water-table response. The equation may be solved analytically for simple problems, though for the ash disposal problem, piece-wise approximation of the equation is usually obtained through the finite element method, because of the complexity of the problem. Once the water-level distribution is available over the whole of the area in question, groundwater flow velocities can be calculated according to the following equation:

$$v = k(\partial h/\partial l)/n$$

where

v = Seepage velocity (m/d)

k = Hydraulic conductivity (m/d)

n = Effective porosity

$\partial h/\partial l$ = Groundwater gradient

On the basis of the hydraulic gradients and flow velocities, movement of pollutants through the aquifer may be calculated next. The mass transport equation is of the following type in its one-dimensional form:

$$D_x(\partial^2 c/\partial x^2) - V_x(\partial c/\partial x) = R(\partial c/\partial t)$$

where

D_x = Dispersion coefficient in the x-direction (m^2/d)

$\partial^2 c/\partial x^2$ = Flux direction

V_x = Seepage velocity (m/d)

$\partial c/\partial x$ = Change in concentration with distance

R = Retardation coefficient

$\partial c/\partial t$ = Change in concentration with time

This equation may be expanded into two or three dimensions, depending on requirements.

It is clear from the above, that coupling of the mass transport equation with the flow equation becomes rather complex in the real life situation, because of the additional variables present. The dispersive properties of the soil and aquifer, degree of convection within these systems, as well as chemical reactions that may take place, must be known before the mass transport equation can be applied successfully.

Certain simplifying assumptions may be made with respect to the problem to be modelled, without jeopardising the value of the model. Examples are:

- The vertical dimension in the equations may be eliminated. Most of the flow is taking place within the weathered upper 10 - 15 m. Laterally, aquifers extend over several kilometres, to the nearest streams and beyond. The vertical dimension is therefore very small in comparison to the lateral dimensions. The time for the pollutant to travel vertically into the aquifer is therefore negligibly small in comparison to the time for the pollutant to disperse regionally. Elimination of the vertical dimension in the problem will therefore have no effect on the arrival time for the pollutant at some distant point. Elimination of the vertical dimension from the simulation reduces required computational time by an estimated factor of ten.
- For modelling purposes, chemical constituents within the ashing environment may be grouped into various categories, depending on their reactivity, attenuation and decay properties. As an example, calcium is unstable in the ash water, since it will precipitate as calcium carbonate when carbon dioxide enters into the water. Another mineral that may also precipitate from the ash water is ettringite, because of the specific ash water

chemistry. In terms of mass transport modelling, constituents that precipitate need not be modelled, because they will not be available for transport. A typical constituent that is likely to persist within a power station ashing environment is sodium. Sodium is introduced into the fly ash through effluent from the demineralisation plant. At modern power stations, about 300 kg of sodium is used daily and the effluent is disposed of in the ashing system. Sodium is therefore a convenient constituent to study the movement of pollution from an ash dam, because:

- Sodium is very soluble and will not precipitate within the ashing system or the aquifer.
    - Sodium does not adsorb readily onto clay particles within the soil or the aquifer.
    - Sodium does not decay with time.

Apart from sodium, no other constituents really need to be considered in a mass transport simulation. Most heavy metals will not be mobilised because of the high pH of the ash water and the neutral pH-character of the groundwater surrounding the ashing facilities. Neither are other mobile elements such as chloride, chromium, molybdenum and boron present in significant concentrations.

The complexity of the problem to simulate mass transport from a fly ash dam may therefore, through logical reasoning, be reduced significantly.

It should, however, be emphasised that even in its simplest form, mass transport simulation is still complicated, because of the many other hydraulic variables that partake in the solution.

4.4.2.3 Hydraulic constraints

Other variables and constraints that act upon the movement of pollutants through an aquifer typically are:

- The transmissivity and hydraulic conductivity of the underlying strata.
- The storativity and effective porosity of the underlying strata.
- The hydraulic gradient, dispersion and convection characteristics of the aquifer.
- Boundaries such as dolerite dykes, catchment and surface.
- Other sources of water in the area such as streams, pans, dams and lakes.
- Sinks within the area where groundwater is abstracted or naturally emanates on surface in the form of fountains.

Transmissivity and hydraulic conductivity

Pumping tests on more than 100 boreholes have been done during the Olifants Catchment investigation in the Eastern Transvaal to determine the hydraulic characteristics of the Karoo strata. Transmissivity characteristics of the strata are generally low. The rate at which pollution will be able to travel through aquifers of the Eastern Transvaal is therefore limited. It should, however, be kept in mind that two modes of travel for pollutants are available. For the purpose of the model, a transmissivity value of 0,125 m^2/d has been assumed. This translates into a hydraulic conductivity of 0,01 m per day. The latter is a feasible value for water movement through the weathered strata.

Storativity and effective porosity

The storativity of aquifers within the Eastern Transvaal has been tested by pumping test methods and an average value of 10^3 can be assumed for the fractured aquifer. The reason for this relatively low storativity value lies within the assumption that only a small proportion of the pores and fractures within the fresh aquifer can contribute to water movement. In the upper, weathered aquifer, the effective porosity is an order higher and a value of 10^{-2} can easily be achieved. This is due to the fact that much of the calcium that normally binds the sedimentary grains has been leached from the weathered horizon. Water can therefore permeate through the weathered matrix with relative ease. Almost all the water within the weathered strata therefore partakes in mass transport and a value of 10^{-2} for the effective porosity of the aquifer is realistic.

Regional water-table gradient, dispersion and convection

As a start for the determination of the regional water-table gradient, measurement of static water levels in a selection of monitoring boreholes is necessary. Since the regional water-table gradient within the Olifants Catchment is controlled by the surface topography, the latter may be used as a controlling factor within a Bayesian estimation model to infer groundwater levels in areas where monitoring boreholes are not available. This provides a well-defined distribution of the water tables, which, in turn, is essential for calculating groundwater flow directions and velocities.

Boundaries

Groundwater boundaries exist in various forms in nature and have to be accounted for in the models. Typical boundaries are dolerite dykes and sills, which may act as impermeable barriers in the transverse direction or as conductive zones along intrusive contacts. Other boundaries are, for instance, catchment boundaries, where an abrupt change in the direction of the water-table gradient may occur. Surface boundaries, above which the groundwater level cannot rise without decanting, should also be considered. Boundaries of these types can be accommodated within flow and mass transport models with the prerequisite that knowledge about these boundaries must be available in the form of spatial co-ordinates.

Sources of water

Sources of water may, under specific circumstances, also be treated as boundaries within groundwater computer models. They may be classified as constant or specified flux boundary. A good example of such a source in the modelling exercise to follow, is that of the constant water supply at the two ash dams.

Sinks

Points where water is taken from the system, such as boreholes, drains or fountains, are usually referred to as sinks. The finite element model to be used in this exercise can accommodate sinks at any position within the model and the effect of water abstraction or water loss at these points, can be simulated. This facility may be used to predict the response of a pollution plume during groundwater abstraction.

4.4.2.4 The finite element network

Because of the numerous variables within the governing equations, as well as geometric constraints that may be placed on the solutions, modelling of mass transport through an aquifer is complex. It is therefore common practice that the area to be modelled is subdivided into smaller areas. If a finite element model is to be used, these smaller areas are commonly referred to as elements. Constraints that apply to each of these elements are:

- Characteristics within each element should apply for the whole element.
- Elements should be of a shape and size that numeric conversion is achieved.
- Elements should depict the geometric constraints that are to be superimposed onto the model.

4.4.3 SIMULATION OF POLLUTION TRANSPORT

4.4.3.1 Introduction

The results of the final simulations are discussed in this section. Figure 4.37 shows the finite element network that has been used in this modelling exercise. The following characteristics of the network are of interest:

- The geometry of the ashing facilities has been accommodated within the finite element network by the introduction of small transverse elements in the area of ashing.
- The stream below the ashing area with its tributary can also be seen in the design of the network.

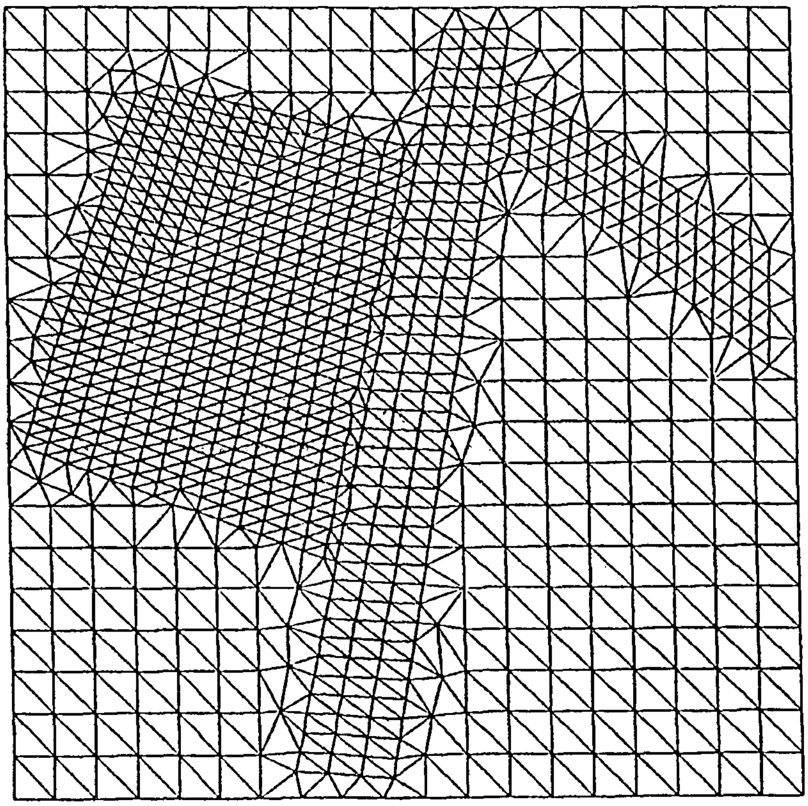


Figure 4.37. Finite element network for Model I.

4.4.3.2 Initial water levels and flow velocities

The initial water-level distribution before ash disposal commenced, has been obtained from water-level measurements in selected boreholes and the conversion of point information to spatial information by means of the Bayes technique (Van Tonder, 1990). The latter improves the accuracy of the water-level contours, since the water-level contours are referenced to surface elevations. Figure 4.38 provides an idea of the water-table contours before ashing operations commenced. Natural groundwater flow velocities before ashing are depicted in Figure 4.39. Arrows of various lengths show directions and flow velocities for each node in the finite element network. It is clear that groundwater flow generally occurs towards the streams. Significant variations in flow velocities exist in the area, as can be seen from the lengths of the vectors. The highest flow velocities are encountered in areas of steep hydraulic gradients. As the groundwater approaches the streams, flow velocities decrease and groundwater evaporation losses in the area alongside streams exist. Within the streams themselves, fountains exist, which account for further discharge of the lateral groundwater flow.

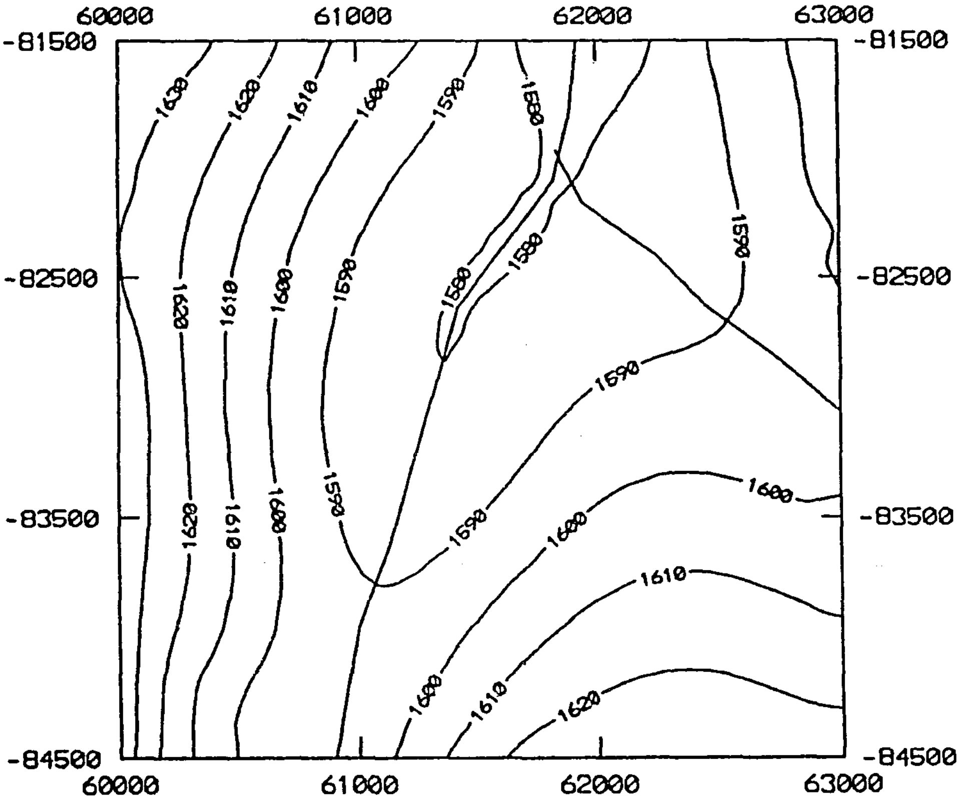


Figure 4.38. Natural groundwater level contours before ashing commenced.

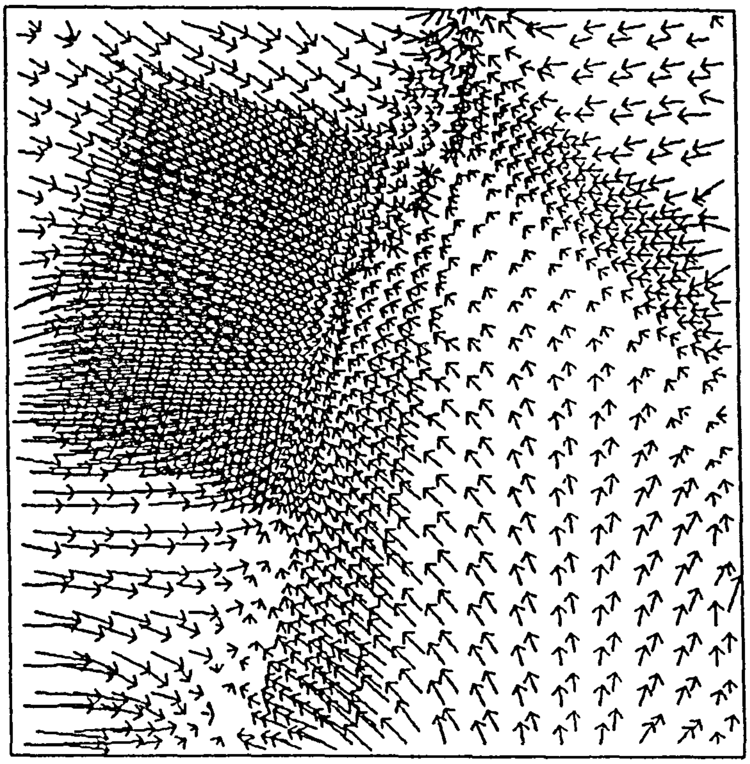


Figure 4.39. Natural groundwater flow vectors before ashing commenced.

4.4.3.3 Groundwater levels and flow velocities during ash disposal

As a result of wet fly ash disposal, the hydraulic gradients in the vicinity of the ash dams have increased.

For the purpose of the present simulations, it has been assumed that the hydraulic conductivity of the fly ash is higher than that of the underlying clays. This is a reasonable assumption in view of the fact that the average hydraulic conductivity of fly ash is in the order of 10^{-1} m/d, whereas that of the underlying clays is usually less than 10^{-4} m/d.

Because of this increased hydraulic head in the vicinity of the ashing operations, a distortion of the natural groundwater contours will occur, as indicated in Figure 4.40. Based on new groundwater contours, groundwater flow vectors have been reconstructed in Figure 4.41.

From Figures 4.40 and 4.41, it can be seen that the regional hydraulic effect which ash disposal has on the groundwater system is negligible. The increased groundwater flow velocities are only effective within short distances around the ashing facilities. Along the western perimeter of the two ashing facilities, the hydraulic gradient is reversed for short distances.

The rapid decrease in the flow velocities between the ash dams and the streams implies that a large percentage of the ash water will surface at the line where the flow velocities decrease.

It is common practice that toe drains are installed at wet ashing facilities in South Africa. These drains are usually not deeper than 2 m and they merely serve to keep the surface immediately surrounding the ash dams, dry. From these drains onwards, seepage water will flow with the groundwater regime at a rate controlled by the hydraulic conductivity of the aquifer.

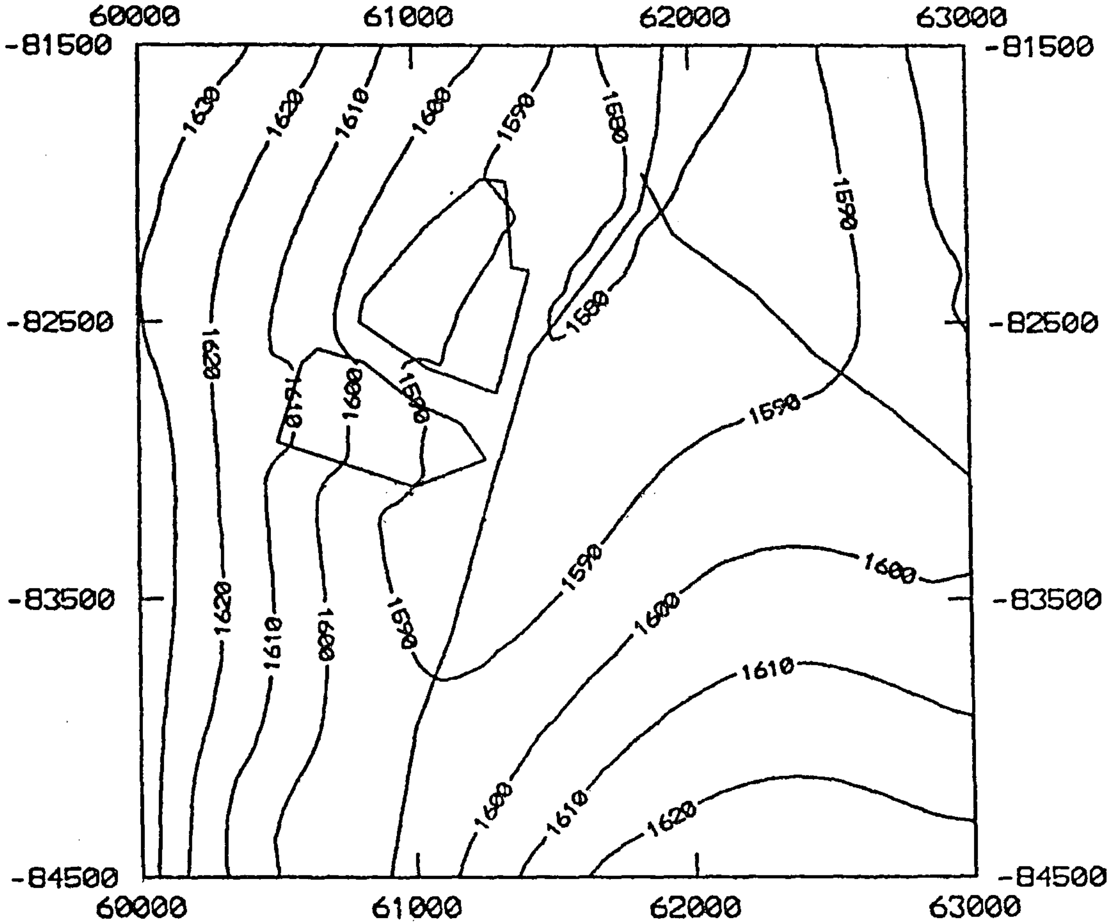


Figure 4.40. Groundwater level contours following commencement of ashing operations.

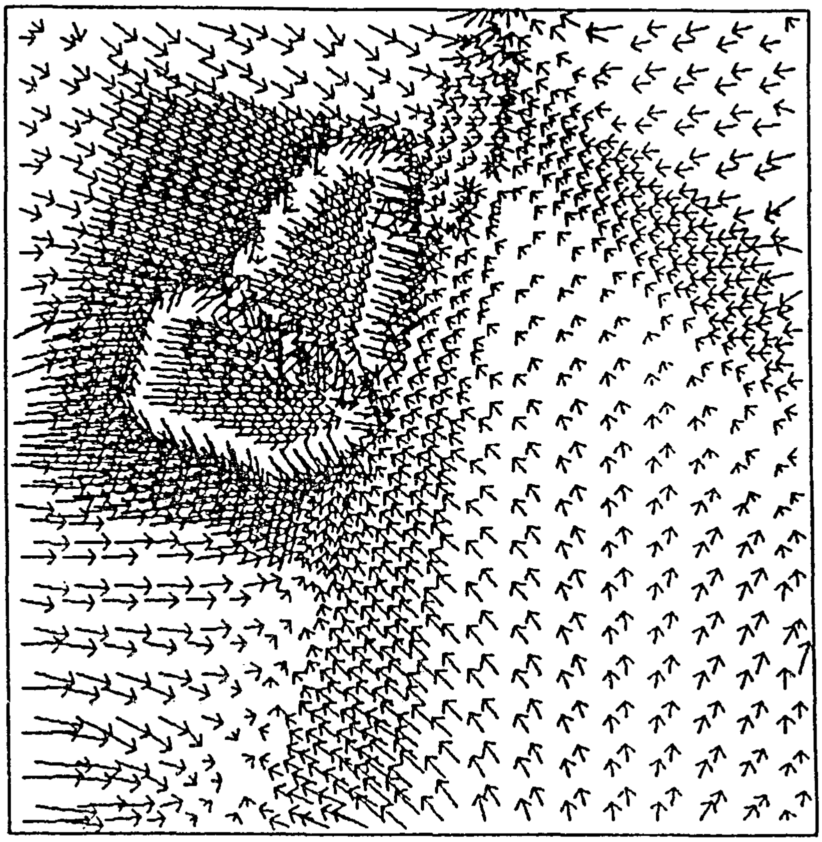


Figure 4.41. Groundwater flow vectors following commencement of ashing operations.

4.4.3.4 Simulation of the pollution plume

For the purpose of this simulation, it has been assumed that a concentration of 150 mg/ℓ sodium exists within the ash water. This is a reasonable assumption based on observations at the power stations. Five simulations have been carried out. The results of these simulations are indicated in Figures 4.42 - 4.45.

The following conclusions can be drawn:

- The movement of sodium pollution from the ashing facilities is extremely slow.
- The rate of movement is limited by the relatively low hydraulic conductivity of the aquifer.
- The lateral spread of pollution during the first few years is dominantly towards the stream.
- From the differences in flow velocities, it can be inferred that more than 80 per cent of the ash water seepage rises to surface in the area between the ash dam and the stream, and in the stream itself.
- Significant dispersion of sodium pollution occurs beyond the first stream, into the area between the two streams. This is ascribed to the very low natural groundwater flow velocities in this area.
- No dispersion of sodium pollution occurs beyond the second stream, because of the relatively high groundwater flow velocities in the north-eastern corner of the area.
- Rapid dilution of the pollution plume occurs further downstream.
- This dilution is sufficient to prevent serious groundwater contamination further downstream.

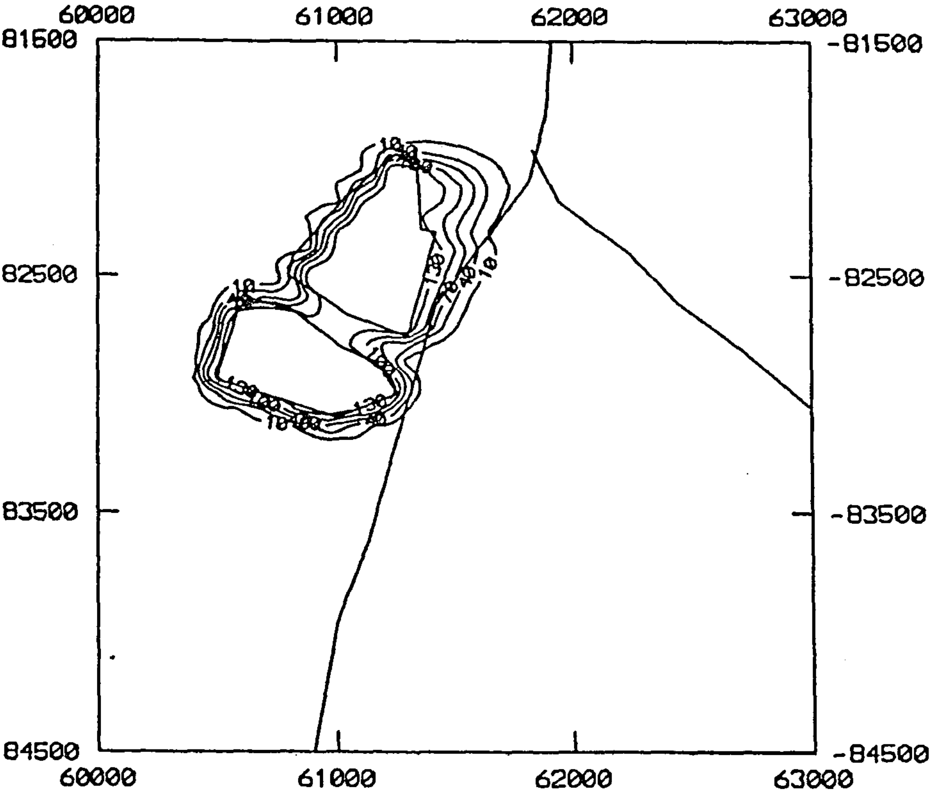


Figure 4.42. Sodium concentrations 2 years after commencement of ashing operations.

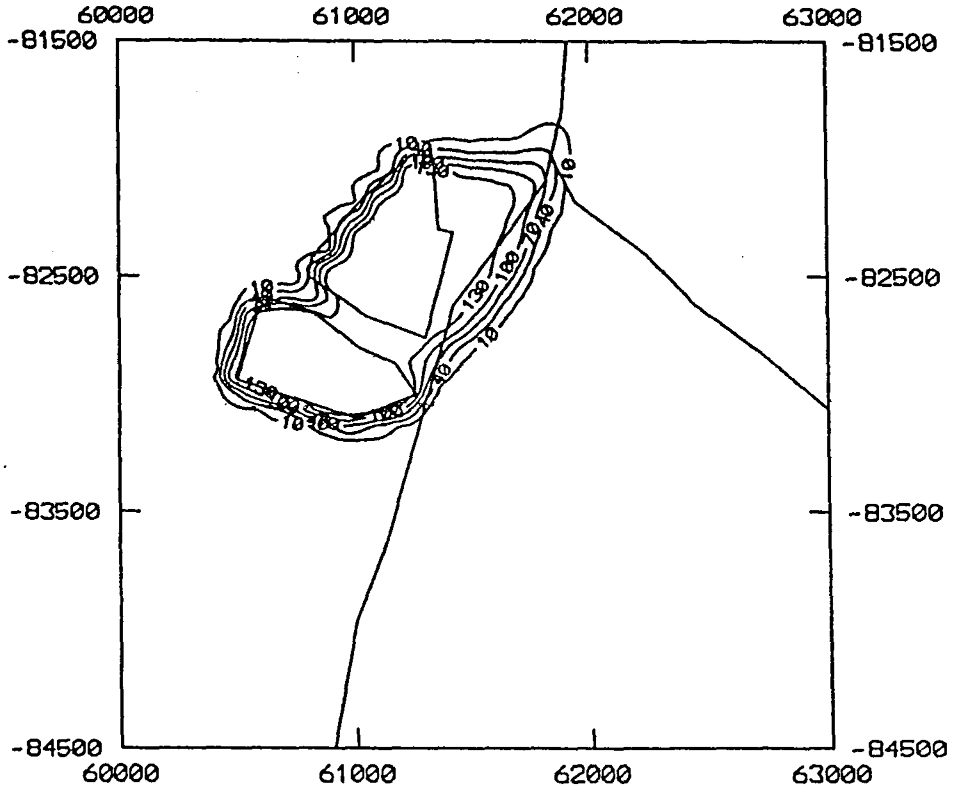


Figure 4.43. Sodium concentrations 10 years after commencement of ashing operations.

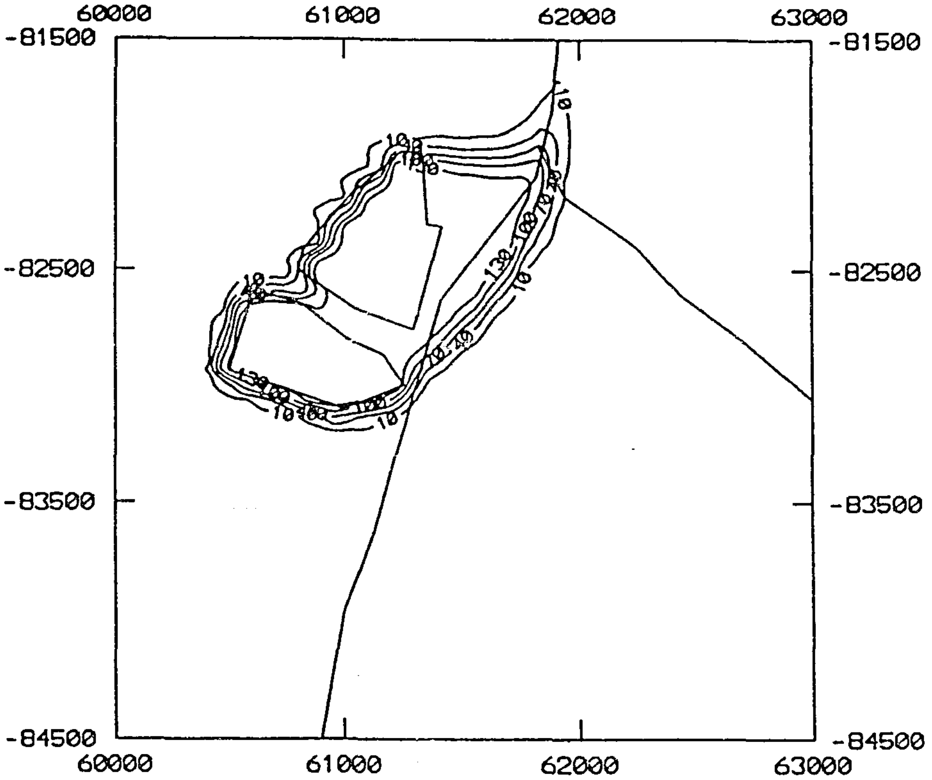


Figure 4.44. Sodium concentrations 20 years after commencement of ashing operations.

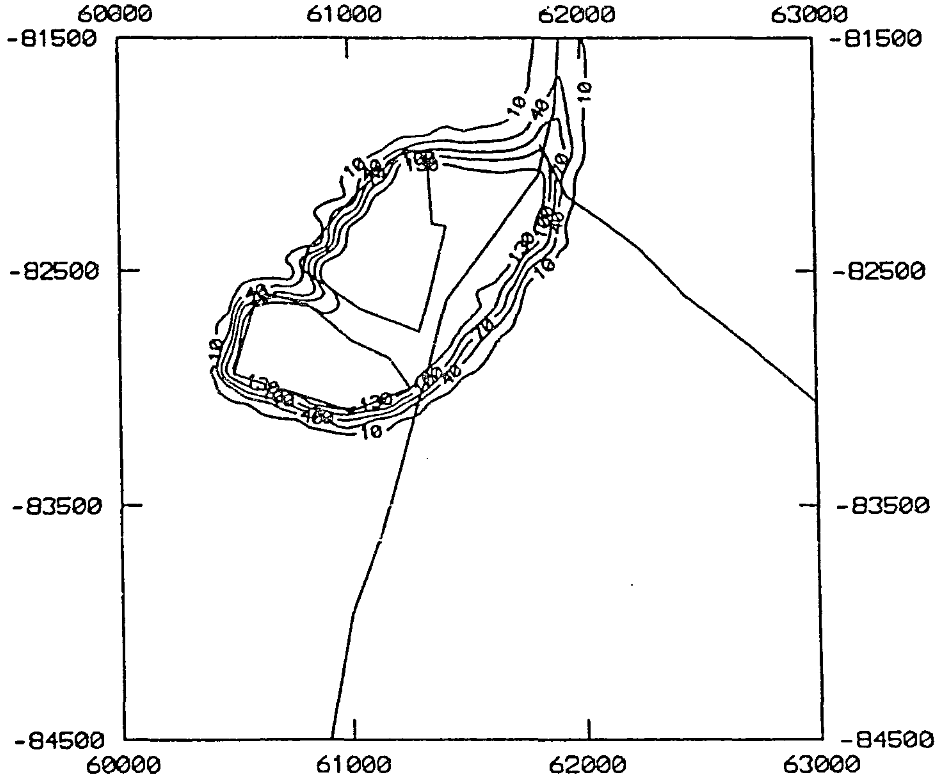


Figure 4.45. Sodium concentrations 40 years after commencement of ashing operations.

The above conclusion is significant in terms of groundwater pollution management. It emphasises that, for the situation as modelled in this exercise, very little precaution needs to be taken by Eskom to prevent the spread of contaminants within an aquifer. The aquifer between the ashing facilities and the stream should clearly be written off as a source for domestic water supply. This is not a serious loss, since the distance between the ashing facilities and the nearby stream is small. At none of the power stations do water-supply boreholes exist between ashing facilities and streams.

Important though, is the fact that the total sodium load passing through the groundwater system, in terms of tonnage per day, could still be significant. Interception of these salts can be accomplished by the incorporation of an agricultural drainage trench in the stream. Two possibilities exist for such a drain. The smallest cost would be entailed if the trench is placed across the stream in position A, as indicated in Figure 4.46. At this point, an automatically activated sump pump can be installed, thus keeping the hydraulic head in the sump lower than that in the ground. Because the groundwater level is at surface within the stream, the drain itself needs only be three metres deep for instance, with the pump keeping the groundwater level down at two metres below surface.

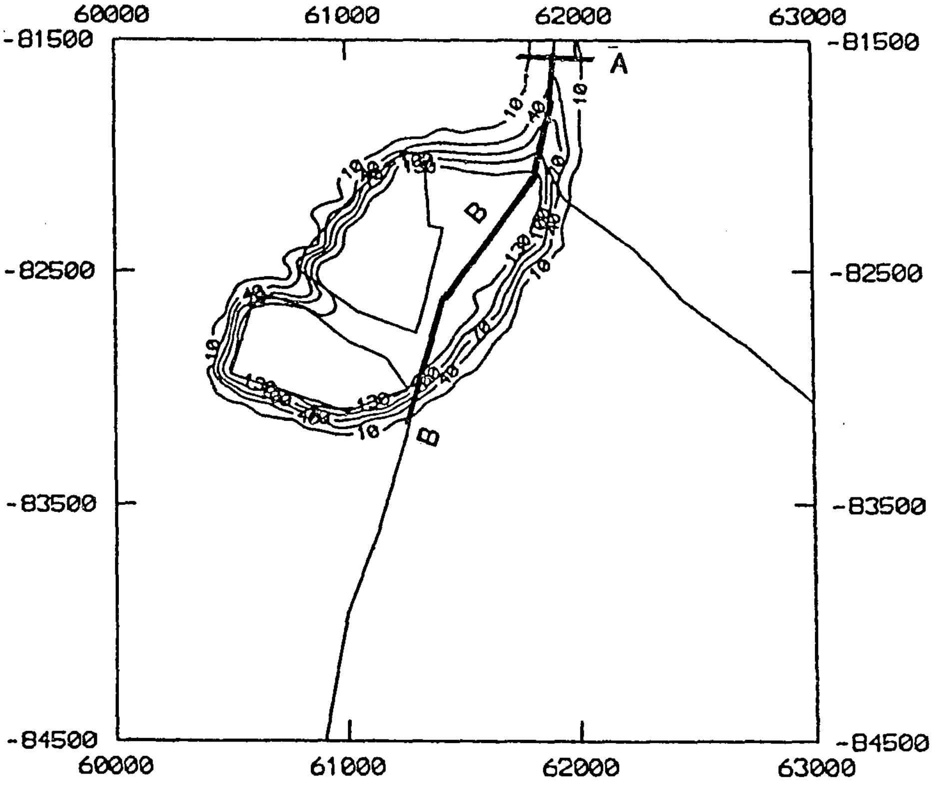


Figure 4.46. Suggested position of pit for interception of pollution plume.

Another option for pollution plume management, is the installation of a drain underneath the stream, for the distance indicated by line B in Figure 4.46. This drain should be installed two metres deep and covered by a layer of low hydraulic conductivity. The intention of this drain is to intercept groundwater seepage from both sides of the stream, while surface water passes unhindered across the drain. Water that enters into the drain can be returned to the ash dam. In conjunction to this drain, an additional herringbone drainage system should be installed in the area between the ash dam and the stream (Figure 4.46), to keep the water levels in this area down. As has been demonstrated in the model, a gradual decrease in the groundwater hydraulic gradient is present in the area between the ash dam and the stream. If a herringbone drainage system is not installed in this area, ash water will rise to surface with the resultant crystallisation of salts on surface. These salts will be mobilised during rainfall events and end up in the stream. The herringbone drains can all lead into the final cut-off drain in the stream, thus minimising long-term operational costs.

The limited propagation of the groundwater pollution plume in the simulation emphasises the necessity that the spread of surface pollution should be kept to a minimum. A single spillage of ash water or ash into the stream will have an impact far greater than that possible through groundwater pollution.

Two further simulations of pollution plume movement have been made, to illustrate and emphasise specific aspects that have come to light in the first simulation. The first of the two models illustrates a pollution source on a continuous slope. The groundwater pollution plume will migrate, with time, over a significant distance (Figures 4.47 and 4.48). It is clear from this simulation that the only dilution effect is that of lateral dispersion within the aquifer.

A similar situation, but with a converging groundwater gradient, has been simulated in Figures 4.49 and 4.50. The dilution effect from clean groundwater coming in from the sides is obvious in this model and the pollution plume is not nearly as pronounced.

For the second modelling example of pollution migration from an ash disposal site, a real life situation has been chosen, namely ash disposal at Hendrina Power Station.

Ash disposal at Hendrina Power Station has many of the elements that have been built into the first model. Figure 4.51 provides a site layout. Specific differences from the first simulation are present, of which probably the most important is the position of the ash dams with respect to the catchment divide. A comparison of the two site plans (Figures 4.37 and 4.51) will show that a continuous gradient exists underneath the ash dams for the first model, whereas in real life, the ash dams lie near the catchment divide. Groundwater level contours before and during ash disposal are presented in Figures 4.52 and 4.53. Figure 4.54 shows flow vectors for groundwater during ash disposal. The flow vectors in northerly and westerly directions are important in that, even though the seepage velocities are low, this could result in wide dispersion of the pollution plume.

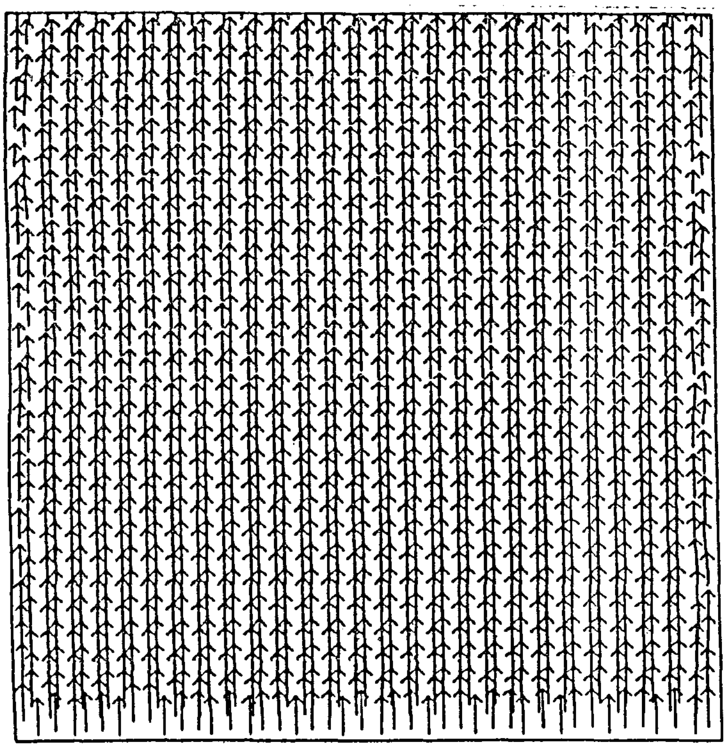


Figure 4.47. Groundwater flow vectors on a continuous slope.

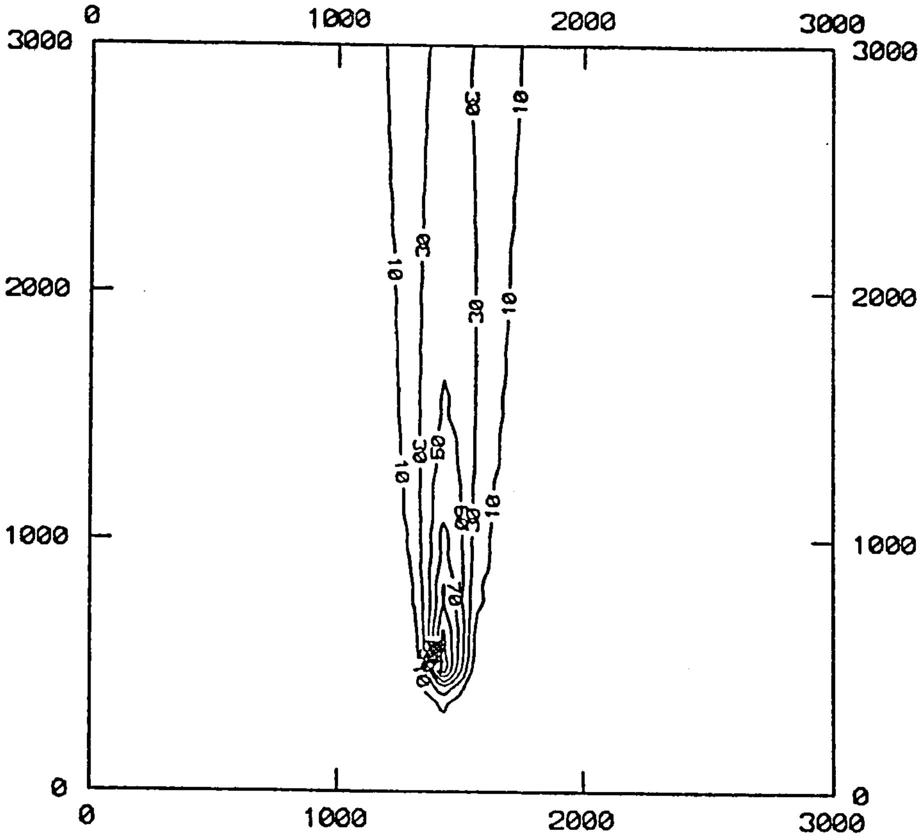


Figure 4.48. Pollution plume on a continuous slope.

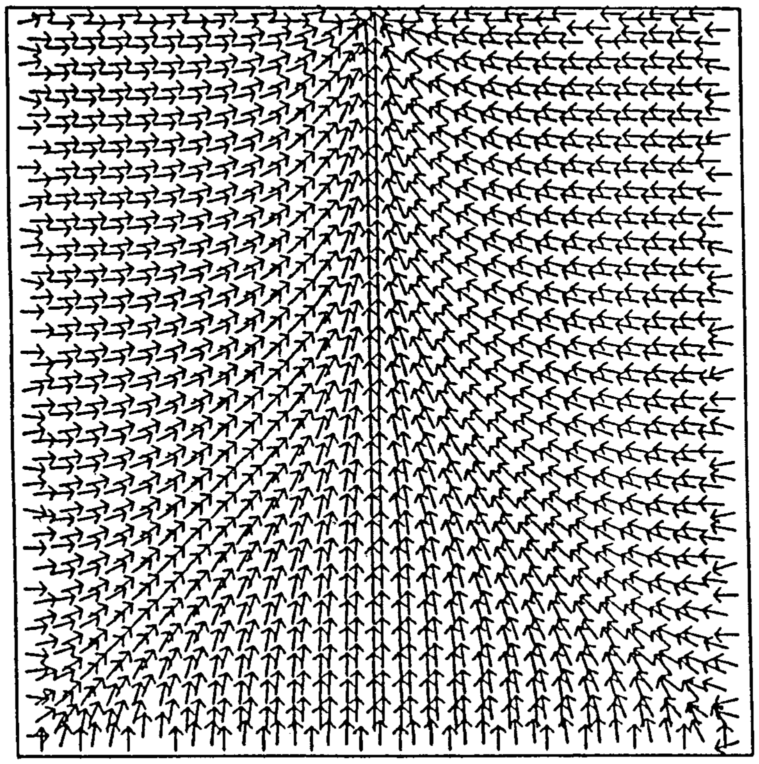


Figure 4.49. Flow vectors on a converging slope.

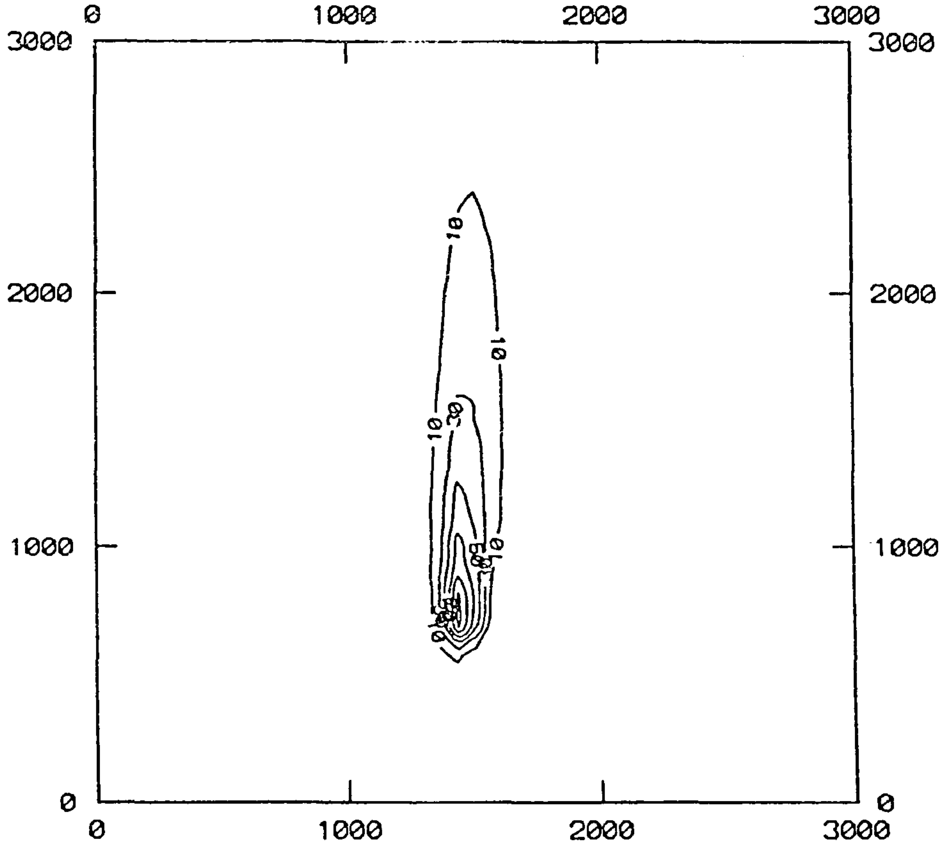


Figure 4.50. Pollution plume on a converging slope.

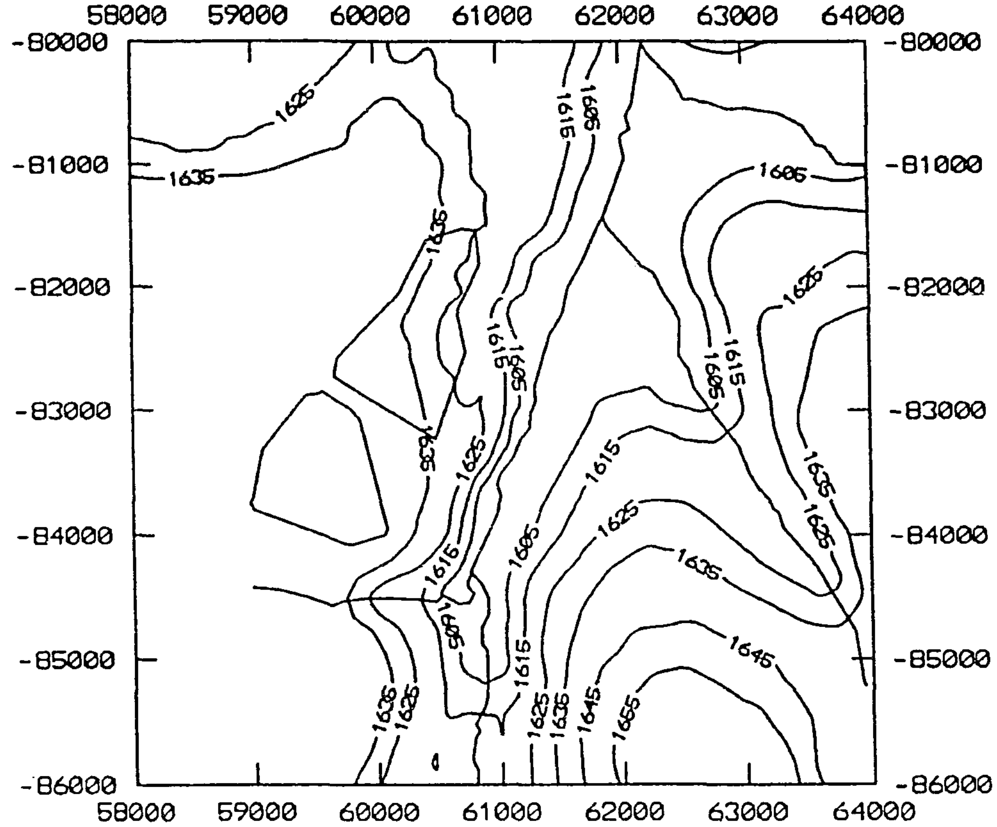


Figure 4.51. Site layout and surface contours for ashing facilities at Hendrina Power Station.

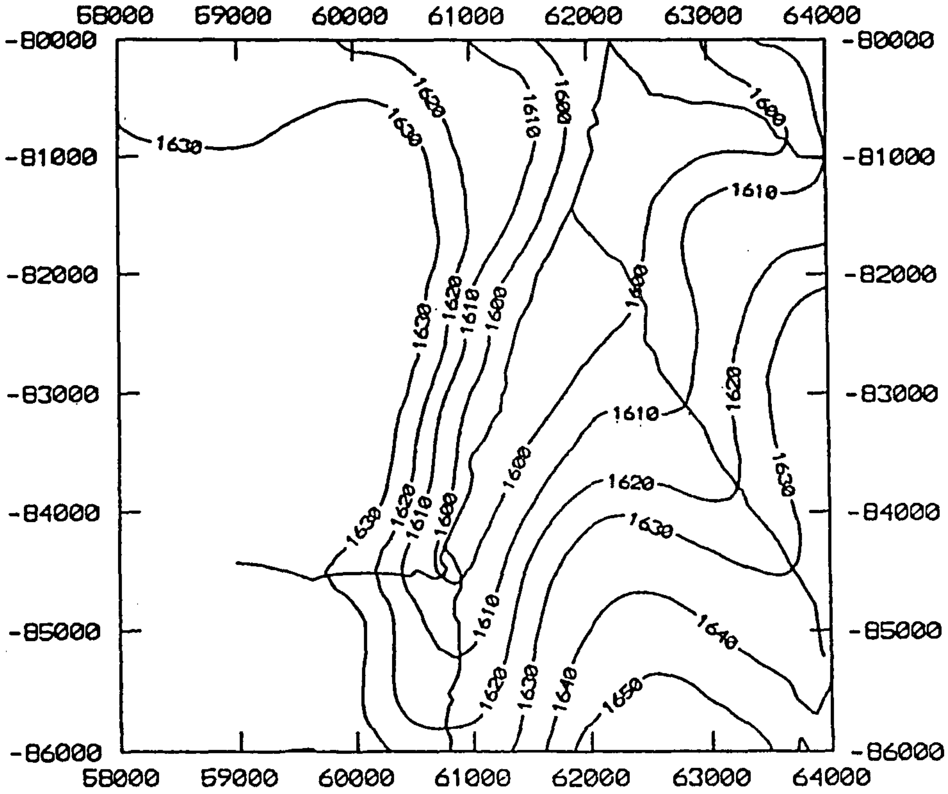


Figure 4.52. Natural groundwater level contours before ashing commenced.

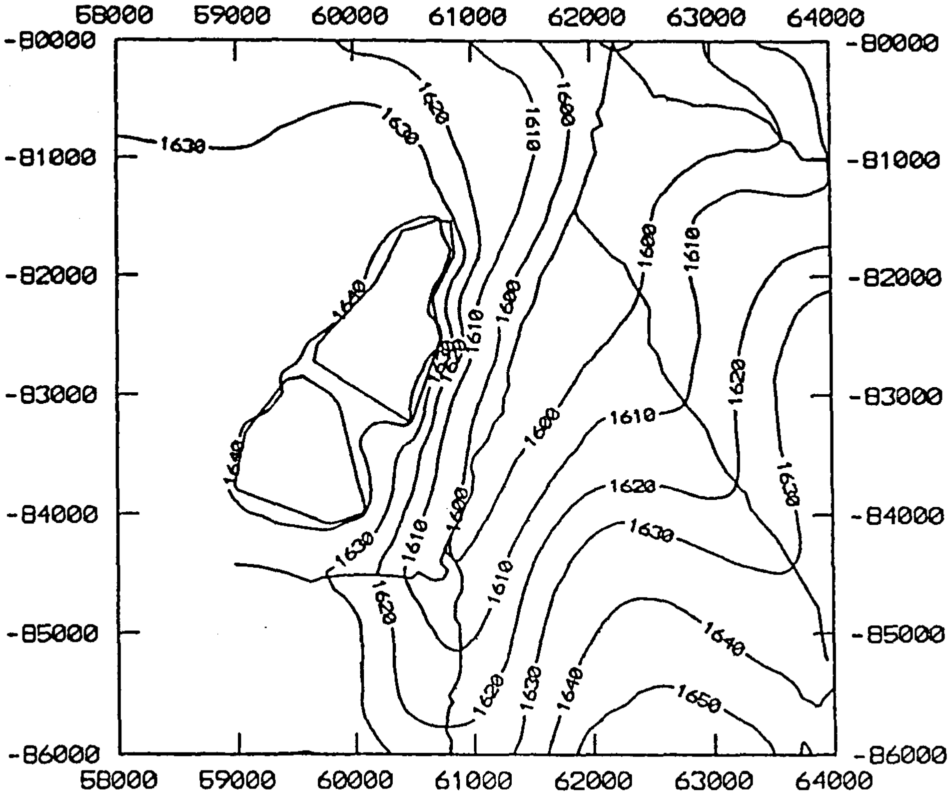


Figure 4.53. Groundwater contours after ashing commenced.

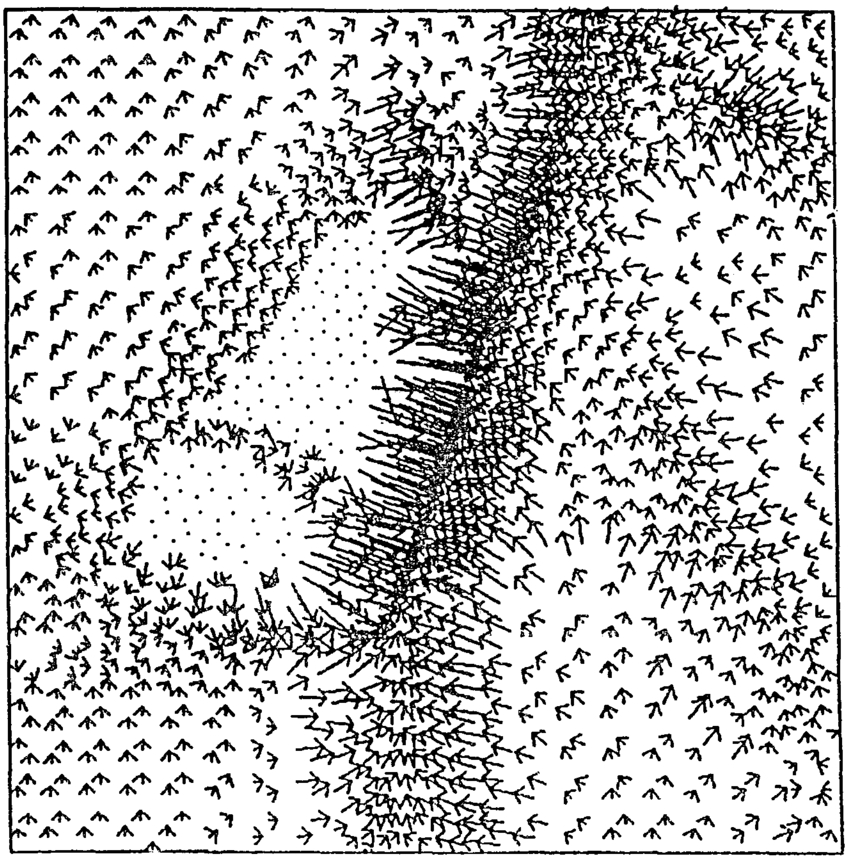


Figure 4.54. Groundwater flow vectors after ashing commenced.

The finite element network that has been used in this simulation is reproduced in Figure 4.55. As in the case of the first model, simulation of the pollution plume movement was done for the life of the power station. Figures 4.56 - 4.58 show some of the results. Flow during the early stage is dominantly towards the stream, because of the high flow velocities in this direction. Dispersion in other directions is minimal. After about 20 years, dispersion to the west becomes noticeable. After 40 years, a breakthrough of the pollution plume to the west occurs, followed by wide dispersion to the north in later years.

The following conclusions are drawn from this modelling exercise:

- Waste management facilities at Eskom Power Stations have very limited potential to pollute regional aquifers, if due consideration is given to their placement.
- Interception trenches, of the nature described above, should be installed at all ash dams and coal stockyards, where crystallisation of salts occurs on surface or where pollution of stream water occurs.

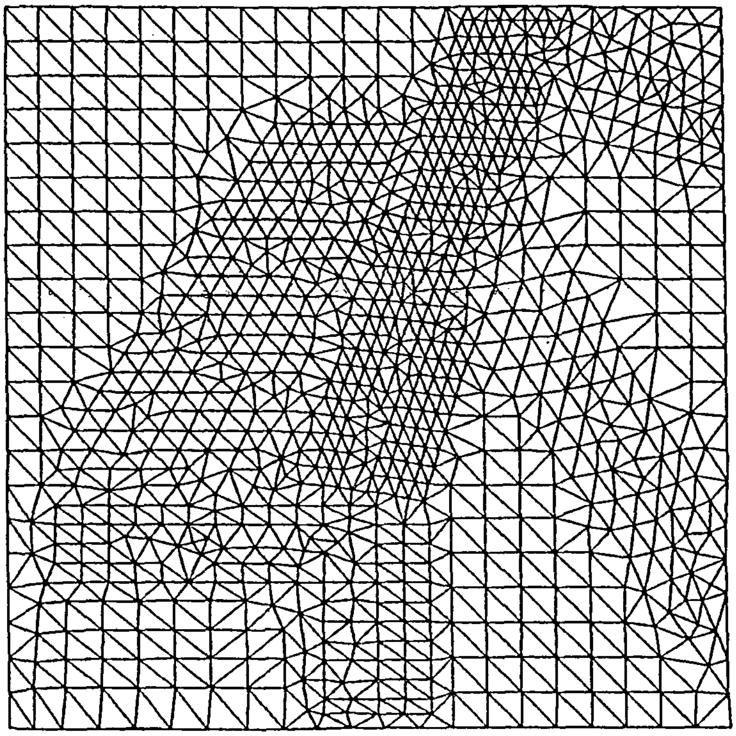


Figure 4.55. Finite element network for ashing at Hendrina Power Station.

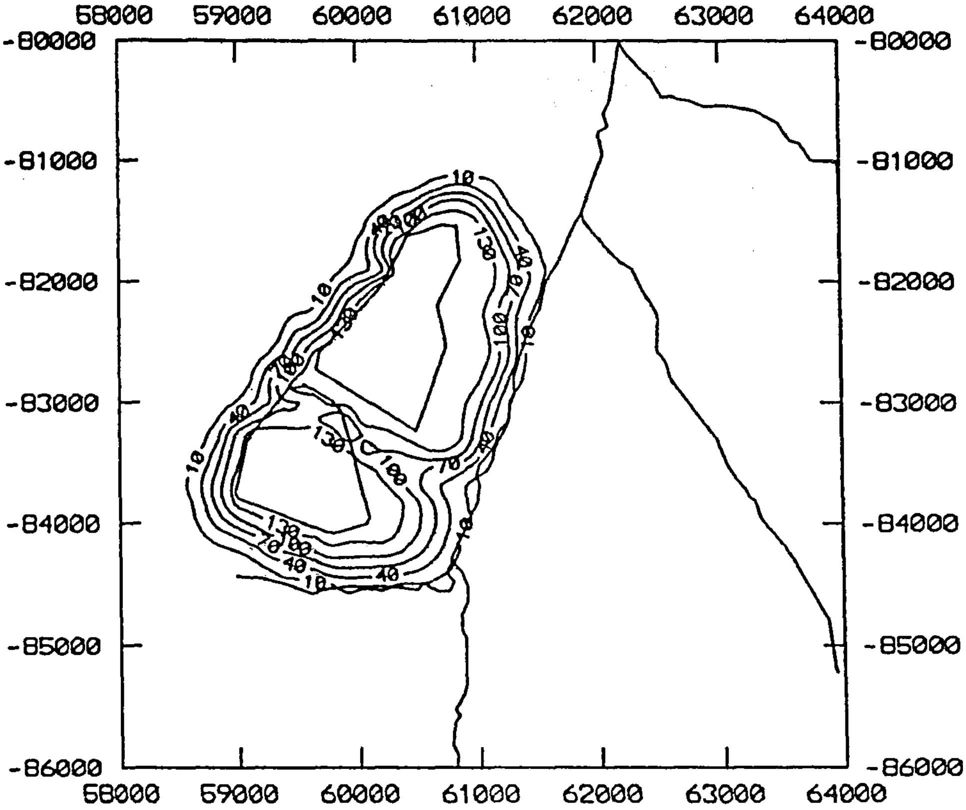


Figure 4.56. Sodium concentrations 2 years after commencement of ashing operations.

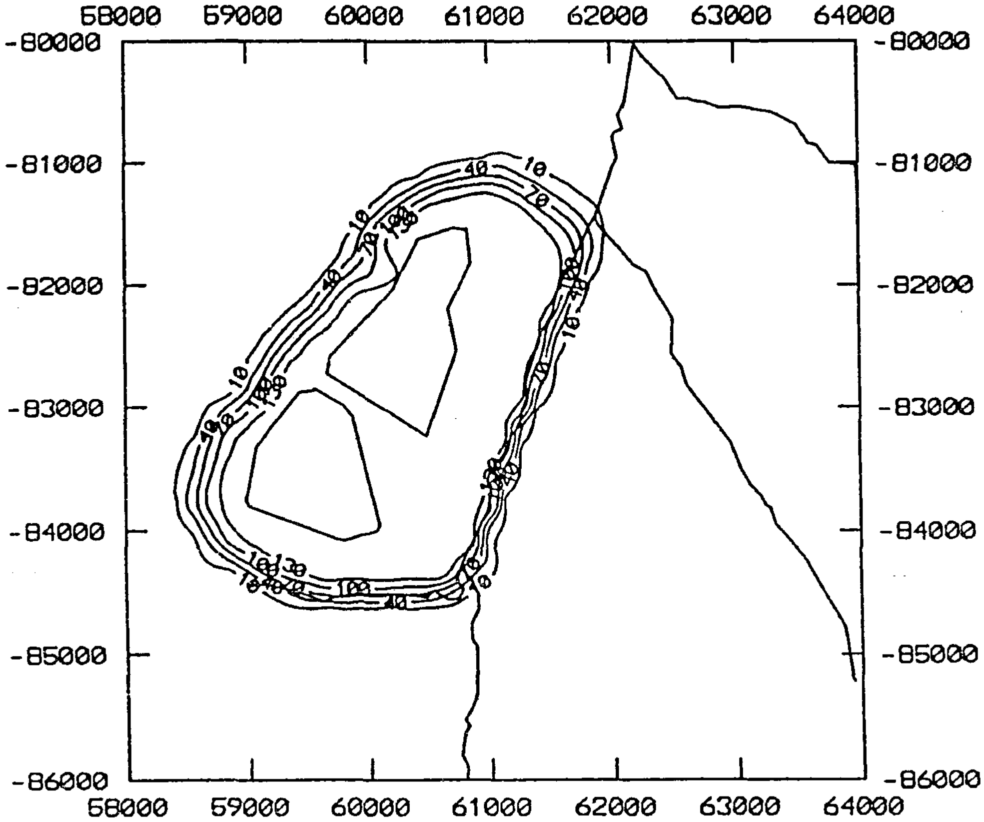


Figure 4.57. Sodium concentrations 10 years after commencement of ashing operations.

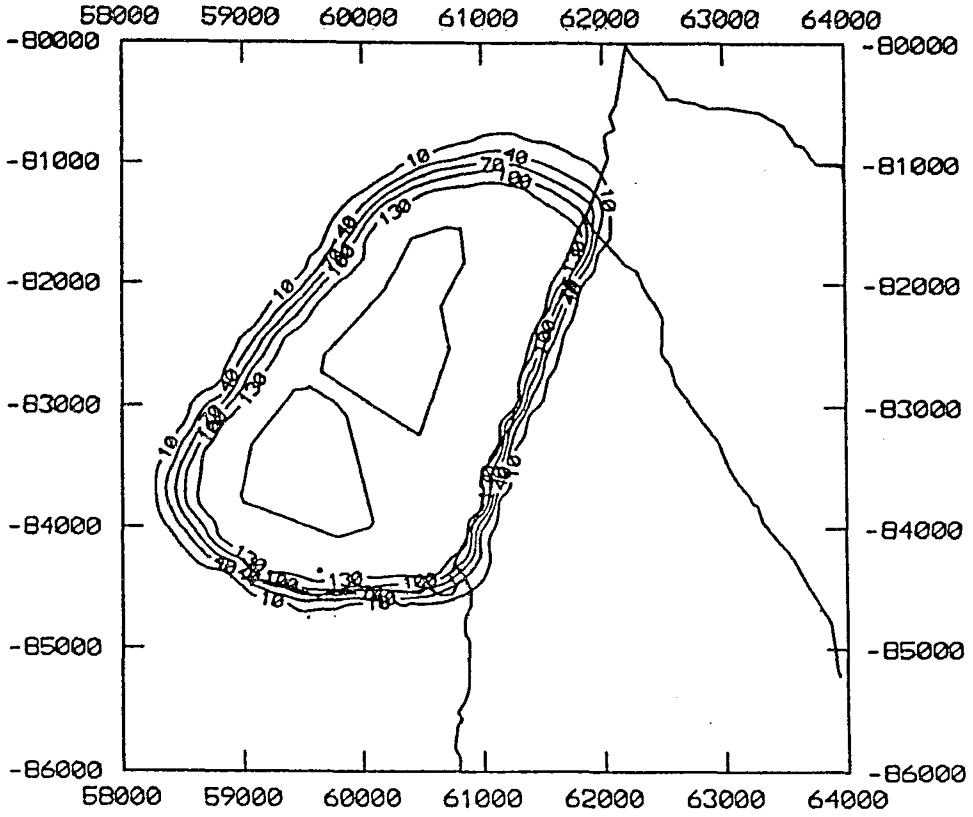


Figure 4.58. Sodium concentrations 20 years after commencement of ashing operations.

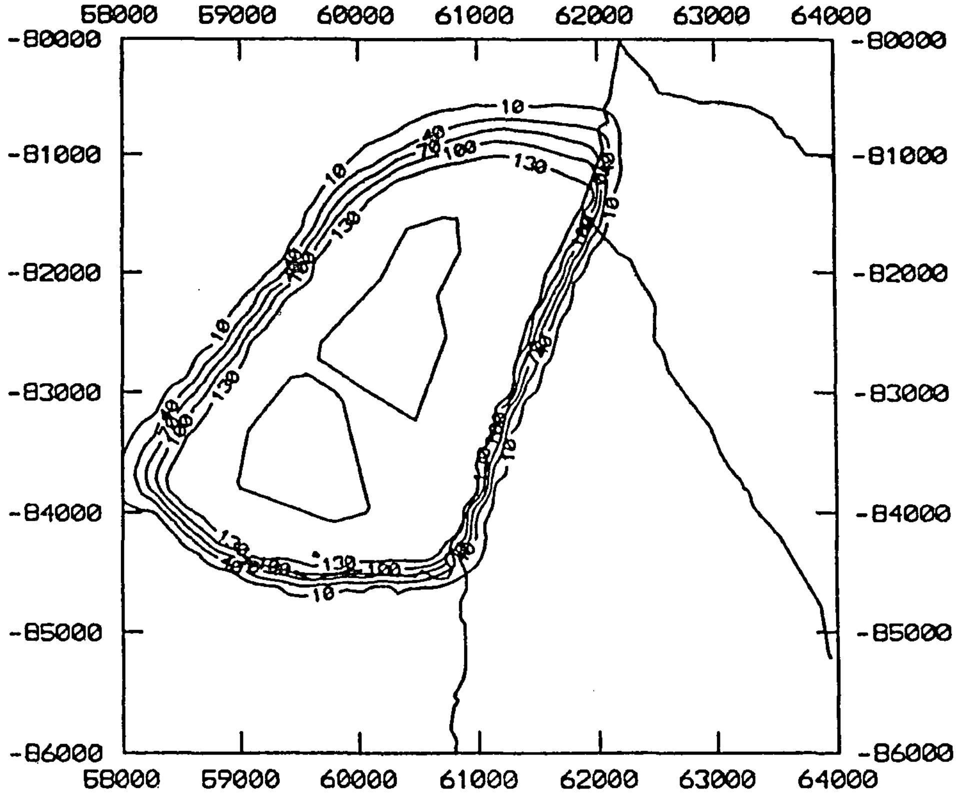


Figure 4.59. Sodium concentrations 40 years after commencement of ashing operations.

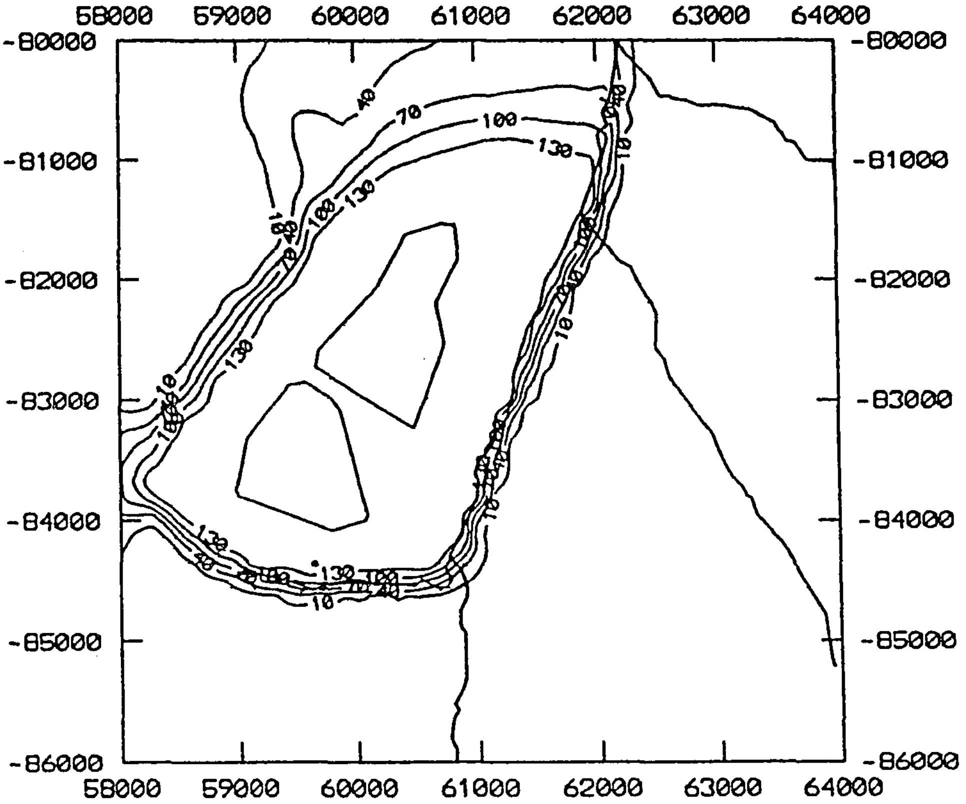


Figure 4.60. Sodium concentrations 80 years after commencement of ashing operations.

4.5 CONCLUSIONS

The following conclusions are drawn from this section of the Olifants investigation:

Groundwater monitoring

- All eight power stations have installed groundwater quality monitoring systems. Monitoring is carried out on a regular basis, mostly by personnel at the power stations themselves. All power stations have HydroCom software for the storage and processing of data which are generated during routine monitoring. Individuals from the power stations have attended courses in the usage of the HydroCom software.

Geohydrology at the power stations

- All power stations are underlain by shallow weathered rocks, ranging in thickness from 5 - 15 m. Water easily penetrates into the weathered rocks. Deeper down alternating layers of sandstone and shale exist. The shale is impermeable to water movement.
- Leachates from waste disposal sites mostly seep only into the weathered strata, whereafter it is deflected along with the regional groundwater gradient towards nearby streams.
- At the streams, the water within the weathered horizon surfaces. The complete water cycle, from where leachate enters into the ground to where it reappears in streams, is in the range of 1 - 5 years, depending on the distance of the waste facility from the stream.

Sources of possible pollution

Sources of possible pollution at the power stations are:

- Fly ash disposal.
- Coal stockpiling.
- Dirty water dams.
- Domestic solid waste.
- Disposal of treated sewage effluent.
- Other minor sources such as oil, grease, pesticides, herbicides, hospital waste and asbestos.

Perspective of possible local problem areas

The following problem areas have been identified:

- A build-up of sodium, originating from the demineralisation process, within the water systems at the power stations.
- Shallow water level conditions around fly ash dams, that could result in white encrustation of sodium salts on surface.
- Possible groundwater contamination from the coal stockyards.
- Periodic excess water at the power stations, that has to be discharged into public streams.

Regional impact

Through observations, substantiated by modelling, it has been demonstrated that the regional impact on the groundwater quality is negligible. The same does not necessarily apply to the surface water resources, for the following reasons:

- Groundwater resources in the areas between disposal sites and streams will, with time, become polluted because of the permeable character of the weathered underlying strata. Modelling has indicated though, that most of the groundwater surfaces at, or before it reaches the stream. For that reason, groundwater pollution plumes will always be restricted in extent.
- Polluted groundwater that surfaces is carried along with the natural run-off, thus contributing to the overall salt load within the catchment. Measures to intercept polluted groundwater before it surfaces have been suggested in this document.
- Once these measures have been implemented, there is no reason why any of the potential groundwater pollution plumes at the power stations should contribute to the overall catchment salt load.

Recommendations

Recommendations for further research and measures to be undertaken to reduce groundwater pollution may be subdivided into two categories namely those which are already being investigated by Eskom and those which are under consideration:

Already being studied or implemented

- An investigation into the so-called self-cleansing action at some of the fly ash dams, during possible ettringite precipitation.
- An investigation into alternative raw water demineralisation techniques, thus eliminating a build-up of sodium in the demineralisation effluent.
- An investigation into alternative ways of brine disposal.
- External environmental audits at regular intervals at all power stations, upgrading existing monitoring facilities, when necessary, to comply with regulations.

Currently under consideration or in the planning phase

- An investigation into water penetrating characteristics and pozzolanic action of fly ash at dry ash disposal sites.
- A study into the feasibility of integrated water management strategies at power stations and coal mines that supply coal to the power stations.
- An investigation into the leaching characteristics of coal stockyards through the installation of leachate collectors at all coal stockyards.
- Modelling the intermediate and long-term effect that waste management has at each of the power stations, thus adding a quantitative aspect to the current qualitative monitoring.
- Implementation of pollution plume interception strategies to minimise environmental impacts, where necessary.

DOMESTIC GENERAL WASTE DISPOSAL AND ITS EFFECT ON GROUNDWATER

5.1 INTRODUCTION

The Environmental Conservation Act, Act 73 of 1989, stipulates that all general waste disposal sites should be licensed. In this respect, the general waste disposal industry is significantly ahead of all other waste handling and disposal industries.

Landfilling is the cheapest method of general waste disposal (Ross, 1990). However, percolating water will leach soluble organic and inorganic chemicals from the waste tip (Thirumurthi, 1991). The resulting leachate may thus pose a threat to the local groundwater resources.

The largest general waste disposal sites in the Olifants Catchment are concentrated within the Witbank and Middelburg regions. According to the respective municipalities, there are no measurable leachates that emanate on surface at the current dump sites. This was confirmed by visits to the various waste disposal sites. Less obvious is the effect which general waste disposal may have on groundwater quality. The following discussions consider this aspect.

5.2 BACKGROUND INFORMATION OF GENERAL WASTE DISPOSAL

Siting, design and operation of landfill sites are well-documented in the literature (Cancelli, 1990; Memon and Downey, 1990; Merry, 1990). The effect of waste composition and co-disposal of waste on the anticipated long-term leachate quality of a dump site has also been discussed (Cheremisinoff and Shah, 1990; De Bruin, 1990). Past environmental impact studies at landfill sites have focused primarily on the generation, migration and impact of leachates on surface and groundwater quality (Husian *et al.*, 1989; Blight *et al.*, 1989, Foerst, *et al.*, 1989; Ross, 1990; Johnston *et al.*, 1991).

The current policy on the classification, siting, design, management and closure of landfill sites in South Africa is described in the "Minimum Requirements for Waste Disposal Facilities", which is currently under preparation for the DWA&F.

For the purposes of this study, the discussion on landfill sites will focus primarily on the method of leachate generation and the consequences of the resultant leachate chemistries on groundwater quality.

5.2.1 LEACHATE DEVELOPMENT

The quantity of leachate that emanates from a landfill site depends on the absorption capacity of the general waste. This capacity has been estimated at 125 l/m^3 for domestic refuse by Marriott (1981). Leachate is usually evident before the field capacity of the landfill is reached. On average, leachate is likely to emanate at 55% of the landfill field capacity (Senior and Shibani, 1990). The refuse composition determines the field capacity, since a high paper content may absorb water in the order of 250% of its own weight. Deeper fills tend to adsorb more moisture before leaching occurs.

Management practices that involve the shredding of material result in a threefold increase in the adsorptive capacity by providing additional void spaces. Highly compacted landfills (>0.7 t/m^3) have a leachate production rate in the order of 15% - 20% of the mean annual rainfall (Senior, 1990). Low compacted waste has an equivalent leachate production rate of between 25% - 50% of the mean annual rainfall. Although these figures are unrealistic for most South African climatic conditions, they do indicate that infiltration may be sufficient for the production of leachate at landfill sites, despite good management practices.

The method of waste collection also affects the moisture content of the landfill since publically collected waste has an average moisture content which is double that of privately collected waste.

The DWA&F (1994), in their Waste Management Series No. 2, has recommended the following water balance formula to determine whether a landfill site will produce leachate:

$$B = R - E$$

where

- B is the climatic water balance in mm of water.
- R is the rainfall in mm.
- E is the evaporation from a soil surface taken as 0,7 x A-pan evaporation in mm or 0.88 x S-pan evaporation in mm.

According to the DWA&F, it will not be necessary to design for leachate collection mechanisms in future non-leachate landfill sites. This has serious implications for future local groundwater quality, should such sites eventually produce leachate. Remediation of such sites will be costly in view of the difficulty in collecting the leachate once the landfill is in place.

A further area of concern is that the recommended water balance formula does not always hold true. For example, the northern municipal dump site in Bloemfontein has a deficient water balance according to the above formula. However, the leachate flow rate has been observed at a rate of 1 ℓ/s (personal communication, Van Tonder, 1993).

The water balance formula should rather cater for the actual evaporation from the proposed landfill site. Holmes (1980) has proposed an alternative water balance equation:

$$\Delta W = R + E - P$$

where

- ΔW is the mean storage change over the site.
- R is the mean site run-off, which is maximised by the emplacement of capping.
- E is the mean site actual evaporation, which includes transpiration of vegetation cover and is estimated to reduce the infiltration rate to 30% of the applied rainfall.
- P is the mean site catchment precipitation (at least 50% of the annual precipitation falling on an operational landfill will infiltrate).

After the storage capacity of the site has been reached, the leachate flow equals the infiltration rate. The quantity of leachate may be improved by vegetating the landfill sites. However, reductions in the leachate volume are not necessarily desirable, since high moisture content promotes the refuse decomposition process. The effect of decomposition on the leachate quality will be discussed below.

5.2.2 LEACHATE COMPOSITION

The fluid migration through a landfill site mobilises the soluble components which are either initially present in the refuse or are formed during the chemical and biochemical processes. Other constituents in the leachate include bacteria, protozoa and viruses (Englebrecht and Amirhor, 1975). Keenan *et al.* (1984) identified three distinct phases of leachate generation.

- An initial period where the leachate reaches field capacity.
- An extended period of leaching of contaminants.
- A period in which the leachate no longer represents a pollution threat.

The chemical quality of leachates is site specific and will depend on the composition of the waste, whether the waste is compacted or shredded, the age of the landfill and environmental factors such as temperature and precipitation.

Christensen *et al.* (1987) characterised the pollutants in landfill leachates as follows:

- Common inorganic cations: calcium, magnesium, sodium, potassium, ammonium, iron and manganese.
- Common inorganic anions: chloride, sulphate and bicarbonate.
- Heavy metals: cadmium, zinc, lead, copper, nickel and cobalt.
- Organic matter expressed as chemical oxygen demand: volatile fatty acids and more refractory compounds.
- Specific organic compounds from household or industrial waste: aromatic hydrocarbons, phenols and chlorinated aliphatics.

Other compounds that are associated with landfill sites are normally of secondary importance.

Deep landfill sites tend to decompose at a slower rate. In these instances, potentially adverse leachate will be generated for a longer time before the general waste has reached the stabilisation phase. Laboratory experiments by Raveh and Avnimelch (1979) have shown that for a given refuse type or density, the concentration of pollutants within the leachate was the same regardless of the infiltration and leachate volumes. However, the leachate quality is affected by a variation in the degree of compaction or density of the material. For example, the leachate from high density baled waste is characterised by low concentrations of total dissolved solids, nitrates, sulphides and BOD.

Leachate compositions vary with time during the stabilisation or decomposition of the waste. The various decomposing mechanisms that operate in a landfill site are listed in Table 5.1.

Table 5.1. Major Attenuation Mechanisms in Landfill Stabilization (after Ross, 1990).

CHEMICAL	PHYSICAL	BIOLOGICAL
Neutralization	Ion Exchange	Hydrolysis
Precipitation	Adsorption	Catabolism
Oxidation	Absorption	Fermentation
Reduction	Filtration	Nitrification
Complexation	Evaporation	Denitrification
Ionization	Extraction	Sulphate Reduction
Acid-Base Reactions	Encapsulation	Mineralization

Biological processes are important since they control the physical and chemical changes via the effect that they have on parameters such as pH and redox potential.

The design of landfills has tended towards the anaerobic digestion of waste because it is, in most instances, possible to apply some degree of compaction (Cossu *et al.*, 1987). Compaction reduces the volume of the waste and inhibits the influx of water and air into the landfill. Figure 5.1 illustrates the progressive changes in indicator parameters during the general waste degradation phase.

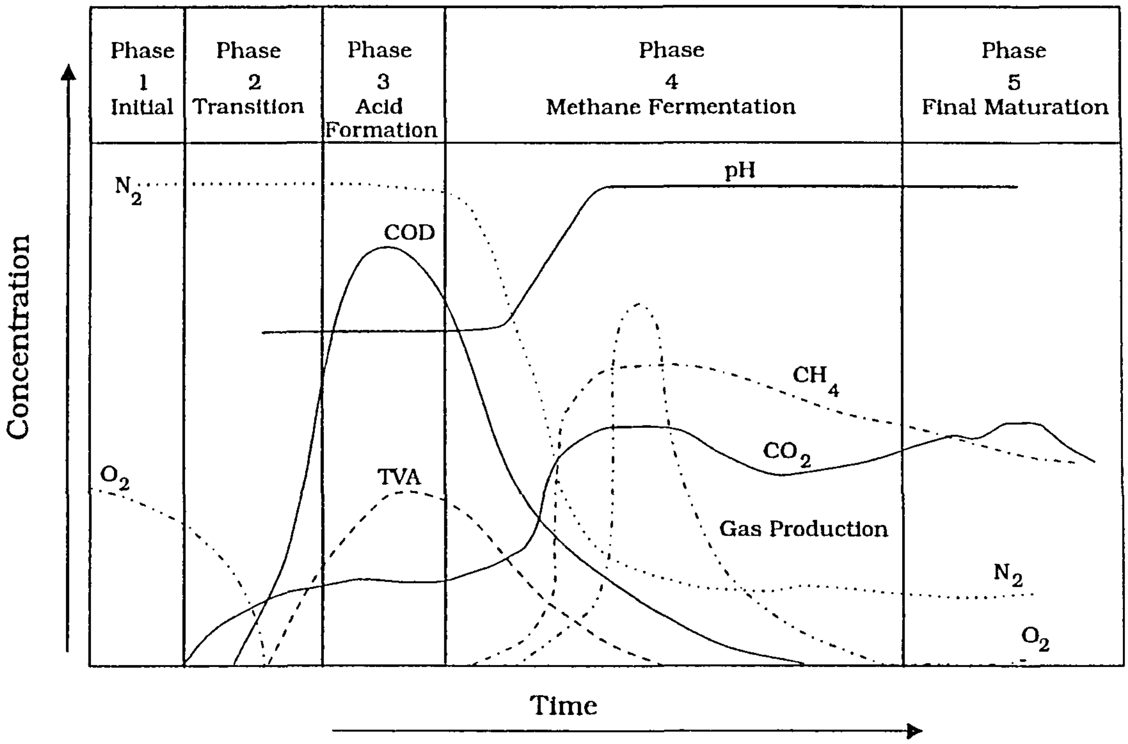


Figure 5.1. The changes in a selection of indicator parameters with the progressive degradation of general waste (after Pohland et al., 1983). Note that there is no specific time or concentration constraints associated with the graph since these parameters are related to the individual on-site conditions.

There are two main decomposition stages in the anaerobic process, namely the acid and methanogenic phases (Cossu *et al.*, 1987). The change from the acid phase to methane degradation will alter the quality of the leachate. The experiments of Robinson (1987) and Cossu *et al.* (1987) indicate that the qualities of the leachates for the two phases are as follows:

- Acid phase: Leachates have a pH-range around 6, are characterised by high COD- and BOD-values and contain abundant volatile fatty acids (TVA). More complex acids are degraded to acetic acid, which is a well-known substrate for methanogenic bacteria. Methane concentrations may be in the order of less than 20% by volume. Acidic conditions in the landfill promote the mobilisation of heavy metals.
- Methanogenic phase: Methane and carbon dioxide are produced and the pH of the leachate tends to be more alkaline. The COD and BOD of the leachate are considerably reduced as a result of the degradation of organic matter.

The cost of disposing of acid phase leachate is considerably higher than that of the methanogenic phase (Beker, 1987). Leachate which is produced during the final maturation or stabilised phase will have the least impact on the environment. Poor management practices result in the retardation of the degradation process. This prolongs the duration of the acid phase and results in the generation of highly polluted leachates.

5.2.3 ENVIRONMENTAL IMPACT OF LANDFILL LEACHATES

Theoretical redox zones associated with the leakage of leachate from a landfill site are described by Christensen *et al.* (1987). The aerial distribution of the zones is depicted in Figure 5.2.

The three zones are as follows:

- Zone 1. An anaerobic zone with a redox potential suitable for methane generation will develop close to the leakage point.
- Zone 2. The anoxic zone develops further away from the leakage point and is characterised by the absence of methane formation and dissolved oxygen. Nitrate, manganese, ferric compounds and sulphate are reduced in that order. Nitrate reduction may dominate in areas of intensive agricultural activities.
- Zone 3. Aerobic zones located in areas where the leachate is sufficiently diluted or attenuated. Free dissolved oxygen is indicative of this zone.

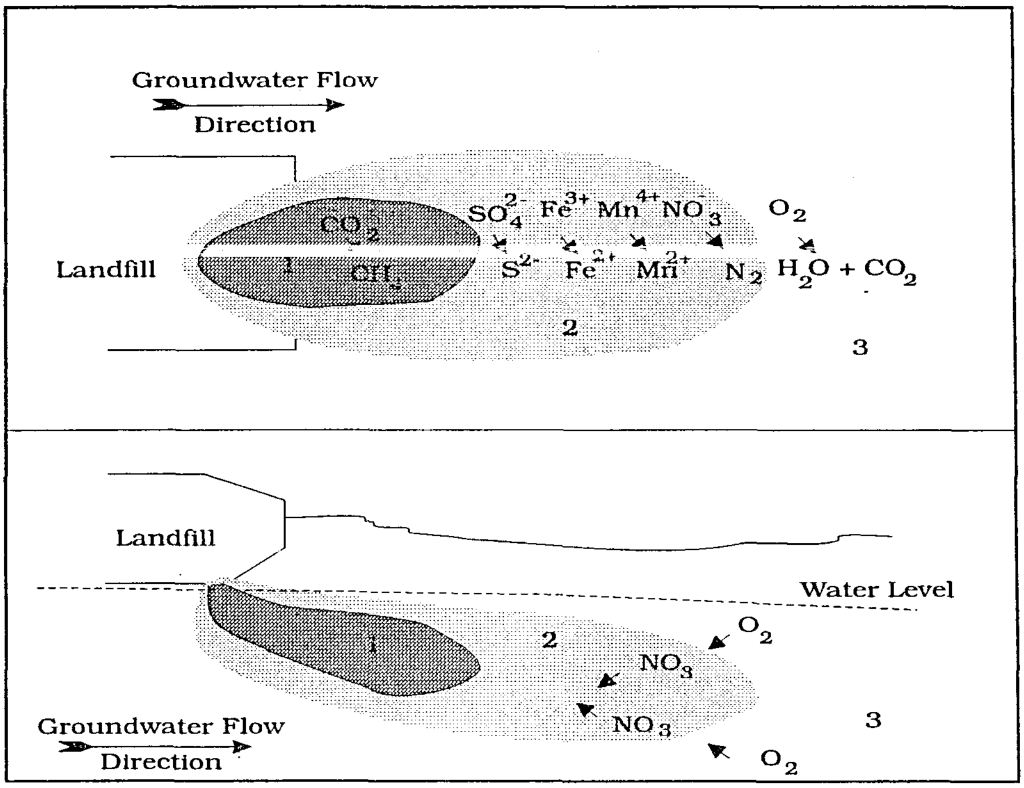


Figure 5.2. Theoretical redox zones downstream from a leachate point (Christensen, 1987).

Continuous leakage from the landfill site may result in the outward migration of these zones. However, as the leachate chemistry changes from the acid phase to the methanogenic phase, the reducing ability of the fluid will decrease due to the lack of degradable organic matter. The reducing zones will consequently be eliminated.

The experiments of Christensen *et al.* (1987) have identified the mechanisms which result in the attenuation of the leachate pollutants. These mechanisms will be discussed in terms of their effect on the various leachate constituents.

The majority of the pollutants are governed by sorption and precipitation/dissolution, e.g. calcium, magnesium, cadmium and zinc. Only a few of the inorganic constituents were governed solely by sorption processes, e.g. sodium, ammonium and, in some soils, potassium and iron. Redox processes also affect the mobility of manganese and in some cases, iron.

It is possible to establish the relative velocity of the components (Table 5.2), although there may be exceptions as a result of on-site conditions.

Table 5.2. Relative velocities of inorganic leachate components (after Christensen, 1987).

VERY MOBILE	MODERATELY MOBILE	PARTLY MOBILE
Chloride	Sodium	Zinc
Total Solids	Ammonium	Cadmium
	Potassium	Iron
	Magnesium	Manganese
	Iron	

Organic matter, particularly volatile fatty acids, is retarded through the process of sorption. However, fatty acids are degraded anaerobically provided the carbon dioxide content does not inhibit bacterial activity. Non-fatty acid fractions are unaffected by the anaerobic environment. Many of the studied components are subject to degradation in the aerobic zone within 280 days. This emphasises the necessity of an aerobic zone underneath a general waste site.

5.3 DOMESTIC WASTE SITES IN THE OLIFANTS CATCHMENT

5.3.1 MAJOR SITES

The major municipal general waste sites in the Olifants Catchment are situated in the Witbank and Middelburg Areas. Witbank Municipality currently operates building rubble and garden refuse sites at Clewer and Reynoridge. An additional large dump site at Blesboklaagte caters for the domestic refuse waste in Witbank. Middelburg Municipality has a large general waste site which accepts building rubble, garden refuse and domestic waste for the entire municipality.

The landfill sites at Witbank and Middelburg Municipalities generally comply with the "Minimum Requirements for Waste Disposal Facilities", as under compilation by the DWA&F. The following aspects which pertain to the retardation of leachate generation, warrant mentioning:

- Temporary daily covers are used at all the sites.
- A seven metre thick clay liner was laid down before waste disposal at the Witbank Blesboklaagte landfill site. The purpose of the layer is to prevent fire, which may occur in the dump from spreading to the underlying coal mines. High density clay was also used to seal defunct mine shafts in the area. The Middelburg landfill site is situated in a clay quarry. The presence of these clay layers should retard the migration of leachate.
- There was no observed leachate flow at any of the sites at the time of the investigation. Middelburg Municipality has a groundwater monitoring point which is situated down gradient from the landfill site. This will facilitate early detection of leachate seepage from the landfill.
- A methane monitoring shaft has been installed at the Middelburg dump site. This will enable the authorities to monitor and evaluate the stabilisation process within the landfill profile. It will therefore be possible to determine the onset of methane production and the generation of less polluted leachates.

Practices that promote leachate generation were identified as follows:

- Water accumulation was observed on the top of the temporary cover at one of the sites after a period of rainfall. These occurrences promote infiltration of water and the subsequent generation of leachate.
- A garden refuse and building rubble landfill has a fountain of water which flows from the one corner of the site during periods of heavy rainfall. This will accelerate the migration of soluble elements and will pollute the surrounding water resources if the nature of the waste is altered at the site.

5.3.2 MINOR SITES

There are numerous minor general waste sites in the Olifants Catchment. These are typically associated with towns, villages, mines, power stations and farms. It is possible to classify such sites into two categories, namely:

- Sites where an attempt is made to dispose of the waste in accordance with the recommendations by the DWA&F (1994) and for which disposal permits will be applied , and
- Sites where waste disposal is disorganised and in contradiction to that recommended by the DWA&F (1994).

Without necessarily identifying individual sites, the following statements can be made:

- The Eskom general waste disposal sites generally comply with the recommendations of the DWA&F. However, clean-up of historic disposal sites is necessary in certain instances.
- Many of the mines are aware of the consequences of unorganised general waste disposal. Nonetheless, general waste is still disposed of in the coal discard dump at several mines. This practice is unacceptable because of the acid conditions within the coal discard dumps and the resultant leaching of heavy metals from the general waste.

5.4 CONCLUSIONS

The following conclusions, regarding the potential for general waste to pollute groundwater in the Olifants Catchment, are drawn:

- Landfill sites do not impact on the regional groundwater quality within the Olifants Catchment. This is a function of the small leachate volumes that are currently generated from general waste disposal, in comparison to other sources of leachate in the catchment.
- This does not imply that local groundwater pollution does not exist. It is suggested that water quality is monitored in compliance with suggestions by the DWA&F (1994) in their Waste Management Series No. 1, to facilitate early detection of any leachate seepage.
- It is disappointing that only one landfill site in the Olifants Catchment has been licensed since the inception of the licensing of landfills five years ago. The problem seems to be twofold. Firstly, applications for licensing are submitted at a very slow rate and secondly, the processing of the applications by the DWA&F appears to take a considerable period of time.
- A concern with respect to future general waste disposal is the classification of landfills according to their leachate generating potential. The formula, which is proposed in the "Minimum Requirements for Waste Disposal" (< biblio >) does not appear to hold true

for all landfill, sites in the Olifants Catchment. This is mainly because of mismanagement at the current operating sites. Most landfill sites are in proximity of urban development. If the correct precautionary measures are not taken, the cost of remediation may be significant. Further work is necessary to establish a reliable method of predicting whether or not sites will generate leachate.

Section 6

TREATED SEWAGE EFFLUENT AND ITS EFFECT ON GROUNDWATER QUALITY

6.1 INTRODUCTION

The four major sewage treatment works within the Olifants Catchment are located within the Witbank and Middelburg Areas. Many smaller sewage treatment plants are located at villages, power stations, mines and other industries within the area of investigation. Treatment methods, which are most commonly used within the study area, comprise activated sludge processes and the use of biological filters. The discharge of treated sewage effluent is controlled by the DWA&F. Local authorities are responsible for monitoring the quality of the effluent and taking corrective action, where necessary. This ensures that the effluent meets the specifications as laid down by the DWA&F. The effluent from the majority of the large water treatment works is monitored on a weekly basis. Sewage is discharged directly into the nearest surface water body.

Informal settlements, particularly at Witbank, lack the necessary infrastructure or facilities to deal with sewage that is generated within these densely populated areas. The potential for sewage pollution from these communities may be significantly higher than that of well-managed municipal areas (Wimberely and Coleman, 1993). These settlements are often located in the immediate vicinity of streams. It is anticipated that surface run-off from such communities will adversely impact on the surface water quality of the catchment. Groundwater models (see Section 4.4) indicate that the impact on groundwater quality, of a pollution source that is situated on a slope adjacent to river systems, will be localised in nature. The attenuation and the dilution of the pollution plume by the converging groundwater flow beneath the river, improve the water quality downstream.

A detailed description of the various mechanisms of sewage treatment is beyond the scope of this study.

6.2 BACKGROUND INFORMATION

Impact studies of sewage on water quality abound in literature (Benett, 1989; Skibbe, 1991; Hrudey *et al.*, 1991; Baird and Pereyra-Lago, 1992).

The accelerated eutrophication of river systems as a consequence of sewage disposal is a well-documented phenomenon (Best, 1982; Wiechers, 1985; Roberts and Ashton, 1988). The transport of pathogenic bacteria within sewage sludge has also been described (Alföldi, 1988).

However, case studies have been published where sewage has effectively reduced the impact of acid mine drainage on catchment areas (Kardos *et al.*, 1979; Sopper and Kerr, 1981; Joost *et al.*, 1987; Davison, 1990).

6.2.1 THE IMPACT OF SEWAGE ON RIVER SYSTEMS

The major consequences of sewage disposal within a catchment area are:

- Eutrophication.
- Increased suspended solids.
- Transportation of pathogenic bacteria.

6.2.1.1 Eutrophication

The major water quality problems in South Africa have been identified as salination and eutrophication (Du Plessis and Van Veelen, 1991). According to Ryding and Rast (1989), eutrophication applies to the state of a water body where the excessive growth and accumulation of algal and other aquatic plant matter occur. The term is more traditionally applied to the natural ageing process of a lake or reservoir as a result of the cumulative impact of sedimentation, water quality and biological accumulations within a lake. For the purposes of this investigation, the process of eutrophication is used in the context as defined by Ryding and Rast (1988).

Six of the eleven eutrophic impoundments studied by Du Plessis and Van Veelen (1991) received substantial quantities of secondary purified sewage effluents. An increased input of plant nutrients in the form of phosphate and nitrate as a result of sewage discharge may stimulate algal and aquatic plant growths. This may result in poor taste and odour characteristics of drinking water, even though it has been treated and filtered prior to use.

Furthermore, as the algal populations die and settle to the bottom of a water body, their decay by bacteria may reduce oxygen concentrations in the lower portions of the aquatic profile. Living organisms are at risk, should the oxygen levels become sufficiently depleted. In extreme instances, the resulting anoxic conditions may promote the accumulation of iron and manganese within the water. These elements may interfere with the subsequent water treatment process. Anoxic conditions may also promote the mobilisation of toxic trace elements contained within the organic solids. Up to 50% of toxic metals such as copper, zinc, nickel, cadmium and chromium may be associated with the solid organic particles within sewage discharge (Hellawell, 1986).

The problem of biological growth may be accentuated where raw or inadequately treated sewage is discharged to the nearest body of water. Despite the efforts of the municipalities to reduce the nutrient pollution in river systems, the development of informal settlements has increased the danger of raw sewage run-off into the rivers within the Olifants Catchment.

However, the increased biological productivity inherent in the eutrophication process may be useful in countering acidic drainage within the catchment system. The initial results of Schindler (1985) and Schindler *et al.* (1985) suggest that acidic water may be ameliorated through the artificial fertilisation of suitable river systems.

6.2.1.2 Suspended solids

Suspended solids are detrimental to the environment where they exist at abnormally high levels for a prolonged period of time. The settling of organic solids will contribute to the substrate blanketing effect which may result in the death of benthic organisms (Hellawell, 1986).

Organic solids are also subject to oxidation by micro-organisms within the aquatic environment. The discharge of sewage therefore results in the increase in BOD within the water body. These processes have the greatest impact in deep slow flowing river systems where the rate of gaseous exchange is greatly reduced in comparison to shallow more active water courses.

6.2.1.3 Pathogenic bacteria

Domestic waste water and discharges from abattoirs often contain infectious bacteria from sick animals and humans. The pathogens are subsequently transported into the waste water infrastructures and form part of the sewage discharge (Kristensen and Bonde, 1977).

Disease such as typhoid, cholera and hepatitis are associated with sewage or the organic pollution of water (Alföldi, 1988).

6.3 SEWAGE EFFLUENT WITHIN THE OLIFANTS CATCHMENT AREA

Representative chemistries of treated sewage effluent at the four major water treatment works within the catchment area are shown in Table 6.1.

Table 6.1. Representative treated sewage chemistries of the four major water works in the Olifants Catchment.

Parameters (mg/l where applicable)	Middelburg	Naauwpoort	Witbank Ferrobank	Riverview
Daily approx. flow (Ml)	12	4	17	7
TDS	385	562	618	628
Suspended solids	1.0	6.4	13.2	11.6
Conductivity mS/m	76	93	96	99
COD	3.0	53	55	64
Ammonia as N	0.6	4.8	9.6	21
Nitrate as N	4.8	<0.1	5.9	0.2
Nitrite as N	*	0.14	0.35	0.23
Orthophosphate as P	4.3	2.1	5.8	7.8
Chloride as Cl	62	45	54	79
Total Alk as CaCO3	130	182	124	210
pH	7.7	7.9	7.8	7.9
Na	*	88	108	96
K	*	15	13	22
SO4	250	182	249	192

* = no value available

Sulphate and the electrical conductivity are in the region of the recommended limit as defined by Kempster and Smith (1985). However, these parameters are below the maximum permissible limit. Ammonia is above the crisis limit for the three Witbank water works. The toxicity of ammonia depends on the pH of the water. Apart from these parameters, it is evident that the final sewage effluent quality for the respective water works falls within the recommended drinking standards of South Africa.

The alkalinity and neutral pH of the sewage are important characteristics with respect to the amelioration of acid mine drainage. When mine drainage is diluted by sewage, the pH is raised and this allows the precipitation of heavy metals. Not only is the toxicity of the drainage reduced, but the water is also diluted. The Naauwpoort water works has a diluting and sweetening effect on the saline mine water from upstream collieries (Van Niekerk, 1993).

A further example is the reduction in aluminium, sulphate and manganese concentrations in the Klip Spruit as a result of treated sewage discharge. The improvement in the water quality is indicated in Table 6.2.

Table 6.2. Improvement in the water chemistry in the Klip Spruit as a result of treated sewage discharge.

Parameters (mg/l) where applicable	Upstream Water chemistry mg/l	Downstream Water Chemistry mg/l
pH	<5.5	>6.0
Al	>0.5	<0.15
Mn	>1.0	0.05 - 1.0
SO4	>600	200 - 600
F	>1.5	1.0 - 1.5

A more detailed chemistry along the Klip Spruit is presented in Figures 6.1 to 6.4. In this study, 17 water samples were taken along the Klip Spruit. Some of the localities are described in Table 6.3.

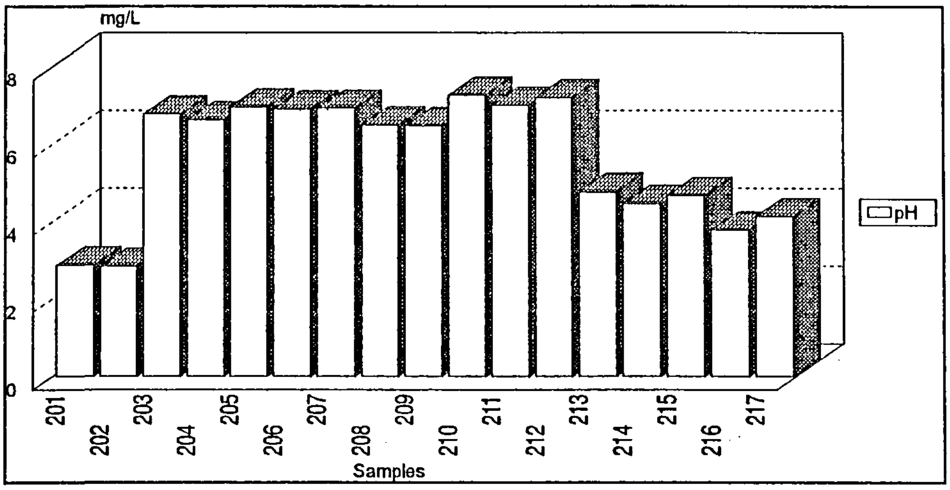


Figure 6.1. pH-levels measured in association with the Klip Spruit, below the Ferro Bank sewage works.

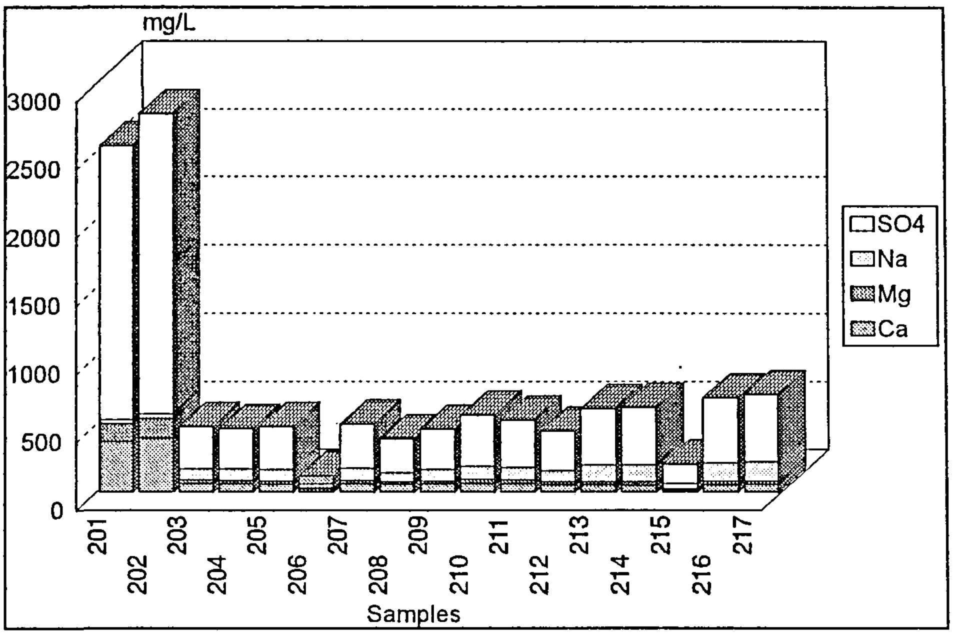


Figure 6.2 Macro constituent levels measured in association with the Klip Spruit, below the Ferro Bank sewage works.

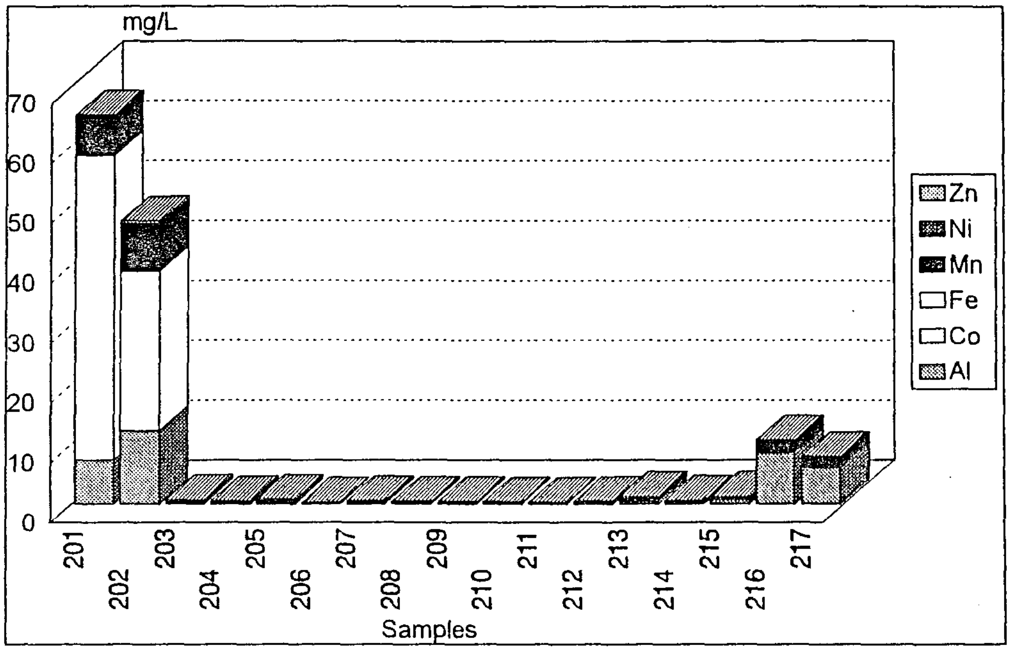


Figure 6.3. Micro constituent levels measured in association with the Klip Spruit, below the Ferro Bank sewage works.

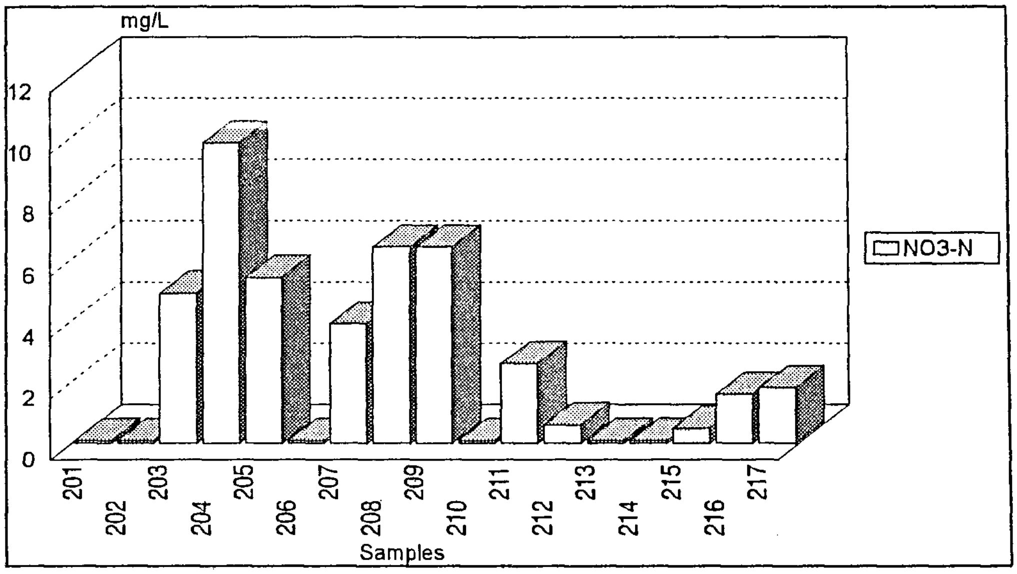


Figure 6.4. Nitrate levels measured in association with the Klip Spruit, below the Ferro Bank sewage works.

Table 6.3. Description of the sampling localities along the Klip Spruit.

Sample Numbers	Locality
C1 - C2	Acid mine drainage stream that flows into the Klip Spruit.
C3 - C5	Klip Spruit below confluence with acid mine drainage stream
C6	Minor stream into Klip Spruit
C7 - C11	Klip Spruit
C12 - C14	Klip Spruit below inflow from Ferro Bank Area
C15	Minor stream into Klip Spruit
C16 - C17	Klip Spruit further downstream

In the field, the ratios between acid water and treated sewage effluent mixing were estimated at 1:17. To test this assumption, a mixing model (PHREEQE) has been run, with the discharge of treated sewage effluent as a given value. It is evident from Table 6.4 that the physical chemistries which were measured in the Klip Spruit correspond closely to the PHREEQE simulation which assumes an acid stream flow of 3,5 Ml/d. However, it should be noted that there is a poor correlation with respect to sulphate and alkalinity.

A better sulphate correlation exists between the physical chemistry and the simulation for an acid flow of 1 Ml/d. The calcium and magnesium discrepancy between the latter model and the physical chemistry may be attributed to the fact that the analysis for the treated sewage by Witbank Municipality was incomplete in this regard. This emphasises the need for accurate and complete chemistries when attempting to predict the impact of sewage discharge on future water qualities.

Table 6.4 A comparison of physical and simulated chemistries of the Klip Spruit, below the Ferrobank sewage works.

PARAMETER	ACID STREAM (1 Ml/d)	TREATED SEWAGE (17 Ml/d)	MIXED WATER MODEL	ACID FLOW (1 Ml/d)
Ca mg/l	394	***	66	22
Mg mg/l	147	***	26	8
Na mg/l	32	94	81	91
K mg/l	7	15	10	15
Cl mg/l	7	61	51	58
SO42- mg/l	2205	214	308	325
TOT ALK mg/l	0	178	70	160
NITRATE (N) mg/l	0.1	5.1	4.9	4.7
pH	2.8	7.8	6.8	7.4

Sewage has the greatest ameliorating effect during the winter months when the stream flow consists predominantly of sewage effluent and seepage from rehabilitated opencast mines. The dilution effects of the sewage waste water serve to improve the quality of the stream water in terms of the salinity and/or acidity.

The remediation of mine spoils by the application of sewage sludge has been extensively researched in the United States. The advantages and limitations of such management options are discussed in Section 3.3.3.1.3.3 and will not be reiterated here.

All the streams which receive treated sewage effluent are called gaining streams. The latter term implies that the groundwater gradient is towards the respective surface water systems. Since the groundwater flow converges at the streams, it will be impossible for surface water contaminants to pollute the groundwater.

6.4 CONCLUSIONS

The following conclusions are drawn from the information in this section:

- The quality of the sewage effluent from the four major water works in the study area is good and generally complies with the recommended drinking-water standards in South Africa.
- The impact, which the discharge of treated sewage effluent into streams has on groundwater quality, is negligible. All the streams in the Olifants Catchment have positive groundwater gradients towards them and can therefore not be polluted from the streams themselves.
- It is anticipated that informal settlements will impact on the surface water quality. The effect on the groundwater quality should be minimal because of the proximity of these settlements to the streams.
- The treated sewage waste water could play an important role in the amelioration of acid groundwater seepage from coal mines within the Olifants Catchment.

Section 7

METAL INDUSTRY AND ITS EFFECT ON GROUNDWATER QUALITY

7.1 INTRODUCTION

Metal industries in the Olifants Catchment are primarily concentrated around Middelburg and Witbank. These plants generate considerable solid waste and sludge. It is estimated that the waste generated per ton of steel may be in the order of 500 kg (Philipp *et al.*, 1986). Some of this discard is potentially hazardous. Metal industries therefore represent a potential threat to the groundwater resources within the Olifants Catchment.

Several in-house environmental impact studies of the steel industries within the Olifants Catchment have been conducted by various institutions. These investigations show that there is some groundwater pollution associated with specific metal processing facilities.

The management policy of the co-disposal of liquid and solid waste promotes the generation of leachate. The resulting fluids allow interaction between various forms of waste products and facilitate the subterranean pollution of the environment (Philipp *et al.*, 1986).

7.2 BACKGROUND INFORMATION

7.2.1 INTRODUCTION

Impact studies have documented the effect of steel industries and heavy metals on the environment (UNEP, 1984; Verriopoulos and Dimas, 1988; Jallen and Pandey, 1991).

Laboratory leaching methods have also been designed to predict the quality and characteristics of the final leachate (Jecko and Raguin, 1981; Philipp *et al.*, 1986; Hall *et al.*, 1987). Methods of treating the wastes and leachate have been utilised in other countries in an effort to reduce contamination (Price *et al.*, 1984; Bagnyuk, 1987).

A pro-active management approach towards metal industry contamination involves, in part, the reduction in the final dump volume. This is normally achieved through recycling or other alternative uses of the waste (Raguin, 1982; Sweeney *et al.*, 1982; UNECE, 1990; Akinmusuru, 1991).

Other forms of pollution associated with metal processing such as phenolic compounds fluorides and salination appear to have received little attention in literature.

7.2.2 SOURCES OF POLLUTION WITHIN THE METAL INDUSTRY

Most of the pollution associated with the metal processing industry originates from solid or liquid waste. Typical sources of possible pollution are as follows:

7.2.2.1 Slag Dumps

These features have been identified as a likely source of groundwater pollution. Heavy metals are progressively released through the weathering of the discard material. These elements may be subsequently transported into the groundwater systems by acidic leachates. Potentially harmful leachates from dumps at stainless steel plants are characterised by nickel and hexavalent chromium. Phenols have also been observed in the groundwater in the close proximity to these features. This is the product of co-disposal of phenol and slag at some of the facilities. The groundwater systems also contain elevated salinities where neutralised acid is deposited on slag dumps.

7.2.2.2 Sludge

Solid particles, including the insoluble organics that are formed during the manufacturing process, form sludge when mixed with water. The composition of the sludge depends on the nature of the metal processing operation and the stage of the process from which the sludge originates. The sludge normally consists of an amalgamation of the chemicals that are utilised or formed as by-products during the manufacturing process. These may include elements such as chromium, nickel, vanadium, manganese and fluoride.

7.2.2.3 Slimes Dams

Pollution of groundwater in the vicinity of slimes dams is usually characterised by elevated salt loads such as sulphate, sodium and chloride concentrations. Elevated chrome values are also observed in addition to lowered pH-values. The tendency of leachate chemistry towards acidic pH-values is a cause for concern, since these fluids mobilise heavy metals which are toxic to the environment.

7.2.2.4 Airborne particles

This atmospheric discharge may provide a diffuse source of pollution, which will degrade the quality of groundwater. Dust extraction systems do not usually trap the finer particles (< 100 microns). These particles contain the most reactive and harmful chrome and nickel components as well as soluble contaminant forms of fluorine, silicon-fluoride and manganese. These particles will therefore easily be assimilated into the water systems, once they settle.

7.2.2.5 Sewage plants and domestic solid waste dumps

These sources of pollution are relatively minor and will not be discussed here. Further details regarding the contamination associated with these disposal and water treatment facilities are described in Sections 5 and 6.

7.2.2.6 Surface run-off

This potentially polluted water seeps into groundwater systems via poorly maintained surface run-off trenches. Other avenues of contaminant transport include the infiltration of run-off from dumps. None of the dumps in the steel industries have been rehabilitated to the authors' knowledge.

7.2.2.7 Accidental Spillages

The spillage of untreated raw materials or effluent expose potentially hazardous material to leaching processes by surface run-off and rainfall. Mobilisation of hazardous material is most likely to be the product of leaching by acid rain. This promotes the formation of soluble chromic sulphate, chromates and nickel sulphate which are toxic compounds.

7.2.3 TOXICITY OF CHEMICAL CONTAMINANTS

The contamination associated with the metal processing industry may be subdivided into two categories:

- *Inorganic pollutants.* These chemicals modify the pH and toxicity of the water. Inert soluble inorganics may also result in the formation of sludge deposits. Inorganic pollutants such as heavy metals may have a major impact on the environment.
- *Organic contaminants.* These compounds have a potential to deplete the surrounding aquatic resources of dissolved oxygen. The bacteria oxidise the organic matter to produce carbon dioxide and water while consuming the necessary quantity of oxygen and nutrients. Certain organic substances are also toxic.

However, there are also inorganic substances that deplete the dissolved oxygen in the receiving environment.

Caution should be exercised when labelling a chemical as toxic, since small quantities may be essential for human bodily functions. Elements only assume toxic characteristics if the quantity needed for optimum biological growth is exceeded. This concept is illustrated in Figure 7.1. However, one cannot allow effluent to be discharged if the constituents subsequently form toxic compounds through the processes of oxidation or reduction.

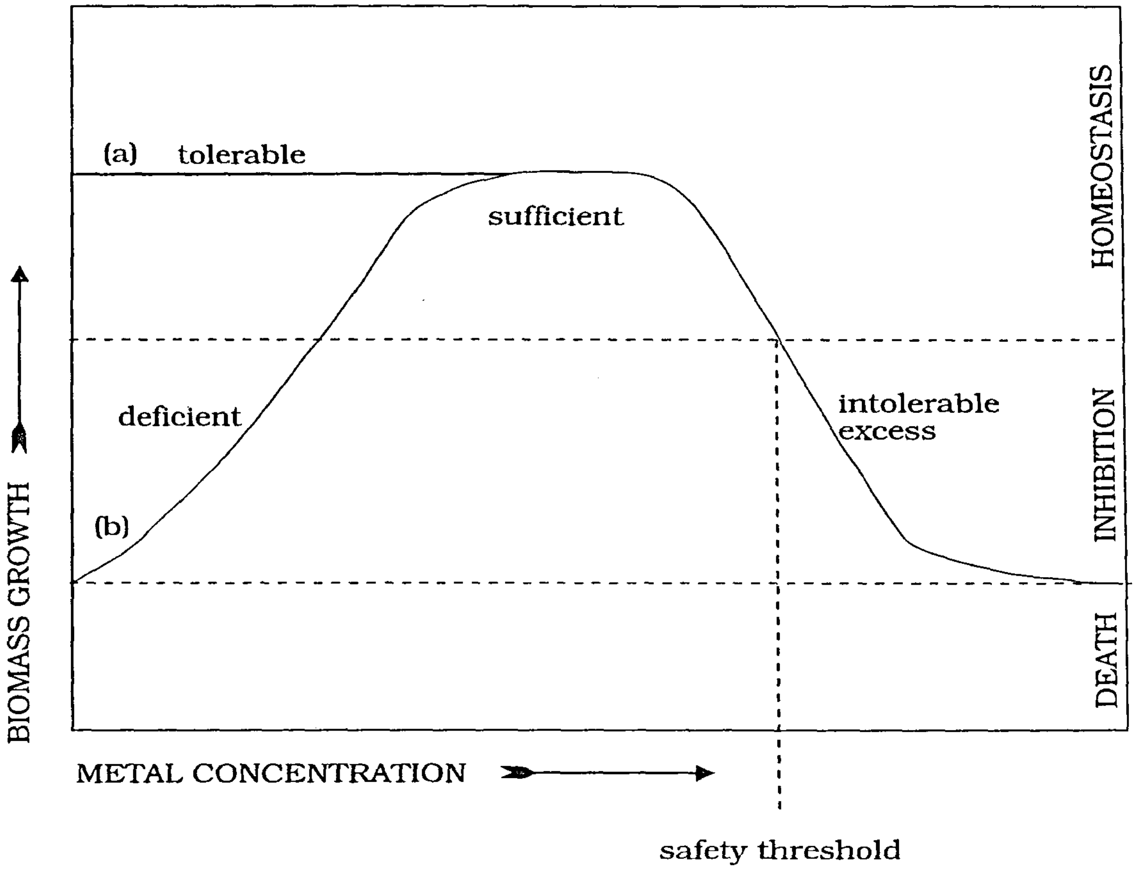


Figure 7.1. Effects of increasing metal concentration on biomass for (a) a non-essential element and (b) for an essential element (after Rudd, 1987).

There is a tendency for metal industry leachates to exceed the recommended drinking standards as outlined by Kempster and Smith (1985) with regard to the following constituents:

Inorganic chemicals including:

- Heavy metals, particularly hexavalent chromium at stainless steel facilities
- Fluoride
- Ammonia
- Sulphate
- Sodium

Organic chemicals including:

- Phenolic compounds

The above components increase the toxicity and/or salinity of the water. These parameters will be discussed in terms of their impact on groundwater quality.

7.2.3.1 Inorganic contaminants

7.2.3.1.1 Heavy metals

For the purposes of this investigation, the discussion of heavy metals will focus on chromium, nickel, vanadium and manganese.

Chromium

Chromium is commonly used in the production of alloys and corrosion-resistant steel and iron (Browning, 1969).

This metal is particularly dangerous since it accumulates in many organisms (Duffus, 1980). It is said that chromium is involved in fatty acid metabolism in humans. It is also an essential element needed for the proper functioning of insulin (Kempster and Smith, 1985). Chromium (III) is less readily adsorbed via the gastro-intestinal tract than chromium (VI). Consequently, the toxic effects of chromium are usually caused by the hexavalent form. Acute effects from oral exposure include the corrosion of the gastro-intestinal tract and kidney necrosis (Rudd, 1987). Furthermore, the hexavalent form of chromium is recognised as a mammalian carcinogen. It should be noted that chromium (III) compounds are more toxic to fish than the hexavalent compounds.

Chromium is normally present in the environment as poorly soluble trivalent salts of low toxicity (Rudd, 1987). Since chromium (III) may be oxidised to the more toxic hexavalent form, it is suggested that the standards for drinking water should be expressed in terms of the total chromium concentration (Schroeder and Lee, 1975). The rate at which chromium (III) is oxidised under normal conditions is uncertain. However, oxidation of this element is likely to be accelerated during the chlorination of drinking water.

Nickel

Nickel is used in the production of corrosion-resistant steel and iron. This metal is an essential nutritional element (Kempster and Smith, 1985). Oral intake of nickel is relatively non-toxic due to the low absorption via this route (Rudd, 1987). It is also rapidly excreted from the body via urine, sweat and hair. However, the ingestion of soluble salts may cause nausea, vomiting and diarrhoea (Budavari*et al.*, 1989).

Vanadium

Vanadium is used for its strong oxidising capacity that increases the hardness, malleability and resistance to fatigue in steel. It lowers blood cholesterol and may be an essential nutritional element (Kempster and Smith, 1985). The normal daily dietary intake is in the order of 1 000 to 4 000 μg/*l*. Vanadium forms inorganic orthovanadate and oxyanions in solution, which are poorly absorbed in the gastro-intestinal tract (Rudd, 1987). Vanadium is therefore considered to be relatively non-toxic to man.

Manganese

Manganese is used in the manufacturing of steel to provide added strength and hardness (Browning, 1969). It is an essential nutritional element for humans and plants (Rudd, 1987). Colour/staining problems may arise where the concentration in water exceeds 50g (Kempster and Smith, 1985). Excessive concentrations of this element result in cramps, tremors, hallucinations, manganic pneumonia and renal degeneration in animals (Duffus, 1980).

7.2.3.1.2 Other inorganic compounds

Salination has been identified as one of the major degenerating aspects of groundwater quality, due to the leaching of the slag dumps and waste disposal points. This is aggravated by the deposition of neutralised acids on top of the solid waste heaps. The following elements contribute to the salt load and in certain instances to the toxicity of the water systems.

Fluoride

Fluorspar functions as a fluxing agent in the production of steel (Waldbott, 1973). The fluorspar is converted to fluoride during the manufacturing process and is usually emitted into the atmosphere. However, since this element is very soluble in water, it is considered to have a potential impact on groundwater quality.

Humans adsorb fluoride most easily from water. However, the presence of elements such as calcium, aluminium and magnesium will inhibit this process. People with normal kidneys may excrete up to 61% of the ingested fluoride through their urine.

The benefit of fluoride is its capability to prevent dental cavities when optimum concentrations exist in water (Benefield *et al.*, 1982). Fluoride that is present in small amounts decreases the incidence of dental caries (Kempster and Smith, 1985). However, the margin between beneficial and toxic amounts of fluoride is narrow.

The strong affinity of fluoride for magnesium, manganese and other metals results in the interference of the activity of enzymes. Fluoride also damages the pituitary gland, which regulates the water balance in the body. This results in excessive thirst and urination. Bluish-brown lesions, which simulate bruises on the skin, may also occur. This may be accompanied by muscular pains and gastro-intestinal upsets in the most affected cases (Waldbott, 1973).

7.2.3.1.3 Organic Compounds

Phenolic Compounds

Phenols are obtained from coal tar (Budavari*et al.*, 1989) and are a by-product of the metal manufacturing industry. These chemicals impart an unpleasant taste/odour to water (Kempster and Smith, 1985). Phenols are highly toxic as described by Budavari*et al.* (1989): "Ingestion of even small amounts may cause nausea, vomiting, circulatory collapse, paralysis, convulsions, coma, greenish or smokey-coloured urine, necrosis of the mouth and gastro-intestinal tract, death from respiratory failure and sometimes cardiac arrest. The average fatal dose is 15g, but death from as little as 1g has been reported. Chronic poisoning with renal and hepatic damage may occur from industrial contact."

It is evident from the above information that contaminants associated with the metal industry may be toxic in nature and could be detrimental to the environment. Species that are relatively non-toxic degenerate the water quality, by contributing to the total salt load. The non-biodegradability of inorganic compounds results in a progressive accumulation of these elements within water systems. Although certain levels of concentrations may still be within the legal limit, the water quality will deteriorate where water is recycled through the manufacturing process.

7.2.4 MIGRATION OF CONTAMINANTS WITHIN THE GROUNDWATER SYSTEM

7.2.4.1 Dissolution of hazardous material

Trace metals, toxic salts and many toxic organics will dissolve on contact with water (Devinny *et al.*, 1990). Low pore water velocities and large surface areas of the waste particles enhance this process. The dissolution of heavy metals is also accelerated by the presence of acidic leachates. Alkaline solutions tend to precipitate heavy metals as hydroxides.

7.2.4.2 Precipitation

It is anticipated that the ameliorating effect of the receiving environment will raise the pH of the fluids with the coincidental precipitation of the toxic heavy metals. However, the assimilating capacity of the receiving environment is finite. A subsequent pH-decrease may remobilise the deposited heavy metals.

7.2.4.3 Adsorption

Heavy metals and other contaminants may become adsorbed onto clay, silt-sized particles or iron and manganese hydroxides. For trace or heavy metals, ion exchange forms the dominant method of adsorption. The pollutants will migrate rapidly once the available adsorption sites have been occupied.

An increase in salinity increases the number of species that compete for adsorption sites (Devinny *et al.*, 1990). Furthermore, the work of Bailey *et al.* (1986) showed that the presence of organics such as phenols decreases the adsorption of heavy metals.

Adsorption of metal cations is also determined by the pH of the leachate solution. The H^+ ions compete with the metal species for adsorption sites. Previously adsorbed metal ions may be remobilise under very acidic conditions. The retardation of the pollution plume by the processes of adsorption and precipitation may be a temporary phenomenon. Contaminants may be remobilised into the groundwater system if the leachate or receiving environment characteristics change.

7.3 IMPACT ON GROUNDWATER AND SURFACE WATER IN THE OLIFANTS CATCHMENT

Observations at three metal processing sites within the Olifants Catchment (six boreholes), suggest that seepage and migration of groundwater pollution plumes occur predominantly within the weathered lithologies. This shallow groundwater discharges into the nearest surface water system. The rocks underlying the weathered sediments tend to be relatively impermeable. However, there may be site specific areas where hydraulic continuity exists between the weathered zone and the underlying fractured aquifers. Pollution may migrate along faults or the contacts of igneous intrusions to deeper levels.

Certain areas, such as the Ferrobank industrial site in Witbank, are heavily polluted. Clearly much work will be necessary to rid this area of groundwater pollution.

Most industries are in the process of conducting environmental investigations. In certain instances, significant capital is being spent to improve disposal facilities to meet international standards. It is uncertain to what degree the pollution potential from existing and future sites can be controlled and improved under present South African legislation.

7.4 CONCLUSIONS

The following conclusions are drawn from the above observations:

- Metal processing facilities have polluted the local groundwater systems in the Olifants Catchment. The contaminants include toxic heavy metals as well as inorganic compounds which increase the salinity of the water systems. One of the concerns is the disposal of phenols in association with slag dumps. This has resulted in a noticeable increase in the concentrations of toxic organic compounds in the groundwater.
- Environmental impact assessments have been conducted by various institutions to define the extent and severity of the pollution. They have presented recommendations for the prevention and amelioration of groundwater pollution. The following is a list of remedial actions that are generally recommended:
    - The separation of solid and liquid waste disposal.
    - The separation of hazardous and non-hazardous waste.
    - The separation of the various water qualities which are generated in the plant.
    - The reduction in the area of the active working face at the slag dumps to prevent the infiltration of rainfall.
    - The reduction of the quantity of waste through recycling processes.
    - The reduction in the use of fresh water within the manufacturing process.
    - Provision for rehabilitation and long-term management of waste sites.
- Although the metal industries do pollute the water systems, the impact will be restricted to areas between waste facilities and the nearest streams. It is doubtful whether they can achieve zero effluent disposal. These facilities will therefore have a cumulative effect on the salt load in the Olifants Catchment.
- Monitoring according to the recommendations of the DWA&F (1994).

Section 8

AGRICULTURE AND ITS IMPACT ON GROUNDWATER QUALITY

8.1 INTRODUCTION

The Eastern Transvaal Highveld is one of the most fertile agricultural regions in the country (Wates, Meiring and Barnard, 1993). Maize from this area constitutes a significant proportion of the national staple food supply.

Fertilisers and chemicals are applied to maintain the current high crop yields. However, the intensive use of such chemicals ultimately may result in the degradation of groundwater quality.

8.2 THEORETICAL CHEMICAL CONSIDERATIONS CONCERNING THE AGRICULTURAL SECTOR

Several books discuss the environmental impact of agricultural activities (Richardson, 1991; Briggs and Courtney, 1989; Loehr, 1977). These effects are more evident with the progressive intensification of farming methods.

Wates, Meiring and Barnard (1993) identify the phosphate and nitrogen contribution of the agricultural sector within the Witbank Dam Catchment. These figures give a good indication of the impact of agricultural activities within the study area.

The impact of pesticides has been addressed extensively in overseas literature (Gunn and Stevens, 1979; OECD, 1986; Dinauer, 1974). The only South African study of the impact of pesticides on groundwater, which has been undertaken to the authors' knowledge, is the work of Weaver (1993).

Various computer programs have been developed to simulate the movement and persistence of pesticides in soils (Pennell *et al.*, 1990). However, their description and use are beyond the scope of this study.

8.2.1 AGRICULTURAL SOURCES OF NUTRIENTS THAT ACCELERATE EUTROPHICATION OF WATER SYSTEMS

Nitrogen and phosphorous accelerate the eutrophication of water systems by acting as nutrients for aquatic plant growth. According to Mason (1991), these nutrients are mobilised from the agricultural sector in one of three ways:

- By the percolation of water through the soil thereby leaching the soluble plant nutrients.
- By inefficient disposal of livestock excreta.
- By the erosion of surface soils or by the movement of fine soil particles into subsoil drainage systems.

8.2.1.1 Fertilisers

The development of high-yielding crops has led to an increase in the plant nutrient and water requirements (Pereira and Quelhas dos Santos, 1991). These quantities often exceed the natural availability of the constituents in the soil. An artificial supply of water and nutrients is therefore required in the form of irrigation and fertiliser applications. To ensure high crop yields, the fertiliser applications are usually in excess of the plant requirements. The surplus nutrients are prone to leaching by percolating soil waters. These processes are accentuated when heavy fertiliser treatment takes place on a heavy wet soil in periods of heavy rainfall (Dix, 1981).

The two main elements in artificial fertiliser applications are phosphorous and nitrogen.

8.2.1.1.1 Phosphatic Fertilisers

The efficiency of nitrogen fixing crops depends on the adequate supply of phosphate. Biological phosphate exists at a higher free-energy level than phosphatic minerals within the majority of geological formations and unfertilised soils (Taylor and Kilmer, 1980). According to these authors, the energy requirements for the assimilation of phosphate by plants are derived from photosynthesis. Without this energy, the quantity of available high-energy phosphates declines through the processes of leaching and adsorption.

Fertilisers are required to maintain the biological productivity in areas that are depleted in biological phosphate. Natural phosphate rock is relatively insoluble and is unsatisfactory as a fertiliser except for more acidic soils. However, treatment of this material with sulphuric acid under controlled conditions gives $Ca(H_2PO_4)_2$ which is more soluble in water. The stability range of the $H_2PO_4^-$ ion is restricted and phosphate fertilisers are therefore most efficient over a pH-range of 6 - 7 (Fergusson, 1982). Other phosphate ions tend to form insoluble iron, aluminium and calcium salts which are easily adsorbed by soil or organic particles. The relative distribution of phosphatic ions with respect to pH is shown in Figure 8.1.

It can be seen from the above that any long-term increase in fertility with respect to phosphate may only be achieved through successive fertilisations. The physical limitations with regard to the solubility of phosphate mean that the use of these fertilisers is often expensive and wasteful. However, there is no toxicological reason to define a drinking limit for inorganic phosphate (Kempster and Smith, 1985).

8.2.1.1.2 Nitrogen Fertilisers

Nitrogen is a vital element for the formation of amino acids from which proteins are formed (Fergusson, 1982). The potential for nitrogen leaching in the soil depends on the fertiliser type. Leaching potentials of the various forms of fertiliser are shown in Figure 8.2.

Nitrogen is most readily assimilated as the nitrate ion by plants. Farmers therefore tend to apply nitrogen in this form at peak growing times. However, nitrates are very soluble in water and are not retained within the chemical complex of the soil (Pereira and Quelhas dos Santos, 1991). This has led to the world-wide pollution of groundwater systems.

Although the most significant transformation of nitrogen compounds within a water source is the reduction of nitrate, it is thought that these denitrification processes are negligible in aquifers (OECD, 1986). Exceptions are shallow aquifers that have high concentrations of organic carbon or ferrous iron and are depleted in oxygen. A discussion on the denitrification processes is beyond the scope of this report.

The recommended level of nitrate in drinking water is 6 mg/ℓ with a maximum permissible level of 10 mg/ℓ (Kempster and Smith, 1985). The nitrate level should not be too high because of the risk of the reduction to nitrite in the less acidic stomachs of young children. This results in the development of methamoglobinemia where the NO_2^- oxidises the ferrous ion in the haemoglobin to the ferric iron as well as co-ordinating with the metal (Fergusson, 1982).

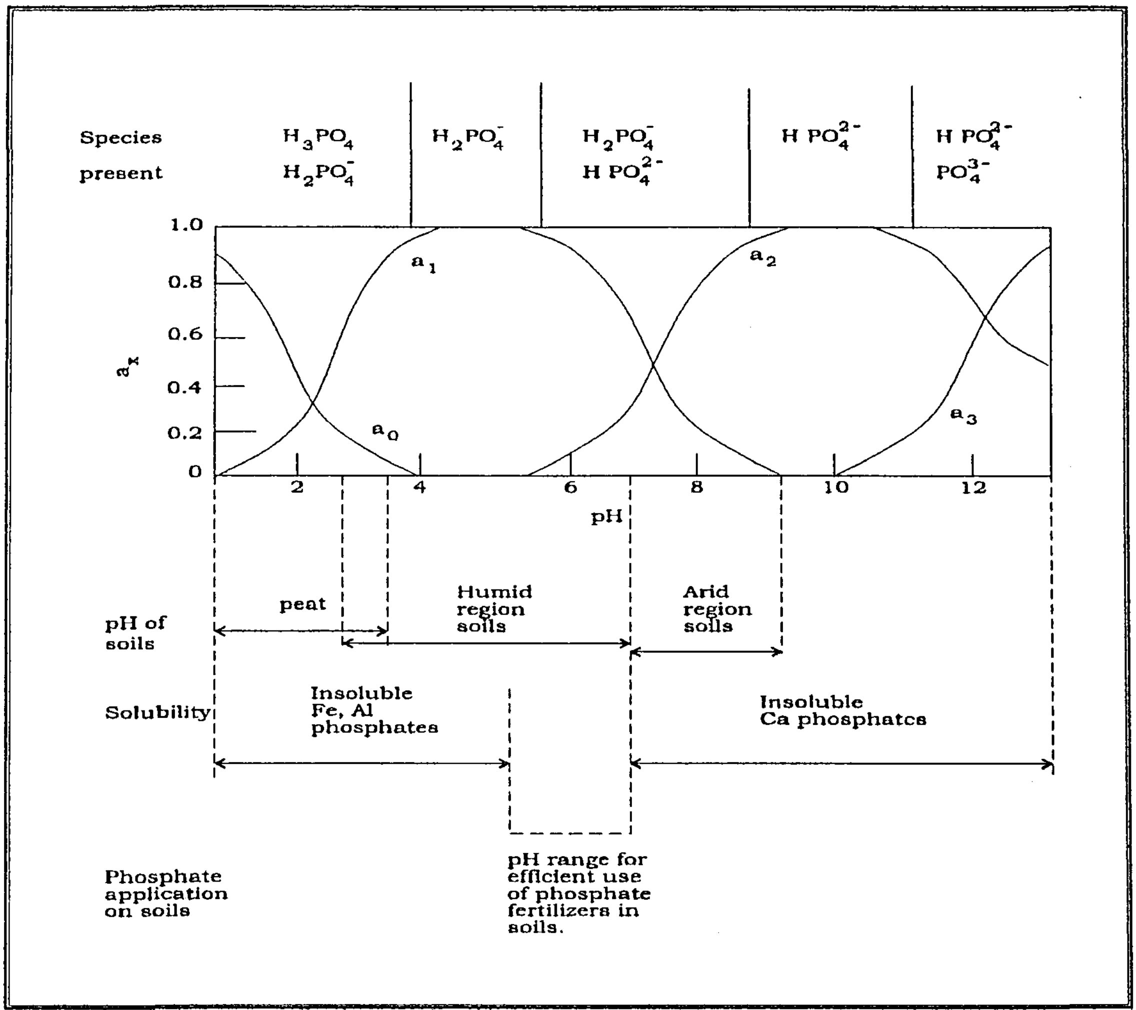


Figure 8.1. Distribution diagram for phosphatic species and their solubilities in relation to the soil pH (after Fergusson, 1982).

8.2.1.2 Feedlots

Free-ranging or outdoor methods of livestock farming have been shown to be inefficient as far as the growth and weight increases of cattle are concerned. Animals are a production unit in the modern agricultural system (Dix, 1981). This commercial approach has led to the use of confinements or feedlots where the temperature, feeding and animal health conditions are more strictly controlled.

Intensive farming requires a high standard of hygiene and therefore involves regular washing and hosing down of materials. This produces a high proportion of liquid waste that consists essentially of urine and excreta. The liquid contains high nitrates, high phosphates and some organic pesticides that may have been present in the original cattle feed. When this slurry is disposed of, particularly in the wet season when the soil moisture content is high, the contaminants are readily transported as surface run-off or percolate into the groundwater. This presents a serious threat to the environment that may result in the eutrophication of water supplies and health risks to the local water users. Undiluted farm slurry may be 80 - 100 times more contaminated than that of untreated sewage sludge (Mason, 1991).

A comparison between selected feedlot effluents from the literature and the raw sewage at a typical sewage works within the Olifants Catchment is made in Table 8.1. Although feedlot slurry may contain phosphate concentrations that are similar to that of fertilised soil, the majority of the species in the slurry may be in the water-soluble form.

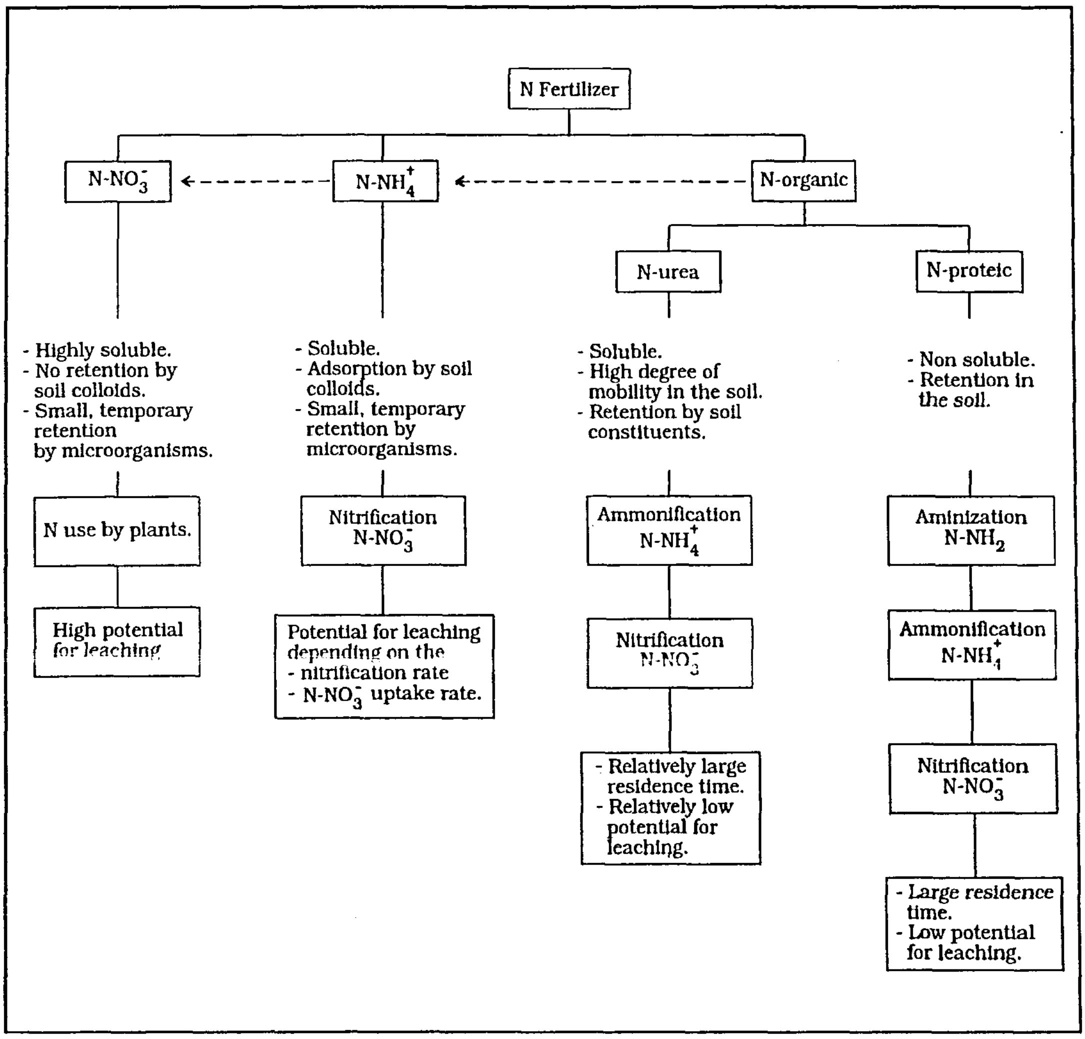


Figure 8.2. Potential for nitrogen leaching in relation to the type of fertiliser (after Pereira and Quelhas dos Santos, 1991).

Table 8.1. A comparison of selected feedlot effluents with raw sewage.

AUTHOR	Hegg et al. (1983)	Young et al.(1980)	Cummings and Jewell (1977)	Booram (1977)
EFFLUENT SOURCE	Cattle feedlot run-off	Cattle Feedlot	Cattle Feedlot	Swine Feedlot
pH	6.5 - 7.2	***	7.3 - 8.6	7.3
N-NO3 mg/l	0.2 - 10	4	***	***
Kjeldahl - N mg/l	10990	41	3400 - 5500	462
N-NH4 mg/l	35440	5	1600 - 2100	434
Tot P mg/l	18264	23	***	***
P PO4 mg/l	***	***	***	69
Cl mg/l	***	***	***	343
COD mg/l	30 - 1400	***	42900 - 57700	1177

The feedlot effluent quality varies considerably between the various studies. This is obviously related to the site-specific factors such as the quantity of animals and sampling points. It is evident from the table that the feedlot effluents are generally more contaminated than raw sewage effluent. However, it should be pointed out that the estimates of Mason (1991), as described above, appear to be somewhat exaggerated.

Silage production, which also generates food for the cattle during the lean months, may also generate effluent that is characterised by a high organic content and a low pH. This effluent may be considered to be 100 times more polluted than any other form of farm waste (Dix, 1981).

8.2.2 PESTICIDES

The term pesticide usually includes herbicides, insecticides and fungicides, most of which are chemical products. The chemicals that are used to supply plant nutrients are not normally included in this category.

Pesticides have made it possible to practise intensive agriculture. However, the application of these chemicals is generally inefficient. Only 10% of pesticide dust will settle on the plant and only 20% of liquid pesticides will attach to the plant surface (Li, 1991). The remaining 30 - 60% of the pesticides will fall onto the ground where it is available for leaching into the groundwater. These chemicals therefore represent a potential threat to the environment.

Pesticides may be subdivided into chemicals that fulfil specific functions. These are as follows:

- Herbicides to eliminate unwanted plants.
- Insecticides to eradicate pests which destroy crops.
- Fungicides to prevent crop deterioration through the presence of fungi.

These various categories will be discussed in more detail.

8.2.2.1 Herbicides

Herbicides have enabled the farmer to practise monoculture and to chemically weed his crops. However, as indicated previously, the majority of a pesticide application ultimately settles on the ground. Soil-acting herbicides therefore tend to be the most efficient weed-killer.

The ideal soil-acting herbicide is one that persists in the soil for a long period of time until the crop can compete with weeds. Ideally, at this point, the chemical should disappear through the process of degradation. The disadvantage of this process is that the weeds rapidly develop resistance to this form of herbicide. This necessitates the rapid development of new brands.

The most commonly used herbicides in the Olifants Catchment may be subdivided into the following categories as defined by Duffus (1980):

8.2.2.1.1 Photosynthetic inhibitors

About 45% of all herbicides may be classified as photosynthetic inhibitors. These chemicals block vital stages in the photosynthetic process resulting in plant death from a lack of synthesised carbohydrate (Dix, 1981). The response of the plants to these herbicides, which include the triazines, is fairly slow.

Simazine and, to a certain extent, atrazine are soil-acting herbicides which are primarily absorbed by the root system. These chemicals have been known to persist in the environment for a year.

8.2.2.1.2 Uncouplers of respiration

Bromoxynil uncouples oxidative phosphorylation. There is a good correlation between the uncoupling power and the herbicidal efficiency. The production of adenosine triphosphate by oxidative phosphorylation is the main source of energy in all aerobic organisms. The interference of this process means that this type of herbicide can kill most living things. Human fatalities have been caused by the misuse of this chemical.

8.2.2.1.3 Growth regulators

The herbicides 2,4-D and MCPA fall into this category. These herbicides cause rapid, distorted growth resulting in eventual plant death (Dix, 1981).

The 2,4-D and MCPA herbicides are water soluble and consequently migrate easily through the soil into groundwater (OECD, 1986).

Most herbicides affect the metabolic pathways in plants that do not exist in animals (Streibig, 1991). Although the toxicity of herbicides in animals is low, they do have some effects on these living organisms. Cases are reported where people have died as a result of excessive exposure to 2,4-D (Duffus, 1980). The limit for herbicides in water in the European Community is in the order of 0,1μg/l (Funari *et al.*, 1991).

8.2.2.2 Insecticides

An insect becomes a pest when it appears in such quantities that it has an adverse effect on the quantity and quality of the crop. Other animals may be harmful because they feed on stored food or because they act as parasites on domestic animals or transmit diseases. The discussion on insecticides will primarily deal with plant applications.

Plants develop defence mechanisms against insects that, in turn, adapt their behaviour to accommodate these different sets of circumstances. Insects may therefore only attack young parts of the plant or alternatively feed during a specific time in the plant's cycle. Each plant has specific enemies. In natural environments, the plant consumes less than 1% of the annual biomass production, whereas in agriculture these estimates rise to 5% - 30% (Eilenberg *et al.*, 1991). It is therefore necessary to use insecticides to avoid excessive crop losses.

Modern insecticides have moved away from the early organochlorines since they are extremely detrimental to animal life for the following reasons:

- They are chemically stable.
- They dissolve in the body fat of vertebrates.
- They can become dispersed over wide areas through the bodies of migrant animals.

When insecticides are tested, their aim is to eradicate 95 - 100% of the insect population to provide the best yield (Eilenberg *et al.*, 1991.) If less than 95% of the insects are killed, it is often necessary to repeat the treatment. Resistance occurs when some insects which survive a first encounter with the pesticide, pass on a decreased sensitivity to the chemical to the next generation. Pesticides that have long residual times also promote resistance among the target insects.

Pyrethroids are commonly used insecticides in the study area.

8.2.2.2.1 Pyrethroids

Pyrethroids have a low toxicity to animals because they are rapidly metabolised to harmless substances. However, they can cause allergic dermatitis and systematic allergic reactions. Large amounts may cause nausea, vomiting, headaches and other disturbances of the central nervous system (Duffus, 1980).

8.2.2.3 Fungicides

Fungicides are either applied to seeds, plants or soil. A major problem with the use of fungicides is that the fungi tend to become resistant. Rapid research is therefore necessary to keep pace with the development of immunity (Briggs and Courtney, 1989).

8.2.3 RETARDATION FACTORS THAT AFFECT THE MOBILITY OF AGROCHEMICALS

According to Funerary *et al.* (1991), the ability of agrochemicals to reach groundwater systems is determined by the following factors:

- Textual, mineralogical, microbiological and physico-chemical properties of the topsoil, which determine the degree of degradation of the substances and also represent a first barrier to infiltrating water.
- Soil and aquifer properties, particularly the hydraulic conductivity, which will control the percolation of the contaminants into the groundwater.
- Depth to the saturated zone, which will increase the assimilation of the pollutants before they contaminate groundwater.
- Availability of water to transport the contaminants.

Experimental work has been undertaken to quantitatively determine the characteristics that will identify a particular element or compound as a contaminant. For example, the US-EPA (1987) has observed that organic contaminants in groundwater generally have:

- A solubility in water greater than 30 ppm.
- K_d (a measure of the compounds adsorption to soil) value of <5 and usually <1.
- K_{oc} (a measure of the compounds adsorption to organic phases) value of < 300-500.
- *KH* (a measure of the compound to disperse into the air) value of $<10^{-2}$ atm m^3/mol.
- A $t_{1/2}$ soil or dissipation half-life (a measure of the compounds persistence in soil) >3 weeks.

The above factors affect the retardation of contaminants and will be discussed in more detail.

8.2.3.1 Solubility

Solubility is often the controlling factor that determines the mobilisation of contaminants into the groundwater system. For example, nitrates tend to be more soluble than phosphate compounds and are therefore more readily leached from the soil profile. The following factors will determine the degree of solubility and leaching of agrochemicals:

- Water volume.
- Residence time of water within the soil.
- Pore-size distribution (preferential pathways).
- Quantity of leachable chemicals within the soil.

It is important to note that chemicals may be leached from the soil profile before the onset of adsorption or degradation processes during severe rainfall events.

The soil moisture content has an effect on the solubility of compounds. For example, phosphates are reduced to more soluble compounds under anaerobic conditions that are associated with high groundwater level or soil moisture content. Pesticides are also easily removed from the soil once they are reduced into water-soluble compounds.

8.2.3.2 Adsorption

Contaminant migration in water is inhibited by the assimilating capability of the soil. An important characteristic in this regard is the cation exchange capacity. This depends on the structure of the alumina-silicate layers within the individual soil fragments. For example, montmorillonite clays have a greater adsorption capacity than kaolinite. Further details of the clay adsorption mechanisms are discussed in the opencast mining section of this document.

Iron and aluminium compounds are important in the sorption of phosphates. Negatively charged pesticide compounds also tend to be sorbed by strongly hydrated iron and aluminium oxides at low soil pH-values. Nitrates, on the other hand, are held by organic processes.

According to Briggs and Courtney (1989), the best sorption of fertilisers and other agrochemicals is achieved in a clay or peat soil of moderate to high pH.

Speciation also plays an important role in the adsorption tendency of chemical compounds. Negatively charged compounds at the relevant soil pH are unlikely to be adsorbed as a result of the predominantly negative surface charge of soils or clays.

The capacity of organic material to adsorb negatively charged pesticide compounds is dependent on pH (Aharonson, 1987). According to this author, as the pH of the soil is lowered, the negatively charged compounds act more like neutral molecules. This results in a greater sorption capacity of the organic material. Chemicals such as 2,4-D act in this manner.

8.2.3.3 Degradation

The term half-life is used to describe the persistence of chemicals within soil. After four half-lives, about 95% of the compound is decomposed and the chemical no longer has an effect on the environment. The length of the half-life is proportional to the resistance of the agrochemical to degradation.

Degradation represents a series of processes leading to the formation of simpler organic molecules which normally end up as CO_2, salts and water. For example, the degradation of 2,4-D is rapid and remains as a residue of CO_2, water, chlorine and organic matter. Pesticide molecules that only partly decompose will result in the formation of residues which may be adsorbed within the soil and organic particles. The persistence of a pesticide may vary considerably from one environment to another. Decomposition rates tend to be slower under cold, dry or acidic conditions. The concept behind degradation is illustrated in the Figure 8.3.

Pesticides that are exposed on soil and plant surfaces will be degraded by ultraviolet light. Once the chemical constituent passes into the soil, it is subjected to abiotic and biotic degradation processes. Abiotic degradation processes include hydrolysis and oxidation-reduction reactions (Aharonson, 1987). However, quantitatively, the most significant factor in the degradation of pesticides is the microbiological activity of fungi and bacteria within the soil structure (Helweg, 1991). The abiotic degradation mechanisms will therefore not be discussed in this report. Helweg (1991) estimated that a typical soil may contain up to 4 - 8 tons of fungi and bacteria in the ploughed layer of one hectare.

Helweg (1991) describes two processes of microbiological degradation of pesticides (Figure 8.4).

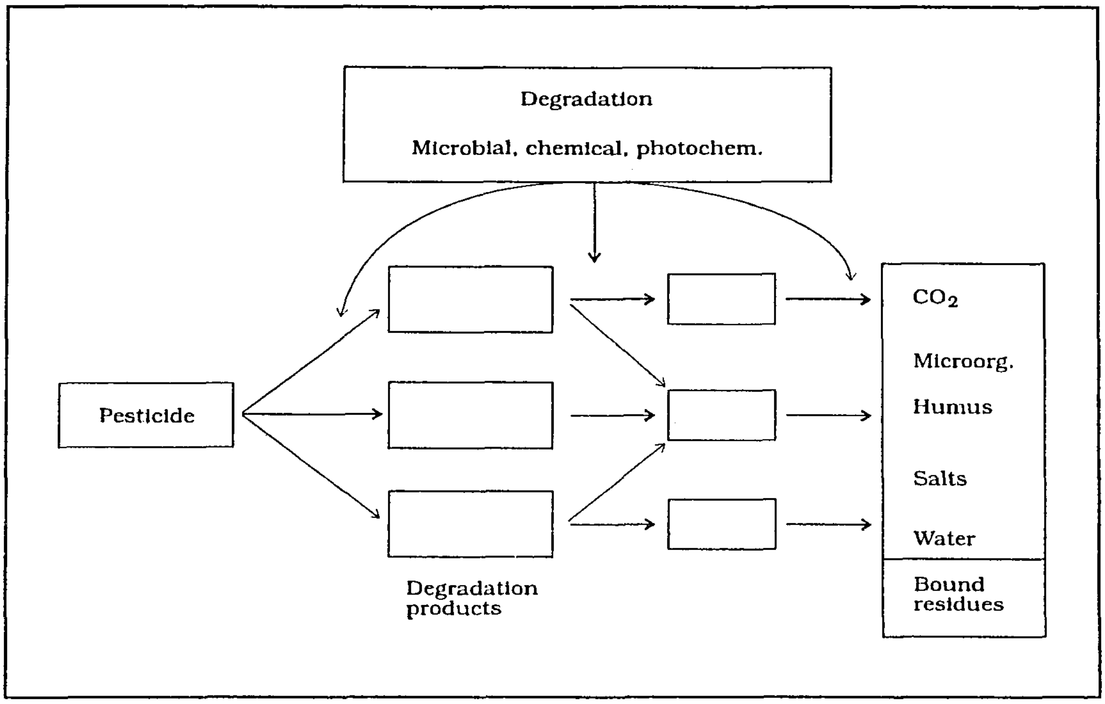


Figure 8.3. Degradation of pesticide molecules into more stable products (after Helweg, 1991).

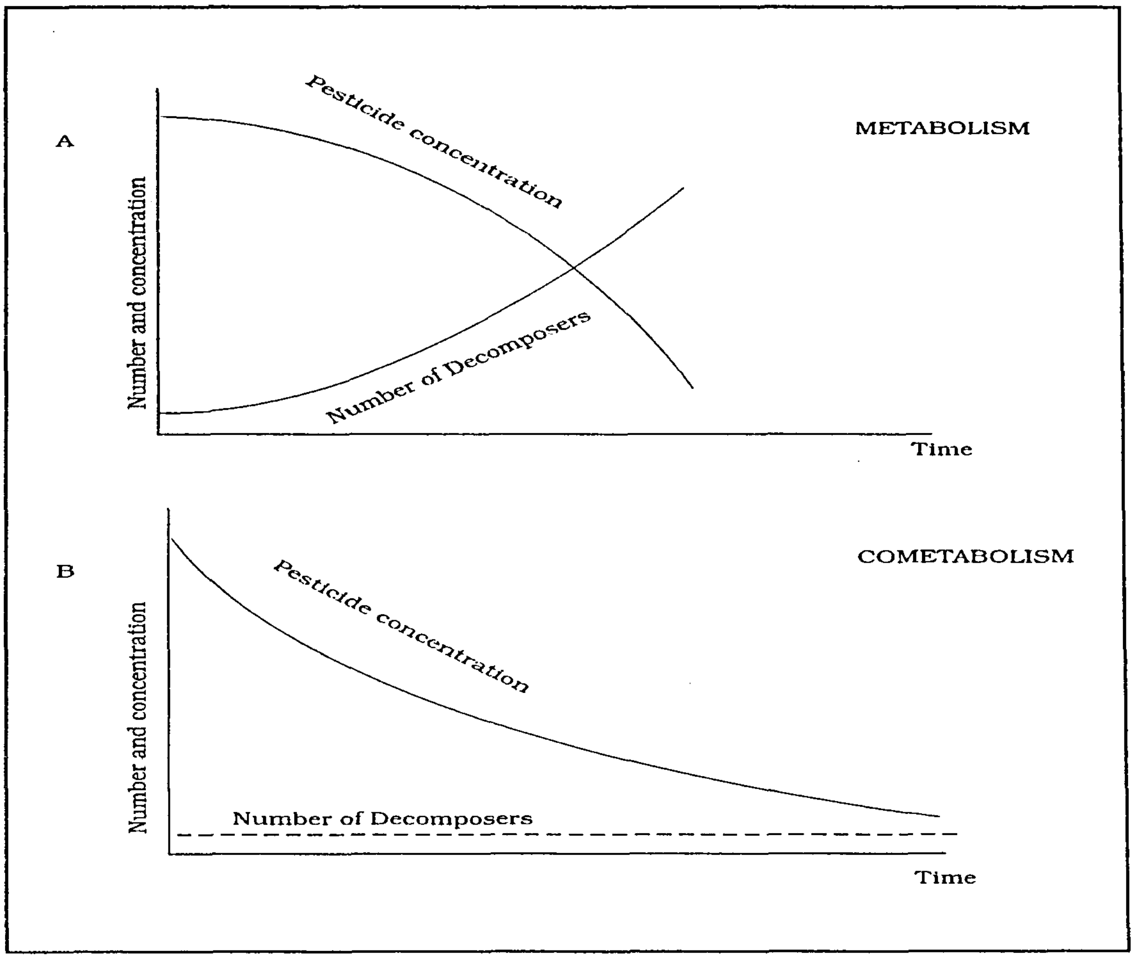


Figure 8.4 Degradation of pesticides by (a) metabolism and (b) cometabolism (after Torstensson 1977 in Helweg 1991).

These reactions are as follows:

- *Metabolic reactions.* Initially, the rate of degradation is slow due to the limited number of micro-organisms. However, the micro-organisms multiply rapidly as they feed on the pesticide. This results in the acceleration of the degradation process. The phenoxyherbicides MCPA and 2,4-D are decomposed in this manner. Soil enrichment with respect to the micro-organisms may last several years, since the fungi and bacteria have the capacity to survive in a dormant state.
- *Cometabolic reactions.* During these reactions, the number of decomposers is not affected by the presence of the pesticide, since micro-organisms survive on other substances in the soil. The rate of degradation is fastest in the initial stages. Once the pesticide concentration drops, the probability of contact between the decomposers and the pesticide decreases. The rate of degradation therefore tapers off.

The degradation of a great number of pesticides seems to follow the cometabolism reactions. However, only free unadsorbed compounds appear to be degraded by biotic activity (Aharonson, 1987).

Micro-organisms are primarily concentrated within the surficial sediments (Table 8.2). This effectively means that biodegradation is limited to the topsoil (Table 8.3). The subsoil lacks the nutrients to sustain a significant population of bacteria and fungi. If the pesticides are rapidly flushed from the upper portions of the soil profile, it appears unlikely that they will be subjected to further micro-organism attack. The thickness of the soil, which may support a significant microbiological population, will vary from location to location.

Table 8.2. Relative bacterial and fungal concentrations within the ploughed layer and subsoil (after Helweg, 1991).

SUBSTRATE	Bacteria (millions per g)	Fungi (millions per g)
Ploughed layer	5000 - 10000	200 - 2000
Subsurface	1 - 10	very few

Table 8.3. Degradation of ^{14}C labelled MCPA and TCA within surficial and subsoil sediments (after Helweg, 1991). The figures shown in the table represent the number of days until 5% of ^{14}C has been liberated in the form of CO_2.

PESTICIDES	Sandy clay (surface)	Coarse sand (subsoil)
MCPA	11 days	100 days
TCA	16 days	300 days

8.3 SOURCES OF POLLUTION WITHIN THE OLIFANTS CATCHMENT

8.3.1 SOURCES OF NITROGEN AND PHOSPHATE

It is important to establish the relative nitrogen and phosphate contributions of the various activities within the Olifants Catchment. For this purpose, the figures for the Witbank Dam Catchment which were compiled by Wates, Meiring and Barnard (1993) are presented in Table 8.4.

Although the figures for the nutrients in the table below are listed for the Witbank Dam Catchment, they give a good indication of the relative proportion of the nutrients that are generated by the various sectors in the Olifants Catchment Area.

Table 8.4 Plant Nutrient Discharges to the Catchment Upstream of the Witbank Dam (after Wates, Meiring and Barnard, 1993).

Plant Nutrient Sources	Load (t P/yr)	% (P/yr)	Load (t N/yr)	% (N/yr)
Natural anthropogenic weathering and agriculture	40.3	53.5	22.3	18.8
Municipal sewage treatment plants	28.9	38.4	54.9	46.4
Power station effluents	0.3	0.4	1.6	1.4
Coal-mining operations	5.8	7.7	39.5	33.4
TOTAL	75.2		118.3	

It is evident from the table that the agricultural sector is an important generator of phosphatic nutrients. On the other hand, the nitrogen that is generated by agricultural activities is relatively insignificant, in comparison to the other sectors within the Olifants Catchment. According to Wates, Meiring and Barnard (1993), the considerable nitrogen load from the coal mining industry is generated by the rehabilitation of old areas and the use of explosives.

Phosphate tends to form insoluble salts within the soil horizon. These salts are adsorbed onto the soil particles. Nitrates, on the other hand, are readily dissolved in water and are therefore leached into water systems.

Several studies have identified feedlots as a major source of pollution. The scale of the problem in South Africa is illustrated by the example used by Pieterse *et al.* (1982). The largest cattle feedlot system at that time in South Africa contained approximately 30 000 animals. Feedlot effluent generation in this confinement is equivalent to the waste produced by a city of 180 000 to 300 000 people. Treatment of sewage is required by a city of 300 000 people. However, similar legislation does not exist for feedlots.

Although the quantity of phosphate in faeces is approximately equivalent to that in fertile soil, the greater fraction in the faeces may be in water-soluble forms (Taylor, 1967).

In terms of groundwater, neither of the two compounds is present in noticeable quantities (Figure 8.5). Phosphate is not presented in this diagram because it is usually present in concentrations below 1 mg/*l*. The conclusion is drawn that groundwater contamination as a result of fertiliser application is almost non-existing in the Olifants Catchment. This is attributed to the dynamic throughflow nature of the upper weathered aquifer. It can, for instance, be demonstrated that for an annual groundwater throughflow of 3,6 Mm^3 in the Witbank Catchment, the nitrogen contribution to the Witbank Dam is in the order of 1,7 tons per annum.

8.3.2 SOURCES OF AGROCHEMICALS

A selection of the more common chemicals is presented in Table 8.5.

Table 8.5. A selection of pesticides which is commonly in use in the Olifants Catchment. Application rates are from personal communication with Hartwanger (1993).

PRODUCT NAME	CHEMICAL FUNCTION	ACTIVE INGREDIENT	CONC. OF ACTIVE INGREDIENT	APPLICATION RATE
Galleon	Herbicide	Sulcotrione	125 g/*l*	600 ml/ha
		Atrazine	291 g/*l*	
		Other related triazines	9 g/*l*	
Terbo	Herbicide	Bromoxynil	150 g/*l*	1250 ml/ha
		Terbuthylazine	333 g/*l*	
Dual S	Herbicide	Metolachlor		1000 ml/ha
Karate	Insecticide	Pyrethroid	50 g/*l*	100 - 250 ml/ha
Mancozeb	Fungicide	Dithiocarbamate	800 g/kg	2 - 3 kg/ha
Score	Fungicide	Difenoconazole	250 g/*l*	300 ml/ha

Crop spraying in the Olifants Catchment is most active during December, January and February (personal communication, Hartwanger, 1993). It is anticipated that this list is not complete due to site-specific applications and preferences.

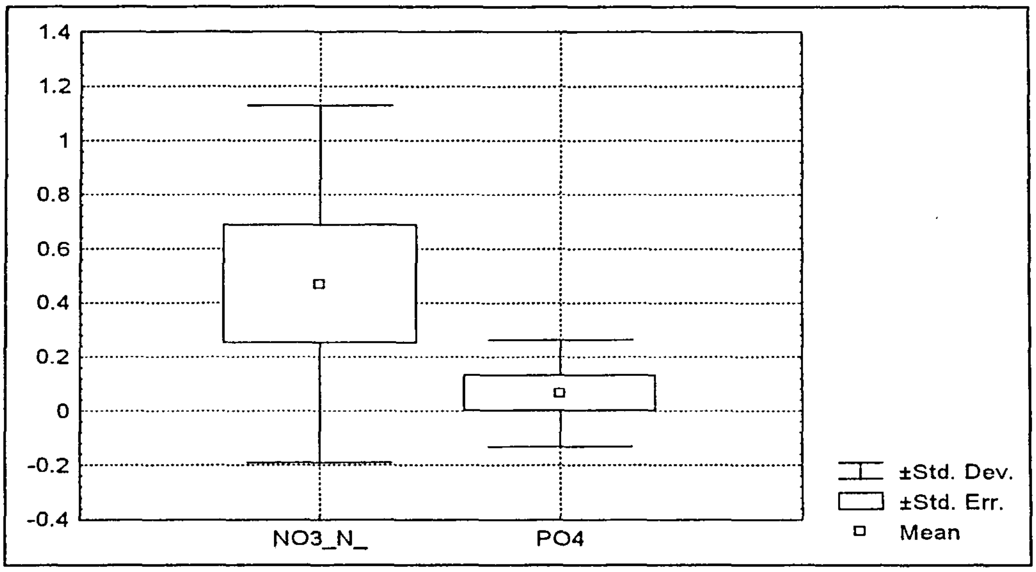


Figure 8.5. Box and whisker plot for nitrate (mg/l N) and phosphate (mg/l) in groundwater samples from the weathered aquifer.

8.4 MIGRATION OF THE POLLUTANTS THROUGH THE CATCHMENT

Contamination from the agricultural sector that is not transported by surface run-off, may be leached by percolating soil water into the groundwater systems. Groundwater flow occurs primarily within the shallow weathered Karoo aquifer that discharges into local streams.

The natural water/lithological interaction assimilates much of these pollutants. Wates, Meiring and Barnard (1993) have estimated that the catchment may assimilate 68% of the phosphorous and 58% of the nitrogen phase. It should be emphasised that these figures relate to surface waters. Groundwater values may differ significantly, especially where reducing conditions produce compounds that readily migrate in aqueous solutions. The effect of the actual export for the Witbank Dam Catchment is summarised in Table 8.6.

Table 8.6 Observed catchment discharges and export of phosphorous and nitrogen to Witbank Dam (after Wates, Meiring and Barnard, 1993).

	Total Phosphorous	Total Nitrogen
Catchment Sources	75.2 t P/yr	118.26 t N/yr
Catchment export to Witbank Dam	23.95 t P/yr	49.17 t N/yr

It can be seen from the above table that most of the phosphorous has either been assimilated or precipitated within the environment.

Nielsen (1991) established that streams had higher concentrations of phosphates and nitrates than lakes or reservoirs. The author attributes this to the loss of nitrogen and phosphorous by the processes of sedimentation and denitrification within standing bodies of water. It is likely that these processes also operate within the Olifants Catchment.

As discussed previously, the mobility of pesticides depends primarily on the solubility and persistence of the chemicals. The available data on chemicals that are used in the study area are presented in Table 8.7. Chemicals, for which data are unavailable, have been omitted from the table.

Table 8.7. Solubility and degradation rates for selected commonly used pesticides in the study area (after the Royal Society of Chemistry, 1987).

COMMON NAME	SOLUBILITY (mg/l at 20 °C)	DEGRADATION RATE (soil half-life)
Atrazine	28.0	5-7 months
Propazine	8.6	***
Simazine	3.5	***
Terbuthylazine	8.5	***
2,4-D	620.0	6 weeks
MCPA	825	3-4 months
Lamda-cyhalothrin	0	4-12 weeks
Mancozeb	0	***
Metalochlor	530.0	***

Although some of the chemicals may be persistent, it is evident from their solubility that they will not be transported into the water systems. On the other hand, other chemicals which are water-soluble are less persistent. These pesticides are therefore unlikely to act as contaminants in water systems.

It is possible to apply the criteria of the US-EPA (1987) (see Section 8.2.3), to determine which chemicals will act as contaminants in the groundwater systems. The chemicals that have solubility greater than 30 ppm and a soil half-life greater than three weeks, are Atrazine, 2,4-D, MCPA and Metolachlor. It is therefore anticipated that these pesticides will be present as contaminants in the water systems.

8.5 IMPACTS OF CONTAMINANTS ON THE WATER QUALITY WITHIN THE OLIFANTS CATCHMENT

8.5.1 PHOSPHOROUS AND NITROGEN NUTRIENTS

According to Taylor and Kilmer (1980), the normal concentrations of phosphate within groundwater may be in the order of 0,5 ppm (mg/l). Exceptions do exist where the waters flow through phosphatic rich lithologies or areas of decaying organic matter. This may be compared to surface water samples within the Olifants Catchment as shown in Table 8.8.

Table 8.8. Phosphate and nitrate concentrations within the Olifants Catchment, during January 1994.

Sample Number	Nitrates (mg/l)	Phosphates (mg/l)
PC1	0.4	1.2
PC2	0.3	0.3
PC3	0.5	0.5
PC4	1.1	0.7
PC5	0.1	0.4
PC6	1	1
PC7	0	<0.1
PC8	0	0
PC9	0.1	<0.1
PC10	0.1	<0.1

It is evident that the phosphate and nitrate concentrations at the time of sampling are low.

Phosphorous concentrations within freshwater lakes correlate with the amount of phytoplankton which is present in the water (Fergusson, 1982). Nielsen (1991) postulates that phosphate is the limiting factor of aquatic growth. According to Wates, Meiring and Barnard (1993), this is also the case within the Witbank Dam Catchment. This accounts for the apparent absence of aquatic growth in the water systems at the time of sampling.

The Olifants River is characterised by higher nitrate and phosphate concentrations than the Witbank and Loskop Dams. This trend corresponds to that described by Nielsen (1991) (see Section 8.4). This trend is not apparent in the Klein Olifants River at the Middelburg Dam. It is felt that site specific conditions may have an effect in this area.

8.5.2 PESTICIDES

The world-wide pesticide concentrations are generally low in relation to the toxicity of the chemicals. Nonetheless, a limit of 0,1 µg/*l* has been established for most EC countries (Helweg, 1991). Table 8.9 shows the maximum concentration of pesticides in Swedish streams and USA groundwater samples.

Table 8.9. Maximum pesticide concentrations in Swedish streams and USA groundwater samples (after Helweg, 1991).

Pesticide	Swedish Streams (ppb)	SA groundwater samples (ppb)
Atrazine	1.0	3.0
Simazine	***	2.0
MCPA	3.0	***

Water samples were taken on three occasions from surface water sources and once from boreholes in the Olifants Catchment. A description of the sampling localities for surface waters is presented in Table 8.10. Figure 8.6 is a map showing all the sampling positions. Concentrations of selected pesticides within the water are presented in Figures 8.7 - 8.10. Pesticide analyses were limited by the available analytical facilities at the time of sampling. Potential contaminants such as metolachlor were therefore not included in the analysis schedule. The concentrations within the study area compare favourably with those in Table 8.9. However, it should be noted that these concentrations are generally in excess of the 0,1 µg/*l* limit within the European communities.

Table 8.10 Co-ordinates of surface water sampling positions.

SAMPLE NUMBER	SAMPLE DESCRIPTION	Y COORD	X COORD
1	Middelburg Dam Wall	-54840	2851450
2	Middelburg Dam Inflow	-60300	2857520
3	Olifants at Van Dyksdrift	-32350	2888450
4	Olifants at R544	-29350	2876650
5	Loskop Dam Wall	-36300	2811600
6	Klein Olifants at Slaghoek	-18750	2829500
7	Olifants at Slaghoek	-18650	2829400
8	Witbank Dam Wall	-30400	2864850
9	Wilge River	-9800	2829500
10	Bronkhorst Spruit near Premier Mine Dam	17880	2855600
11	Bronkhorst Spruit Dam	27850	2864000
12	Botshabela	-41100	2843200
13	Olifants Hotel	-31350	2850000
14	Loskop Dam Inflow	-27800	2816500

Note: Three sampling episodes have been performed and they have the suffixes B, C and D in Figures 8.7 - 8.9, denoting different sampling episodes.

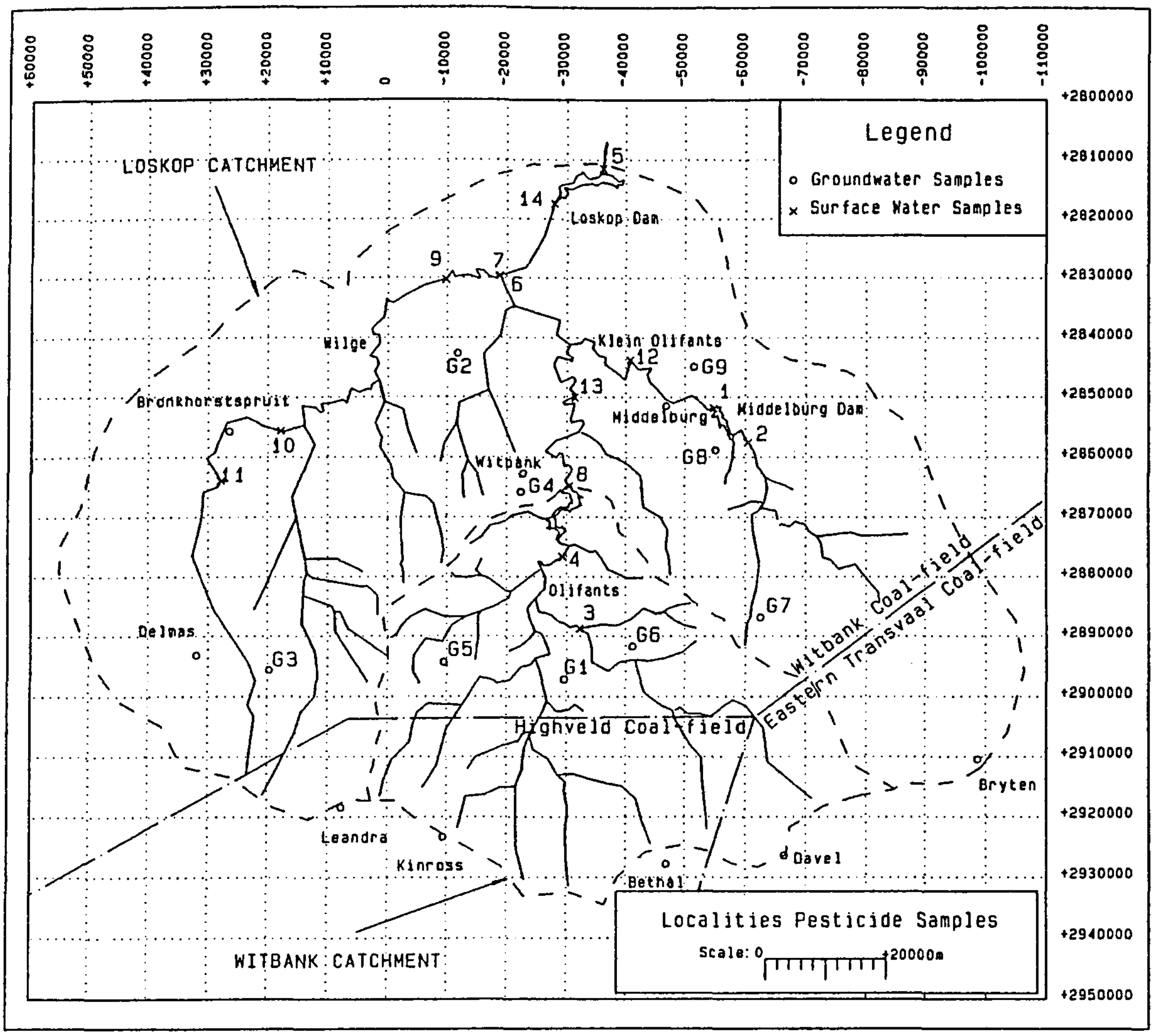


Figure 8.6. Sampling localities of water for pesticide analyses.

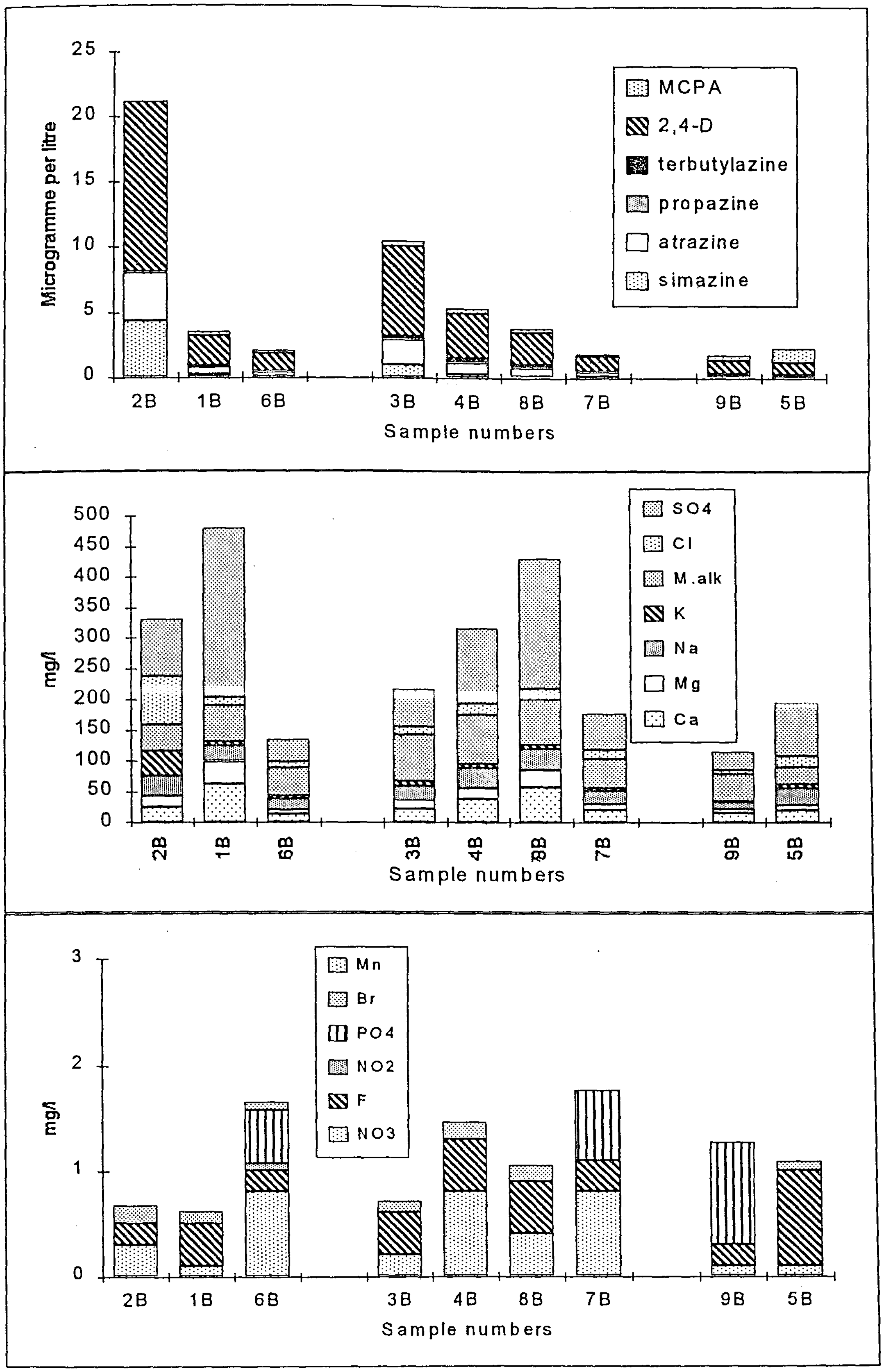


Figure 8.7. *Chemistries of water samples in rivers within the Olifants Catchment on 20.11.93.*

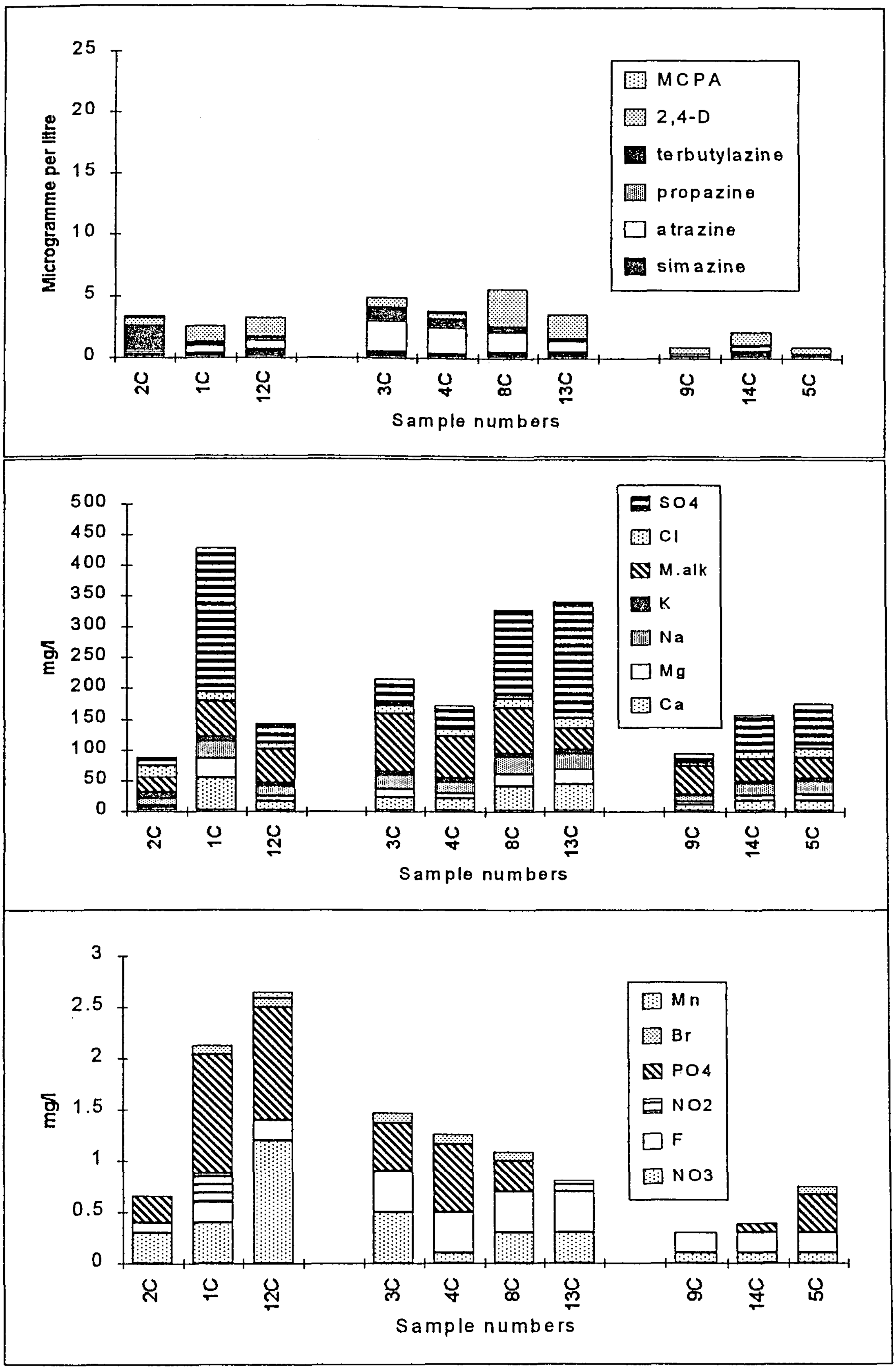


Figure 8.8. Chemistries of water samples in rivers within the Olifants Catchment on 02.01.94.

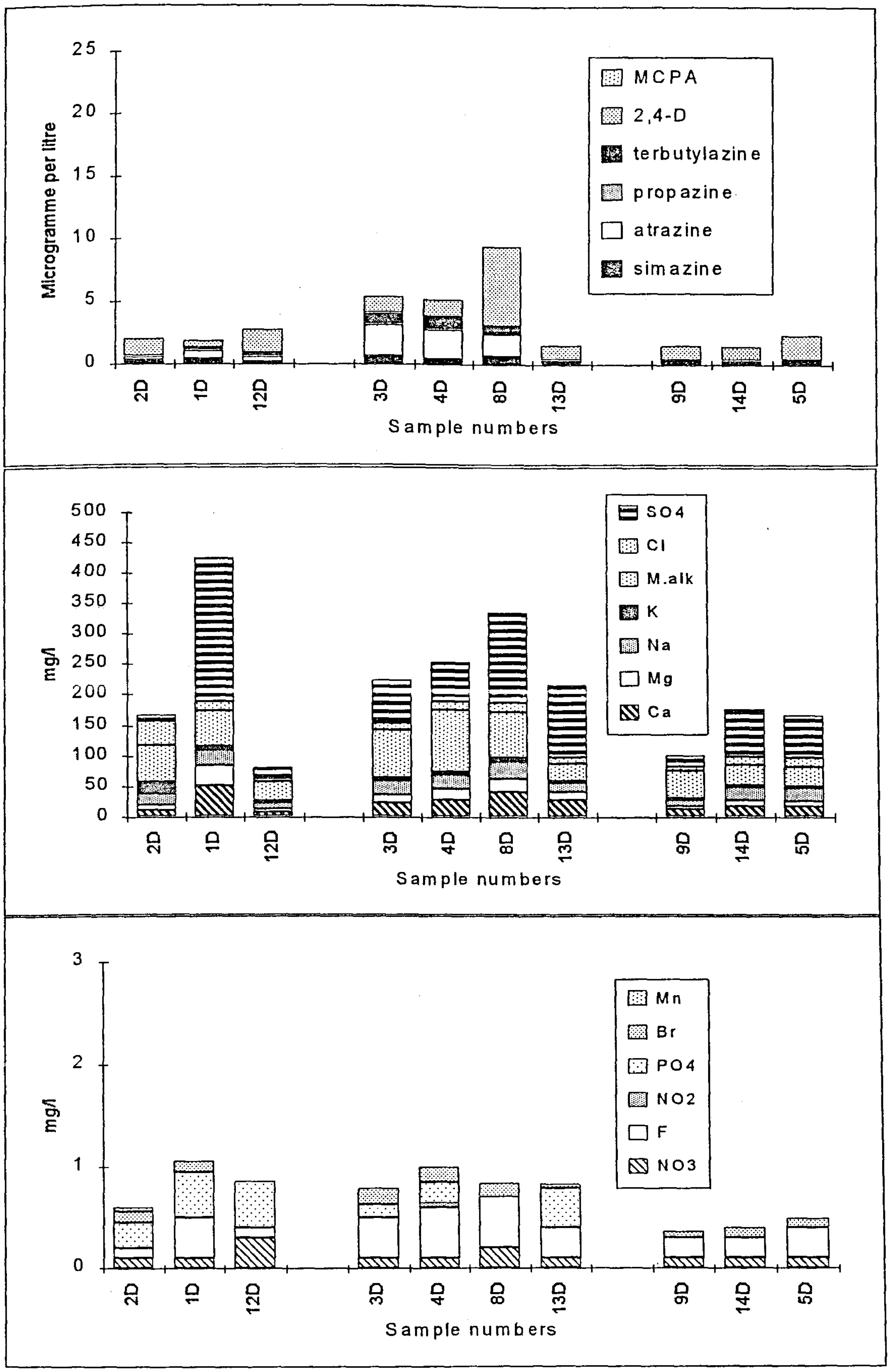


Figure 8.9. Chemistries of water samples in rivers within the Olifants Catchment on 25.01.94.

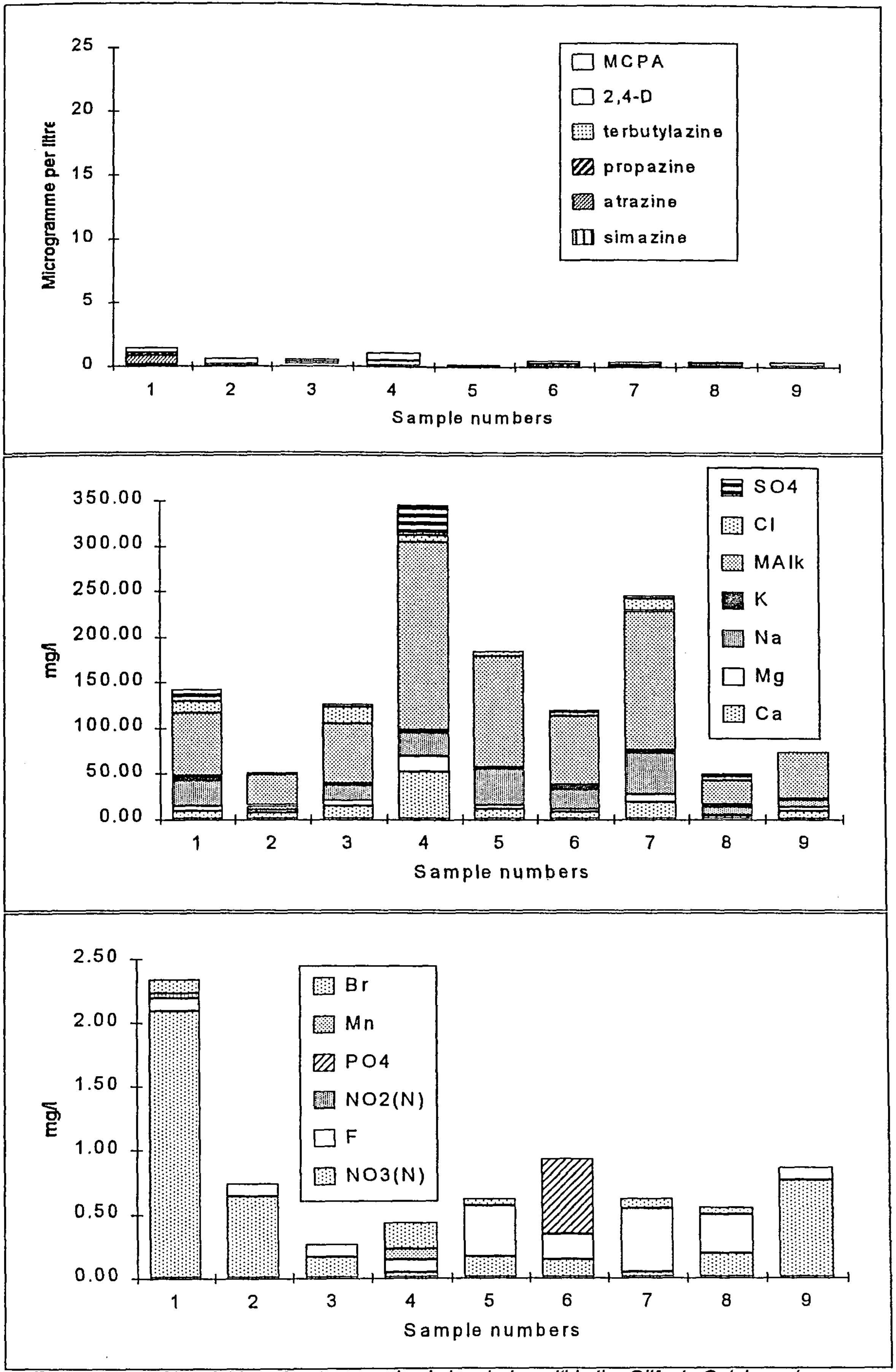


Figure 8.10 Chemistries of water samples in boreholes within the Olifants Catchment on 7.04.94.

The following conclusions may be drawn from the diagrams:

- Although the application rate of terbuthylazine is twice that of atrazine (see Table 8.5), the latter pesticide is generally more abundant. This is attributed to the greater solubility of atrazine. Degradation rates may also play an important role.
- The soil degradation rate of MCPA is considerably slower than 2,4-D (see Table 8.7). However, the decomposition rate for both chemicals in groundwater is approximately six months (Howard *et al.*, 1991). This may account for the fact that 2,4-D is more abundant than MCPA. The latter chemical may also be present in smaller quantities within the pesticides.
- Atrazine is more abundant than the remainder of the triazines. This is attributed to the greater solubility of atrazine and the fact that this chemical is present in greater concentrations within the pesticides (see Table 8.7).
- Concentrations of pesticides in the groundwater are clearly significantly lower than in surface waters. Traces of pesticides are, however, present in both the weathered and fractured Karoo aquifers.

The toxicity of the pesticides that were screened for in the Olifants Catchment sampling program, is included in Table 8.11.

The majority of the pesticides are not highly toxic to animals. Furthermore, the majority of the chemicals are rapidly eliminated from the body within 24 hours. However, the case studies included within this report do suggest that the long-term effects of pesticides may be equally detrimental to the environment.

Table 8.11. Toxicity of the pesticides which were screened during the Olifants Catchment sampling program (after the Royal Society of Chemistry, 1987).

Pesticide	Acute oral toxicity in rats (LD50 mg/kg)	% pesticide eliminated in mammals within 24 hr.
Atrazine	3080.0	88
Propazine	>7000	42-46
Simazine	>5000	65-97
Terbuthylazine	2160.0	80-84
2,4-D	375.0	rapid
MCPA	700	***

It is necessary to develop analytical techniques for the anticipated pesticides within the Olifants Catchment. It is recommended that research into these methods be prioritised in order of toxicity.

8.6 ENVIRONMENTAL IMPACT OF THE UTILISATION OF SULPHATE RICH WATERS FOR AGRICULTURAL PURPOSES

Lime treated acid mine waters have a neutral pH, contain virtually no ferruginous compounds, but remain high in dissolved solids. Since calcium sulphate or gypsum are used for the reclamation of sodic soils, Du Plessis (1983) suggested the use of lime treated waters for the irrigation of crops. This author suggested that this practice would preserve other water resources while effectively disposing of undesirable mine effluent.

Du Plessis (1983) concluded that the environmental impact of using sulphate rich or lime treated mine waters would be as follows:

- Irrigation with a calcium sulphate water will result in a lower soil salinity than with chloride water of otherwise similar ionic composition. This is attributed to the possible precipitation of gypsum.

- Greater yields are expected from a wider variety of salt-resistant crops.
- It is anticipated that there will be no serious sodium-related physical soil problems as a result of the relative increase in sodium due to the precipitation of calcium.
- The reduction in the salinity of the drainage water is expected to be greater for the sulphate than the chloride rich waters. This effectively means that the risk of contaminating other water resources will be relatively minor, particularly in areas under lower leaching conditions.

It must be pointed out that the above assumptions are based on the fact that there is sufficient calcium to achieve gypsum saturation levels. If gypsum saturation is not achieved, the risk of contaminating the underlying groundwater systems will be that much greater.

8.7 CONCLUSIONS

The following conclusions are drawn with respect to this section:

- Overseas research has shown that the agricultural sector has the potential to pollute the environment. These trends are less obvious in the Olifants Catchment at the present time. It is, however, anticipated that future problems may arise as the pressure for greater productivity increases.
- The greatest problems in the study area are related to the eutrophication of water systems. Wates, Meiring and Barnard (1993) have suggested that the most practical method of reducing the nutrient input into the water systems is by the treatment of point sources such as sewage and industrial effluent. Intensive livestock farming in the form of feedlots also presents a problem.
- Methods of ameliorating feedlot effluent include the use of algal ponds (Pieterse*et al.*, 1982), the use of vegetated buffer strips (Young*et al.*, 1980) and the irrigation of crops with waste water (Olson *et al.*, 1982). However, treatment of waste at small to medium farms is unlikely when the relatively high labour and treatment costs are considered.
- Although pesticide concentrations are similar to those of Sweden and the United States, they exceed the limit as laid down by the EEC. The question arises whether this limit is reasonable or whether a higher limit might be set which still prevents the effects of long-term pesticide exposure.
- The future of agrochemicals will be in the efficient use of fertilisers and pesticides and the implementation of non-chemical control techniques. For example, the loss of fertilisers to groundwater would be minimal if they are applied at the correct time in optimum quantities. This practice would reduce the quantity of residual fertiliser that was left in the soil for leaching.
- Further research may be necessary to determine the long-term effects of agrochemical applications.

Section 9

CONCLUSIONS ON REGIONAL MATTERS

The conclusions in the chapter all relate to regional issues. Local issues are dealt with in the individual sections. The term 'waste handling facilities' is used in this section to describe all activities in the Olifants Catchment that impact on groundwater.

9.1 RISK ASSESSMENT

Numerous activities exist in the Olifants Catchment that may impact on groundwater quality and quantity. Several risk assessment procedures are available to assess the potential risk that a specific activity may hold for an aquifer. The EMPR suggests the evaluation of systems according to four issues, namely: duration of impact; significance of impact; probability of significance; status of impact. Built into this evaluation are aspects of:

- The type of waste,
- The amount of waste,
- The potential of the waste to generate leachate,
- The vulnerability of groundwater resources, and
- The potential for groundwater usage in the area.

This corresponds with the procedures for risk assessment recommended by Parsons and Jolly (1994) and the DWA&F (1994) in their document on monitoring requirements.

9.1.1 AQUIFERS

The risk associated with local pollution of aquifers in the Olifants Catchment is significant. It can safely be stated that at all waste sites, the underlying shallow weathered aquifer is already polluted. Regional groundwater flow in the shallow aquifers occurs towards local streams. Areas between waste handling facilities and streams should therefore be regarded as potentially polluted zones. Groundwater utilisation from this area is not advised. Observations, followed by modelling, have also indicated that most of the groundwater flow within the weathered aquifer surfaces as it approaches the streams. A fair proportion of all surface water in the catchment is therefore groundwater. This proportion varies between summer and winter. In winter, all low-flow in streams is regarded to be groundwater.

The conclusion is drawn that shallow groundwater pollution is of a local nature and that almost all polluted groundwater surfaces within short distances from their sources. Monitoring of the surface water quality is therefore an essential part of understanding the degree to which groundwater has already been polluted.

Deeper aquifers in the Olifants Catchment are not polluted to the same degree as the shallow weathered aquifer. Reasons for this are:

- The stratified nature of the Karoo Sediments, thus retarding vertical permeation of fluids.

- The fact that deeper aquifers are fully saturated with water. This counteracts infiltration of fluids.
- Mining has intensively punctured the deeper aquifers and regional flow is currently towards the mining activities, ignoring historical regional flow patterns.

For the time being, pollution migration along the deeper aquifers therefore has no regional impact on groundwater quality.

9.1.2 UNSATURATED ZONE

Waste handling facilities in the Olifants Catchment can be subdivided into those where excess water is present, such as slimes and ash dams, mining activities and treated sewage effluent disposal. Other wastes, such as general waste, slags, dry ash disposal, coal discards and coal stockpiling are usually deposited under water deficit conditions.

In the case of water excess, it is safe to assume that the unsaturated zone has been destroyed. Evidence to this effect is present in the whole of the Olifants Catchment. Even in many of the so-called water deficit systems, excess water is present. Examples are general waste sites in quarries, dry ash disposal where excessive dust suppression or fluid disposal is done and co-disposal of coal discard and coal slimes.

In many instances, the amount of leachate from waste is also so great, that most of the attenuation properties of the subsoil are quickly destroyed. In other instances, at coal discards and coal stockpiling for example, the leachate is acid and the unsaturated zone has no beneficial effect.

It can be concluded that the amelioration effect of the unsaturated zone can only be utilised at general waste sites. At these sites, precautions are necessary to protect the unsaturated zone by effective rehabilitation and the channelling of water away from the sites.

9.1.3 QUANTITIES OF WASTE AND TOXICITY

The risk of water pollution in the Olifants Catchment can therefore be reduced to the type of waste, quantities thereof and the possibility that leachate migration cannot be controlled.

In terms of toxicity, waste within the catchment constitutes all wastes identified by the DWA&F in their document on minimum requirements for monitoring, except nuclear waste. Small amounts of hazardous wastes, such as at some of the metal industries and hospital waste, are definitely present within the catchment. These are, however, not considered significant on a regional basis.

In all, it is therefore mainly a case of the size of a waste site. Larger waste handling facilities pollute more than smaller ones, because they have a greater potential to generate uncontrollable leachate.

It is possible, on a regional as well as local basis, to draw a comparison between the potential impact that the various waste handling facilities may have. Table 9.1 summarises the regional and local risks. In this table the numeric values have been normalised, with 100 the highest and 1 the lowest scores. On the local scale, these values should be taken as general indices and not as absolute values, since circumstances may differ from one locality to the next. The regional impact on groundwater is minimal because of the layered nature of the aquifers and dynamic throughflow in the upper aquifer.

Pollution moves with the natural groundwater gradient towards streams where it discharges.

Table 9.1 Relative risk ratings for various current activities to pollute water.

Activity	Regional impact	Risk rating	Local impact	Risk rating
Opencast mining	Very High	100	Very High	100
Coal discard disposal	Medium	40	Very High	90
Shallow bord-and-pillar mining	Medium	30	Very High	90
High extraction underground mining	Low	10	Very high	80
Metal industry	Very Low	5	Very High	80
Coal stockpiling (power stations)	Very Low	3	Medium	40
Fly ash disposal (wet)	Very Low	3	Medium	40
Deep bord-and-pillar mining	Very Low	3	Medium	30
Dirty water dams (power stations)	Very Low	2	Medium	30
General waste disposal	Very Low	2	Medium	30
Sewage effluent disposal	Very Low	2	Low	20
Agriculture	Very Low	2	Low	10
Fly ash disposal (dry)	Very Low	1	Very Low	5

It should be stressed that this table is based on the authors' understanding of the risk that groundwater and later surface water may become contaminated in the Olifants Catchment as a result of a specific activity. Clearly, the regional scores will increase as the intensity of waste activity increases, for instance by starting up a new mine or industry. The relative values for local risks may vary according to specific circumstances. The local risk ratings should be used in accordance to the suggested monitoring procedures in Appendix E, to determine the type, intensity and frequency of monitoring.

The individual regional scores need some explaining:

Opencast mining

The high score of opencast mining is attributed to:

- The scale of operations.
- A tendency of the spoil towards the acid side.
- The volumes of water entering into these mines.
- The presence of internal drainage areas in rehabilitated spoils.
- The high magnesium content of the base minerals in the spoil.
- The dragline mining method whereby selective spoil handling cannot effectively be done.

Coal discard disposal

Coal discard disposal has, by virtue of the smaller volumes, a lower rating than opencast mining. However, many of the newer coal discard dumps are not yet contributing towards the regional pollution. It is envisaged that the pollution load from this source will increase with time. Precautionary measures must be instituted on a local basis.

Shallow bord-and-pillar mining

Shallow bord-and-pillar mining invariably contributes acid water into catchments. The main area of concern lies west of Witbank. However, within the catchment several other shallow underground mines exist that do not yet seep into adjacent aquifers. This source of pollution will therefore increase with time.

High extraction underground mining

None of the longwall, shortwall or stooped areas are currently decanting. The impact from these areas will only become evident when these mines fill with water over the next few decades. The relatively low regional risk is due to the low intensity of these mining activities in the catchment.

Other

The other items listed in Table 9.1 all have very low regional risks. This is due to a variety of reasons, such as:

- Immobility of leachates.
- Relatively small quantities of waste.
- Sufficient precautions being taken.
- No potential groundwater users.

9.2 CATCHMENT LOAD

The current catchment load is known from water and salt balances for the Witbank and Middelburg Dams. It has been suggested by Wates, Meiring and Barnard (WM&B) (1993) that the annual average sulphate load into the Witbank Dam is currently in the order of 33 t/d.

In the current investigation, it has been suggested that the sulphate generation by opencast mining is in the order of 5 - 10 t/ha/day. When projecting this to current opencasting (9 000 ha) in the Witbank Dam Catchment, a total sulphate production of 45 - 90 t/d (average 70 t/d) becomes possible. In this respect, it is important to note that only two of the pits in the Witbank Catchment are full to the extent that they decant onto surface. Most of the future impact of existing opencast mining therefore still has to be superimposed onto the current load of 33 t/d, suggested by WM&B. Extrapolation of this to include future opencasting at existing mines, the sulphate contribution can escalate an anticipated value of 120 t/d. The latter translates into a sulphate concentration in the Witbank Dam of 450 mg/l. On top of this come the contributions from all other future mines and new sources. It is apparent that, given present circumstances, the present target of 200 mg/l sulphate for the Witbank Dam (WM&B, 1993) cannot be achieved without dramatic changes in the opencast mines.

To instigate effective changes within past opencast areas is practically impossible. The systems were simply not designed to achieve containment. It should be emphasised that actions at the opencast collieries to minimise throughflow of water will lead to concentration of sulphate and magnesium to extreme levels within pit water. If this water is not totally contained, the impact of this magnesium sulphate water on the catchment below the mines will be far greater than the current situation, because this water will be useless.

To alleviate the long-term threat of salination within the mines, an option whereby the mines are actively flushed, is therefore propagated. This will lead to water of a better quality being made available for irrigation locally. The emphasis in this scheme is on the use of the water, otherwise the salt load will merely be transferred into dams and beyond. It should be stressed though, that flushing is not an option at acid mines. Here, containment or treatment of the water is advised.

Other sources of sulphate, such as coal discards and underground mining also need to be addressed. These issues are site-specific and their contribution can, through improved design, be greatly reduced.

Other sources of sulphate also exist in the catchment. Very little can or need to be done to minimise contributions from groundwater pollution at these sites, because their contribution is minute compared to that of the mining activities.

Other contaminants that may become a problem on a local basis in the catchment are magnesium and manganese from mining environments and sodium from power stations and metal industry. Current sodium application for demineralisation purposes in the Olifants Catchment is in the order of 4 t/d. Similar amounts of sulphate are introduced in the form of sulphuric acid. Most of this sodium and sulphate are disposed of in fly ash or slimes dams. Seepage from these facilities occurs. Even if all this sodium and sulphate would leach into streams, it would effectively mean an introduction into the Witbank Dam of 12 mg/ℓ sodium and sulphate. This provides a perspective on the difference in potential impact between opencast mining and power generation.

9.3 CATCHMENT MANAGEMENT PROGRAM

The DWA&F has, during the past four years, developed a catchment management program for the Olifants Catchment. One of the aims in this program was to lower the sulphate content in the Witbank Dam. Through the introduction of this program, a general awareness of water pollution has come about. Supported by the EMPR, this program will not only fulfil an important role in the future water management of the catchment, but also on the local scale, to direct mines, industries and municipalities towards better water quality management.

The suggestion is made that the catchment management model should be made available to researchers and planners. Courses should be presented in the use of the model. Through this kind of interactive exposure, the model as well as the community that are dependent on predictions by the model, will benefit.

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THEORY OF CHEMICAL REACTIONS IN OPENCAST MINES

A.1 IMPORTANT CHEMICAL DEFINITIONS AND DESCRIPTIONS

Successful proactive water quality management requires a thorough understanding of the pertinent chemical processes. Important chemical concepts that are referred to in the study are therefore discussed under this Appendix, for the reader's convenience. Reactions that contribute to water deterioration in opencast mines are examined in detail. Natural mechanisms that aid in the amelioration of water quality are also reviewed. Some relevant definitions are included to clarify any discrepancies that may exist with respect to the terminology.

Aqueous speciation is the formation of ion pairs through the association of individual cations and anions in solution. This often leads to the formation of soluble complex ions.

Solubility product constants (K_{sp}). Reactions proceed to a state of equilibrium where the concentration of products and reactants is invariant. When the solid mineral phase and the dissolved mineral constituents are in equilibrium, the solution is saturated with respect to that specific mineral. The solubility product constant quantifies the concentrations of the mineral constituents at this point.

Minerals with high solubility products are more soluble. For example, gypsum ($2{,}6 \times 10^{-5}$ K_{sp}) is more soluble than calcite ($4{,}8 \times 10^{-9}$ K_{sp}) (Skoog *et al.*, 1992).

The principle of Le Chatelier (Skoog *et al.*, 1992) states that the equilibrium position will shift to compensate for any stress which is applied to the system. Stress may take the form of temperature, pressure or concentration variations. For example, if a solution is undersaturated with respect to calcite, dissolution will take place until saturation is reached. Likewise, if the solution is oversaturated, calcite will precipitate until equilibrium is attained.

Redox reactions involve the transfer of electrons. Oxidation reactions represent the loss of electrons from atoms. The gain of electrons is termed reduction. Oxidising agents take on electrons, i.e. they are reduced whereas reducing agents give up electrons, i.e. they are oxidised. Since free electrons do not exist in solution, every oxidation is accompanied by a reduction in order to maintain the electron balance. These simultaneous reactions are termed redox reactions.

Acid-base reactions concern the transfer of protons. According to the Brønsted-Lowry definition, acids are proton donors, while bases are proton acceptors (Lloyd and Heathcote, 1985). The products of the reversible acid-base reaction are salt and water (Metcalfe *et al.*, 1978). Hydrolysis is an obsolete term used to describe the proton transfer between water and an acid or base. This expression is still used when the acid in the reaction is an aqueous metal ion.

Alkalinity may be defined as the capacity of carbonate and bicarbonate in water to neutralise acids (Matthess, 1982). In reality, many other components such as the hydroxide ion may also contribute to the alkalinity of the water.

Adsorption-desorption. Surficial charges on a crystal surface are attributed to unsatisfied valences. For example, charged oxide ions are situated at the surface of silicates and metal oxides. Since these oxide ions react with water, the nature of the charge is pH-dependent. Crystal surfaces are positively ionised in acidic solutions and negatively in alkaline solutions. A zero point of charge exists at some intermediate pH. This position varies between pH 9 for basic oxides and pH 2 for clay minerals.

The interaction between charged colloidal particles and ionic solutions creates a diffuse adsorbed layer of counterions. Due to their negative surficial charge, clay minerals adsorb cations under most hydrochemical conditions, hence the term cation exchange capacity (CEC). Cations are adsorbed onto clays in the order of $Al^{3+} > Ca^{2+} > Mg^{2+} > K^+ > Na^+$ (Fergusson, 1982). The order is proportional to the decrease in charge and hydrated ionic size.

Ion exchange. Adsorbed cations are exchangeable due to the weak electrostatic forces that bind them to mineral surfaces. The cation exchange capacity (CEC) is proportional to the density of the surface charge.

Open and closed systems. Carbon dioxide is exsolved during the neutralisation reactions in an open system. Closed systems are characterised by the dissolution of CO_2 and the subsequent formation of the carbonate species, such as $CO_3^=$, HCO_3^- and H_2CO_3.

A.2 CHEMICAL PROCESSES THAT DICTATE WATER QUALITY

The chemical reactions within the mining environment can be subdivided into five categories, namely:

- Redox reactions.
- Acid-base reactions.
- Adsorption-desorption phenomena (ion exchange).
- Dissolution-precipitation.
- Aqueous speciation.

A.2.1 REDOX REACTIONS AND HYDROLYSIS

The required reactants for acid generation in mining environments are:

- Sulphide and/or ferric ions.
- Water or a humid atmosphere.
- Oxygen or some other oxidant.

Since pyrite is the dominant sulphide mineral found in coal deposits, this phase is generally used to describe acid formation. In the Witbank Coalfield, the No. 2 Seam contains 0,96% sulphur on average (Cairncross *et al.*, 1990).

The pyrite oxidation model taken from Stumm and Morgan (1981), will be used as a discussion framework. The reactions of the model are as follows:

(1) $FeS_2\,(s) + 3{,}5O_2 + H_2O = Fe^{2+} + 2SO_4^{2-} + 2H^+$

(2) $Fe^{2+} + 0{,}25O_2 + H^+ = Fe^{3+} + 0{,}5H_2O$

(3) $FeS_2 + 14Fe^{3+} + 8H_2O = 15Fe^{2+} + 2SO_4^{2-} + 16H^+$

(4) $Fe^{3+} + 3H_2O = Fe(OH)_3(s) + 3H^+$

A graphical presentation of the model is outlined below:

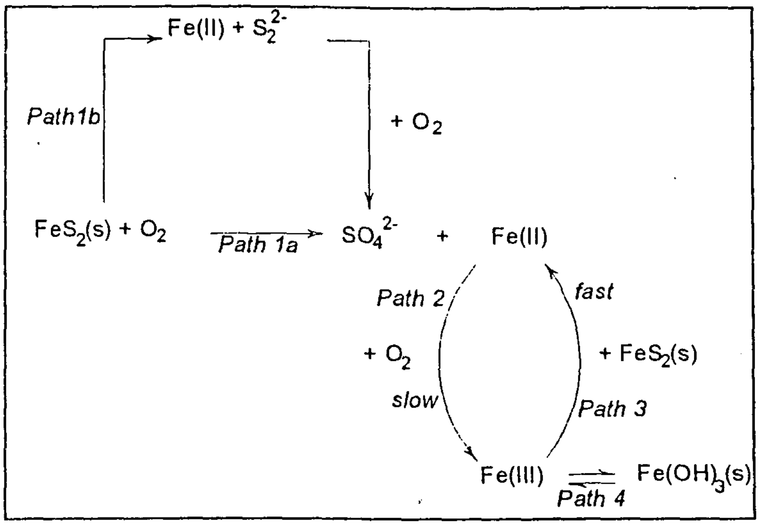


Figure A.1. Pyrite oxidation model, after Stumm and Morgan (1981).

Reaction 1

Pyrite is oxidised directly by oxygen. A variation of this reaction involves the oxidation of the aqueous species that originate from pyrite dissolution. However, this is questionable since pyrite is relatively water insoluble (4,9 mg/l) (Lovell, 1983).

The constituents required for further acid generation are liberated. Aqueous species, namely Fe^{2+}, SO_4^{2-} and H^+, increase the total dissolved solids and acidity of the water. Levinson (1974), Kempe (1983) and Lovell (1983) include $FeSO_4$ as one of the products in reaction 1. However, Lovell (1983) admits that $FeSO_4$ is quite soluble in water. It is therefore improbable that any durable $FeSO_4$ precipitate will form in the dynamic spoil waters associated with mining activities.

Reaction 2

Although reactions 3 and 4 dominate the acid production by virtue of their rapid reaction rate, they are limited by the availability of ferric iron. The oxidation of ferrous iron (reaction 2) is therefore the rate-determining step for the acid formation model.

At pH 3, the half-time for the oxidation of Fe^{2+} is in the order of 1 000 days (Stumm and Morgan, 1981). However, the presence of bacterial catalysts accelerates the acid-generating process.

Reaction 3

Ferric ions are utilised in the further oxidation of pyrite. The relatively fast reaction (half-time 1 700 minutes) may also occur under anaerobic conditions (Garrels and Thompson, 1960). Ferric ions cannot exist in the presence of pyrite due to the spontaneity of the reaction. This is the prominent mechanism of pyrite oxidation, once the acidification sequence is initiated (Stumm and Morgan, 1981). The importance of reaction 1 is therefore limited to the initiation of the reaction sequence.

Another set of reactions completes the chemical cycle (Jaynes*et al.*, 1984):

(a) $FeS_2 + 14Fe^{3+} + 8H_2O = 15Fe^{2+} + 2SO_4^{2-} + 16H^+$

(b) $14Fe^{2+} + 3{,}5O_2 + 14H^+ = 14Fe^{3+} + 7H_2O$

The ferric iron that is consumed by reaction (a) is regenerated by reaction (b). By adding up reactions (a) and (b), it can be seen that one mole of pyrite plus one mole of water and 3,5 moles of oxygen, produce two moles of acidity (Jaynes*et al.*, 1984). This is identical to the oxidation of pyrite by oxygen. It is therefore the relative reaction rates that dictate the dominance of one reaction over the other.

Reaction 4

Excess ferric iron that has not reacted with pyrite will precipitate as ferric hydroxide. Further acidification of the system takes place as a result of this hydrolysis. The insolubility of the $Fe(OH)_3$ above the threshold value of 3,1 - 3,7, maintains the acid levels.

The dehydration of ferric hydroxide to goethite, as described by Levinson (1974), concludes the set of reactions:

$Fe(OH)_3 = FeO(OH) + H_2O$

An alternative pyrite oxidation model

Levinson (1974) mentions the reaction of sulphuric acid with pyrite:

(1) $FeS_2 + H_2SO_4 = FeSO_4 + H_2S + S$

According to Levinson (1974), this accounts for the presence of small amounts of native sulphur at outcrops. It could also explain why H_2S is a common gas around coal mines and mine dumps, other than the H_2S which is ascribed to bacterial sulphate reduction. If reaction (1) occurs in the anaerobic zone, the products of the reaction will reform pyrite with the aid of bacterial catalysts (Fergusson 1982).

Other reactions that complete the process are as follows:

(2) $FeSO_4 + 0{,}25O_2 + 2{,}5H_2O = Fe(OH)_3 + H_2SO_4$

(3) $H_2S + 0{,}5O_2 = S + H_2O$

(4) $2S + 3O_2 + 2H_2O = 2H_2SO_4$

Reactions (3) and (4) (Fergusson, 1982) are accelerated by catalysts which are described below.

Other mechanisms for the oxidation of H_2S include the following reactions:

(5) $H_2S + O_3 = H_2O + SO_2$

(6) $SO_2 + H_2O + 0{,}5O_2 = H_2SO_4$

The oxidation of sulphur dioxide is inhibited as the solution becomes more acidic. It is therefore unlikely that reactions (5) and (6) will generate substantial acid.

It is significant that the acid generation is activated when gaseous oxygen becomes available. Summing the first four equations, gives four moles of acidity generated by 3,75 moles oxygen, 3,5 moles of water and one mole of pyrite. The component requirements for the generation of acidity are therefore identical to that of Stumm and Morgan (1981).

A.2.1.1 Factors which affect the rate of acid generation

Grain-size

Disseminated, amorphous sulphide precipitation commonly forms in bogs and peats or in environments of rapid deposition (Steffen, Robertson and Kirsten, 1989). The initial minerals (e.g. greigite) may subsequently alter to framboidal pyrite through sulphurization. Oxidation of such fine-grained pyrite occurs at a substantially faster rate than that of euhedral pyrite (Barton, 1978; Houghton *et al.*, 1985; McCurry and Rauch, 1986).

Bacterial catalysts

Bacterial catalysts are crucial for the acceleration of the acidification process. There are two bacterial species, namely *Thiobacillus ferrooxidans* and *Thiobacillus thiooxidans,* which are relevant to the acid-generating models, described previously.

Thiobacillus ferrooxidans is a bacterial species that is known to accelerate the oxidation of iron (Kleinmann and Erickson, 1983). Barton (1978) mentions that the bacteria may accelerate the acidification process by more than a factor of 10^6 at a pH of 2,5 - 3,5.

Favourable environments ensure the survival of the micro-organisms. Steffen, Robertson and Kirsten (1989) list the following factors that affect bacterial activity within the spoil:

- Optimum pH (between 2,5 - 3,5).
- Optimum temperature (between 30 - 50°C).
- Population density of the bacteria.
- Rate of population growth.
- Nitrate concentration (which is low in South African Collieries).
- Ammonia concentration (which is low in South African Collieries).
- Phosphorous concentration (which is low in South African Collieries).
- Carbon dioxide content (which is variable in South African Collieries).

According to Miller and Murray (1988), biologically catalysed acid generation occurs in three discrete stages:

- Stage 1 is the chemical oxidation of sulphide minerals, which slowly produces acid. This acid may be neutralised by carbonates or other minerals present and there is little, if any, decrease in the pH. The duration of this stage is variable. Techniques that simply measure the pH will not detect the oxidation of sulphides at this point.
- Stage 2 begins once the carbonates and other neutralising agents are consumed in local pockets within the rock or spoil, e.g. at a pH below 4,5. The pH drops and acidophillic bacteria multiply.
- Stage 3 is characterised by a drop in the pH below 3,5. At this point, bacterially catalysed oxidation of sulphide minerals is the main acid-producing process. Acid generation is rapid and is only limited by the concentration of ferric iron. Oxidation of the latter is dependent on the availability of oxygen.

Oxidation of hydrogen sulphide and elemental sulphur is accelerated by *Thiobacillus thiooxidans,* to levels where acid generation is significant in the Levinson (1974) model.

Availability of other reagents

Oxygen and water are the two other reagents required for the oxidation of the pyrite. Oxygen ingress into the unsaturated zone is facilitated by barometric variations. Similarly, water is available from recharge by the infiltration from rainfall.

It can therefore be concluded that the reactivity of pyrite and the pH of the spoil water in the unsaturated zone dictate the rate of acid generation. The latter determines the activity of bacterial catalysts.

A.2.2 ACID-BASE REACTIONS

Two mechanisms exist for the amelioration of acid within natural systems, namely neutralisation and adsorption or ion exchange. Neutralisation reactions may be classified as acid-base reactions since they involve the transfer of protons.

A.2.2.1 Neutralisation

Sources of acid-consuming components are:

- The alkalinity of the water itself.
- Minerals contained within the spoil.
    - Carbonates
    - Clay minerals

A.2.2.1.1 Alkalinity of natural water

The main source of alkalinity in natural ground water is carbonate dissolution. Although the solubility of carbonates in pure water is low, the dissolution is substantially increased in the presence of carbon dioxide according to the reaction (Matthess, 1982):

$$CaCO_3 + H_2O + CO_2 = Ca^{2+} + 2HCO_3^-$$

The pH increases as a result of the addition of the calcium carbonate. As the pH and bicarbonate concentrations increase, so does the $CO_3^=$ concentration until equilibrium conditions are reached (Lloyd and Heathcote, 1985).

The concept of alkalinity and carbonate reactions with acid is described in the following manner by Metcalfe *et al.* (1978):

$$CO_3^= + 2H^+ = H_2O + CO_2$$

In systems where carbon dioxide is dissolved, the buffering reactions are (Metcalfe *et al.*, 1978):

$$CO_3^= + H^+ = HCO_3^- + H_2O\text{, and}$$

$$HCO_3^- + H^+ = H_2CO_3 + H_2O$$

The relative distribution of the H_2CO_3, HCO_3^- and $CO_3^=$ species with respect to the pH-range is shown in Figure A.2. The principle of the coexistence of two or more carbonate species is demonstrated in this diagram.

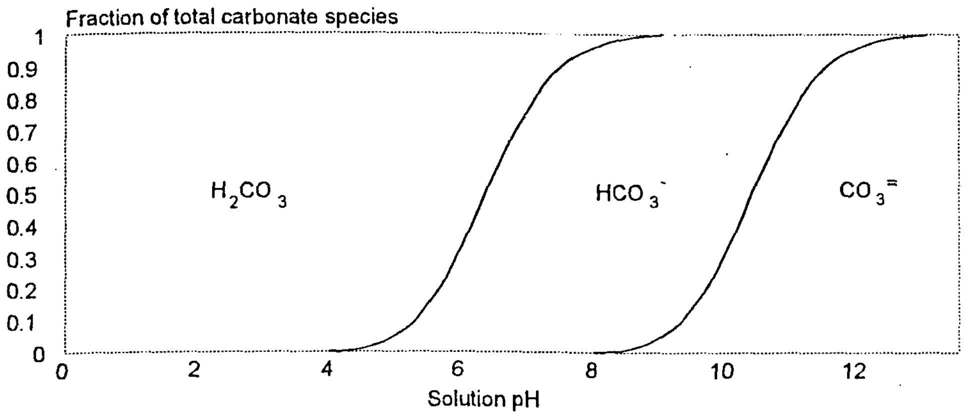


Figure A.2. Distribution of carbonate species with respect to the pH-range, after Lloyd and Heathcote (1985).

A.2.2.1.2 Minerals within the spoil

Neutralisation through reaction of base minerals with acidic leachate is the second acid-consuming mechanism. Calcite, then dolomite, kaolinite, hydroxide and finally limonite/goethite sequentially buffer spoil waters. This important concept is illustrated in Table A.1.

Table A.1. Buffer pH-levels for a selection of acid-consuming minerals, after Steffen, Robertson and Kirsten (1989) and PHREEQE simulations.

Mineral	**Composition**	**Buffer pH**
Calcite	CaCO3	5.5 - 6.9
Dolomite	CaMg(CO3)2	5.3 - 6.8
Siderite	FeCO3	5.1 - 6.0
Kaolinite	Al2Si2O5(OH)4	3.7 - 4.3
Gibbsite	Al(OH)3	3.7 - 4.3
Ferric hydroxide	Fe(OH)3	3.3 - 3.7
Goethite	FeO(OH)	2.1 - 2.2

Carbonates

The lower and upper bounds suggested for pH in terms of South African drinking water are 5,5 - 9,5 (Kempster and Smith, 1985). It is evident from Table A.1 that carbonates are the only base minerals which can be considered if the South African specifications are to be adhered to.

Acid neutralisation by carbonate minerals generates carbon dioxide. The carbon dioxide may exsolve into the atmosphere (open system) or dissolve in water (closed system). The reactions which typify these systems may be written as follows (Lloyd and Heathcote, 1985; Cravotta *et al.*, 1990):

Open System

(1) $FeS_2 + 2CaCO_3 + 3{,}75O_2 + 1{,}5H_2O = Fe(OH)_3 + 2SO_4^{2-} + 2Ca^{2+} + 2CO_2$

Closed System

(2) $FeS_2 + 2CaCO_3 + 3{,}75O_2 + 3{,}5H_2O = Fe(OH)_3 + 2SO_4^{2-} + 2Ca^{2+} + 2H_2CO_3^*$

$H2CO_3^*$ denotes a composite variable that reacts with carbonate in the following reaction:

$H_2CO_3^* + CaCO_3 = Ca^{2+} + 2\ HCO_3^-$

A second reaction may therefore be written as:

(3) $FeS_2 + 4CaCO_3 + 3{,}75O_2 + 3{,}5H_2O = Fe(OH)_3 + 2SO_4^{2-} + 4Ca^{2+} + 4HCO_3^-$

The maximum calcium carbonate requirements for acid neutralisation in a closed system are therefore depicted by reaction (3). Realistic conditions for the coal mines lie somewhere between the open and closed systems, with a tendency towards the closed system.

Some carbonates, e.g. siderite, are ineffective neutralisation agents (Cravotta*et al.*, 1990). The reaction between siderite and acid drainage may be written as follows:

1) $FeCO_3 + 2H^+ = Fe^{2+} + H_2O + CO_2$ (open system)

2) $FeCO_3 + 2H^+ = Fe^{2+} + H_2CO_3^*$ (closed system)

3) $Fe^{2+} + 0{,}25O_2 + H^+ = Fe^{3+} + 0{,}5H_2O$

4) $Fe^{3+} + 3H_2O = Fe(OH)_3 + 3H^+$

It is evident that the acidity which is consumed by the carbonate component is regenerated by the subsequent oxidation of iron and the precipitation of $Fe(OH)_3$. Moreover, the siderite may contribute further acidity by the formation of $H_2CO_3^*$ in a closed system. This may result in the further depletion of calcium carbonate as follows:

$FeCO_3 + CaCO_3 + 0{,}25O_2 + 2{,}5H_2O = Fe(OH)_3 + Ca^{2+} + 2HCO_3^-$

Other ineffective acid-consuming phases include ferric and aluminium hydroxide. These phases neutralise acidity through dissolution. If these minerals precipitate again, acid is regenerated. Their net acid consumption potential is therefore zero.

Clay minerals

The kaolinite group is probably the most common and widespread clay mineral (Fitzpatrick, 1980). Garrels and Christ (1965) mention that the dissolution of kaolinite consumes H^+ ions leaving a gibbsite residue. However, Berner (1971) describes the dissolution of kaolinite by the following reaction:

$Al_2Si_2O_5(OH)_4 + 5H_2O = 2H_4SiO_{4aq} + 2Al(OH)_3$

In this reaction, there is no acid consumption if the gibbsite precipitates. Geochemical models confirm this for alkaline waters. However, two processes occur in acidic solutions (pH < 4,0).

Firstly, gibbsite is undersaturated with respect to the solution. The Al(OH)3 remains in the aqueous phase and it has a similar acid-consuming effect as the dissolution of gibbsite. Secondly, computer models also indicate that quartz may become oversaturated within this acidic pH-range. The precipitation of quartz is described by (Berner, 1971):

$H_4SiO_{4aq} = SiO_2 + 2H_2O$

This process drives the kaolinite reaction to the right.

With increased dissolution of the clay mineral, acid consumption of the solution is enhanced. However, this apparent acid consumption is temporary. Once the solution mixes with alkaline aqueous systems, the precipitation of gibbsite will regenerate acidity. Kaolinite, like siderite, is therefore an ineffective acid-consuming mineral.

A.2.3 ADSORPTION

Two important properties of clay minerals in terms of water quality management are their cation-exchange capacity and their ionic double layer (Fitzpatrick, 1980). These characteristics enable clay minerals to ameliorate waters.

The cation-exchange capacity of clay minerals varies according to the structure of the mineral, the pH of the solution and the increased number of adsorption sites due to isomorphous replacement (Fitzpatrick, 1980). Table A.2 lists a variety of clay minerals and their associated cation-exchange capacities.

Table A.2. Cation exchange capacities of clay minerals, after Fitzpatrick (1980).

Clay Mineral	CEC (meq/100g clay, pH 7)
Kaolinite	3 - 15
Halloysite	40 - 50
Mica	14885.0
Montmorillonite	80 - 150
Vermiculite	100 - 150
Chlorite	14885

Minerals such as the kaolinite family have a rigid 1:1 structure consisting of one aluminium hydroxide and one silicate sheet (Fitzpatrick, 1980). This rigid structure prevents ionic substitution of silica by aluminium (Berner, 1971). Exchange sites are therefore restricted to the damaged edges of crystals. It can be seen from Table A.2 that kaolinite has a low exchange capacity due to the relatively few broken bonds at these locations. Such minerals are unlikely to have a significant effect on the pH of any solution (Finkelman and Giffen, 1986).

Acidic soils usually contain kaolinite and are dominated by the hydronium and aluminium ions (Retallack, 1990). Such soils form in humid climates where active leaching produces protonated clays (Fergusson, 1982):

$$\text{Clay-H} + H_2O = \text{Clay-} + H_3O^+$$

The montmorillonite family, on the other hand, has a flexible structure consisting of one aluminous octahedral layer sandwiched between two tetrahedral silicate layers (Berner, 1971). A sizable negative charge is therefore created as a result of extensive isomorphous substitution. The large cation exchange capacity for montmorillonite is therefore a function of the considerable counterions that are required to balance these great surficial charges.

The soil and solution pH-values are affected by the cations which are adsorbed onto the clays (Fergusson, 1982). Dominant Ca, Mg, Na and K ions elevate the pH of a solution when they are desorbed from clay surfaces. The reaction may be written as follows (Fergusson 1982):

$$\text{Clay-Na} + H_2O = \text{Clay-H} + OH^- + Na^+$$

Iron and aluminium, on the other hand, tend to produce acidic solutions when they are desorbed from clay surfaces:

$$\text{Clay-Al} + 5H_2O = \text{Clay-H}_3 + Al(OH)_4^- + H_3O^+$$

The exchange capacity of the clay minerals within the spoil may be reduced by the acidification of the spoil waters. This is due to the presence of tightly adsorbed iron and aluminium ions that block exchange sites. The assimilative capacity of the environment is degraded under these conditions.

In conclusion, the beneficial effect of adsorption/desorption is pronounced in alkaline soils which are rich in exchangeable cations such as Ca and Mg. Acidic soils will not influence the pH of leachates as they are characterised by limited exchange capacity and the dominance of the hydronium ion.

A.2.4 DISSOLUTION AND PRECIPITATION

Dissolution and precipitation of minerals is a function of their solubility product. Water reaches saturation once the concentration of mineral components satisfies their solubility product constants. In the case of gypsum, for instance, the solubility product constant may be written as follows (Skoog *et al*, 1992):

$$K_{sp} = 2{,}6 \times 10^{-5} = [Ca^{2+}][SO_4^{2-}]$$

where [ ] brackets denote actual concentrations of the individual ions.

The effective concentrations within solutions vary with ionic strength and a term called 'activity' is introduced in the above equation. For this purpose, the equation is rewritten as follows:

$$K_{sp} = 2{,}6 \times 10^{-5} = (Ca^{2+})(SO_4^{2-})$$

where ( ) brackets denote the activity of the species.

An index of saturation for gypsum may therefore be written as follows:

$$\text{Saturation index} = \log((Ca^{2+})(SO_4^{2-})/K_{sp})$$

A negative saturation index denotes mineral undersaturation, whereas a positive saturation index denotes oversaturation. Chemical equilibrium models such as WATEQ, PHREEQE and MINTEQ (Trusdell and Jones, 1974; Parkhurst *et al.*, 1980; Felmy *et al.*, 1984) are available to perform these calculations.

The concept of activity has important implications for the dissolution and precipitation of minerals. An increase in salinity will decrease the activity of all the dissolved species. As the activity decreases, the saturation index drops. Using the principle of Le Chatelier, equilibrium will be attained by the dissolution of that specific mineral. This phenomenon is also reflected in the increased solubility of the minerals.

In a complex environment, such as a coal mine, a multitude of ions with different ionic strengths may be present. These all interact upon each other, affecting the saturation indices for various mineral assemblages.

For example, sodium chloride increases the solubility of gypsum by decreasing the effective concentration of the calcium and sulphate ions at low concentrations (Figure A.5). Although the actual species concentration is theoretically sufficient for the precipitation of gypsum, the latter remains in solution.

A pyrite oxidation model illustrates the basic processes and complexity of systems in the opencast environment below. The following assumptions are pertinent to this model:

Assumptions:

- The spoil represents an open system with a pCO_2 pressure of -3,5 (log value) as suggested by Stumm and Morgan (1981).
- Spoil water is contained within the spoil. The oxidation products, the acid-consuming minerals and the precipitation products remain in contact throughout the simulation.
- There is sufficient oxygen to complete the oxidation of iron within the spoil.

- The oxidation of the iron occurs in areas where the alkaline material predominates. Precipitation of ferric hydroxide is therefore favoured to the oxidation of pyrite by ferric ions.
- The amount of pyrite and acid-consuming material was set at an arbitrary value of 21 g/l pyrite, 5,2 g/l dolomite and 1,5 g/l calcite. These quantities were reacted in increments to produce the concentrations illustrated in the diagrams below.

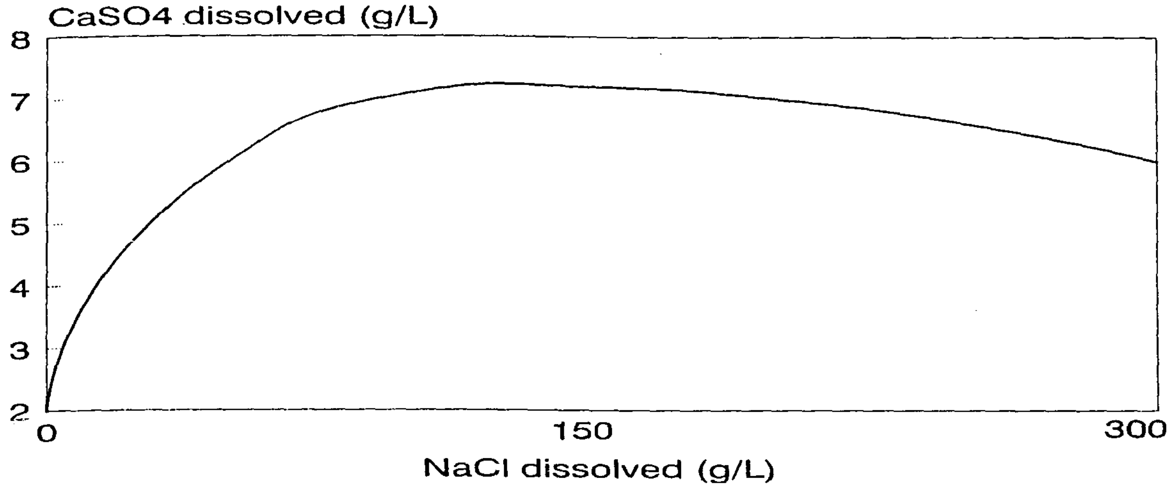


Figure A.3.. Gypsum solubility in the presence of NaCl, after Freeze and Cherry (1979).

Model reactions:

Three diagrams are included (Figures A.4 - A.6), which demonstrate the interactive relationship between the various components, during oxidation of pyrite.

- Point A, in these figures, represents the onset of pyrite oxidation, the simultaneous oxidation of iron and the precipitation of ferric hydroxide. The solution is initially undersaturated with respect to gypsum. The acidity produced by these reactions between A - B is neutralised by carbonate material.
- The carbonate content has totally been consumed at point B. This is reflected by a constant calcium concentration, the relatively stable saturation indices for gypsum and the significant drop in pH after point B.
- At point C, the ferric hydroxide which has precipitated, dissolves again. This releases hydroxide ions into the solution, which buffers the pH until the ferric hydroxide is exhausted at point D.
- The pH continues to drop until the end of the simulation at point E.

The above simulation is only a sample of the many reactions that do occur within the mining environment. The real situation in the field is obviously much more complex, because of the many variables that may be introduced. A typical example of a more complex model would be the introduction of time and spatial variables. Computer models have not yet adequately simulated the hydrochemical evolution within opencast mines, while taking all the contributing factors into account. Although it is theoretically possible and potential models to do this exist (Yeh and Tripathi, 1991), it is prohibitively expensive to calibrate such simulations.

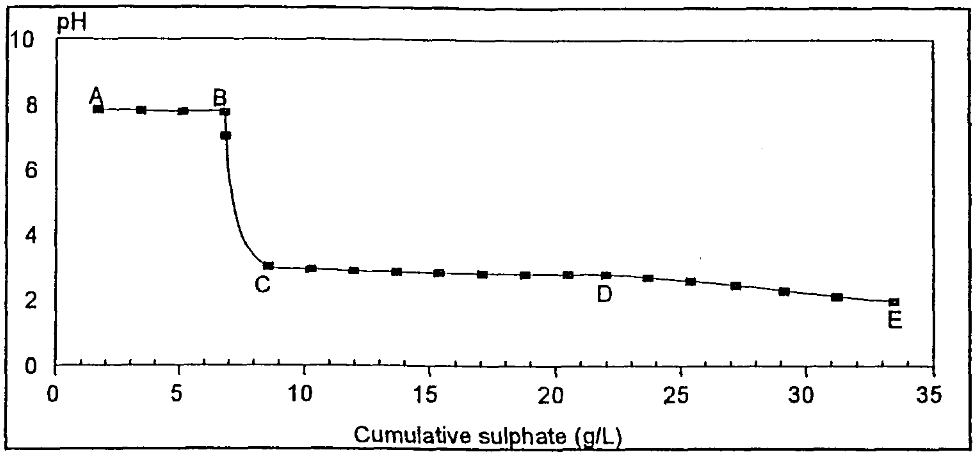


Figure A.4. Drop in pH as a result of pyrite oxidation.

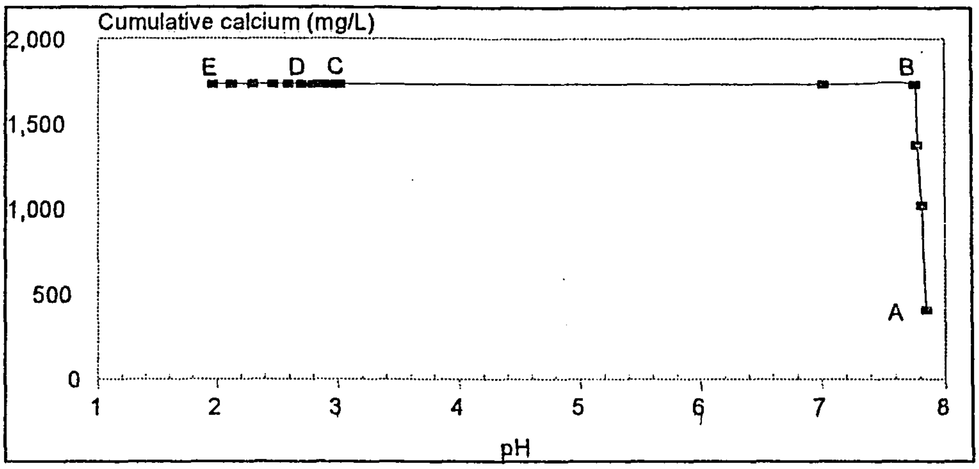


Figure A.5. Calcium introduced as a result of carbonate dissolution.

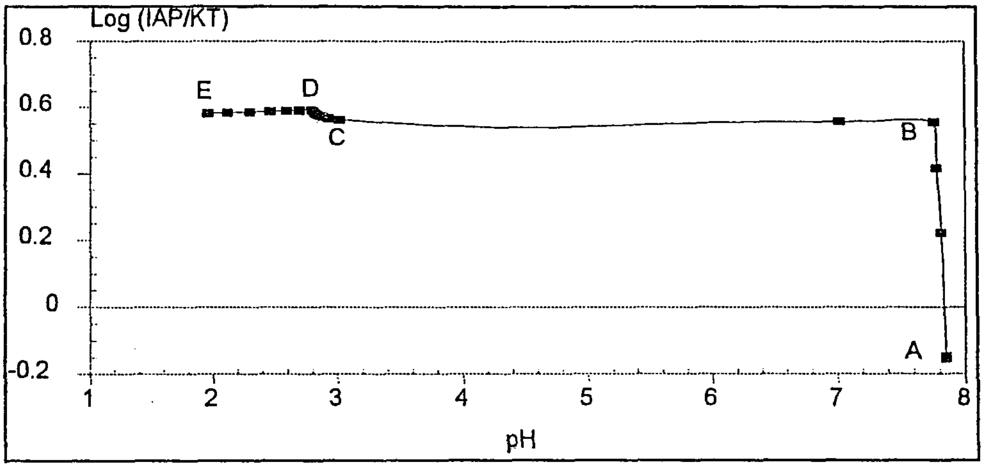


Figure A.6. Gypsum saturation indices.

A.2.5 AQUEOUS SPECIATION

Aqueous speciation is the formation of ion pairs through the association of individual cations and anions. This often leads to the formation of soluble complex ions. For example, the following sulphate species may be formed: $CaSO_{4aq}$, $MgSO_{4aq}$, $NaSO_4^-$, KSO_4^-, $AlSO_4^+$, and $AlSO_4^{2-}$. The formation of these aqueous species reduces the net effective concentration of the sulphate ion. Gypsum will become more soluble under these conditions.

A selection of the various possible aqueous species which may coexist within a typical acid spoil water, has been included in the following tables:

Table A.3. Example of chemistry for acid mine water for the calculation of aqueous speciation distribution.

Variable	Value
pH	2.7
Sulphate (mg/l)	2990
Calcium (mg/l)	431
Magnesium (mg/l)	92
Manganese (mg/l)	39
Potassium (mg/l)	2
Sodium (mg/l)	14
Iron (mg/l)	357
Aluminium (mg/l)	70

This water has been selected because of its high metal content

Table A.4. Aqueous species which should be present in a water with the above chemistry, calculated from PHREEQE.

Species	Percentage
HSO4-	5.7
CaSO4	14
MgSO4	5
NaSO4-	0
FeSO4	8
MnSO4	1
AlSO4+	4
AlSO42-	2
SO42-	61

Noteworthy is the fact that only about 60% of the sulphate in the water occurs as SO_4^{2-}. The potential for gypsum to precipitate from this water is thus greatly reduced.

Appendix B

ACID-BASE EXPERIMENTS AND ACCOUNTING

B.1 INTRODUCTION

Static tests in a laboratory may be used to determine the capacity of spoil heaps to generate and neutralise acidity.

The results of the static tests may be reported as:

- Acid potential.
- Base potential.
- Net neutralising potential.

B.2 ACID POTENTIAL EXPERIMENTS

The determination of the total sulphur is used as the basis for the prediction of the acid-producing potential of a rock sample. This may be achieved through the use of the Leco analyser (Steffen, Robertson and Kirsten, 1989).

However, other methods such as the determination of the reactive sulphur through the treatment of the sample with peroxide, have also been examined (Finkelman and Giffen, 1986). This latter method was chosen above that of the Leco analyser for the following reasons:

- The peroxide laboratory technique only measures the reactive sulphur within the sample.
- A final pH of the solution may be determined in the laboratory. This will provide an estimate of the future water quality that will emanate from the spoil. This information may be used in geochemical models such as PHREEQE to predict the future impact of the mine on the receiving water systems.
- It is possible to correlate the final and initial pH with the NNP of the sample, thereby providing a check for the total sulphur and base values.
- The chemical and toxic characteristics of the final solution may be determined, thereby providing a worst case scenario for the future anticipated salt load.
- The peroxide experimental technique is cost-effective and utilises equipment that is available in the majority of laboratories. The equipment may also be used in the base potential determinations. It also avoids the considerable initial capital outlay for a Leco analyser.

The experimental technique for the peroxide experiment was successfully established at the IGS through the determination of sulphur percentages for pure pyrite samples.

On the basis of the above, the peroxide technique is recommended for acid-base determinations.

B.2.1 STANDARDISATION OF EXPERIMENTAL TECHNIQUE

Standardisation of the experimental technique was as follows:

- A single crystal of pyrite was pulverised to <325 mesh size in a disc mill.
- Three batches of pyrite powder, equivalent to 0,6%, 1% and 5% sulphur for a 5 g spoil sample, were prepared.
- Different volumes of peroxide (30%) were added to portions from each batch of pyritic powder, to determine the optimum volume of peroxide required for complete oxidation.
- The samples were left for 24 hours to complete their reaction.
- Sulphur concentrations in the reacted fluids for each of the samples were determined by ICP.

Analysis of the data indicates that 93% - 100% of the pyrite was oxidised for all three standards at a volume of 50 ml concentrated peroxide (Figure B.1). This was confirmed by the unreacted pyrite in the beakers, which constituted 7% of the original weight on average.

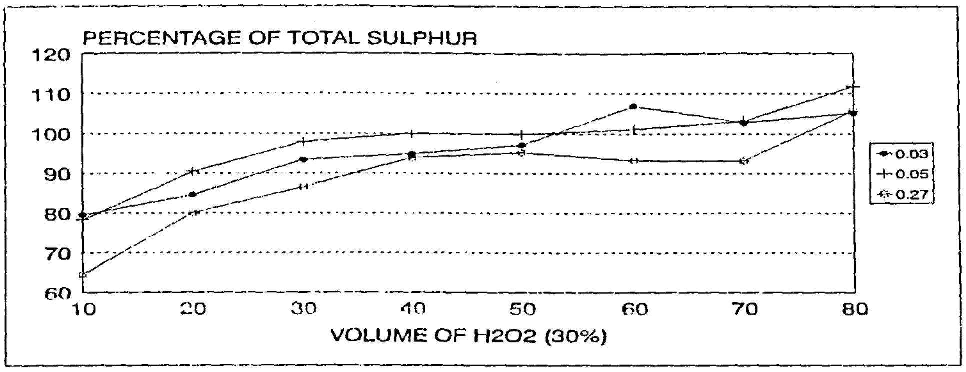


Figure B.1. Volume of peroxide vs varying quantities of pyritic sulphur.

These values are an improvement on the 28% unreacted pyrite reported by Finkelman and Giffen (1986). These researchers used <325 mesh size pyrite and 15% peroxide. Comparative analyses with 15% peroxide (Figure B.2) over 24 hours oxidation time, yielded a higher oxidation rate than the previous researchers. However, this peroxide concentration was insufficient for total sulphur recovery.

Figure B.1 indicates that there are sulphur concentrations in excess of 100% total pyritic sulphur for peroxide volumes greater than 50 ml. This is ascribed to a concentration effect through volume loss. This loss is a consequence of boiling during the reaction of the pyrite with the peroxide. Measurements of the solution volume indicate that the volume loss is negligible at 50 *ml* peroxide.

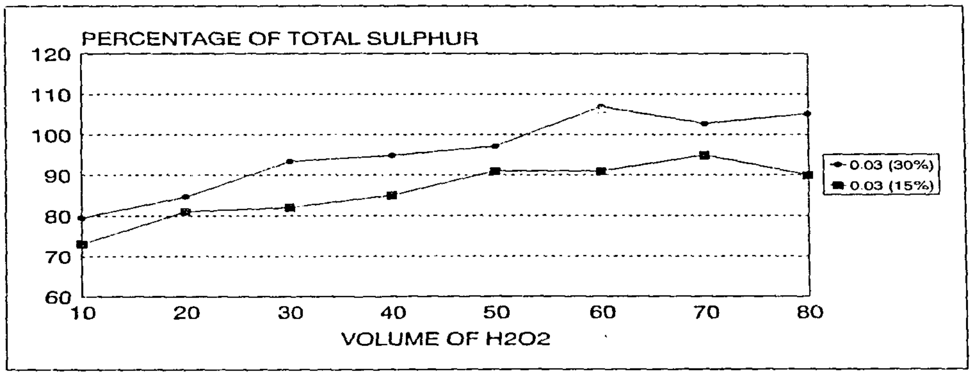


Figure B.2. A comparison of the pyritic sulphur recovery for diluted and concentrated peroxide.

B.2.2 EXPERIMENTAL PROCEDURE

The field and laboratory procedure for the determination of the acid potential were as follows:

- Core samples from boreholes, representing the total stratigraphy, were logged, subdivided into units and individually crushed in a jaw crusher.
- A disc mill was used to pulverise the crushed fragments to <325 mesh size.
- A 5 g sample was weighed into a beaker and 50 m*l* distilled water was added, stirred for 5 minutes by magnetic stirrer and left to stand for 24 hours. The solution was filtered and a fraction was kept aside for pH-measurement. The remainder was acidified with nitric acid to a pH below 2 and analysed by ICP.
- A second 5 g powder sample was weighed into a beaker and 50 m*l* concentrated hydrogen peroxide (30%) was added. The beaker was covered with a watch glass to prevent excessive volume loss due to boiling. For samples with estimated sulphur content above 3%, a 1 g sample was used and the peroxide was added in 10 m*l* increments to prevent violent reactions. Once the beaker cooled, the sample was covered with parafilm and left to stand for 24 hours. This waiting period allows the resultant solution to equilibrate with the minerals present in the spoil. The solution was filtered and a fraction was kept aside for the pH-measurement. The remainder was acidified with nitric acid to a pH below 2 and analysed by ICP method.

An additional benefit of the hydrogen peroxide oxidation of the samples is that an elemental distribution of the heavy metals, which are released during the oxidation process, may be obtained. In mines that could become acid, knowledge of the potential heavy metal release is an important component in the prediction of the long-term leachate chemistry.

B.3 BASE POTENTIAL EXPERIMENTS

Base potential experiments quantify the neutralising capacity of the spoil. Two important aspects, which should be considered during this experiment, are:

- Ineffective or temporary acid-consuming minerals should be excluded from the base potential, and
- Base potential minerals should be capable of buffering mine drainage to an environmentally acceptable pH.

Although various methodologies for the determination of the base potential of spoils were experimented with during this investigation, only one is recommended in this document, based on practical considerations.

B.3.1 STANDARDISATION OF THE EXPERIMENT

Standardisation of the procedure was as follows:

- Excess standardised sulphuric acid was added to different weights of analytical grade $CaCO_3$ and the samples were left to stand for 24 hours. The solution was subsequently titrated with standardised sodium hydroxide to a pH of 7. The pH-level of 7 is the recommended endpoint for the titration, although not all the acid-consuming minerals are capable of buffering solutions to this level (Steffen, Robertson and Kirsten, 1989). The temporary base potential of the heavy metal precipitates is removed at this point. Statistics on the efficiency of this procedure are summarised in Table B.1.

Table B.1. Statistics on the repeatability of base potential determinations for analytical grade $CaCO_3$.

Statistics	Value
Mean (% CaCO3 base potential recovery)	99
Standard Deviation (% CaCO3 base potential recovery)	3
Minimum (% CaCO3 base potential recovery)	98
Maximum (% CaCO3 base potential recovery)	100
Number of tests	8

The 99% base potential recovery, as opposed to 100%, is attributed to the difference between the equilibrium pH of water with $CaCO_3$ for open systems, which is 8,3 and the experimental endpoint of 7.

The methodologies for the determination of acid and base potential are based on reaction efficiencies of the same order. This facilitates the direct comparison of the parameters determined by these techniques.

B.3.2 EXPERIMENTAL PROCEDURE

- Core samples from boreholes, representing the total stratigraphy, were logged, subdivided into units and individually crushed in a jaw crusher.
- A disc mill was used to pulverise the crushed fragments to <325 mesh size.
- A 5 g sample was weighed into a beaker and 15 m*l* of 0,06N standardised sulphuric was added. The beaker was covered with parafilm and allowed to stand for 24 hours whereupon the pH was recorded by using a paper test strip. Additional acid was added if the sample still registered a pH above 3. The procedure was repeated until the pH-level was below 2.
- After an additional 24 hours, the sample was titrated with standardised sodium hydroxide to a pH of 7.
- Once the solution had stabilised at a pH of 7, the beaker was covered with parafilm and the samples were left to stand for a week. This allowed sufficient time for the precipitation of heavy metals, such as aluminium, copper, iron and zinc.
- The pH of the solution was checked after this period and additional base was added to raise the pH to 7 where necessary. The base potential was calculated once the solution was in equilibrium with the sample at a pH of 7.

The above procedure eliminates the effect of minerals, which only contribute a temporary base potential to the system such as siderite, clay minerals and ferric hydroxide.

B.4 ACID-BASE ACCOUNTING CALCULATIONS

The acid potential of a spoil sample is the amount of calcium carbonate that will be consumed by the acidity of the sample according to the following reaction:

$FeS_2 + 2CaCO_3 + 3{,}75O_2 + 1{,}5H_2O = Fe(OH)_3 + 2SO_4^{2-} + 2Ca^{2+} + 2CO_2$

The net neutralising potential (NNP) is determined by subtracting the acid potential (as $CaCO_3$) from the base potential (as $CaCO_3$). If the NNP is negative, the leachate will be acid, whereas a positive NNP indicates alkaline water.

It is convenient to recalculate the acid and base potentials in units of kilogram $CaCO_3$ per tonne of spoil. The amount of base additive which is required for the neutralisation of acidic environments may easily be calculated from these NNP units and compensated for purity by an efficiency factor. This provides an application quantity for the base additive, in terms of kg base additive per ton of spoil.

The above calculation is valid for open systems. However, the spoil may also behave as a semi-closed or closed system. Some or all of the carbon dioxide will dissolve under these conditions as explained in Appendix A. The reaction may then be written as follows:

$FeS_2 + 4CaCO_3 + 3{,}75O_2 + 3{,}5H_2O = Fe(OH)_3 + 2SO_4^{2-} + 4Ca^{2+} + 4HCO_3^-$

in which case twice the base potential will be required to neutralise all the acidity within the spoil.

The conclusion is drawn that the base potential of the spoil should exceed the acid potential by a factor of more than two to compensate for the dissolution of carbon dioxide.

B.5 TOTAL ACID-BASE POTENTIAL FOR A MINE

The calculation of the total acid-base potential for the total mine is the next stage. The following is a step by step procedure list:

- Relate the laboratory acid-base values to the field situation by calculating the acid-base distribution for the mine, based on lithostratigraphic information available from prospect boreholes. This value is calculated as follows. The normalised strata NNP is obtained by multiplying the sample NNP with the normalised strata thickness. Strata thickness is only available where core samples could be obtained. The mine NNP is obtained by summing all the normalised strata NNP's.
- Quantify each source of water to the pit, and calculate dilution factors to various portions of the pit.
- Identify areas and calculate the volumes of spoil which will be flooded when the pit has filled to its maximum capacity, after pit closure. This spoil will not contribute to the total acid potential of the pit provided that the flooded material does not come into contact with acidic ferric-rich waters.
- Subdivide the pit into areas of uniform spoil characteristics. Delineate areas where coal discards, coal slurry or other material with a high acid-generating potential have been dumped. The latter will be so-called 'hot spots' where significant acid generation will be initiated.
- Identify mixing cells and flow paths within the spoil, along which spoil waters of different compositions will flow and mix. Apply chemical equilibrium models to each cell, to predict the salinity and elemental compositions for the spoil water as it evolves as a result of mixing, on its way to the decanting point of the pit.

In conclusion, it may be said that the laboratory acid-base experiments constitute but a small portion of the total work involved in the prediction of the acid-base potential and the long-term chemistry for an opencast pit.

Each pit is different and the many factors that influence the final outcome of the pit chemistry contribute to varying degrees in each instance.

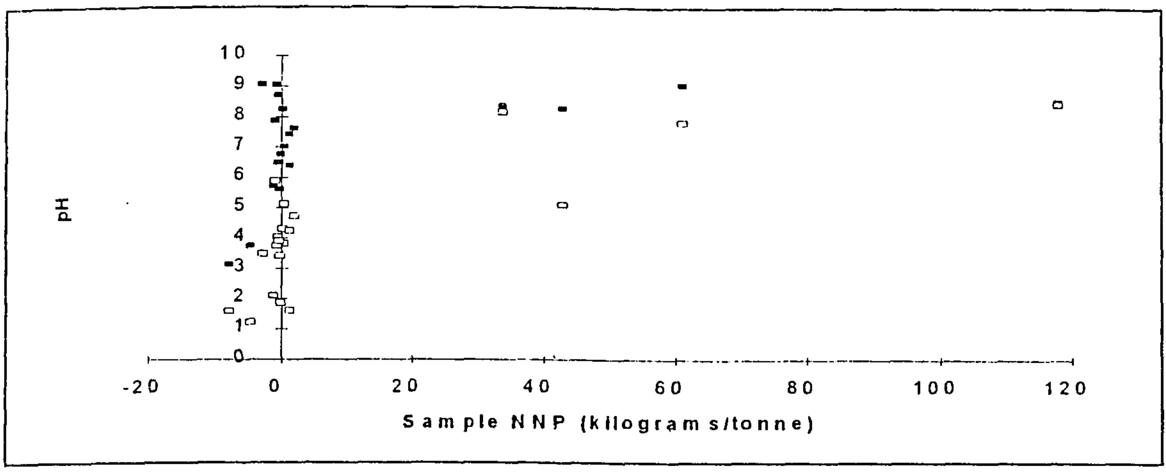
Appendix C

ACID-BASE INFORMATION

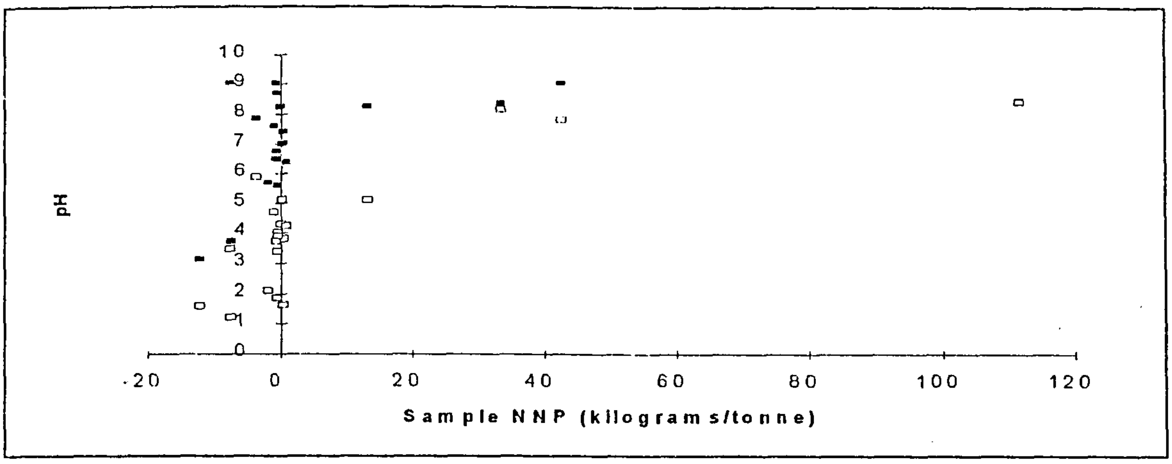
This Appendix contains the graphs and tables for the acid-base determinations at individual mines. The following is a short discussion of the format for the graphs and tables:

- The first two graphs for each mine depict the drop in pH during oxidation of the sulphide minerals within the pulverised samples. The first graph shows the drop in pH under open conditions, i.e. carbon dioxide is allowed to exsolve. The second graph is for confined conditions, i.e. the carbon dioxide is contained within the reaction environment and allowed to dissolve into the water. The NNP for each sample is plotted on the x-axis. Theoretically, a positive NNP should be reflected by an alkaline final solution pH. Correspondingly, a negative NNP should be reflected by an acidic final pH. Most samples follow this trend. However, there are some exceptions. Four of the samples with slightly positive NNP's still went acid during the oxidation tests. The close approximation of the remainder of the experimental data, to the theoretical relationships, confirms the accuracy of the experimental assumptions and procedures.
- The normalised strata NNP is obtained by mutiplying the sample NNP with the normalised strata thickness. The strata thickness is only available where core samples could be obtained. The normalised strata NNP's for all the mines could therefore not be calculated. The mine NNP is obtained by summing all the normalised strata NNP's.
- The tables include statistics on elements that are released by dissolution and the complete oxidation of the spoil material. Since minerals such as cadmium, copper, iron, nickel and zinc commonly occur as sulphides, they are entirely dissolved during oxidation. Aluminium commonly occurs in nature as hydroxides and silicates. Its solubility is therefore a function of the final pH of the solution. The concentrations of the aluminium in the tables therefore do not reflect the entire quantity of aluminium that may be released under field conditions.

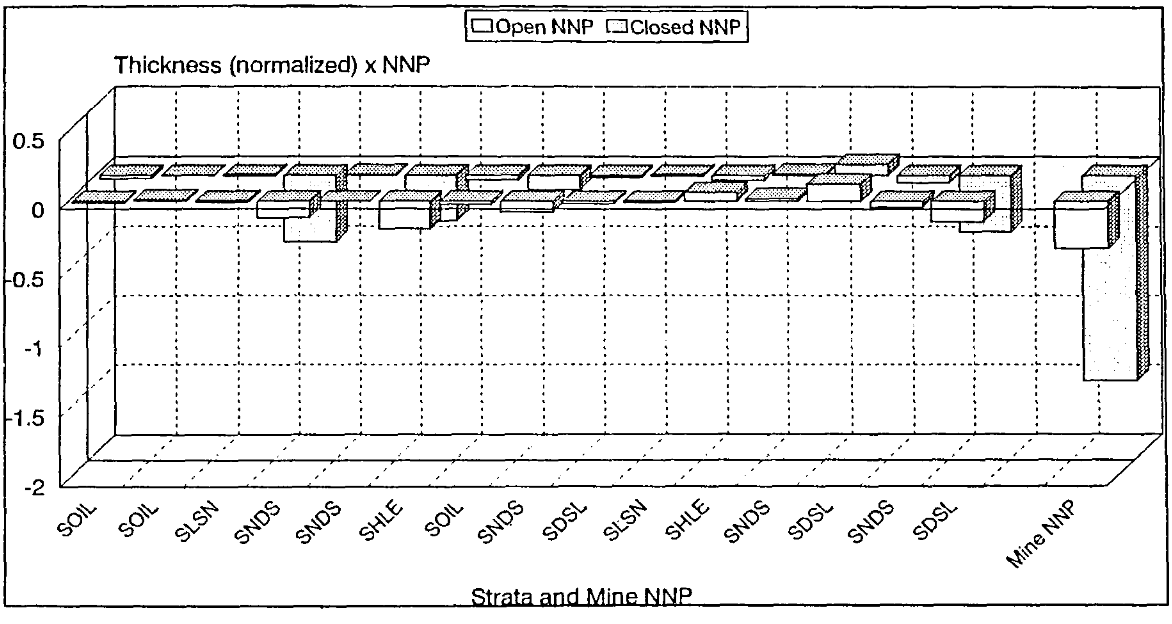
ACID-BASE INFORMATION FOR MINE 1



Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Normalised thicknesses times NNP for individual strata and for the total mine.

Statistics on soluble elements within the spoil samples.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g
Mean	0.00865	0.00000	0.00189	0.19803	0.00000	0.00034	0.00048	0.00678	0.03834
Median	0.00000	0.00000	0.00054	0.12154	0.00000	0.00000	0.00000	0.00120	0.01901
Std. Dev.	0.03545	0.00000	0.00589	0.28065	0.00000	0.00098	0.00158	0.01913	0.05795
Minimum	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000
Maximum	0.16271	0.00000	0.02746	1.15814	0.00000	0.00429	0.00647	0.08659	0.24899
Count	21	21	21	21	21	21	21	21	21

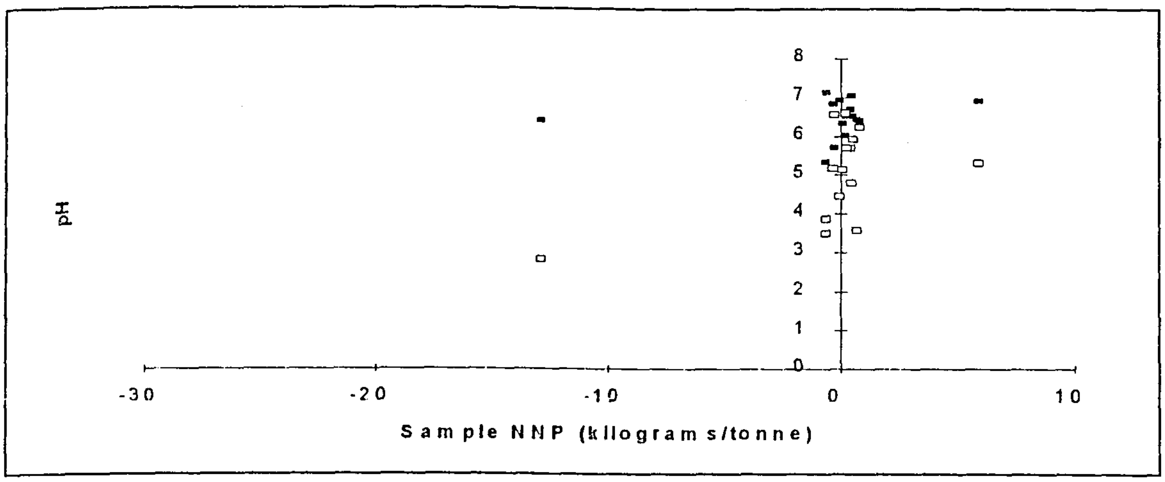
Statistics	Mg mg/g	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	Cr mg/g	
Mean	0.07119	0.01088	0.05230	0.00062	0.00079	0.01020	0.24662	0.00340	
Median	0.06036	0.00092	0.04131	0.00000	0.00064	0.00075	0.11748	0.00000	
Std. Dev.	0.07480	0.03304	0.02344	0.00282	0.00080	0.03118	0.30355	0.01533	
Minimum	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	
Maximum	0.23100	0.15232	0.08149	0.01294	0.00263	0.14108	0.95247	0.07030	
Count	21	21	21	21	21	21	21	21	

Statistics on elements released during complete oxidation of spoil samples, NNP and pH change.

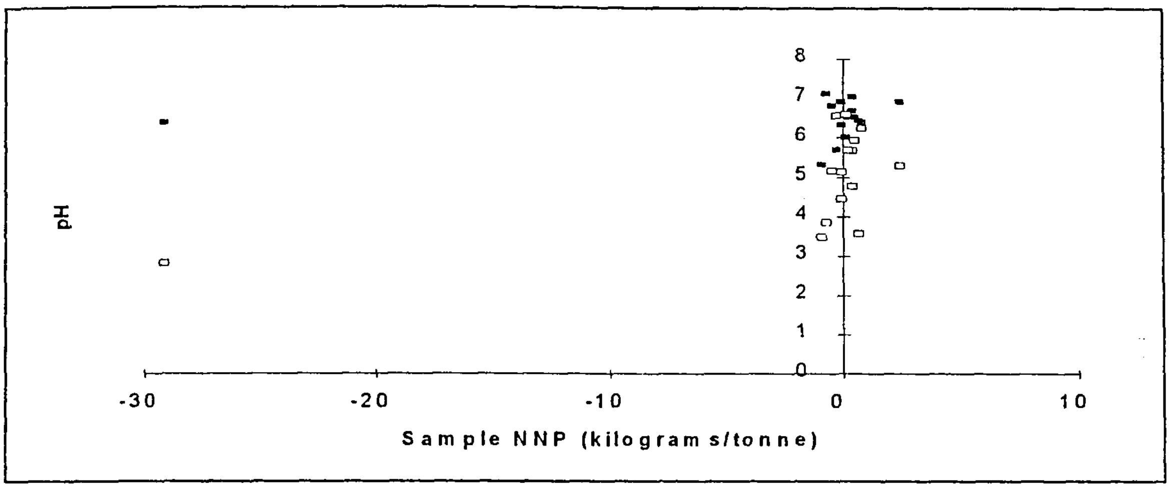
Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.09884	0.00049	0.00391	0.60389	0.00010	0.00136	0.00397	0.54705	0.41928	0.4754
Median	0.04789	0.00000	0.00120	0.16715	0.00000	0.00063	0.00102	0.01283	0.02337	0.07698
Std. Dev.	0.11340	0.00096	0.00923	1.26799	0.00024	0.00229	0.00575	0.96786	1.82343	1.24038
Minimum	0.00000	0.00000	0.00000	0.02956	0.00000	0.00000	0.00000	0.00000	0.00000	0.00314
Maximum	0.36506	0.00283	0.04228	5.59392	0.00089	0.00954	0.02137	4.17202	8.37690	5.61744
Count	21	21	21	21	21	21	21	21	21	21

Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP close	Initial pH	Final pH
Mean	0.02234	0.07616	0.00181	0.00304	1.61771	1.17966	3.90142	0.21498	7.17	4.2
Median	0.01058	0.07724	0.00000	0.00068	0.00234	0.17853	-0.27071	-0.57596	7.41	3.9
Std. Dev.	0.03547	0.03810	0.00426	0.00531	7.33372	2.32144	13.51151	13.31977	1.64	2.08
Minimum	0.00000	0.00806	0.00000	0.00028	0.00000	0.00583	-6.15101	-23.05917	3.14	1.21
Maximum	0.15702	0.14922	0.01434	0.02006	33.62429	9.48889	55.86866	49.64347	9.06	8.45
Count	21	21	21	21	21	21	21	21	21	21

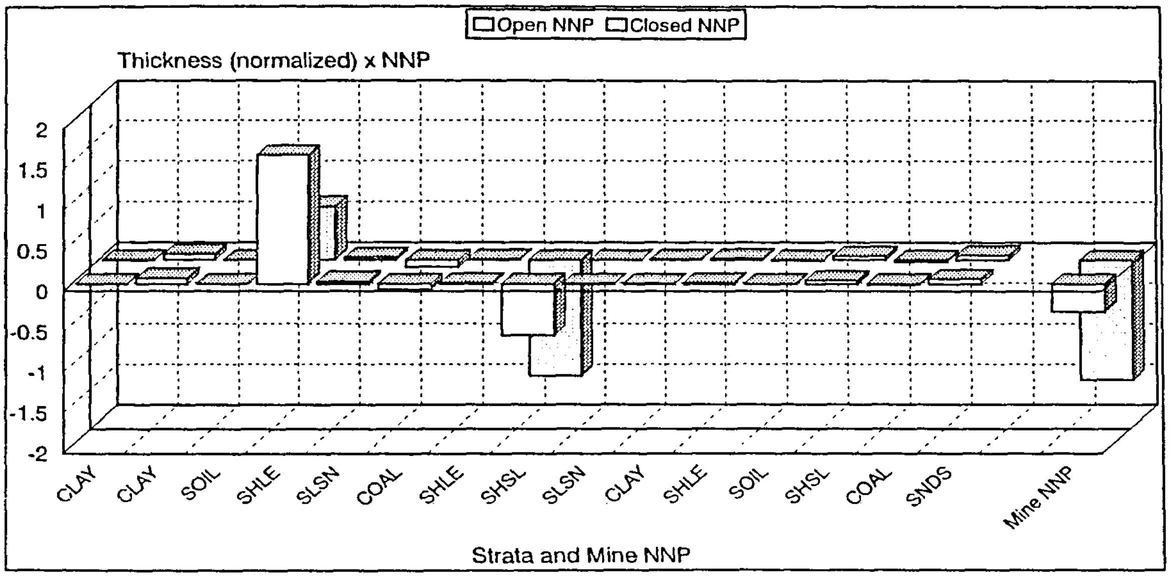
ACID-BASE INFORMATION FOR MINE 2



Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Normalised thicknesses times NNP for individual strata and for the total mine.

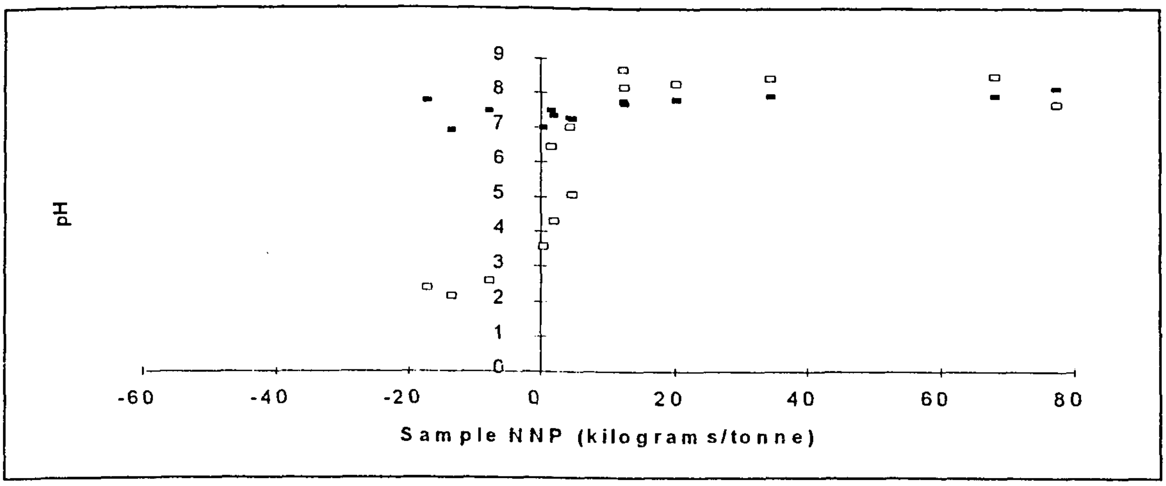
Statistics on soluble elements within the spoil samples.

Statistics	**Al mg/g**	**B mg/g**	**Ba mg/g**	**Ca mg/g**	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.00355	0.00000	0.00043	0.03833	0.00001	0.00007	0.00039	0.00145	0.04804	0.01898
Median	0.00000	0.00000	0.00041	0.02674	0.00000	0.00000	0.00042	0.00000	0.04421	0.01154
Std. Dev.	0.00882	0.00000	0.00029	0.02415	0.00004	0.00012	0.00011	0.00275	0.01314	0.02375
Minimum	0.00000	0.00000	0.00007	0.01275	0.00000	0.00000	0.00019	0.00000	0.03282	0.0047
Maximum	0.03193	0.00000	0.00119	0.09147	0.00015	0.00040	0.00056	0.00728	0.07506	0.09739
Count	15	15	15	15	15	15	15	15	15	15
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g				
Mean	0.00079	0.07054	0.00039	0.00034	0.00033	0.04695				
Median	0.00050	0.04669	0.00037	0.00019	0.00021	0.02004				
Std. Dev.	0.00092	0.04463	0.00041	0.00037	0.00040	0.06171				
Minimum	0.00000	0.03579	0.00000	0.00008	0.00000	0.00668				
Maximum	0.00324	0.19182	0.00143	0.00158	0.00172	0.20814				
Count	15	15	15	15	15	15				

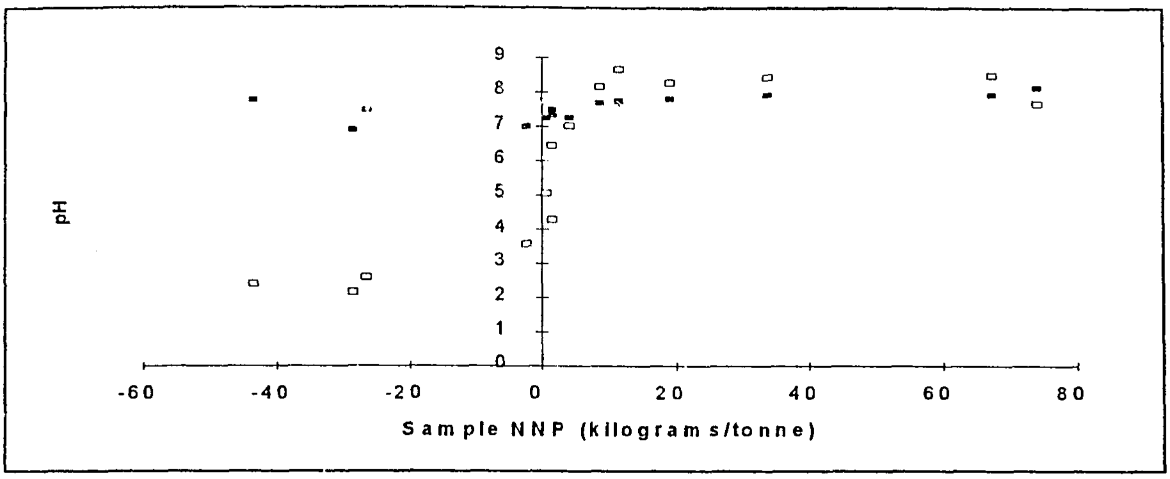
Statistics on elements released during complete oxidation of spoil samples, NNP and pH change.

Statistics	**Al mg/g**	**B mg/g**	**Ba mg/g**	**Ca mg/g**	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.10913	0.00000	0.01051	0.41299	0.00025	0.00236	0.00365	0.79198	0.04860	0.16845
Median	0.02586	0.00000	0.00229	0.05373	0.00000	0.00048	0.00006	0.00245	0.03197	0.02605
Std. Dev.	0.19292	0.00000	0.02255	0.86253	0.00052	0.00512	0.00741	1.89943	0.04497	0.28952
Minimum	0.00000	0.00000	0.00000	0.01038	0.00000	0.00000	0.00000	0.00000	0.00088	0.00459
Maximum	0.70982	0.00000	0.08431	2.87703	0.00156	0.01576	0.02112	5.95217	0.15962	0.83162
Count	15	15	15	15	15	15	15	15	15	15
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP Close	Initial pH	Final pH
Mean	0.02386	0.04000	0.00695	0.00328	0.01073	0.44039	-0.87002	-2.24622	6.48	5.02
Median	0.01362	0.03061	0.00000	0.00049	0.00045	0.01990	0.06182	-0.05267	6.52	5.16
Std. Dev.	0.03112	0.03699	0.01644	0.00590	0.02614	1.34644	3.81134	7.93239	0.5	1.17
Minimum	0.00073	0.00000	0.00000	0.00006	0.00000	0.00234	-14.57525	-30.82741	5.33	2.81
Maximum	0.12001	0.12213	0.05268	0.01872	0.08885	5.20069	1.21161	0.36912	7.12	6.59
Count	15	15	15	15	15	15	15	15	15	15

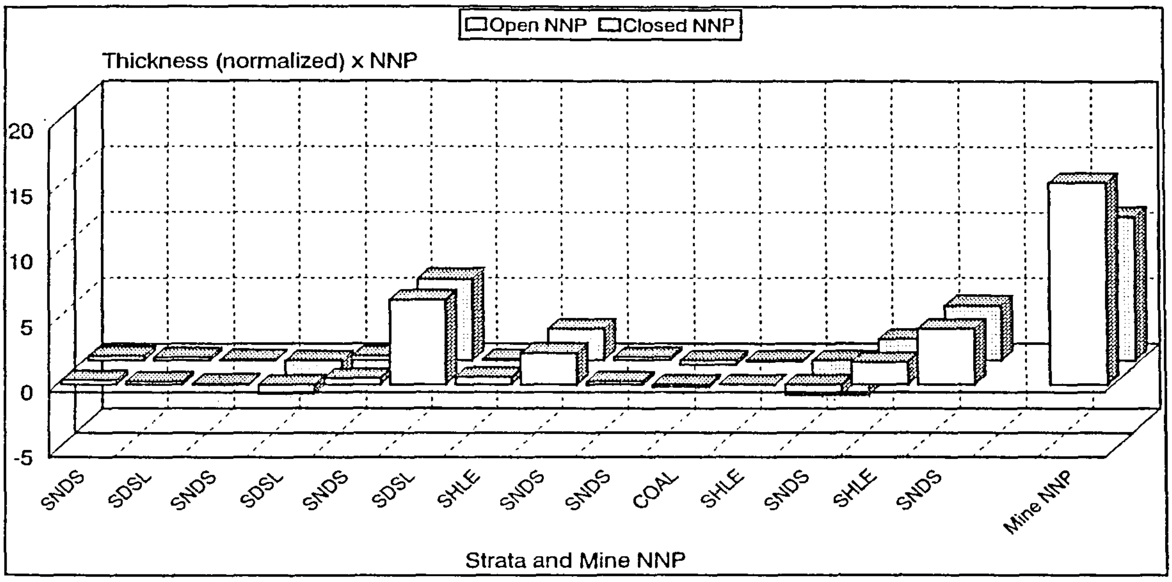
ACID-BASE INFORMATION FOR MINE 3



Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Normalised thicknesses times NNP for individual strata and for the total mine.

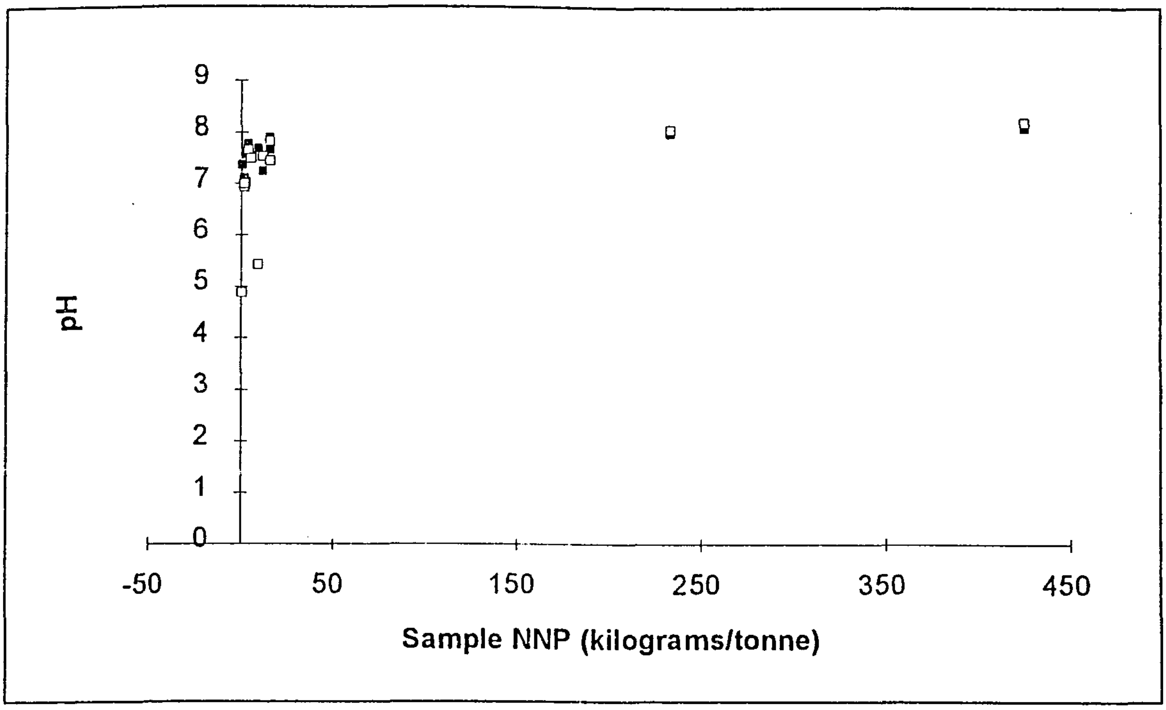
Statistics on soluble elements within the spoil samples.

Statistics	**Al mg/g**	**B mg/g**	**Ba mg/g**	**Ca mg/g**	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.00212	0.00000	0.00041	0.06082	0.00000	0.00006	0.00038	0.00060	0.03848	0.01612
Median	0.00086	0.00000	0.00026	0.05460	0.00000	0.00000	0.00040	0.00000	0.03748	0.01603
Std. Dev.	0.00337	0.00000	0.00032	0.02990	0.00000	0.00015	0.00020	0.00225	0.01400	0.01046
Minimum	0.00000	0.00000	0.00010	0.01700	0.00000	0.00000	0.00000	0.00000	0.01636	0.00259
Maximum	0.01236	0.00000	0.00108	0.11452	0.00000	0.00054	0.00073	0.00843	0.06026	0.03184
Count	14	14	14	14	14	14	14	14	14	14
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g				
Mean	0.00018	0.09832	0.00041	0.00067	0.00005	0.07200				
Median	0.00000	0.08791	0.00040	0.00077	0.00000	0.06257				
Std. Dev.	0.00039	0.05034	0.00032	0.00048	0.00018	0.04690				
Minimum	0.00000	0.03207	0.00000	0.00009	0.00000	0.01764				
Maximum	0.00145	0.20084	0.00103	0.00167	0.00068	0.16868				
Count	14	14	14	14	14	14				

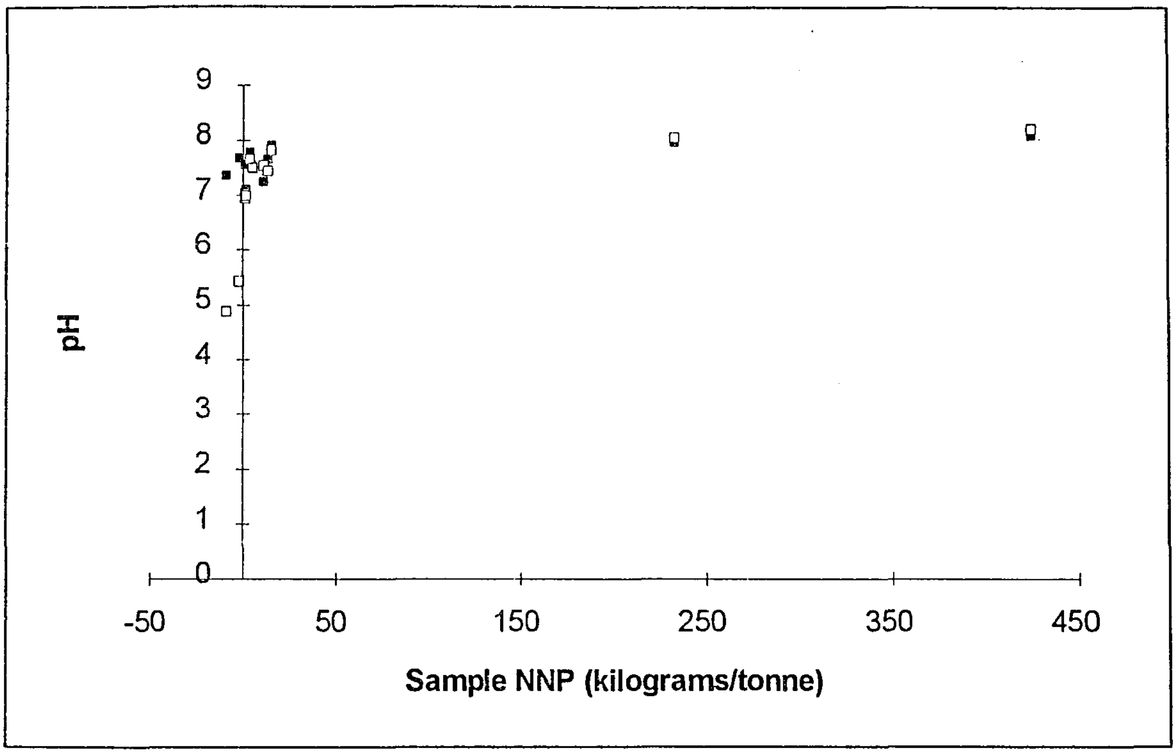
Statistics on elements released during complete oxidation of spoil samples, NNP and pH change.

Statistics	**Al mg/g**	**B mg/g**	**Ba mg/g**	**Ca mg/g**	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.05311	0.00000	0.00359	0.58360	0.00035	0.00233	0.01547	1.69636	0.05091	0.27868
Median	0.02838	0.00000	0.00133	0.48325	0.00000	0.00101	0.00399	0.01949	0.04482	0.27268
Std. Dev.	0.05878	0.00000	0.00829	0.39971	0.00060	0.00294	0.02072	3.05668	0.03632	0.18942
Minimum	0.00000	0.00000	0.00015	0.07585	0.00000	0.00000	0.00000	0.00409	0.00595	0.0399
Maximum	0.17725	0.00000	0.03220	1.48086	0.00213	0.01054	0.06321	9.90252	0.13582	0.63364
Count	14	14	14	14	14	14	14	14	14	14
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP close	Initial pH	Final pH
Mean	0.02214	0.01440	0.00480	0.00524	0.01402	1.78402	4.36452	-1.21055	7.54	5.93
Median	0.00957	0.00154	0.00138	0.00456	0.00510	0.61731	1.45018	0.60171	7.59	6.73
Std. Dev.	0.02996	0.01897	0.00593	0.00345	0.02086	2.64170	16.48151	22.85011	0.37	2.51
Minimum	0.00026	0.00000	0.00000	0.00046	0.00024	0.01586	-21.69110	-47.92513	6.89	2.14
Maximum	0.10549	0.05533	0.01688	0.01106	0.06881	8.39489	37.07621	33.80101	8.12	8.67
Count	14	14	14	14	14	14	14	14	14	14

ACID-BASE INFORMATION FOR MINE 4



Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.

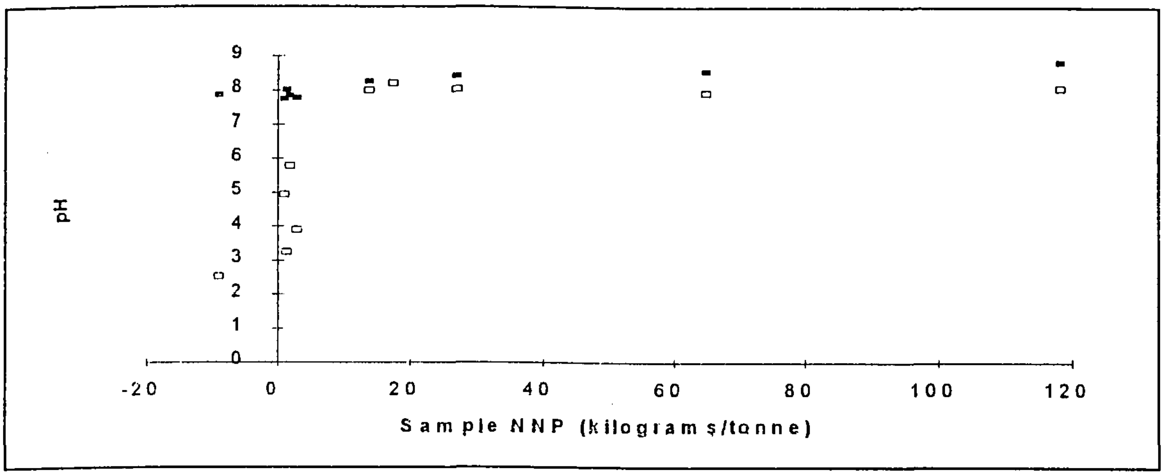
Statistics on soluble elements within the spoil samples.

Statistics	**Al mg/g**	**B mg/g**	**Ba mg/g**	**Ca mg/g**	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.00010	0.00027	0.00043	0.16673	0.00000	0.00000	0.00039	0.00024	0.07349	0.04416
Median	0.00000	0.00000	0.00024	0.16266	0.00000	0.00000	0.00040	0.00000	0.06093	0.02873
Std. Dev.	0.00036	0.00093	0.00043	0.10873	0.00000	0.00000	0.00013	0.00055	0.03770	0.03937
Minimum	0.00000	0.00000	0.00000	0.00547	0.00000	0.00000	0.00015	0.00000	0.03696	0.00194
Maximum	0.00125	0.00321	0.00149	0.31827	0.00000	0.00000	0.00058	0.00148	0.14712	0.11829
Count	12	12	12	12	12	12	12	12	12	12
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g				
Mean	0.00016	0.06847	0.00050	0.00168	0.00007	0.14204				
Median	0.00005	0.05328	0.00047	0.00093	0.00000	0.02766				
Std. Dev.	0.00029	0.04549	0.00022	0.00182	0.00012	0.16644				
Minimum	0.00000	0.03760	0.00000	0.00003	0.00000	0.00000				
Maximum	0.00100	0.20574	0.00082	0.00626	0.00034	0.46764				
Count	12	12	12	12	12	12				

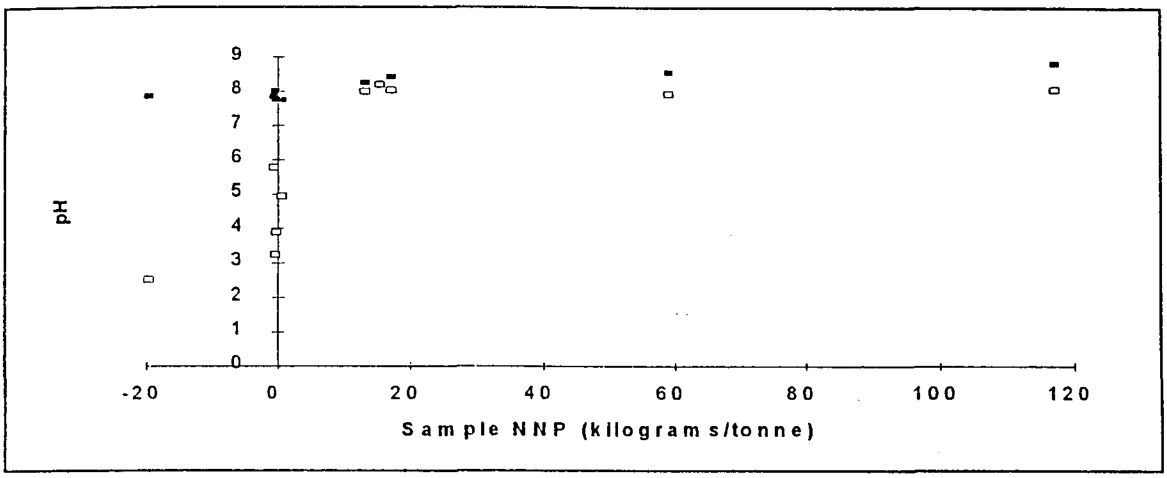
Statistics on elements released during complete oxidation of spoil samples,
NNP and pH change.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.07675	0.00000	0.00163	0.50233	0.00006	0.00075	0.00043	0.14536	0.07971	0.23469
Median	0.01780	0.00000	0.00044	0.33650	0.00000	0.00018	0.00000	0.02097	0.05510	0.1014
Std. Dev.	0.12059	0.00000	0.00216	0.51309	0.00012	0.00139	0.00079	0.30742	0.08159	0.38648
Minimum	0.00130	0.00000	0.00003	0.02627	0.00000	0.00000	0.00000	0.00250	0.00570	0
Maximum	0.36834	0.00000	0.00612	1.47314	0.00031	0.00394	0.00207	1.00852	0.27449	1.27828
Count	12	12	12	12	12	12	12	12	12	12
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP close	Initial pH	Final pH
Mean	0.01826	0.06527	0.00159	0.00556	0.00425	0.66634	29.19370	27.11138	7.58	7.12
Median	0.00158	0.04044	0.00000	0.00160	0.00056	0.04062	2.20133	2.14554	7.61	7.47
Std. Dev.	0.03374	0.04893	0.00406	0.01108	0.00924	1.25799	66.59606	67.70982	0.34	1.01
Minimum	0.00000	0.02532	0.00000	0.00008	0.00000	0.00232	-4.47030	-13.94055	7.07	4.87
Maximum	0.11497	0.17378	0.01355	0.03936	0.03083	3.59782	212.68380	212.56449	8.09	8.21
Count	12	12	12	12	12	12	12	12	12	12

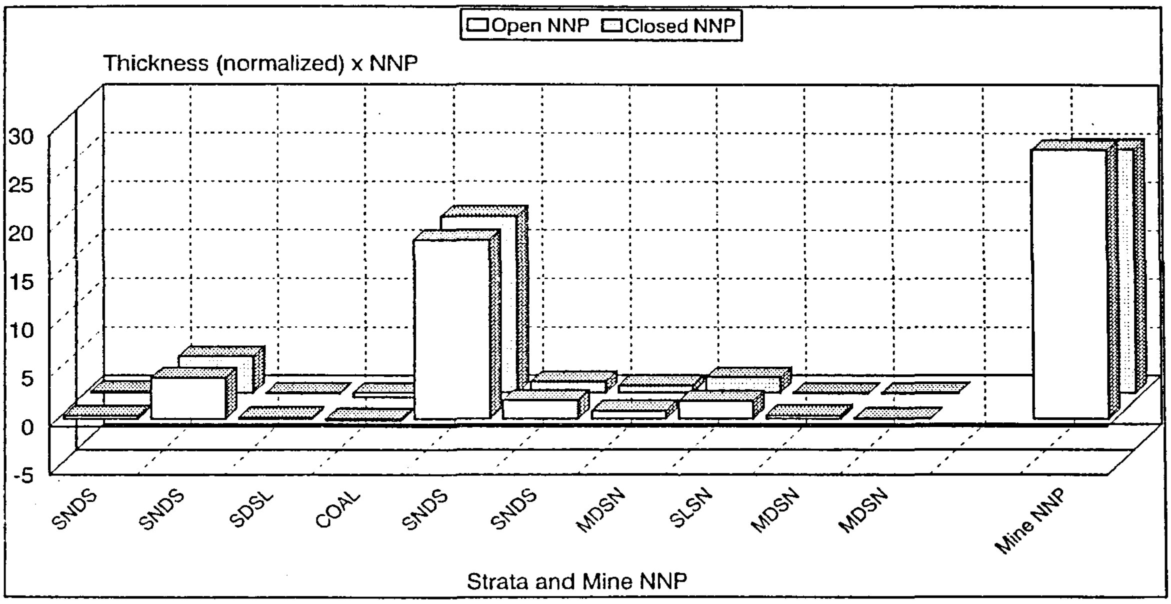
ACID-BASE INFORMATION FOR MINE 5



Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Normalised thicknesses times NNP for individual strata and for the total mine.

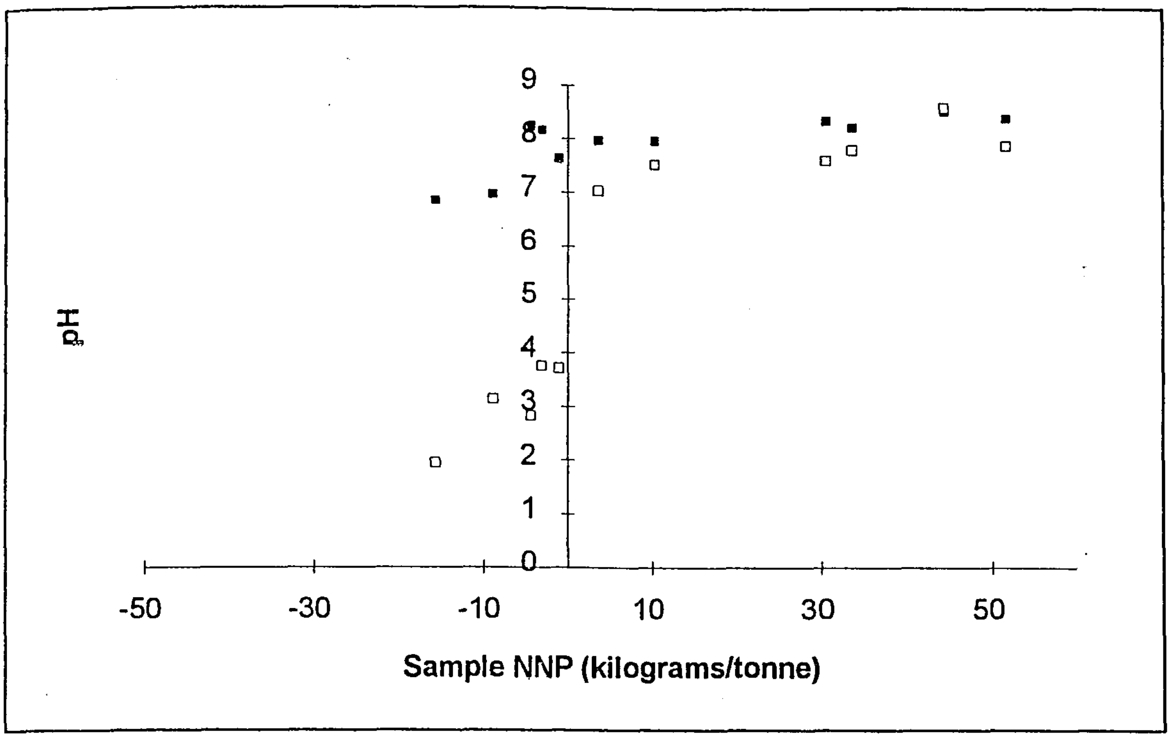
Statistics on soluble elements within the spoil samples.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.00379	0.00000	0.00120	0.09646	0.00000	0.00012	0.00000	0.00118	0.08365	0.01562
Median	0.00312	0.00000	0.00103	0.09483	0.00000	0.00008	0.00000	0.00077	0.08001	0.01262
Std. Dev.	0.00310	0.00000	0.00077	0.04753	0.00000	0.00014	0.00000	0.00187	0.02986	0.00997
Minimum	0.00000	0.00000	0.00020	0.04379	0.00000	0.00000	0.00000	0.00000	0.02685	0.00436
Maximum	0.00978	0.00000	0.00241	0.17762	0.00000	0.00032	0.00000	0.00608	0.12991	0.03268
Count	10	10	10	10	10	10	10	10	10	10
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g				
Mean	0.00004	0.08518	0.00039	0.00101	0.00033	0.04312				
Median	0.00000	0.07496	0.00041	0.00074	0.00027	0.04372				
Std. Dev.	0.00009	0.04332	0.00020	0.00065	0.00016	0.01612				
Minimum	0.00000	0.04984	0.00000	0.00026	0.00012	0.01506				
Maximum	0.00028	0.18939	0.00069	0.00193	0.00063	0.06636				
Count	10	10	10	10	10	10				

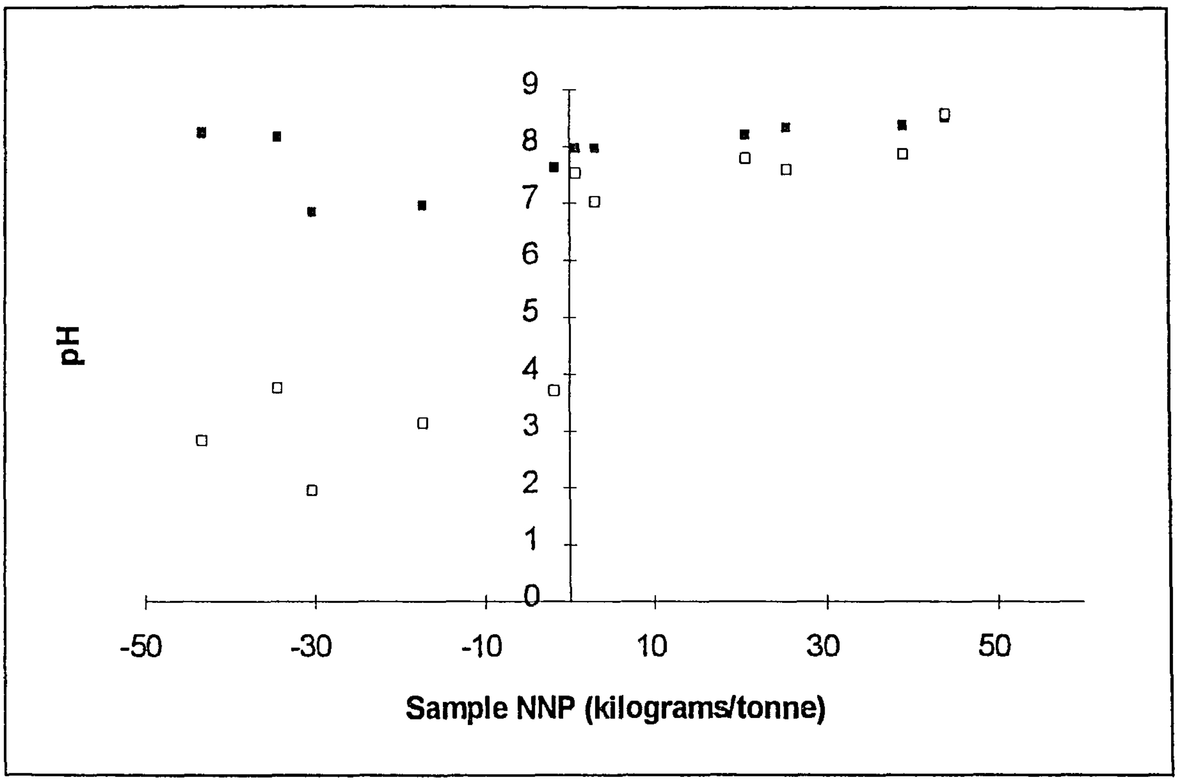
Statistics on elements released during complete oxidation of spoil samples,
NNP and pH change.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.08245	0.00000	0.00242	0.77389	0.00023	0.00351	0.00639	0.68895	0.01997	0.18896
Median	0.01905	0.00000	0.00133	0.58142	0.00000	0.00128	0.00043	0.00457	0.00952	0.15914
Std. Dev.	0.12322	0.00000	0.00311	0.67432	0.00036	0.00443	0.01162	1.35792	0.02614	0.14539
Minimum	0.00000	0.00000	0.00000	0.08585	0.00000	0.00000	0.00000	0.00000	0.00000	0.0219
Maximum	0.36009	0.00000	0.01046	2.24376	0.00102	0.01163	0.03599	3.82000	0.07044	0.49866
Count	10	10	10	10	10	10	10	10	10	10
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP close	Intial pH	Final pH
Mean	0.00809	0.02850	0.00441	0.00553	0.01999	1.22713	10.07177	6.23699	8.15	6.05
Median	0.00444	0.03174	0.00055	0.00524	0.00185	0.73844	3.46201	-0.80710	8.09	6.86
Std. Dev.	0.00949	0.02434	0.00647	0.00284	0.02947	1.19769	19.88879	21.02489	0.37	2.27
Minimum	0.00029	0.00000	0.00000	0.00121	0.00009	0.14558	-9.84462	-20.44193	7.75	2.51
Maximum	0.02968	0.06508	0.01523	0.01118	0.08231	3.39114	58.57112	57.44366	8.83	8.20
Count	10	10	10	10	10	10	10	10	10	10

ACID-BASE INFORMATION FOR MINE 6



Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.

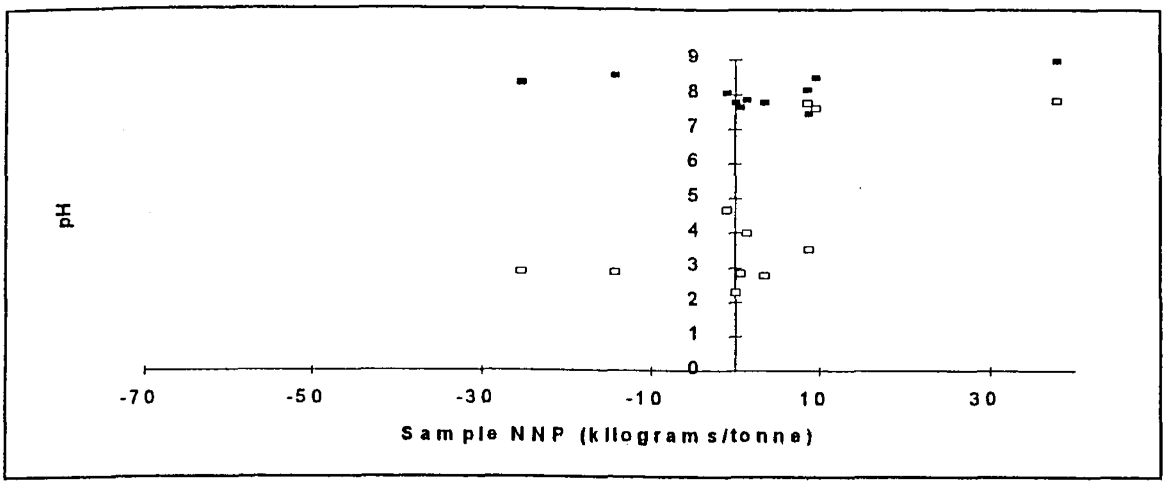
Statistics on soluble elements within the spoil samples.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.00067	0.00000	0.00181	0.10990	0.00000	0.00005	0.00000	0.00081	0.08711	0.02423
Median	0.00000	0.00000	0.00159	0.10510	0.00000	0.00000	0.00000	0.00000	0.08602	0.02354
Std. Dev.	0.00141	0.00000	0.00157	0.07979	0.00000	0.00009	0.00000	0.00204	0.03001	0.01558
Minimum	0.00000	0.00000	0.00020	0.03100	0.00000	0.00000	0.00000	0.00000	0.04785	0.00398
Maximum	0.00415	0.00000	0.00471	0.30272	0.00000	0.00024	0.00000	0.00676	0.14246	0.05499
Count	11	11	11	11	11	11	11	11	11	11
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g				
Mean	0.00023	0.11423	0.00024	0.00208	0.00039	0.08463				
Median	0.00000	0.10274	0.00021	0.00262	0.00030	0.07014				
Std. Dev.	0.00061	0.05239	0.00020	0.00156	0.00021	0.05119				
Minimum	0.00000	0.07042	0.00000	0.00014	0.00014	0.03165				
Maximum	0.00203	0.25679	0.00057	0.00431	0.00073	0.21640				
Count	11	11	11	11	11	11				

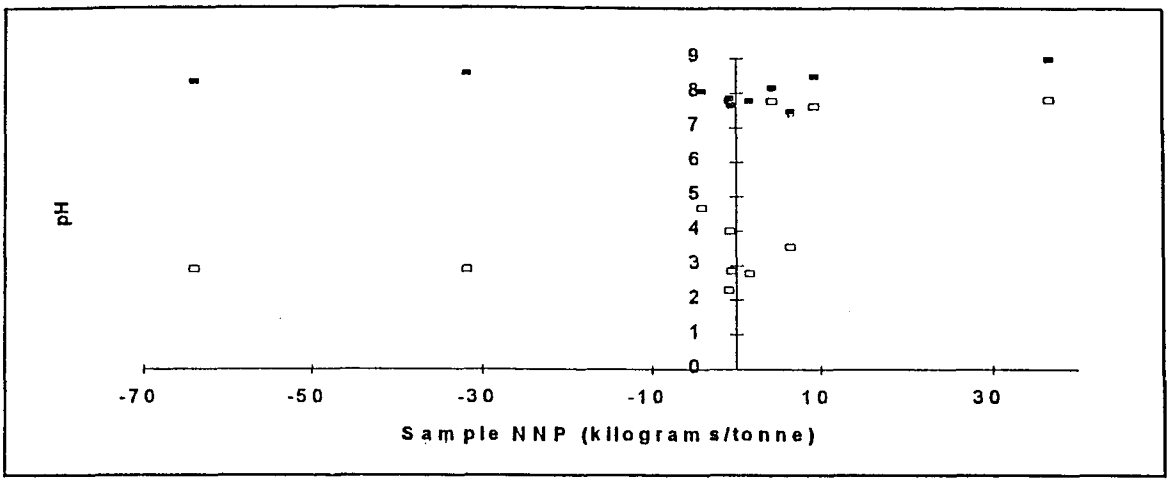
Statistics on elements released during complete oxidation of spoil samples,
NNP and pH change.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.03358	0.00000	0.00134	1.70753	0.00008	0.00163	0.00673	1.42629	0.01707	0.59996
Median	0.00880	0.00000	0.00026	1.39886	0.00000	0.00088	0.00163	0.00491	0.01537	0.29479
Std. Dev.	0.04592	0.00000	0.00229	1.59919	0.00026	0.00187	0.01126	1.97258	0.01769	0.65272
Minimum	0.00000	0.00000	0.00000	0.08227	0.00000	0.00000	0.00000	0.00000	0.00000	0.0205
Maximum	0.14734	0.00000	0.00600	4.48364	0.00087	0.00475	0.03791	4.57000	0.05641	2.03085
Count	11	11	11	11	11	11	11	11	11	11
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP close	Initial pH	Final pH
Mean	0.01828	0.01617	0.00427	0.02011	0.01054	3.93798	0.26844	-12.03773	7.93	5.61
Median	0.00187	0.00000	0.00236	0.00488	0.00598	3.07730	0.36608	-2.34843	8.16	7.03
Std. Dev.	0.02663	0.02555	0.00540	0.02263	0.01375	4.00994	14.85959	25.10868	0.57	2.51
Minimum	0.00000	0.00000	0.00000	0.00104	0.00008	0.10336	-21.65811	-60.69425	6.83	1.94
Maximum	0.07190	0.07216	0.01572	0.05746	0.04101	12.49156	21.99529	21.67230	8.52	8.58
Count	11	11	11	11	11	11	11	11	11	11

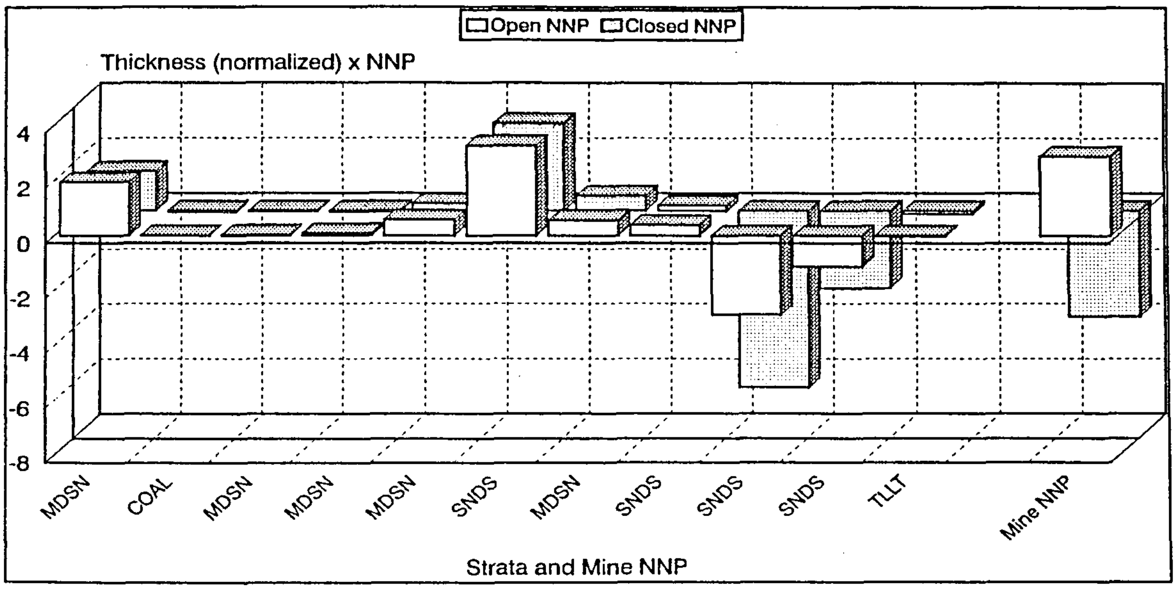
ACID-BASE INFORMATION FOR MINE 7



Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



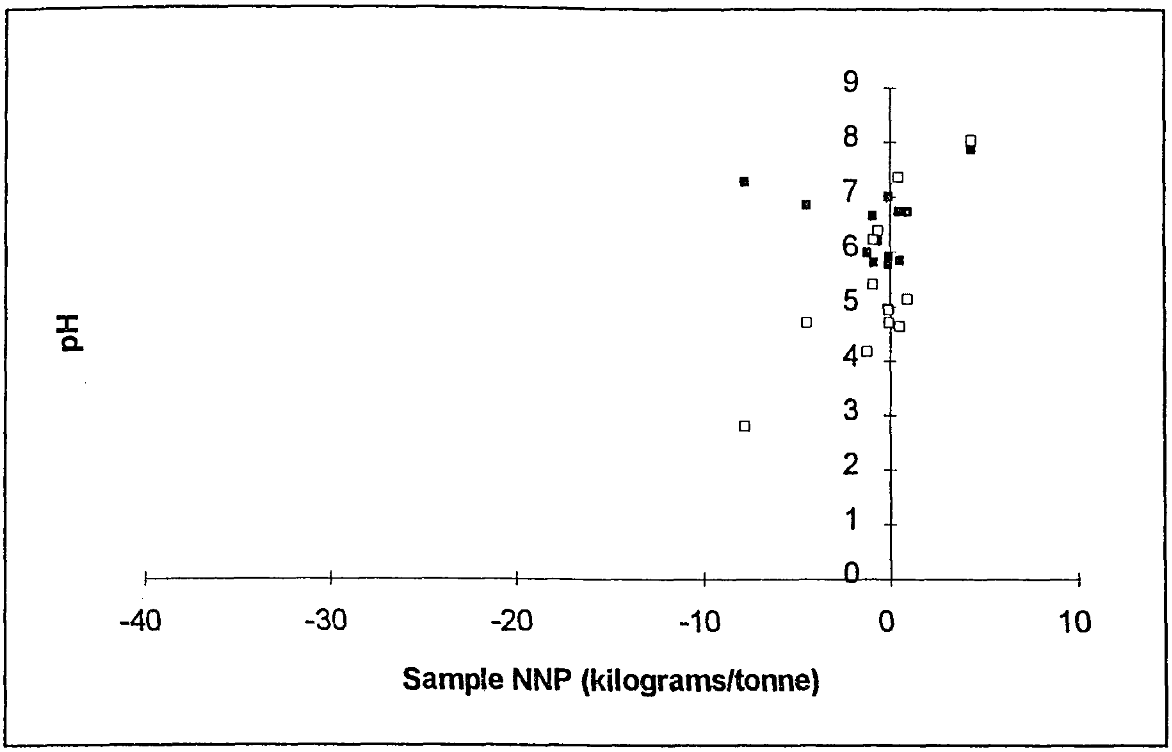
Normalised thicknesses times NNP for individual strata and for the total mine.

Statistics on soluble elements within the spoil samples.

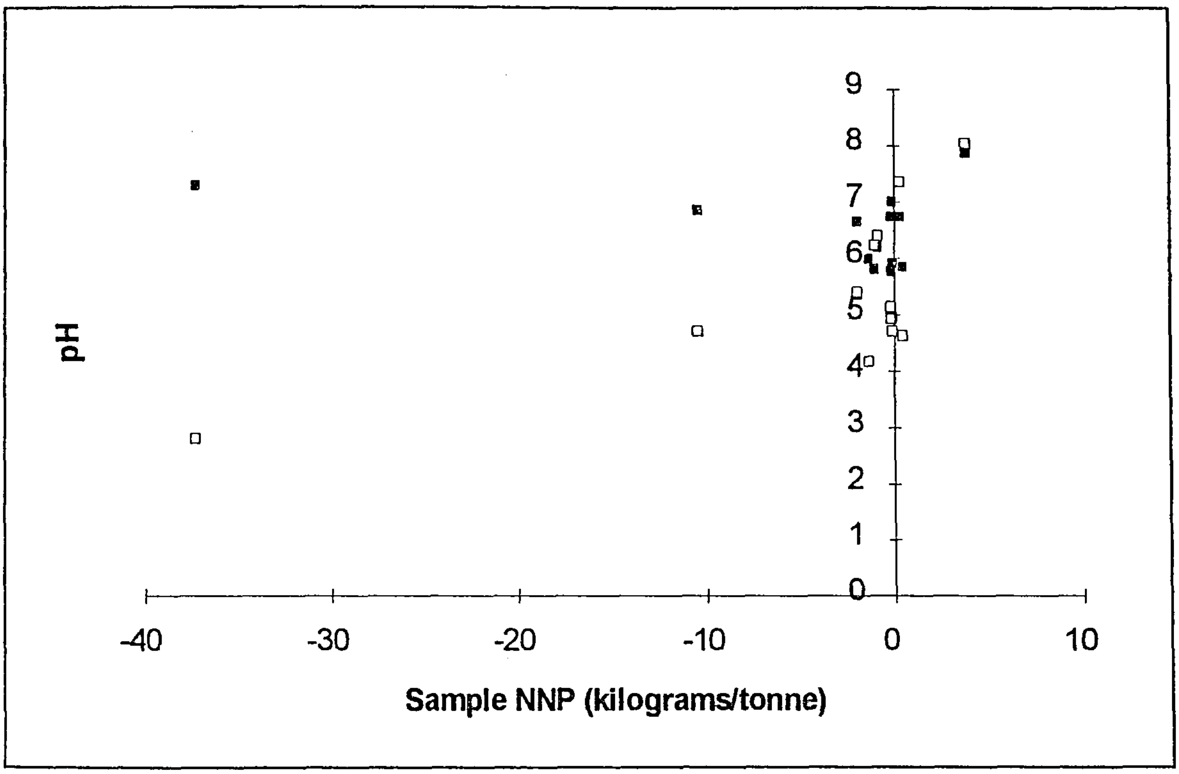
Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.00448	0.00000	0.00055	0.08262	0.00000	0.00007	0.00000	0.00051	0.07476	0.01119
Median	0.00365	0.00000	0.00042	0.08629	0.00000	0.00000	0.00000	0.00000	0.07173	0.01276
Std. Dev.	0.00330	0.00000	0.00029	0.04836	0.00000	0.00013	0.00000	0.00062	0.02371	0.00387
Minimum	0.00000	0.00000	0.00024	0.02334	0.00000	0.00000	0.00000	0.00000	0.03989	0.00554
Maximum	0.01024	0.00000	0.00101	0.19193	0.00000	0.00038	0.00000	0.00152	0.10565	0.0161
Count	11	11	11	11	11	11	11	11	11	11
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g				
Mean	0.00004	0.11255	0.00030	0.00045	0.00033	0.04572				
Median	0.00000	0.10319	0.00035	0.00049	0.00031	0.03467				
Std. Dev.	0.00007	0.04397	0.00022	0.00024	0.00017	0.02351				
Minimum	0.00000	0.04949	0.00000	0.00014	0.00000	0.02486				
Maximum	0.00020	0.20031	0.00062	0.00089	0.00051	0.09659				
Count	11	11	11	11	11	11				

Statistics on elements released during complete oxidation of spoil samples, NNP and pH change.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.26443	0.00000	0.00639	0.92643	0.00049	0.00339	0.01262	1.90265	0.04336	0.16235
Median	0.28186	0.00000	0.00545	0.59128	0.00016	0.00185	0.01079	0.59848	0.04350	0.13828
Std. Dev	0.24487	0.00000	0.00525	0.87367	0.00070	0.00505	0.01122	3.07165	0.03908	0.10135
Minimum	0.00000	0.00000	0.00114	0.37808	0.00000	0.00000	0.00000	0.00002	0.00000	0.06901
Maximum	0.81838	0.00000	0.01579	3.39592	0.00209	0.01704	0.03190	9.74848	0.11694	0.42073
Count	11	11	11	11	11	11	11	11	11	11
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP close	Initial pH	Final pH
Mean	0.03922	0.03157	0.00743	0.00698	0.02922	2.14683	-1.98135	-8.69019	8.08	4.45
Median	0.01227	0.02228	0.00400	0.00457	0.01430	0.65878	-0.28243	-1.52073	8.04	3.52
Std. Dev	0.06078	0.03397	0.00863	0.00751	0.04262	3.72884	12.70404	23.77030	0.45	2.20
Minimum	0.00113	0.00000	0.00000	0.00229	0.00025	0.11322	-32.01443	-70.70265	7.45	2.28
Maximum	0.20174	0.10460	0.02582	0.02910	0.14400	12.38023	18.36390	17.19593	8.96	7.79
Count	11	11	11	11	11	11	11	11	11	11



Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.

Statistics on soluble elements within the spoil samples.

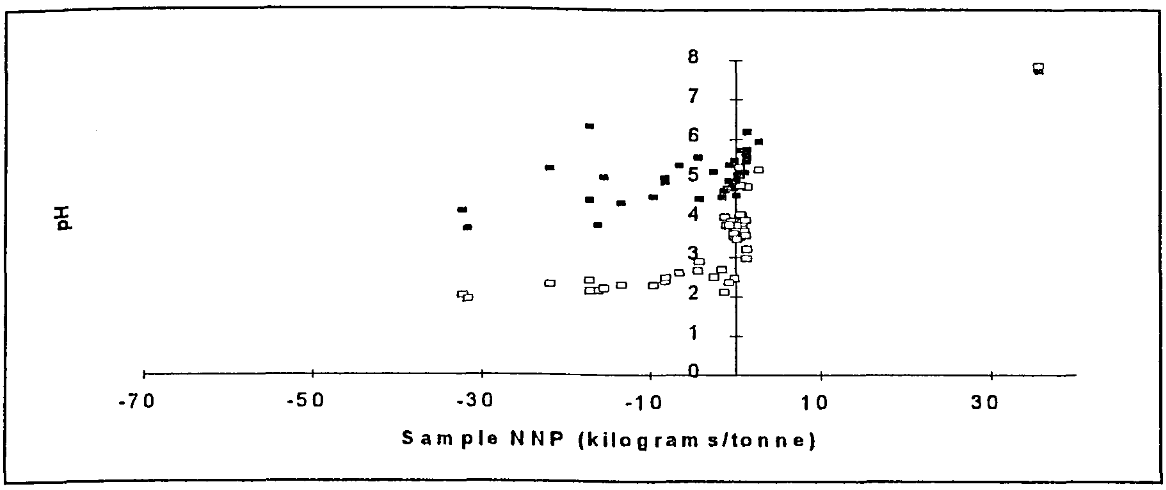
Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.00225	0.00006	0.00047	0.05169	0.00001	0.00000	0.00035	0.00071	0.03708	0.01864
Median	0.00000	0.00000	0.00039	0.01723	0.00000	0.00000	0.00037	0.00000	0.02951	0.00576
Std. Dev.	0.00469	0.00021	0.00041	0.06511	0.00005	0.00000	0.00014	0.00116	0.01740	0.02517
Minimum	0.00000	0.00000	0.00003	0.00498	0.00000	0.00000	0.00000	0.00000	0.01708	0.00082
Maximum	0.01660	0.00077	0.00113	0.19233	0.00017	0.00000	0.00056	0.00313	0.06895	0.07765
Count	13	13	13	13	13	13	13	13	13	13
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g				
Mean	0.00055	0.04973	0.00036	0.00026	0.00020	0.05967				
Median	0.00016	0.03716	0.00038	0.00014	0.00026	0.03127				
Std. Dev.	0.00087	0.02831	0.00023	0.00022	0.00014	0.07627				
Minimum	0.00000	0.02906	0.00000	0.00002	0.00000	0.00000				
Maximum	0.00291	0.13514	0.00078	0.00059	0.00037	0.23716				
Count	13	13	13	13	13	13				

Statistics on elements released during complete oxidation of spoil samples,

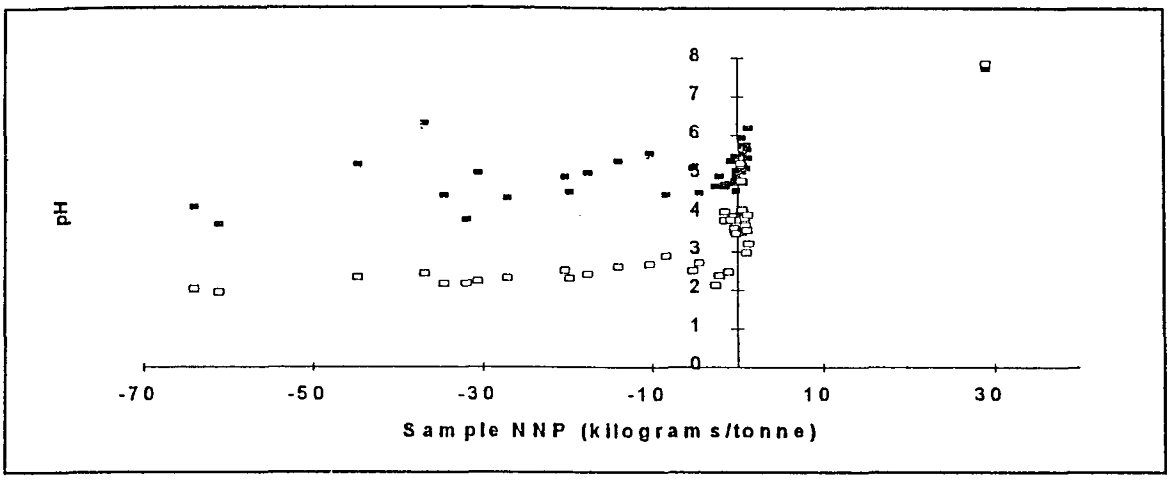
NNP and pH change.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.23408	0.00017	0.00942	0.49206	0.00028	0.00250	0.01041	1.31730	0.06046	0.22214
Median	0.17797	0.00000	0.00180	0.31577	0.00000	0.00073	0.00258	0.23435	0.04866	0.1326
Std. Dev.	0.23489	0.00043	0.01060	0.85304	0.00049	0.00386	0.01200	2.19488	0.03918	0.42691
Minimum	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00109	0.01013	0.00034
Maximum	0.71118	0.00117	0.02845	3.24414	0.00152	0.01216	0.03041	6.07726	0.12473	1.60338
Count	13	13	13	13	13	13	13	13	13	13
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP close	Initial pH	Final pH
Mean	0.03638	0.05601	0.00375	0.00252	0.01870	0.96317	-1.88452	-4.89443	6.51	5.34
Median	0.00932	0.04149	0.00068	0.00208	0.00132	0.06030	-0.08575	-0.67549	6.65	4.95
Std. Dev.	0.05215	0.03741	0.00726	0.00245	0.02874	2.59768	5.28428	13.36156	0.65	1.38
Minimum	0.00021	0.02313	0.00000	0.00003	0.00000	0.00630	-18.65856	-48.15618	5.77	2.80
Maximum	0.15941	0.13090	0.02586	0.00754	0.07432	9.43924	1.90676	1.35463	7.87	8.04
Count	13	13	13	13	13	13	13	13	13	13

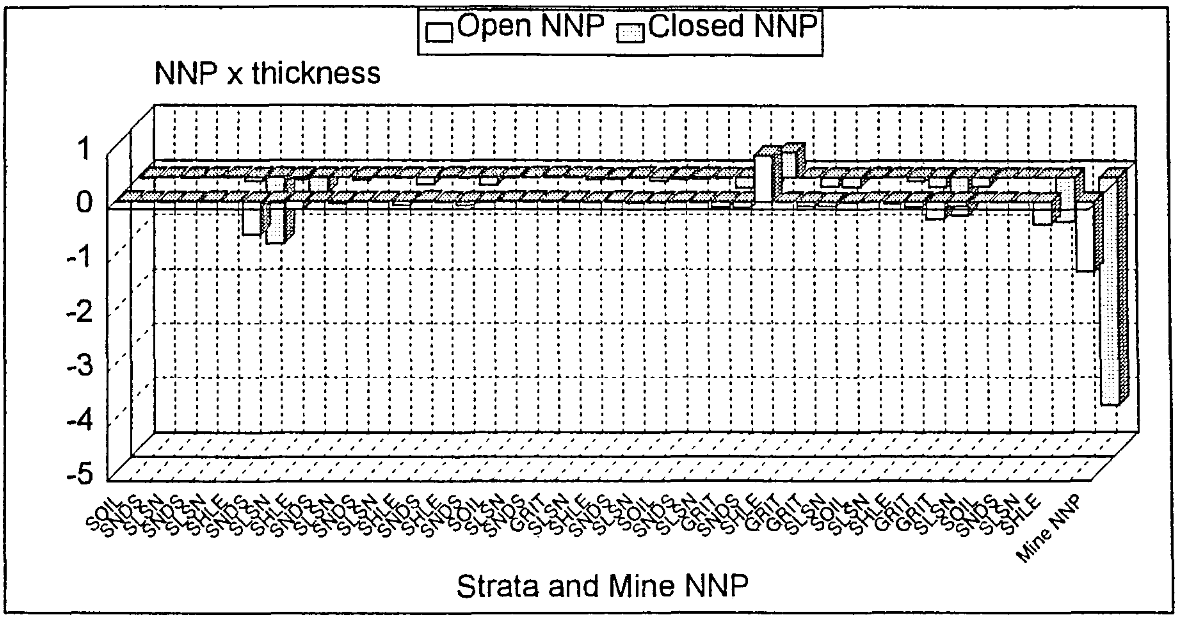
ACID-BASE INFORMATION FOR MINE 9



Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Normalised thicknesses times NNP for individual strata and for the total mine.

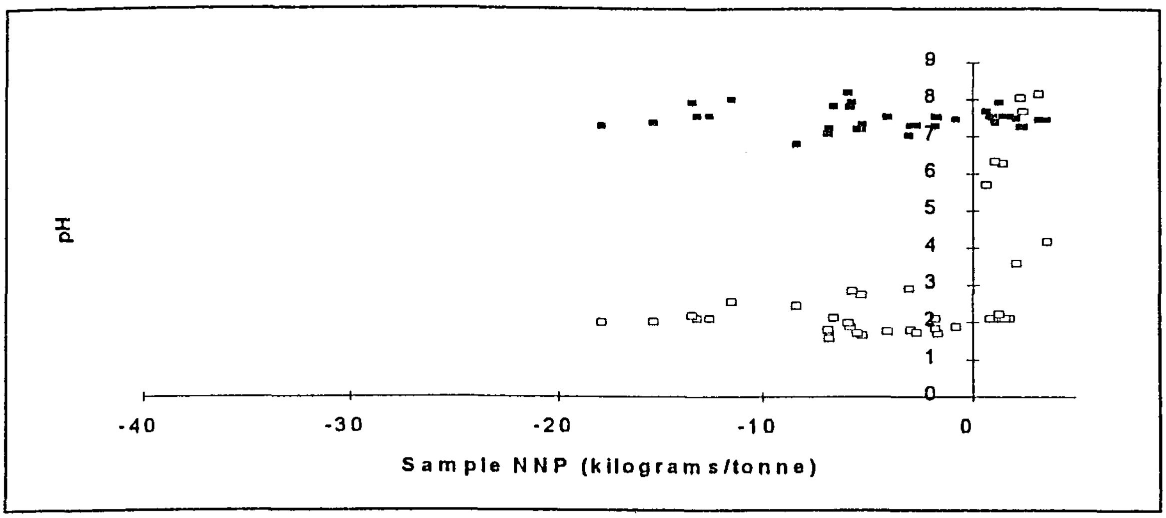
Statistics on soluble elements within the spoil samples.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.00133	0.00000	0.00056	0.03933	0.00001	0.00087	0.00023	0.02994	0.03710	0.01197
Median	0.00000	0.00000	0.00023	0.02184	0.00000	0.00022	0.00000	0.00000	0.03379	0.00336
Std. Dev.	0.00438	0.00000	0.00059	0.05582	0.00004	0.00179	0.00049	0.09078	0.02468	0.02079
Minimum	0.00000	0.00000	0.00000	0.00391	0.00000	0.00000	0.00000	0.00000	0.00000	0
Maximum	0.02670	0.00000	0.00213	0.33346	0.00019	0.00722	0.00237	0.38506	0.14691	0.10284
Count	43	43	43	43	43	43	43	43	43	43
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g				
Mean	0.00107	0.03665	0.00096	0.00021	0.00111	0.08556				
Median	0.00019	0.03407	0.00036	0.00011	0.00023	0.04166				
Std. Dev.	0.00241	0.01201	0.00181	0.00026	0.00227	0.13387				
Minimum	0.00000	0.02718	0.00000	0.00002	0.00000	0.00000				
Maximum	0.01205	0.09791	0.00759	0.00138	0.01258	0.71914				
Count	43	43	43	43	43	43				

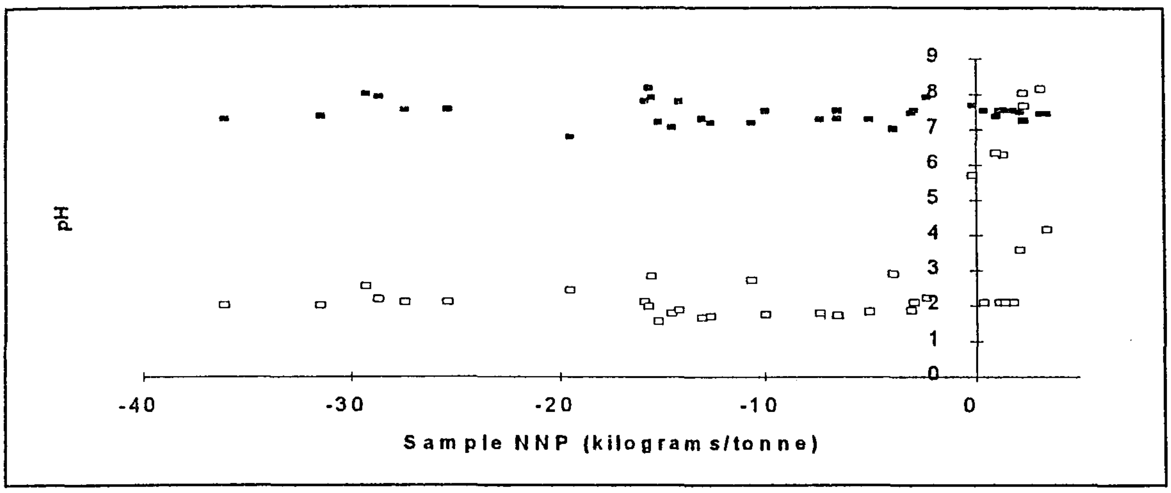
Statistics on elements released during complete oxidation of spoil samples, NNP and pH change.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.07521	0.00000	0.00289	0.12941	0.00039	0.00323	0.00948	1.56404	0.03158	0.0695
Median	0.03651	0.00000	0.00120	0.04141	0.00000	0.00125	0.00147	0.01439	0.01704	0.01877
Std. Dev.	0.08636	0.00000	0.00601	0.26022	0.00063	0.00429	0.01268	2.70002	0.06199	0.13993
Minimum	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00006
Maximum	0.30343	0.00000	0.03775	1.37082	0.00230	0.01973	0.04133	11.43806	0.39593	0.81829
Count	43	43	43	43	43	43	43	43	43	43
Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP close	Initial pH	Final pH
Mean	0.01469	0.02471	0.00716	0.00074	0.02874	1.71492	-4.56060	-9.91972	5.11	3.33
Median	0.00678	0.02304	0.00182	0.00045	0.00218	0.19461	-0.34391	-0.72792	5.10	3.47
Std. Dev.	0.02033	0.01313	0.00901	0.00072	0.04357	2.66502	8.95673	16.99754	0.69	1.13
Minimum	0.00030	0.00000	0.00000	0.00000	0.00000	0.00530	-31.98360	-63.64460	3.69	1.92
Maximum	0.09280	0.05956	0.03126	0.00356	0.19876	10.13151	14.56198	7.94347	7.72	7.84
Count	43	43	43	43	43	43	43	43	43	43

ACID-BASE INFORMATION FOR MINE 10

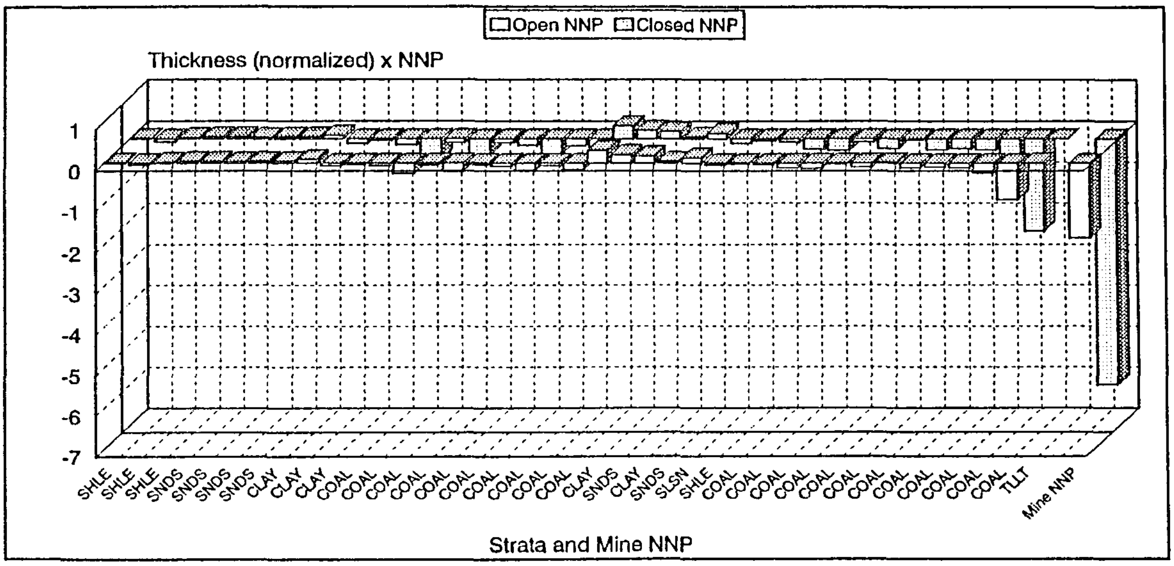


Open system NNP versus initial pH (dots) and oxidation pH (open squares) for spoil samples.



Closed system NNP versus initial pH (dots) and oxidation pH (open squares)

for spoil samples.



Normalised thicknesses times NNP for individual strata and for the total mine.

Statistics on soluble elements within the spoil samples.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.03966	0.00045	0.00054	0.07921	0.00000	0.00022	0.00002	0.01315	0.10520	0.01414
Median	0.00621	0.00054	0.00042	0.06930	0.00000	0.00025	0.00000	0.00000	0.07014	0.01086
Std. Dev.	0.11492	0.00044	0.00059	0.04431	0.00003	0.00015	0.00012	0.03751	0.14140	0.00987
Minimum	0.00000	0.00000	0.00000	0.02104	0.00000	0.00000	0.00000	0.00000	0.01606	0
Maximum	0.68871	0.00148	0.00344	0.21621	0.00016	0.00062	0.00073	0.17569	0.85391	0.04245
Count	38	38	38	38	38	38	38	38	38	38

Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	Cr mg/g	S mg/g
Mean	0.00021	0.16061	0.00036	0.00115	0.00047	0.02014	0.07048
Median	0.00015	0.13481	0.00031	0.00060	0.00045	0.00000	0.06848
Std. Dev.	0.00024	0.08559	0.00064	0.00251	0.00036	0.09819	0.03003
Minimum	0.00000	0.06845	0.00000	0.00013	0.00000	0.00000	0.02852
Maximum	0.00102	0.38936	0.00400	0.01595	0.00136	0.58467	0.19517
Count	38	38	38	38	38	38	38

Statistics on elements released during complete oxidation of spoil samples,
NNP and pH change.

Statistics	Al mg/g	B mg/g	Ba mg/g	Ca mg/g	Cd mg/g	Co mg/g	Cu mg/g	Fe mg/g	K mg/g	Mg mg/g
Mean	0.05144	0.00038	0.00462	0.62209	0.00010	0.00093	0.00495	1.40155	0.01898	0.17212
Median	0.03704	0.00000	0.00196	0.47281	0.00000	0.00059	0.00458	1.05245	0.00000	0.09301
Std. Dev.	0.07074	0.00062	0.00726	0.55140	0.00020	0.00141	0.00477	1.46533	0.03860	0.21766
Minimum	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00000	0.00808
Maximum	0.35204	0.00226	0.03716	1.94237	0.00081	0.00733	0.01729	4.94586	0.19813	1.04374
Count	43	43	43	43	43	43	43	43	43	43

Statistics	Mn mg/g	Na mg/g	Ni mg/g	Sr mg/g	Zn mg/g	S mg/g	NNP open	NNP Close	Initial pH	Final pH
Mean	0.00788	0.05329	0.06250	0.00670	0.00252	1.74024	-3.53832	-8.97657	7.47	2.92
Median	0.00558	0.02532	0.00037	0.00423	0.00222	1.24384	-2.68469	-6.55081	7.52	2.09
Std. Dev.	0.01027	0.06342	0.37043	0.00905	0.00204	1.81885	5.65412	11.23924	0.28	1.88
Minimum	0.00024	0.00000	0.00000	0.00000	0.00000	0.00000	-17.90390	-36.12300	6.79	1.56
Maximum	0.05617	0.29706	2.31604	0.05027	0.00721	5.83013	3.61703	3.45030	8.18	8.14
Count	43	43	43	43	43	43	43	43	43	43

Appendix D

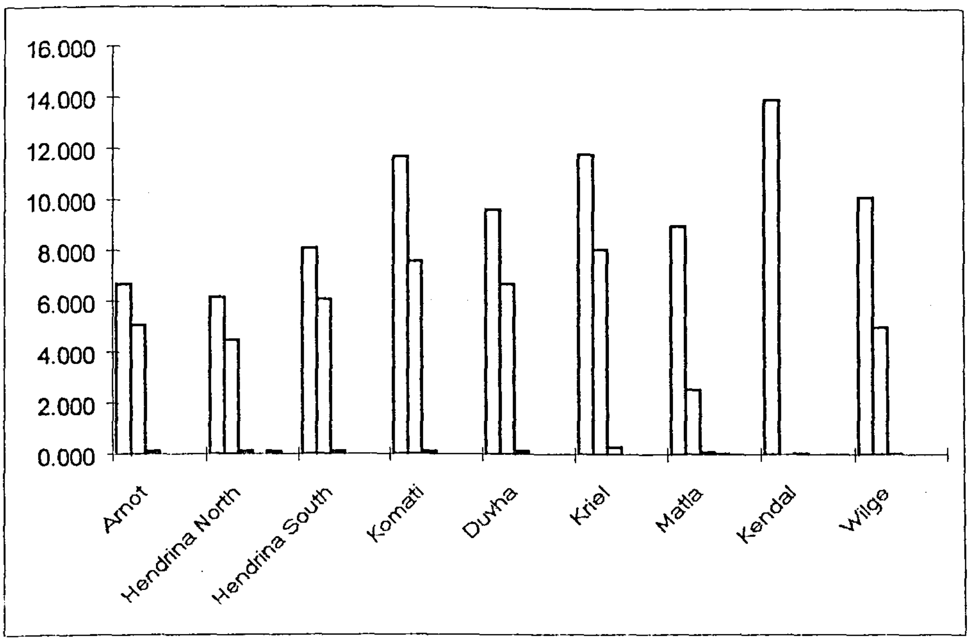
RESULTS OF LEACHING TESTS ON FLY ASH SAMPLES FROM POWER STATIONS WITHIN THE OLIFANTS CATCHMENT

Please note:

Values specified as zero were below the detection limits for ICP analysis

Aluminium in kg/t

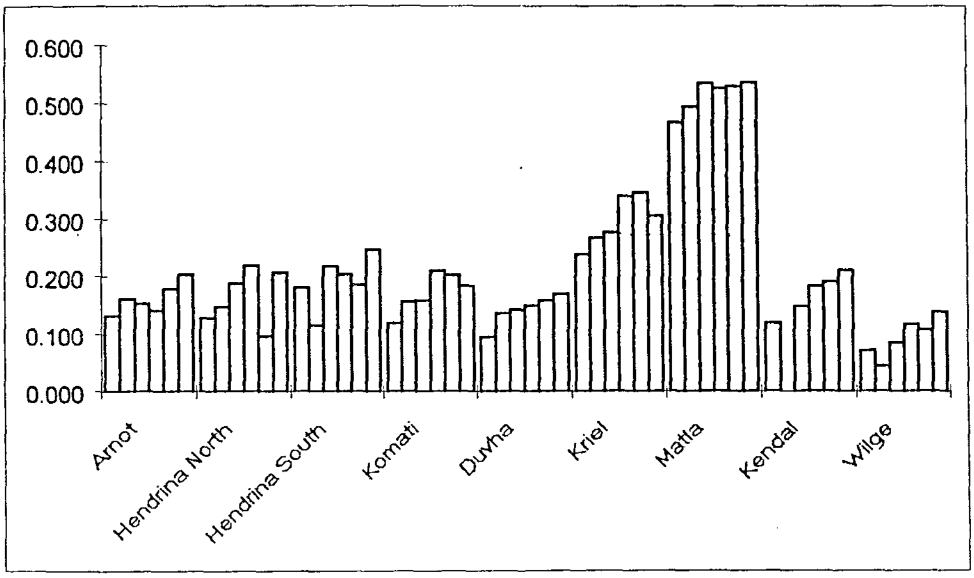
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	6.677	6.154	8.078	11.681	9.631	11.820	8.996	13.981	10.125
4.00	5.078	4.477	6.066	7.560	6.678	8.078	2.556		5.013
5.00	0.120	0.117	0.106	0.114	0.116	0.291	0.100	0.044	0.037
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.025	0.000	0.000
7.00	0.000	0.105	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Boron in kg/t

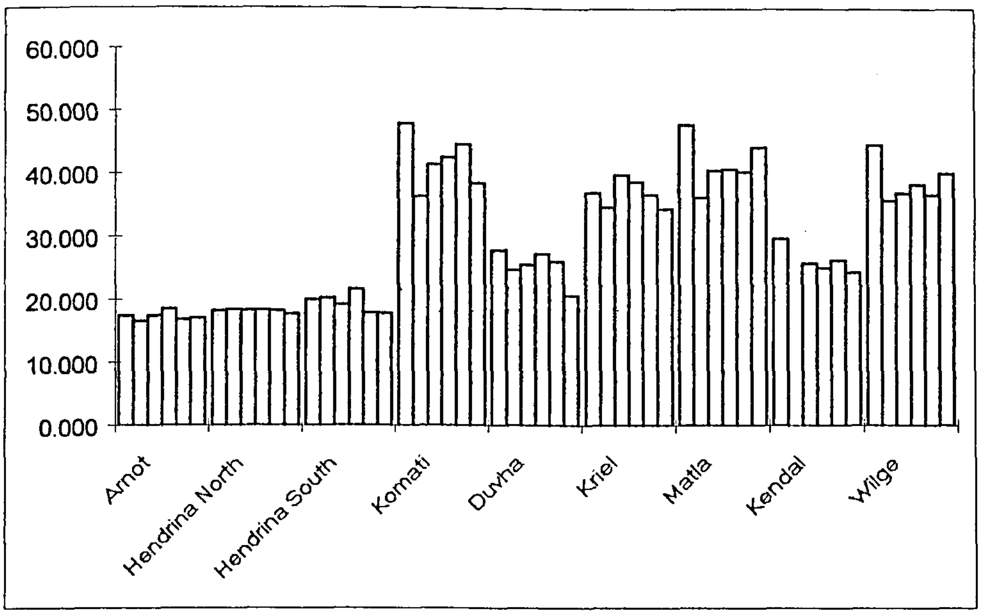
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.132	0.128	0.181	0.119	0.095	0.239	0.468	0.119	0.072
4.00	0.161	0.147	0.115	0.156	0.135	0.267	0.494		0.044
5.00	0.154	0.188	0.218	0.159	0.142	0.277	0.536	0.148	0.085
6.00	0.141	0.219	0.205	0.211	0.149	0.340	0.526	0.183	0.117
7.00	0.179	0.097	0.186	0.203	0.159	0.347	0.530	0.191	0.107
8.00	0.204	0.207	0.247	0.184	0.170	0.306	0.537	0.211	0.138



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Calcium in kg/t

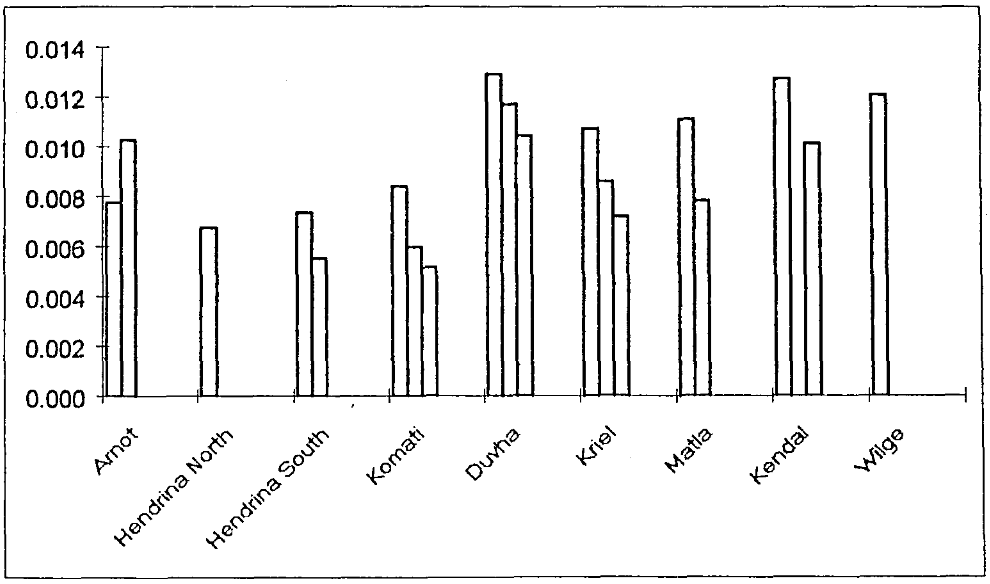
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	17.432	18.230	19.992	47.914	27.901	37.020	47.855	29.792	44.578
4.00	16.597	18.450	20.257	36.288	24.797	34.715	36.278		35.784
5.00	17.442	18.390	19.133	41.385	25.595	39.842	40.562	25.760	36.946
6.00	18.519	18.385	21.613	42.515	27.279	38.696	40.730	25.088	38.258
7.00	16.866	18.232	17.900	44.581	26.099	36.725	40.264	26.296	36.658
8.00	17.055	17.745	17.847	38.465	20.688	34.445	44.183	24.353	40.041
9.00	17.318	18.239	19.457	41.858	25.393	36.907	41.645	26.258	38.711



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Copper in kg/t

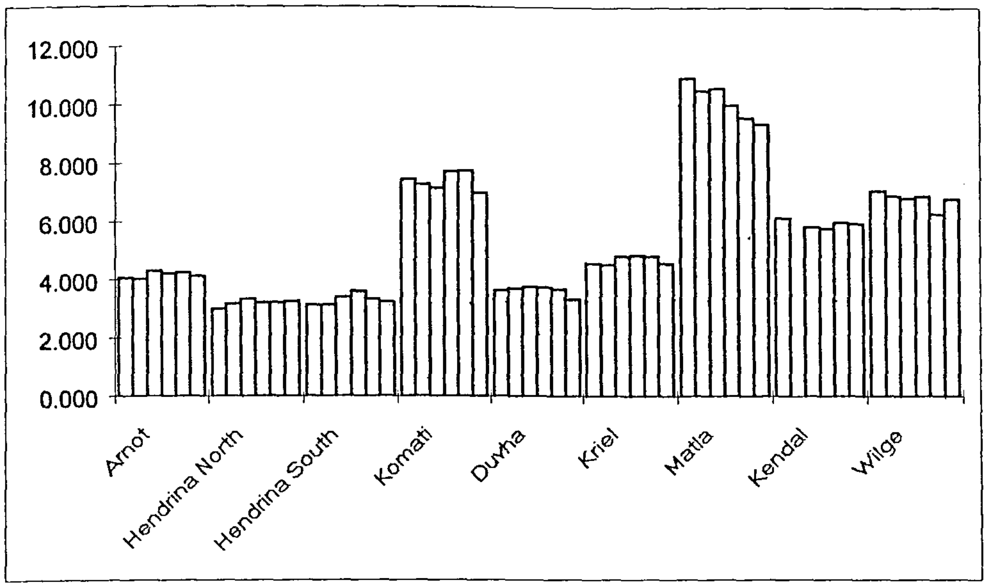
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.008	0.007	0.007	0.008	0.013	0.011	0.011	0.013	0.012
4.00	0.010	0.000	0.006	0.006	0.012	0.009	0.008		0.000
5.00	0.000	0.000	0.000	0.005	0.010	0.007	0.000	0.010	0.000
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Magnesium in kg/t

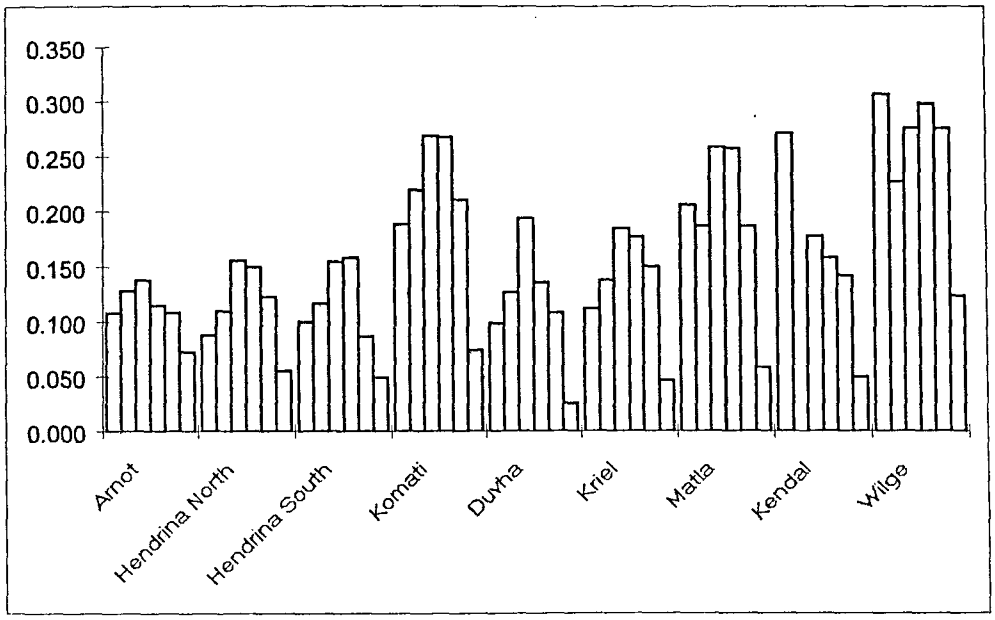
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	4.067	2.995	3.140	7.460	3.636	4.582	10.962	6.149	7.065
4.00	4.035	3.170	3.138	7.277	3.708	4.540	10.516		6.899
5.00	4.317	3.337	3.391	7.146	3.771	4.830	10.607	5.837	6.819
6.00	4.229	3.222	3.600	7.732	3.748	4.864	10.028	5.783	6.885
7.00	4.262	3.212	3.331	7.747	3.686	4.825	9.590	5.997	6.276
8.00	4.123	3.258	3.233	7.000	3.333	4.573	9.377	5.946	6.784
9.00	4.172	3.199	3.306	7.394	3.647	4.702	10.180	5.942	6.788



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatio

Manganese in kg/t

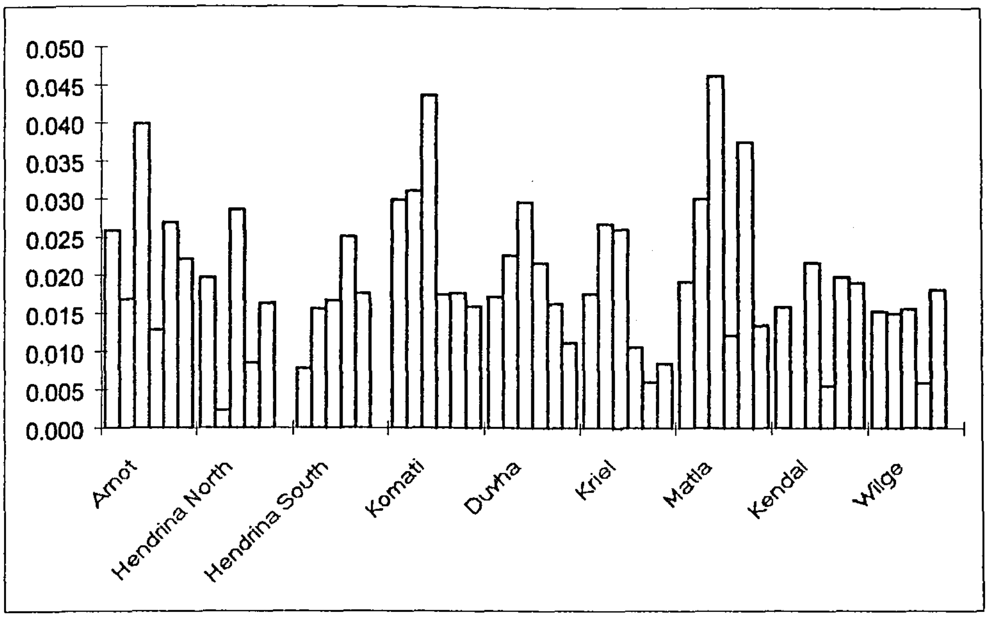
	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.108	0.087	0.100	0.188	0.098	0.112	0.207	0.272	0.307
4.00	0.128	0.110	0.116	0.220	0.127	0.138	0.187		0.228
5.00	0.138	0.156	0.155	0.269	0.194	0.185	0.259	0.178	0.277
6.00	0.115	0.150	0.158	0.268	0.136	0.177	0.258	0.159	0.299
7.00	0.108	0.123	0.086	0.211	0.108	0.150	0.187	0.142	0.276
8.00	0.072	0.055	0.050	0.074	0.026	0.047	0.058	0.050	0.124



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatio

Zinc in kg/t

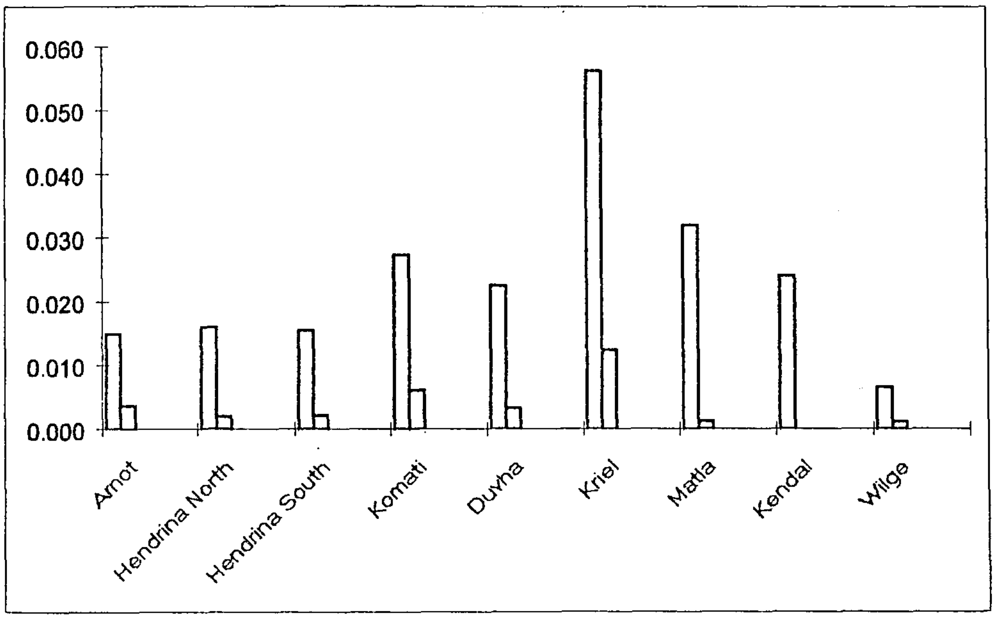
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.026	0.020	0.008	0.030	0.017	0.018	0.019	0.016	0.015
4.00	0.017	0.002	0.016	0.031	0.023	0.027	0.030		0.015
5.00	0.040	0.029	0.017	0.044	0.030	0.026	0.046	0.022	0.016
6.00	0.013	0.009	0.025	0.018	0.022	0.011	0.012	0.006	0.006
7.00	0.027	0.016	0.018	0.018	0.016	0.006	0.038	0.020	0.018
8.00	0.022	0.000	0.000	0.016	0.011	0.008	0.013	0.019	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Chromium in kg/t

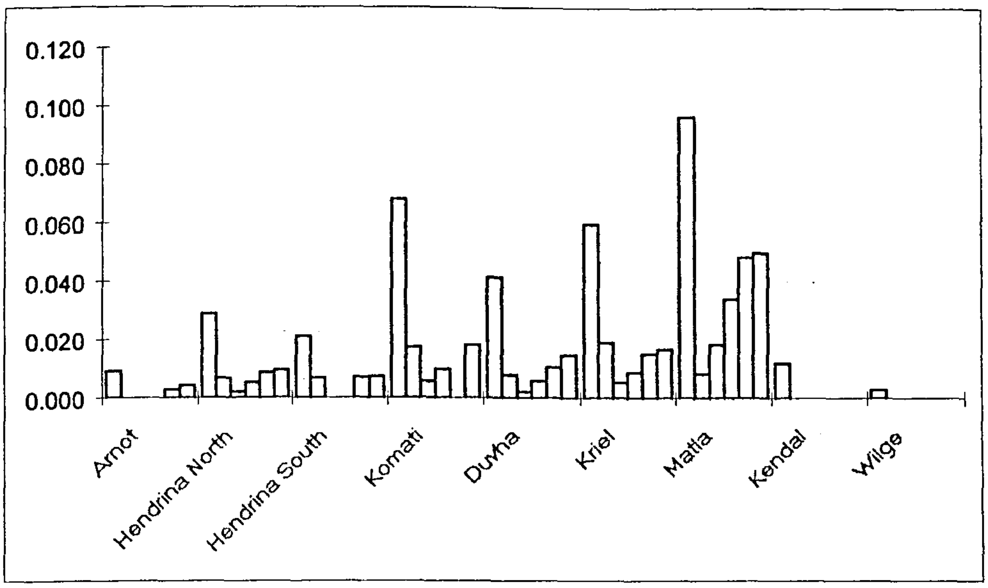
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.015	0.016	0.016	0.027	0.023	0.056	0.032	0.024	0.007
4.00	0.004	0.002	0.002	0.006	0.003	0.012	0.001		0.001
5.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Vanadium in kg/t

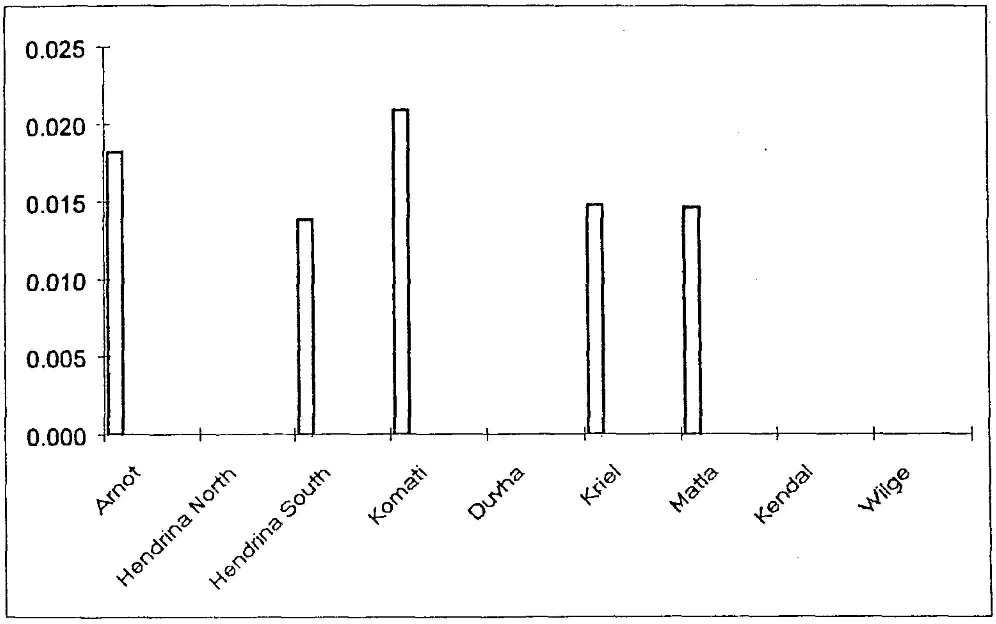
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.009	0.029	0.021	0.068	0.041	0.060	0.096	0.012	0.003
4.00	0.000	0.007	0.007	0.017	0.008	0.019	0.008		0.000
5.00	0.000	0.002	0.000	0.006	0.002	0.005	0.018	0.000	0.000
6.00	0.000	0.005	0.000	0.010	0.006	0.009	0.034	0.000	0.000
7.00	0.003	0.009	0.007	0.000	0.011	0.015	0.048	0.000	0.000
8.00	0.004	0.009	0.007	0.018	0.015	0.017	0.050	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Tin in kg/t

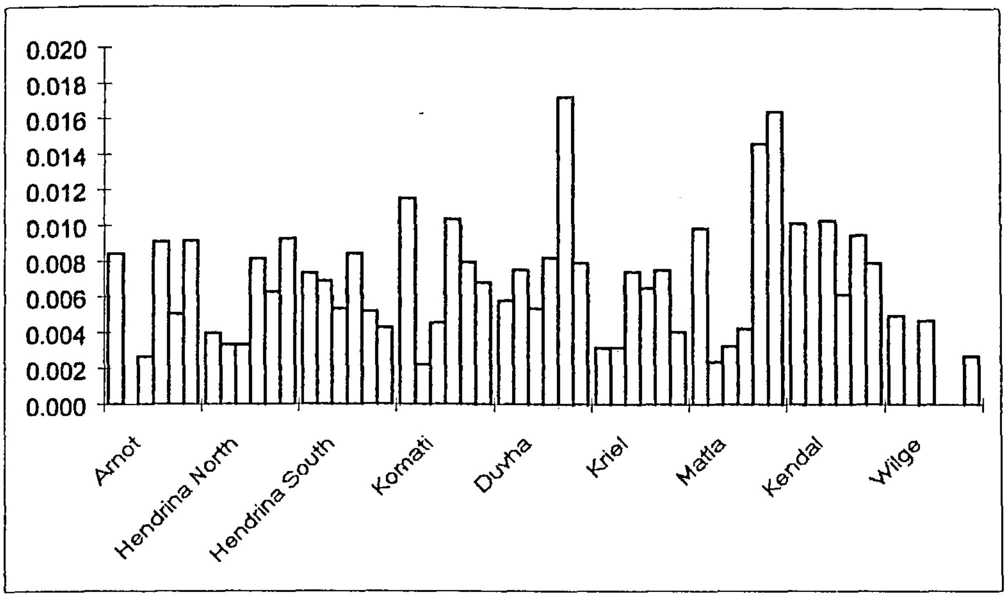
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.018	0.000	0.014	0.021	0.000	0.015	0.015	0.000	0.000
4.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000		0.000
5.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Molybdenum in kg/t

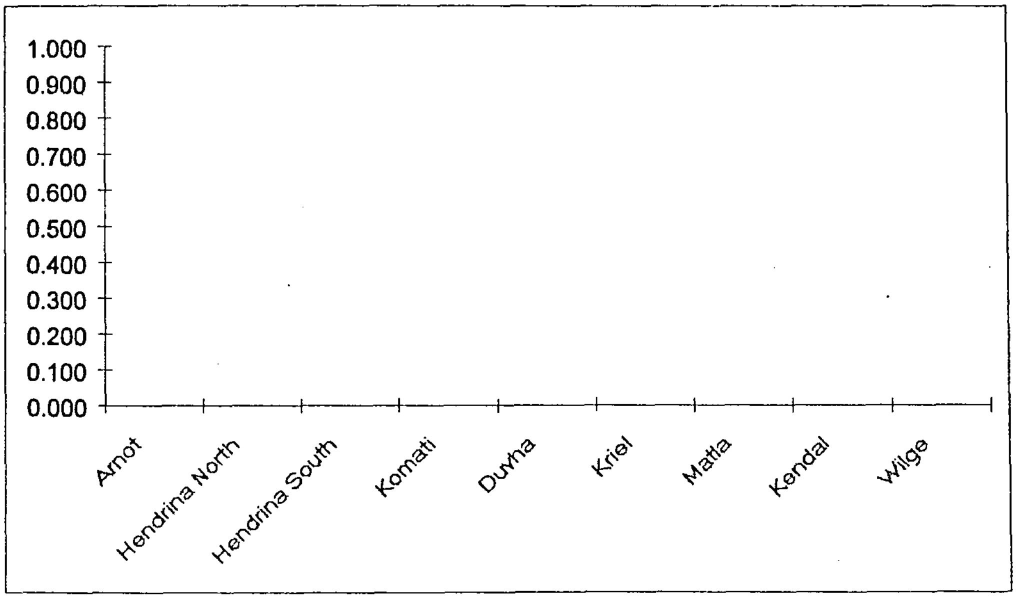
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.008	0.004	0.007	0.012	0.006	0.003	0.010	0.010	0.005
4.00	0.000	0.003	0.007	0.002	0.008	0.003	0.002		0.000
5.00	0.003	0.003	0.005	0.005	0.005	0.007	0.003	0.010	0.005
6.00	0.009	0.008	0.008	0.010	0.008	0.007	0.004	0.006	0.000
7.00	0.005	0.006	0.005	0.008	0.017	0.008	0.015	0.010	0.000
8.00	0.009	0.009	0.004	0.007	0.008	0.004	0.016	0.008	0.003



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Mercury in kg/t

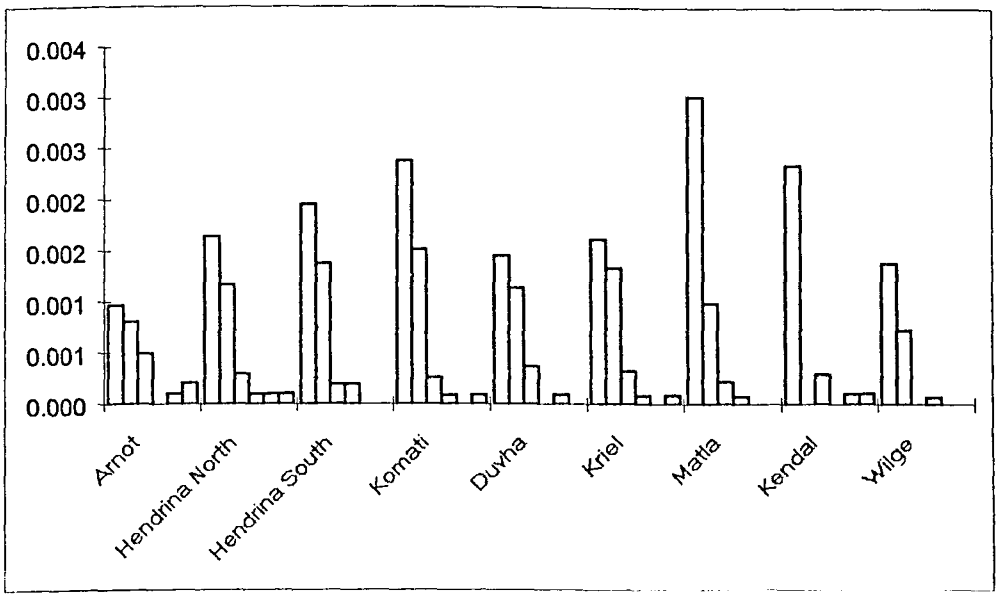
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
4.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000		0.000
5.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Beryllium in kg/t

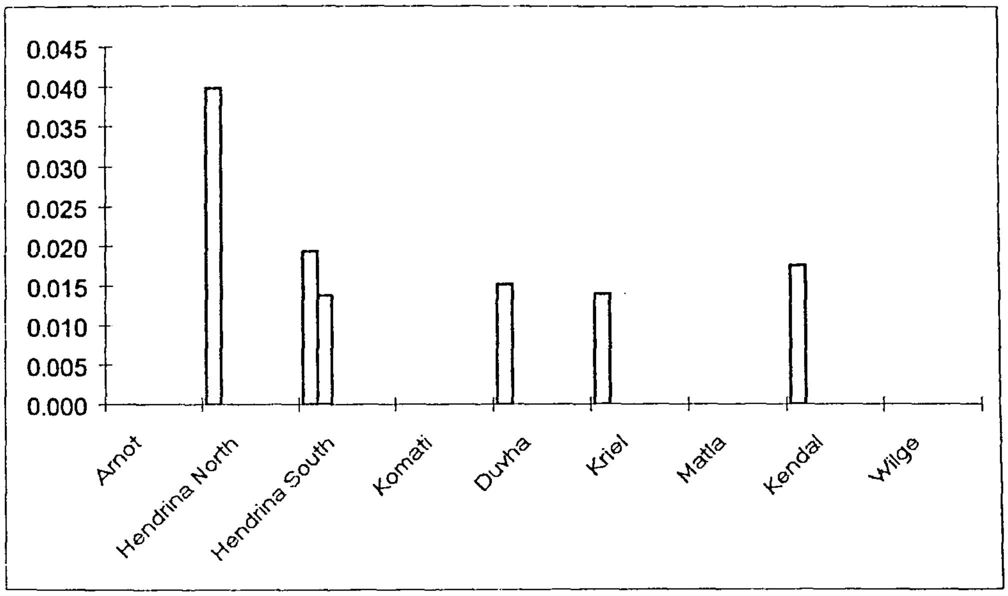
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.001	0.002	0.002	0.002	0.001	0.002	0.003	0.002	0.001
4.00	0.001	0.001	0.001	0.002	0.001	0.001	0.001		0.001
5.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Arsenic in kg/t

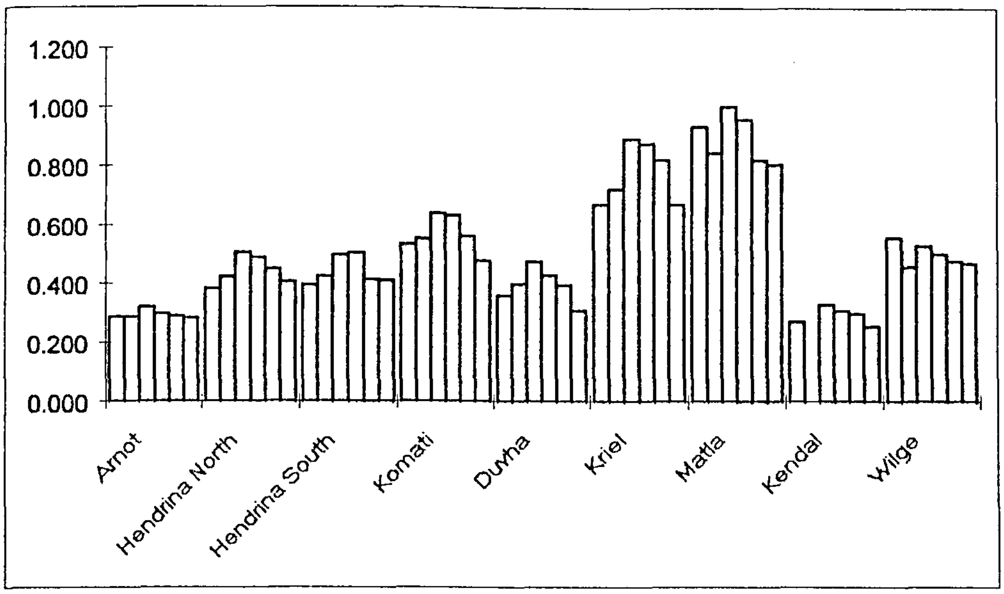
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.000	0.040	0.019	0.000	0.015	0.014	0.000	0.018	0.000
4.00	0.000	0.000	0.014	0.000	0.000	0.000	0.000		0.000
5.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Strontium in kg/t

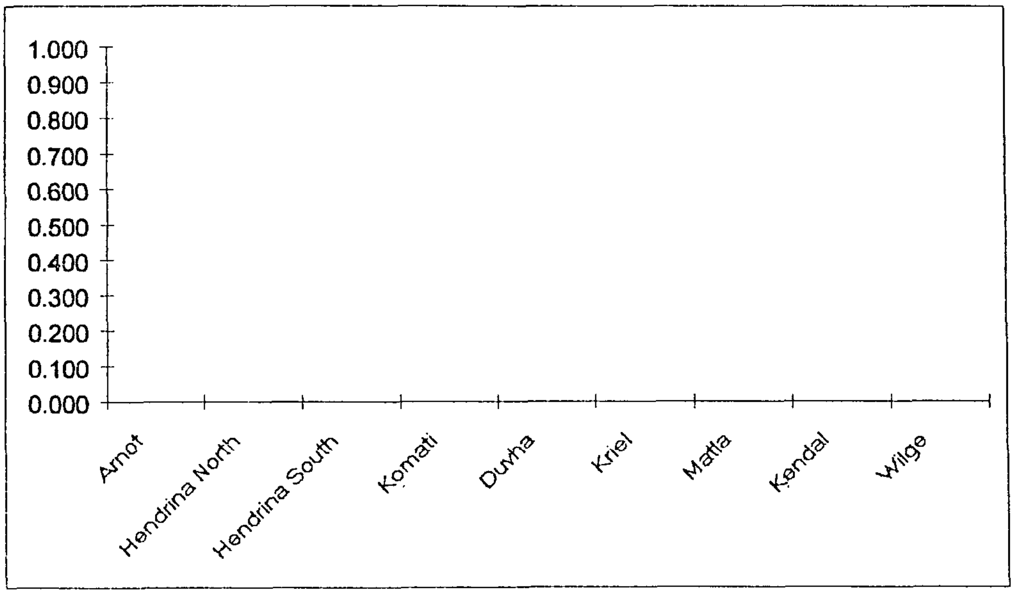
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.286	0.383	0.394	0.534	0.360	0.670	0.935	0.272	0.558
4.00	0.286	0.422	0.424	0.553	0.399	0.723	0.846		0.457
5.00	0.319	0.505	0.497	0.640	0.475	0.894	1.004	0.329	0.530
6.00	0.297	0.488	0.503	0.632	0.429	0.878	0.960	0.308	0.501
7.00	0.289	0.449	0.413	0.562	0.396	0.822	0.821	0.299	0.477
8.00	0.283	0.405	0.411	0.479	0.310	0.669	0.806	0.256	0.470



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more information

Nickel in kg/t

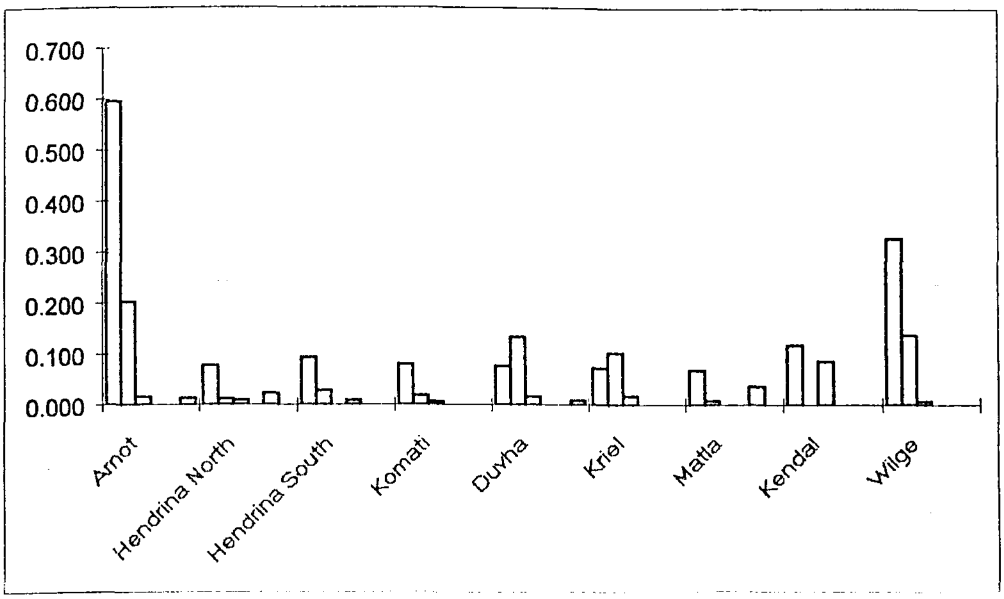
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
4.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000		0.000
5.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more information

Iron in kg/t

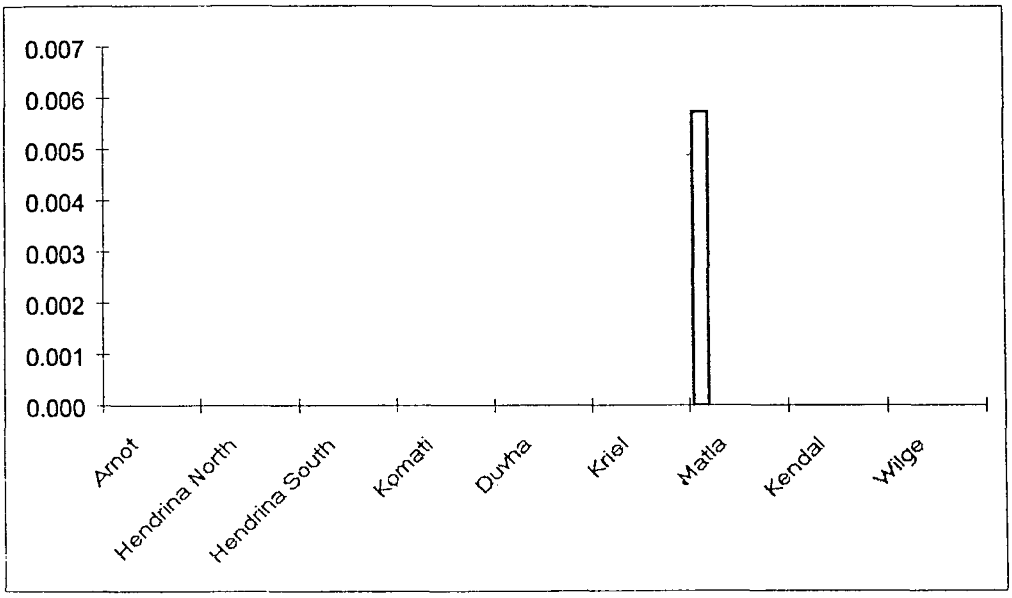
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.596	0.078	0.094	0.079	0.077	0.072	0.068	0.118	0.327
4.00	0.202	0.011	0.028	0.018	0.134	0.102	0.008		0.137
5.00	0.015	0.009	0.000	0.006	0.016	0.017	0.000	0.085	0.007
6.00	0.000	0.000	0.009	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.000	0.022	0.000	0.000	0.000	0.000	0.037	0.000	0.000
8.00	0.013	0.000	0.000	0.000	0.009	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more information

Cobalt in kg/t

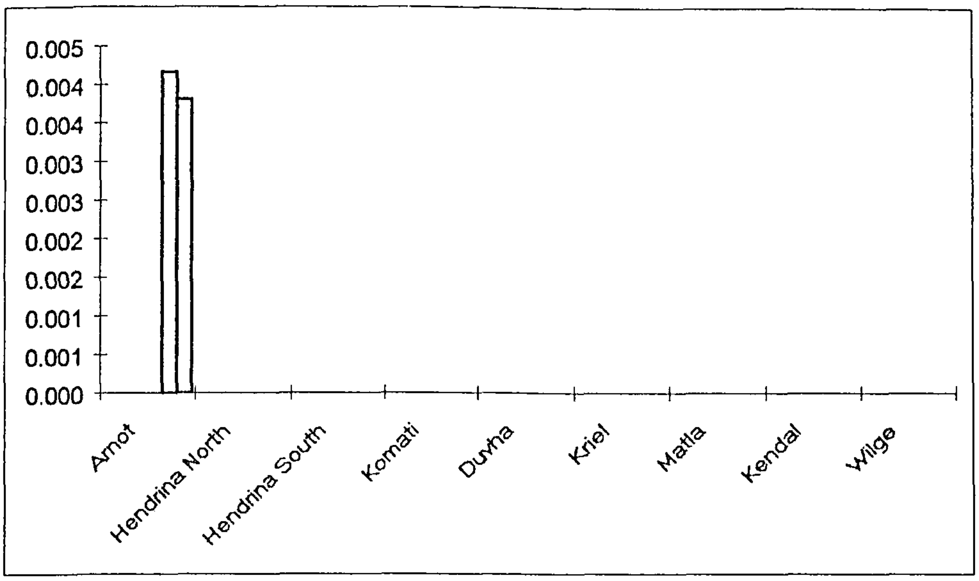
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.000	0.000	0.000	0.000	0.000	0.000	0.006	0.000	0.000
4.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000		0.000
5.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more information

Cadmium in kg/t

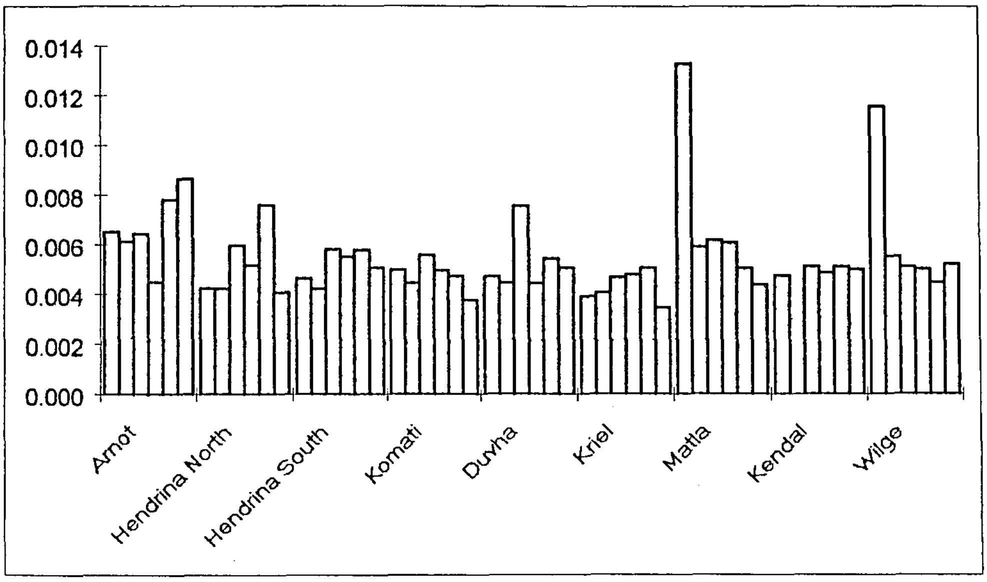
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
4.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000		0.000
5.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
6.00	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
7.00	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
8.00	0.004	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more Informatior

Barium in kg/t

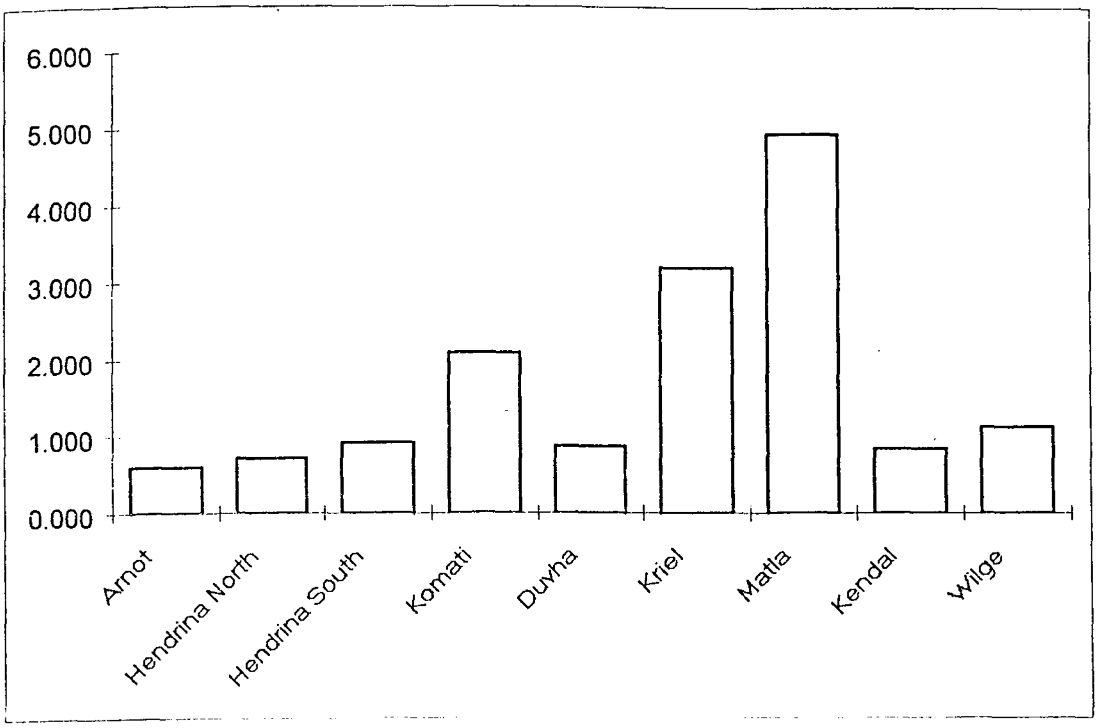
pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
3.00	0.007	0.004	0.005	0.005	0.005	0.004	0.013	0.005	0.012
4.00	0.006	0.004	0.004	0.004	0.004	0.004	0.006		0.006
5.00	0.006	0.006	0.006	0.006	0.008	0.005	0.006	0.005	0.005
6.00	0.005	0.005	0.006	0.005	0.004	0.005	0.006	0.005	0.005
7.00	0.008	0.008	0.006	0.005	0.005	0.005	0.005	0.005	0.004
8.00	0.009	0.004	0.005	0.004	0.005	0.003	0.004	0.005	0.005



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Sulphur in kg/t

pH	Arnot	Hendrina	Hendrina	Komati	Duvha	Kriel	Matla	Kendal	Wilge
2.00	0.600	0.713	0.913	2.090	0.866	3.200	4.940	0.836	1.120



All values in kg/t. Bars are arranged per station and increasing pH from left to right. See Section 4 for more informatior

Appendix E

SUGGESTIONS FOR MONITORING AT WASTE DISPOSAL FACILITIES

E.1 INTRODUCTION

Development inevitably leads to waste production. Waste, on the other hand, when stored, may produce leachate. If this leachate is not contained, pollution of water resources may result.

Monitoring of leachate production is usually done close to possible sources of pollution. Such sources, for the purpose of this investigation, can all be grouped under the general term of "waste management". This broad definition includes all mining, power station, municipal (general), hazardous, sewage, industrial and agricultural waste, whether solid or liquid. This grouping is feasible in terms of monitoring, since monitoring principles are the same for all wastes, irrespective of its origin.

Monitoring the effect which development has on the water quality of surface and groundwater resources is complex and multi-disciplinary. Numerous methodologies exist for monitoring. The type of monitoring required in a specific situation will depend on:

- The type of waste,
- The amount of waste,
- The potential for water usage in the area, and
- The vulnerability of water resources.

South African groundwater systems differ markedly from those overseas. Overseas methodologies for monitoring therefore do not necessarily apply to the situation within the Olifants Catchment. Direct application of overseas methodologies could lead to confusion and result in unnecessary expenditure.

In this chapter on water quality monitoring, the emphasis is on "what could reasonably be achieved in terms of monitoring". The principle of "batneec" (best available technology, not entailing excessive cost) is subscribed to throughout this chapter. Consideration is also given to existing policy documents by governmental departments, such as the Environmental Act, Act 73 of 1989 dealing with general and hazardous waste, and the EMPR (Environmental Management Programme Report) for the mining industry.

Many of the suggestions for monitoring, as presented in this chapter, deviate from those stipulated in classic textbooks. These deviations do not contradict the well-established methodologies, but modifications have been introduced on the "batneec" principle. Those who wish to follow more stringent procedures than that recommended in this chapter should feel free to do so.

E.2 RISK ASSESSMENT

A risk assessment, to determine the risk of water pollution, has to be performed before the design and installation of the water quality monitoring system.

E.2.1 GOALS OF RISK ASSESSMENT

A groundwater pollution risk assessment serves two valuable purposes:

- It provides a numerical value with respect to the groundwater pollution potential. Sites that have been evaluated by these means may therefore be ranked in terms of suitability or remedial priorities.
- Site design and monitoring facilities can be prescribed according to the results of the risk assessment. Particularly, the density and location of monitoring points may be linked to the rating obtained from a risk assessment.

E.2.2 MAIN ISSUES

The first step in the design of a suitable groundwater monitoring system is the assessment of the risk for a facility to pollute the groundwater regime. The risk will differ for each site and monitoring facilities will have to be designed accordingly. The main issues which need to be considered in a risk assessment are the:

- Potential for groundwater usage.
- Aquifer vulnerability.
- Toxicity and other properties of the waste.
- Quantities of waste.
- Potential for leachate generation.

Potential for groundwater usage

The potential for groundwater utilisation from a specific aquifer may change with time. A resource that presently seems unimportant, may, in future, be a valuable asset. In the siting of any waste disposal facility, aquifers should be ranked and those least likely to be of future use, may be considered for waste disposal.

Aquifer vulnerability

Aquifer vulnerability relates to a number of factors, the most important of which are:

- Climate, precipitation and surface water run-off.
- Nature and composition of the unsaturated zone.
- Aquifer characteristics, such as hydraulic conductivity, water quality and regional groundwater flow direction.

In the Olifants Catchment, the upper weathered aquifer is very vulnerable and polluting this aquifer underneath the waste site, is inevitable. The lateral movement of pollutants can, if necessary, be controlled. The degree to which leachate migration will have to be curtailed will depend on the outcome of the overall risk assessment.

Toxicity of the waste

The potential of different wastes to pollute water resources differs greatly, depending on the composition of the waste and its potential for degradation with time. South African legislation broadly classifies waste under two categories, namely general and hazardous waste. In terms of waste in the Olifants Catchment, these two categories form two end members of a continuum, with a vague transition from what could be described as non-toxic to toxic. When referring to the level of toxicity, not only must the constituent itself be considered, but also the possible user of the water, e.g. human, animal, aquatic life or irrigation and the concentration at which the constituent is likely to be present.

Quantities of waste

Toxicity and quantity of waste go hand in hand. It is easier to dispose, manage and contain small quantities than large quantities of waste. The risk for groundwater pollution is therefore usually greater at large waste disposal facilities, where it is often impossible to prevent groundwater pollution because of the nature and scale of operations.

The DWA&F suggest three procedures for risk assessment in the document on monitoring requirements.

E.2.3 DISPOSAL FACILITY DESIGN

It is theoretically possible, by using synthetic liners, to completely contain leachate from a waste site. This is, however, mostly impractical and very costly. The necessity for such extreme measures can be determined by performing a risk analysis, considering the four variables discussed above. Under no circumstances can waste disposal be done without considering all four variables.

E.3 FACILITIES FOR WATER MONITORING

E.3.1 INTRODUCTION

The main purpose of monitoring pollution migration is to:

- Provide reliable and irrefutable data on the quality and chemical composition of groundwater,
- Detect and quantify the presence and seriousness of any polluting substances in groundwater at the very earliest stage possible,
- Detect the possible release or impending release of contaminants from the waste facility,
- Provide a rational comparison between the predicted and actual flow and solute transport rates, and
- Provide an ongoing and reliable record on the performance of the design and control system(s) for effectively controlling pollution.

To achieve the above objectives, it may be necessary to employ two separate monitoring systems in cases where the generation of hazardous leachate may be a problem. The two monitoring systems are:

- Early Warning Monitoring Systems.

- Regional Monitoring Systems.

A schematic presentation of these monitoing options, in relation to a waste site, is depicted in Figure E.1.

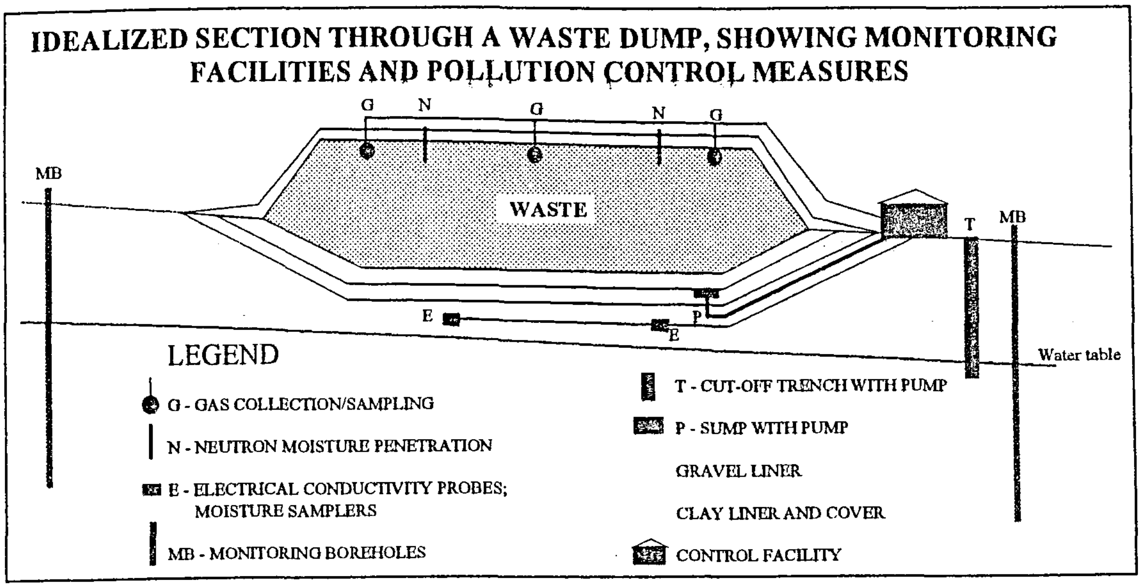


Figure E.1 Monitoring options at waste disposal facilities on surface.

The main difference between these two monitoring systems is that the Early Warning Monitoring System forms part of the disposal design and may be independent of the groundwater regime as well as any direct geohydrological considerations. Regional Monitoring Systems, on the other hand, are entirely dependent on geohydrological considerations and, in fact, cannot be applied successfully in the absence of such knowledge.

E.3.2 EARLY WARNING SYSTEMS

Early warning systems comprise measurements done on top of a disposal facility, within the waste itself and directly underneath the facility in the vadose zone. Such monitoring usually includes:

- Rainfall: Rainfall that infiltrates into a waste facility increases the overall pollution potential from that facility. Rainfall for the past 24 hours must be recorded at 8h00 every morning.
- Evaporation potential: The amount of potential evaporation from free-standing water can either be determined by measuring water losses from a Class A evaporation pan, or be calculated by using a suitable equation such as Penmann. In view of the difficulties which industries normally have in accurately monitoring the potential evaporation for a specific locality, the measurement of pan evaporation has only been suggested at hazardous disposal sites. For all other sites, approximate evaporation potential values can be obtained from the DWA&F.
- Run-off: The amount of water flowing off a disposal site, or a larger complex such as a mine or a power station, is an important component in the calculation of a water and salt balance for the site or complex. Run-off quantities and qualities should ideally be recorded on a continuous basis. Significant teething problems have been encountered at continuous

water quality monitoring points at the mines. pH and electrical conductivity probes are prone to malfunction. Problems have also been associated with pressure transducers that measure water levels in streams. Further research and design are necessary before the cost of accurate continuous recording in streams can be justified.

- Leachate Collection/Toe Seepage: Analysis of leachate from leachate collectors or toe seepage is considered to be the most important early warning indicator. Leachate collectors are part of the design details for certain waste management facilities and have been described by the DWA&F (1993) for general and hazardous wastes. In the case of other wastes, such as coal discard dumps, coal stockyards and ash disposal, the leachate collector should be in the form of a length of a conveyor belt (50 - 100 m), which is secured at a slope on top of the soil, before disposal is done. Leachate that seeps vertically through the dump, will then be deflected on the conveyor belt and can be collected where the belt extends from the dump. This will allow quantification of the amount of the vertical seepage and leachate samples can be obtained for chemical analysis. Samples should be collected, preserved and analysed according to suggestions in this chapter.
- Rehabilitation: Rehabilitation on top of waste should done as soon as is reasonably possible. This will limit the ingress of rainwater, thus reducing the volume of leachate to be dealt with. This will also prevent the ingress of oxygen, which in the case of general waste is essential to sustain a methanogenic environment, thus minimising the risk of acidification.

E.3.3 REGIONAL MONITORING SYSTEMS

Regional monitoring refers primarily to measurements done in the vicinity of the dumping site up to distances as may be required by the specific monitoring system. Monitoring is usually done at:

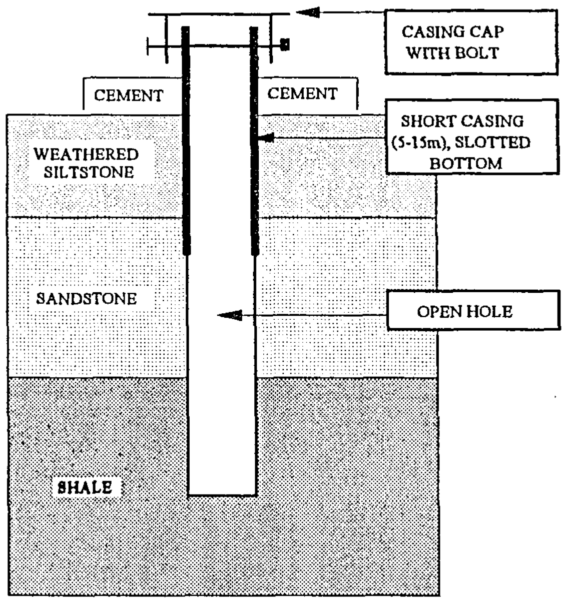
- Boreholes drilled to specific depths into the aquifer.
- Fountains, wells, dams, pans, stream and rivers.
-

E.3.3.1 Monitoring boreholes

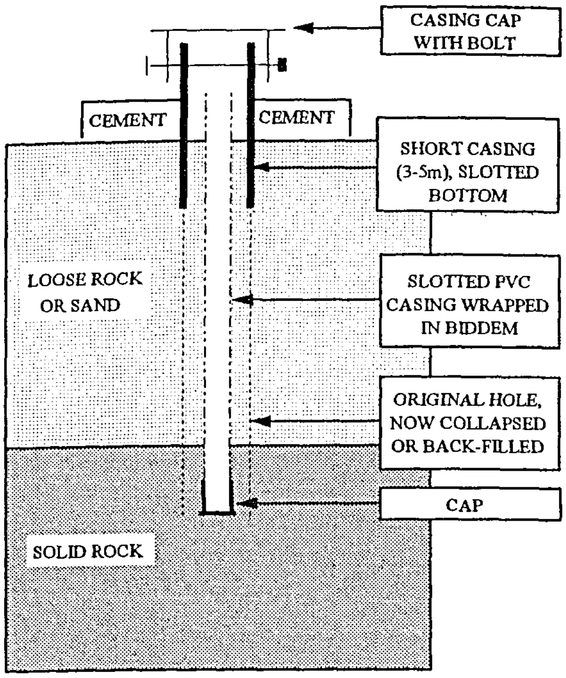
The local geology dictates the way in which boreholes should be constructed. Examples of equipped boreholes are given to the right and below:

Data required from boreholes are:

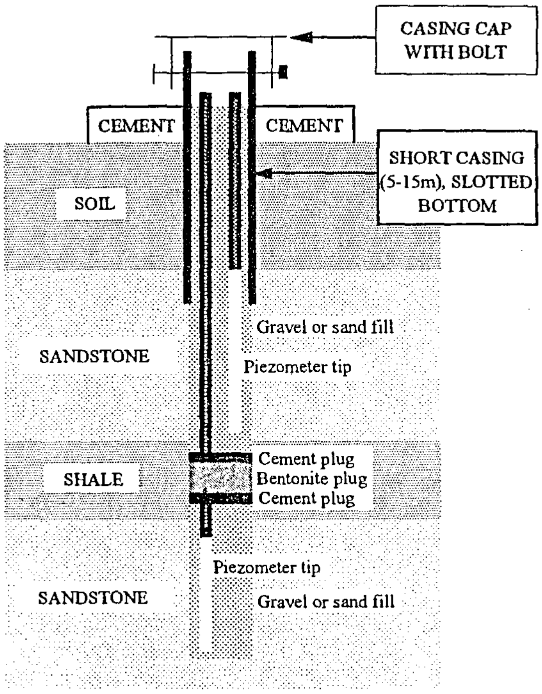
- Geological log.
- Water intersections (depth and quantity).



Typical installation required for monitoring boreholes into solid rock average depth 40 metres.



Typical installation for monitoring boreholes into the loose material such as sand, gravel, discards or rock spoil at the opencast mines



Typical installation of piezometer tubes in a monitoring borehole

- Construction information (depth of hole and casing - solid or perforated, borehole diameter, method drilled, date drilled).
- Use of borehole water, if not solely for monitoring; frequency of abstraction; abstraction rate and whether other water sources are readily available.
- Water quality (see chapter on chemical analyses).

E.3.3.1.1 Borehole type

- Boreholes must be drilled by a drilling technique that will not introduce pollution into the aquifer.
- Air-percussion drilling, without the addition of chemicals, is preferred. This allows the collection of rock chips and measurements of water yields, while drilling.
- In instances of difficult drilling, degradable chemicals may be introduced.
- Upon completion of the hole, water should be flushed through the drill stem or by bailing, to clean the sides of the hole and the inside of the casing.

E.3.3.1.2 Hole diameter

- Monitoring boreholes must be of a diameter that will allow easy access, for the purpose of water sampling and for lowering other test instruments.
- The diameter of the smallest submersible pump available in South Africa is 100 mm. Holes should therefore preferably have diameters larger than 100 mm. In smaller holes or during specialised sampling, pneumatic samplers or special small diameter pumps may be used. For newly drilled monitoring holes, a diameter of 165 mm is suggested.

E.3.3.1.3 Hole depth

- A monitoring hole must be such that the section of the groundwater most likely to be polluted first, is suitably penetrated to ensure the most realistic monitoring results.
- This implies that monitoring holes will at least extend through the weathered zone, the aquifer below and 5 m into the non water-yielding formation below. The latter is intended to act as a sump where material that falls down the hole will accumulate without affecting the performance of the monitoring system.
- The groundwater depth commonly ranges from 5 - 10 m below surface in the Olifants Catchment. Weathering can be recognised by brownish discolouration of the rock. Commonly, weathering extends between 10 - 15 m below surface. A depth of 40 - 50 m for holes, to monitor groundwater quality, should be sufficient, except in special instances.

E.3.3.1.4 Casing, screens and filters

- The materials for the various components in contact with water must be compatible with, and resistant against chemical attack by the water being monitored.
- Casing, screens and filters must allow easy access for monitoring purposes and may in no way block the flow of water through the borehole.
- The top of the casing in the monitoring borehole should rise between 30 - 40 cm above the general ground surface, to ensure that surface run-off does not flow into the borehole, during flood conditions.
- The bottom of the casing should extend only a couple of metres into the solid rock. Deeper casings will interfere with groundwater flow. If the water table is shallow and is likely to rise within the casing, the casing must be slotted to ensure lateral groundwater flow through it. A minimum slot density of 1% is required.
- The casing should preferable be of PVC, protected by a short steel casing at surface. In loose material such as sand, gravel, waste disposed of or rock spoil in the case of opencast mining, slotted PVC tubes, wrapped in biddem, may have to be installed to prevent monitoring holes from collapsing. A 110 mm Class 9 or better PVC casing is recommended. Alternatively, a wire mesh stainless steel casing may be used. In view of the price difference between the two types of installations, a stainless steel casing should only be used if absolutely necessary.
- A cap which can be bolted onto the casing, or which can only be removed by some other mechanical means, should be fitted. Locks are not recommended because they can easily be broken.
- The borehole number should be engraved onto the cap and casing.
- A concrete block around the top of the casing, to protect the casing and borehole, as well as to prevent surface pollution from flowing down the side of the casing, is essential. Recommended dimensions for the concrete block are 750 mm x 750 mm x 150 mm.

E.3.3.1.5 Piezometer tubes

- Piezometer tubes are sometimes installed into monitoring boreholes for various reasons. Piezometer tubes are access tubes that are usually installed to different horizons within a borehole, and sealed off from the other horizons by cement and bentonite clay. Piezometer tubes should be as large a diameter as possible, to allow easy access for water sampling. The minimum recommended diameter is 63 mm.

E.3.3.1.6 Borehole protection

- Destruction of a monitoring facility results in a break of the data sequence. Securing of monitoring boreholes should therefore be a high priority. It is recommended that monitoring boreholes should be fenced in and sufficient markings should be posted to prevent accidental damage. An example of the well-secured monitoring boreholes at Komati Power Station is included in Figure E.3.

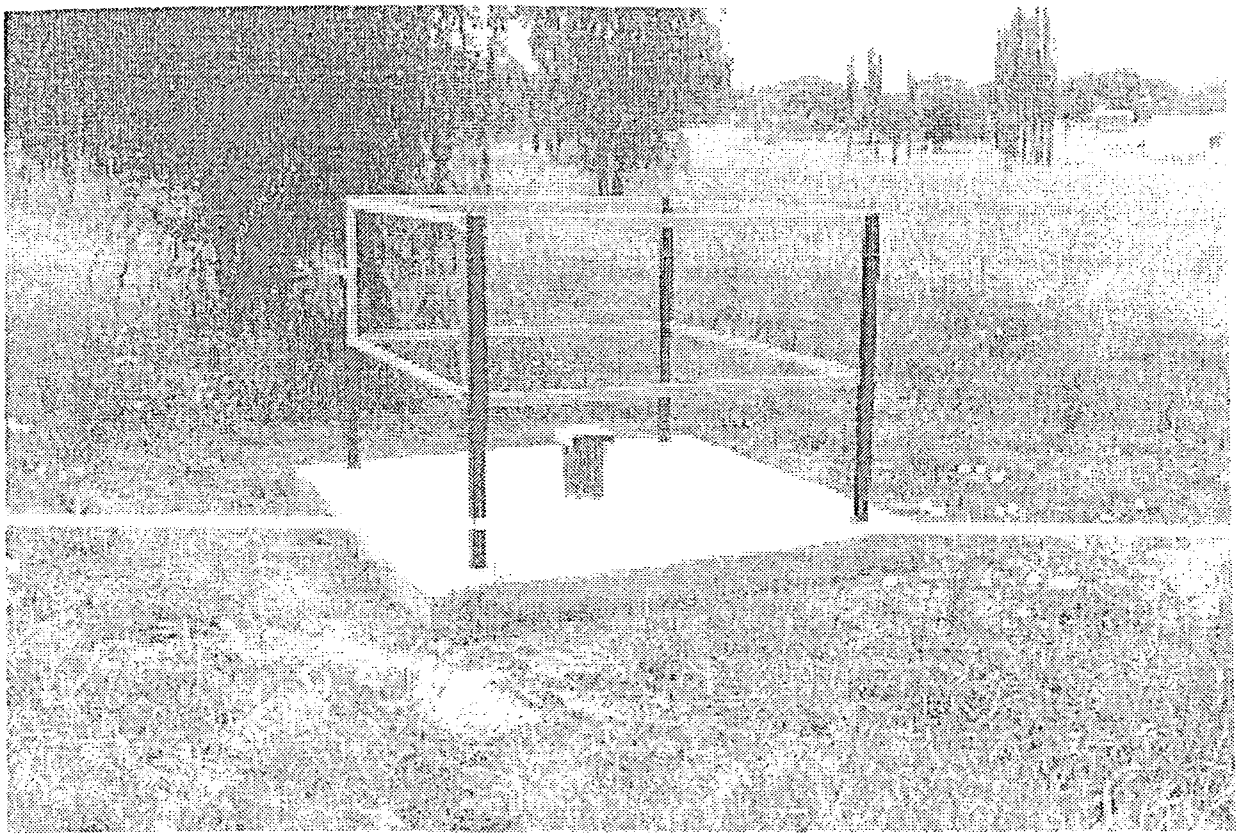


Figure E.3. Example of a secure structure around a monitoring borehole at Komati Power Station.

E.3.3.1.7 Groundwater levels

- Groundwater levels must be recorded to an accuracy of 10 cm or better, using an electrical contact tape, float mechanism or pressure transducer.
- Regional groundwater levels indicate the direction of groundwater flow. A change in the natural water-table gradient indicates that external forces are acting upon the aquifer. Typical such forces may be groundwater abstraction through nearby boreholes, mining or recharge from impoundments.

E.3.3.1.8 Pumping/packer tests

- Where considered necessary by the geohydrologist/design engineer, pumping or packer tests should be carried out on boreholes to obtain additional data on the geohydrological conditions.

E.3.3.2 Fountains, wells, dams, pans, stream and rivers

- Water sources around a waste management facility, within a radius as defined by the risk assessment, should be sampled and preserved for chemical analysis.
- Flow from fountains and in streams should be estimated. If pollution of fountains or streams occurs as a result of waste disposal, continuous recording of flow and water quality is advised.

E.3.3.3 Water sampling

A summary of sampling techniques is presented below. For additional information, the reader is referred to the manual on groundwater sampling by Weaver (1992). Sampling of water from unequipped boreholes is a tedious and difficult task. Two methodologies may be considered, namely stratified sampling or composite sampling.

Stratified sampling is done by removing a small volume of water from a specific depth within a borehole. A prerequisite for stratified sampling is not to disturb the water column unduly while taking the sample. The intention of stratified sampling is to determine the vertical distribution of water quality within a borehole, thus identifying horizons where pollution enters into a borehole.

Composite water sampling is usually done by pumping water from a borehole. Various types of pumps may be used. Two types that may be considered, because of their ease of installation, are the submersible pump and the bladder pump.

It is recommended that:

- Municipalities and industries performing their own sampling on a routine basis, should perform composite sampling.
- Groundwater consultants and others interested in understanding the mechanism, through which pollution may enter into a borehole, should perform stratified sampling.

E.3.3.3.1 Sample bottles and filters

Bottles of plastic, with a plastic cap and no liner within the cap, are required for most sampling exercises. Glass bottles are required if organic constituents are to be tested for. Bottles for sampling organic constituents and*E. coli* should be obtained from the analytical laboratory.

E.3.3.3.2 Sample Frequency

Where waste management permits are issued, the minimum sampling frequency should be prescribed. In other instances, the following guidelines may be adhered to:

Groundwater is a slow-moving medium and drastic changes in the groundwater composition are not normally encountered within days. The frequency with which water samples are to be taken from groundwater access points, is therefore a function of the sampling objectives. At any new groundwater sampling facility, initial sampling should be done at a frequency high enough to obtain statistically valid background information. For any long-term monitoring facility, three initial sampling exercises, all within 90 days, not more frequent than 14 days apart, are suggested. Depending on the variation amongst these values, future sampling may be planned. A three-monthly sampling frequency will mostly be sufficient.

Surface water chemistry may change within minutes, depending on controlled or uncontrolled discharges. The frequency for surface water quality monitoring should therefore range from several times a day to weekly. Continuous monitoring of the discharged flow volume and quality (by electrical conductivity method and pH) is required in instances where polluted water is disposed of into a public stream.

E.3.3.3.3 Sample Preservation

Where indicated in the table below, samples should be preserved. Sample preservation techniques for various tests vary significantly. Check with your analytical laboratory to ensure that the preservation method meets their analytical requirements. The required sample size also differs greatly from one laboratory to the next. The sample size should be sufficient to perform a duplicate analysis, if required.

VARIABLE	ACTION
Carbon dioxide Chloride - residual Oxygen pH	Analyse immediately
Elect. Conductivity Acidity Alkalinity BOD Colour Chromium (VI) Nitrite Silica Sulphate	No additives. Refrigerate. Analyse as soon as can reasonably be achieved
Boron Bromide Chloride Fluoride Potassium Sodium	Analyse when convenient
Hardness Metals (general)	Filter in field. Add HNO_3 to $pH < 2$
COD Grease and oil Nitrogen - NH_4 Nitrogen - NO_3 Nitrogen - Organic Phenols TOC	Add H_2SO_4 to $pH < 2$
Cyanide	Add NaOH to $pH > 12$
Sulphide	Add 4 drops 2N zinc acetate/100ml

No preservatives are required, if analysed within 6 h.

Samples should always be stored or transported at temperatures around 4 degrees centigrade.

E.3.3.4 Analytical Procedures

Analytical procedures differ greatly from laboratory to laboratory. These range from the standard so-called wet methods to the more sophisticated automated and computerised procedures.

Methods commonly used are:

- Titration against indicators, for pH and chloride.
- Specific ion electrode measurement, for pH and fluoride.
- Spectophotometric determination, for nitrate and COD.
- Turbidity measurement, for sulphate and turbidity.
- Conductivity measurement, for electrical conductivity.
- Ion chromatography (IC), for the anions.
- Atomic adsorption (AA) - flame and carbon furnace, for the cations.
- Inductively coupled plasma (ICP), for the cations.
- Gas chromatography (GC), for organic compounds.
- Mass spectrometer (MS), coupled with IC or ICP, for speciation of organic and inorganic substances.
- Other specialised and dedicated procedures, such as dissolved oxygen determinations.

Each of the above analytical procedures represents different levels of sophistication. The following is a brief discussion of the various procedures that are generally available.

The SABS has drawn up a list of unsophisticated analytical procedures. The variables that may be analysed for, as well as the numbers for the analytical procedures, are listed below. Details on these procedures are available from the SABS, upon request. It is recommended that anyone interested in setting up small-scale analytical facilities for water should consider, first of all, the use of these well-proven methodologies.

SABS Method Reference List

1. PHYSICAL

DETERMINAND	**METHOD REFERENCE**
Colour	SABS 198
Conductivity	SABS 1057
Dissolved solids @ 180°C	SABS 213
Dissolved solids @ 550°C	SM 2540 (E)
Suspended solids @ 105°C	SABS 1049
Total solids @ 105°C	SM 2540 (B)
Total solids @ 550°C	SM 2540 (E)
Taste and odour	SABS 241 3.3
Turbidity	SABS 197

2. INORGANIC (NON-METALLIC)

DETERMINAND METHOD REFERENCE

Acidity	SM 2310
Alkalinity	SM 2320
Boron	SABS 1053
Chloride	SABS 202
Chlorine (residual)	SABS 1052
Cyanide (qualitative)	SM 4500 - CN/K
Cyanide (total-quantitative)	SABS 204
Fluoride	SABS 205
Nitrogen:	
Ammonia - N	SABS 217
Kjeldahl - N	SM 4500 - N/B
Nitrate + nitrite - N	SABS 210
Nitrite - N	SM 4500 - NO_2
pH	
Phosphate:	
Ortho-phosphate	SABS 1055
Total phosphate	SM 4500 - P/B
Silica	SM 4500 - Si/D
Sulphate	SM 4500 - SO_4/E
Sulphide	SABS 1056

3. INORGANIC (METALLIC)

DETERMINAND	METHOD REFERENCE
Aluminium	SABS 1169
Antimony	ASTM D 3697
Arsenic	SABS 200
Cadmium	SABS 201
Calcium	SABS 216
Chromium	SABS 1054
Chromium (VI)	SABS 206
Cobalt	SABS 1170
Copper	SABS 203
Iron	SABS 207
Lead	SABS 208
Magnesium	SABS 1071
Manganese	SABS 209
Mercury	SABS 1059
Nickel	SABS 1171
Potassium	SM 3500 - K/D
Selenium	SABS 1058
Sodium	SABS 1050
Zinc	SABS 214

4. ORGANIC

DETERMINAND	METHOD REFERENCE
Chemical oxygen demand	SABS 1048
Oil and grease	SABS 1051
Oxygen absorbed	SABS 220
Phenolic compounds	SABS 211
Surfactants (anionic) - MBAS	SABS 199

Also available commercially and based on similar principles as the SABS procedures, are ready packed liquids, pellets and paper strips, containing exact amounts of reagent required for a measurement. These facilities are particularly useful for use in the field. The advantage of using these pre-packaged facilities, is that no standardisation or calibration is usually required, thus eliminating human error on this side.

The following practical tips on chemical analyses of water are offered for those who are not involved with this kind of work on a day to day basis:

E.3.3.4.1 pH

pH-measurement should preferably be done as the sample is taken. However, pH-probes are sensitive pieces of equipment and high turbidity in water may soon clog the probe, thus rendering it useless. Because of this clogging, average life of a glass pH-probe, used on an everyday basis in the field, is in the order of three months. Daily calibration is also required. In routine measurements, it is usually sufficient to distinguish between acid, neutral or alkaline water. For that purpose, the use of pH-paper in the field is recommended.

E.3.3.4.2 Electrical conductivity

Electrical conductivity is measured in milliSiemens per metre. These measurements are fast, cheap and an easy way of determining the approximate salt concentration in water. By multiplying the electrical conductivity value by a factor between 6 - 9, the total salt concentration may be approximated. The significant range for the multiplication factor is ascribed to the various conductance factors for different constituents in the water. Chloride typically has a high conductance (factor 6), while sulphate has a much lower conductance (factor 9).

E.3.3.4.3 Alkalinity and acidity

Alkalinity and acidity values may change rapidly, after groundwater samples have been withdrawn from confined aquifers. For accurate measurements, these variables should therefore be measured in the field, immediately after the sample has been taken.

Alkalinity and acidity determinations involve titration of the water, using sulphuric acid, to pH end points of 8,3 and 4,5 respectively. Adding pH-indicators to the water could assist detection of these end points. However, handling of strong acids in the field could be cumbersome and it is recommended that, for general applications, these determinations should rather be done within six hours, in a laboratory. Keep samples at 4 degrees centigrade after they have been taken.

E.3.3.4.4 Macro cations

Ca, Mg, Na and K usually occur in significant amounts in groundwater. Although wet techniques are available for their determination, AA or ICP is preferred. Wet determination of particularly Ca and Mg is based on the indirect EDTA method, through which calcium and magnesium hardness is measured. From this, the concentration of elemental Ca and Mg is then calculated. This can introduce errors and is not recommended.

E.3.3.4.5 Heavy metals

The term heavy metals refers to the metals and metalloids in the periodic table, with the exception of the macro cations, listed above. Since these elements usually occur in trace quantities, accurate and sophisticated analytical equipment is required. AA (carbon furnace) or ICP procedures are recommended. Modern carbon furnace equipment allows pre-concentration of elements through multiple injection and extremely low concentrations can be detected. A sequential ICP, coupled with a hydride generator, can also detect heavy metals at satisfactorily low levels.

E.3.3.4.6 Anions

A complete scan of the anions present in water (F, Cl, NO2, Br, NO3, PO4 and SO4) can be obtained within 10 minutes or less, using IC equipment. Detection limits range from less than 1 µg/l to more than 1 000 mg/l. Only two limitations may apply - organic compounds in the water may interfere with the peak reading for fluoride and a multipoint calibration is necessary for accurate work over a wide range of concentrations.

E.3.3.4.7 Organic compounds

Analysis for unknown organic compounds is very difficult, because of the vast range of constituents that may be present. The use of GC equipment for routine analysis of waste of unknown composition is therefore not feasible. At best, certain probable compounds that may be present within the waste can routinely be analysed for. MS enables specification of the compounds and intensive effort usually succeeds in recognition and quantification of the compound involved. Both GC and MS work are highly specialised and expensive procedures.

E.3.3.5 Parameters to be analysed for

The range of elements that may be analysed for within a waste environment, is very extensive. For the purpose of this document, variables to be analysed for are grouped under two headers:

- Comprehensive analysis
- Indicator analysis

E.3.3.5.1 Comprehensive analysis

A comprehensive analysis is recommended for all new monitoring sites. It is essential that accurate background levels for as wide a range of constituents as possible, be established at the outset. This will usually include a complete macro analysis, including trace elements that could reasonably be expected to be present within the environment tested.

E.3.3.5.2 Indicator analysis

In instances where samples are taken frequently, certain indicator variables may be identified and tested for. This will keep analytical costs to a minimum, but still provide enough information upon which further action can be initiated, if necessary. Depending on the type of waste handled, so-called 'pollution indicators' for each of these environments may be identified. Examples are the:

- General and special variables for discharge of industrial effluent into public streams, such as electrical conductivity (EC), Na, SO_4.
- The so-called sewage variables, such as COD, NH_4, PO_4.
- Hazardous waste disposal variables, such as Cr^{6+}, PCB, TCE, BHX.
- General waste disposal variables, such as COD, Cl, NO_3.
- Mine pollution variables, such as pH, EC, Mn, SO_4.
- Power station pollution variables, such as Na, SO_4.

Apart from this distinction, according to the type of waste environment, another dimension can be introduced by classifying contaminants into four classes, namely:

- Physical, such as pH, EC, alkalinity and acidity.
- Esthetical, such as iron, manganese, odour and taste.
- Other inorganic, such as high TDS and heavy metals.
- Other organic, such as toxic or carcinogenic compounds.

Analysis of physical, esthetical and inorganic variables is easily performed. Qualitative and quantitative analysis of organic constituents are extremely complex. In view of these complexities and the high cost associated with chemical analyses, the list of variables which needs to be analysed for, under these minimum monitoring requirements, should be kept as short as possible. Once pollution is detected, more elaborate tests may be performed. The following table suggests minimum monitoring requirements for chemical analyses:

E.3.3.6 Water/salt balances

- Water/salt balances: In instances where excess water is present and this water has to be discharged into public systems, water and salt balances should be calculated. At larger complexes such as a mine, power station or large industry, this usually implies water and salt balances for each of the contributing components, such as raw water intake, materials brought onto, removed from or disposed of on site, rain-water contribution and run-off.

E.4 MONITORING NETWORKS

The key to successful monitoring is the linking of point information into larger systems, referred to as monitoring networks.

In the table below, the same waste environments as that discussed previously, are listed. For each of these environments, borehole monitoring networks are suggested. The typical number of boreholes, their spacing and monitoring frequency are indicated.

Monitoring networks operate on a local, regional and national scale. The local monitoring network is intended for the single waste management facility, whereas regional monitoring relates to a combination of waste management facilities, such as those usually present at mines, power stations, other large industries and large municipalities. The DWA&F do monitoring on a national scale through catchment models and GIS applications.

Monitoring on all these levels is necessary. However, for the purpose of this document, i.e. minimum monitoring requirements at waste management facilities, emphasis will only be placed on the local monitoring network. In many ways though, monitoring on the regional scale merely constitutes combining the information obtained from local monitoring networks.

	At or near surface monitoring								Within waste or unsat. zone						Groundwater monitoring							
Monitoring Requirements → ↓ Waste environment	Rainfall	Evaporation	Run-off (volume, quality)	Water infiltration on waste	Toe seepage from waste	Soil cover on waste	Vegetation on waste or soil	Bioassaying	Pressure vacuum lysimeters	Gas samplers	Electrical conductivity probes	Leachate collectors	Temperature within waste	Special detectors	Special monitoring holes	Other holes	Groundwater levels	Groundwater chemistry	Borehole yield	Groundwater usage	Fountain seepage	Water balance
Mines – Reactive environment	d		d																			m
Slimes (Slurry)				m	m	y	y					m			yes	yes	m	3m	y	y	m	m
Ore discards			d	m	m	y	y					m			yes	yes	m	3m	y	y	m	m
Rock Discards (opencast)				m	m	y	y								yes	yes	m	3m	y	y	m	m
Rock discards (other)					m	y	y								yes	yes	y	y	y	y	y	y
Mine water (impoundment)				m	m										yes	yes	3m	3m	y	y	m	m
Mine water (discharged)			d					y														w
Mines – Inert environment	d		d												no	yes	y	y	y	y	y	m
Slimes (slurry)					m										no	yes	y	y	y	y	y	m
Rock discards					m										no	yes	y	y	y	y	y	y
Ore discards					m										no	yes	y	y	y	y	y	y
Mine water (discharged)			d																			d
Coal fired power stations	d		d												yes	yes	3m	3m	y	y	m	m
Coal stockpiling					m							m			yes	yes	3m	3m	y	y	m	m
Ash disposal (slurry)					m										yes	yes	3m	3m	y	y	m	m
Ash disposal (dry)				m			y								yes	yes	3m	3m	y	y	m	m
Dirty water systems					m										yes	yes	3m	3m	y	y	m	m
Water discharged			d					y														
General waste																						
Large (>500 t/d)	d		d		m	y	y	y		m		m			yes	yes	3m	3m	y	y	m	m
Medium (26 – 500 t/d)	d		d		m	y	y			m					yes	yes	y	3m	y	y	m	m
Small (1 – 25 t/d)					m										yes	yes	y	3m	y	y	m	
Informal (<1 t/d)					m										no	yes	y	3m	y	y	m	
Sewage																						
Unlined maturation ponds			d												yes	yes	y	3m	y	y	m	
Sludge			d																			
Hazardous waste	d	d	d	m	m	m	m	y	m	m		m		m	yes	yes	m	m	y	y	m	y
Waste irrigation	d		d	m	m	m	m	y							yes	yes	m	m	y	y	m	m
Agriculture (feed lots)	d		d												yes	yes	m	m	y	y	y	
Agriculture (diffuse sources)															no	yes	y	y	y	y		
Septic tanks and pit latrines															no	yes	y	y	y	y		
Underground storage tanks															yes	yes	m	m	m	m	m	
Urban development	d		m												no	yes	y	y	y	y		
Industries	Refer to specific waste above, such as general, hazardous, irrigation, impoundment																					
Radioactive waste	As specified by the CNS in collaboration with the DWA&F																					

Explanation of codes: d = daily monitoring; w = weekly monitoring; m = monthly monitoring; 3m = 3-monthly monitoring; y = yearly monitoring

Required Number of Monitoring Holes, Spacing and Monitoring Frequency

Environment	No. Holes	Distance From Waste	Monitoring Frequency
Mines - Reactive environment			Samples from boreholes every 3 months. Samples monthly from streams above and below mine. If pollution from mine occurs, install recorders in streams above and below mine and determine daily flows and EC. Sample farmers' boreholes 1-5 km radius, initially and when problems are expected.
Slimes (slurry)	1-3	50-200 m downstream	
Ore discards	2-5	50-500 m downstream and above	
Rock discards (opencast)	1/100 ha	Into water accumulations	
Rock discards (other)	1-3	50-200 m downstream	
Mine water (impoundment)	2-6	50-1000 m downstream	
Farmers' boreholes		Within 1-5 km from mine workings	
Mines - Inert environment			Monthly samples from streams above and below mine. If pollution from mine occurs, install recorders in streams above and below mine and determine daily flows and EC. Sample farmers' boreholes 1-2 km radius initially and when problems are expected.
Slimes (slurry)	0-1		
Rock discards	0-1		
Ore discards	0-1		
Mine water (discharged)	0-1		
Private boreholes		Within 1-2 km from mine	
Coal fired power stations			Samples from boreholes every 3 months. Samples monthly from streams above and below power station. If pollution occurs in streams, install recorders above and below power station and determine daily flows and EC. Sample farmers' boreholes 1-5 km radius initially and when problems are expected.
Coal stockpiling	2-3	50-500 m downstream	
Ash disposal (wet)	2-3	50-200 m downstream	
Ash disposal (dry)	2-3	50-200 m downstream	
Dirty water systems	2-3	50-500 m along stream	
Private boreholes		Within 1-5 km from power station	
General Waste			Samples from boreholes every 6 months or as specified in permit. Sample water-supply boreholes 1-5 km radius initially and when problems are expected. Sample surface water as specified in permit. Samples monthly from leachate, if any.
Large (>500 t/d)	3-6	20-200 m surrounding	
Medium (>25 - 500 t/d)	2-3	20-200 m downstream	
Small (>1 - 25 t/d)	1-2	20-100 m downstream	
Communal (<1 t/d)	0-1	20 m downstream	
Private boreholes		Within 1-5 km from dumps	
Sewage			Samples from boreholes every 3 months. Samples monthly from streams above and below sewage works.
Unlined maturation dams	1	20-50 m downstream	
Sludge	1		
Hazardous waste	5-10	10-200 m surrounding	Site specific elements at frequency recommended by impact study.
Waste irrigation	3-6	50-500 downstream and above	Borehole samples 3-monthly. Monthly stream samples above and below.
Agriculture - feed lots	2-3	50-200 m downstream	Borehole samples 3-monthly. Monthly stream samples above and below.
Agriculture - diffuse sources	0		Samples from existing water-supply boreholes when problems are expected.
Septic tanks and pit latrines	0		Samples from existing water-supply boreholes when problems are expected.
Urban development	0		Monthly EC in streams above and below development.

Note: The lowest number of special monitoring boreholes is the suggested minimum requirement. The minimum distance for the observation boreholes has been selected in order to protect the borehole from being destroyed during waste disposal. It is intended that the first monitoring borehole should be drilled as close as is feasibly possible to the waste management facility, not exceeding the minimum recommended distance.

E.5 DATA STORAGE, PROCESSING, INTERPRETATION AND REPORTING

E.5.1 INTRODUCTION

Data generated during groundwater quality monitoring are of two types, namely:

- Data generated once off;
- Data generated during repeated measurement.

Typical information which is generated once only, are details on monitoring hole construction, geology, borehole depth and yield from water intersections in the hole.

Repetitious measurements, such as groundwater levels, water chemistry, water abstraction from boreholes, waste composition, tons of waste disposed of, rainfall, surface run-off and rehabilitation progress constitute by far the majority of the observations to be made.

E.5.2 DATA STORAGE

The DWA&F has acquired computerised data storage facilities, into which all information on geohydrological information and waste disposal facilities may be entered. The software, HydroCom and WasteBase, has been developed especially to meet South African requirements.

To date, many of the collieries and all of the power stations have installed HydroCom data bases for the capture and processing of their monitoring data. The information generated during this investigation has also been stored onto HydroCom.

The DWA&F are currently using WasteBase for processing of waste disposal permits.

E.6 SEQUENTIAL STEPS IN THE DESIGN OF MONITORING SYSTEMS

The design of monitoring facilities for waste facilities, by geohydrologists, should follow a certain sequence of events. The following is a summary of actions in this document:

(i) Obtain information from the municipality or industry on disposal practices and volumes.

(ii) Survey of available information on the topography, stream flow, fountains, dams, geology, existing boreholes, wells and excavations. Sampling of surface and groundwater for chemical analyses to determine the spread of pollutants, if any, at existing points.

(iii) Survey of other human activities in the area, which could be affected by the disposal of waste.

(iv) Perform risk assessment and decide on level of impact study and monitoring facilities required.

(v) Perform geophysical tests to locate groundwater barriers. Drilling of boreholes at points as are determined by (i), (ii) and (iii). Log geological and geohydrological information from boreholes. Perform tests in boreholes, if necessary, such as hydraulic conductivity and water quality profiling. Study rain water penetration characteristics into the waste disposed of. Install, if necessary, early warning devices underneath new disposal sites.

(vi) Perform stratified sampling from holes by pneumatic sampler, followed by composite sampling. Analyse for elements typically found within the disposal environment being tested.

(vi) Perform stratified sampling from holes by pneumatic sampler, followed by composite sampling. Analyse for elements typically found within the disposal environment being tested.

(vii) Enter all information into a computeised database for processing and interpretation. Extract tables and graphs for final report.

(viii) Present report, recommend methods and frequency of sampling. Transfer the database to a PC at the municipality or industry for future in-house use. Supply equipment to sample water from boreholes.

(ix) Apply for waste management permit, if so required.

(x) Train personnel at the municipality or industry in the use of the database, the sampling equipment and in the interpretation of the data. Provide facilities for the industry to report to the DWA&F in terms of their permit conditions.

E.7 SUGGESTED READING MATTER

The following is a list of reading matter, suggested for individuals or companies that wish to broaden their knowledge on water quality monitoring and the environment. The numbers below refer to the different chapters in this document, on which the literature has a bearing.

(E.7.1) Background Information

Council for the Environment, 1989. Integrated environmental management in South Africa. *ISBN 0-621-12496-6. Council for the Environment, Private Bag X447, Pretoria 0001*

Council for Industrial Research (CSIR), 1991. The situation of waste management and pollution control in South Africa. *Report to the Department of Environment Affairs by the CSIR programme for the Environment, Pretoria. Report number CPE 1/91. CSIR, P.O. Box 395, Pretoria 0001.*

Department of Water Affairs and Forestry, 1991. Water quality management policies and strategies in the RSA. *Department of Water Affairs and Forestry, Private Bag X313, Pretoria 0001.*

Department of Water Affairs and Forestry, 1994. Minimum requirements for waste disposal sites. Waste Management Series No. 1.*Department of Water Affairs and Forestry, Private Bag X313, Pretoria 0001.*

Department of Water Affairs and Forestry, 1994. Minimum requirements for the management and handling of hazardous waste. Waste Management Series No. 2.*Department of Water Affairs and Forestry, Private Bag X313, Pretoria 0001.*

Environmental Protection Agency, 1984. Groundwater protection strategy. *EPA, Office of Groundwater Protection, Washington, DC 20460.*

Winter, G., 1988. Business and the environment.*McGraw-Hill, ISBN 3-89028-223-7.*

Environmental Conservation Act, No. 73 of 1989.

Minerals Act, No. 50 of 1991.

Water Act, No. 54 of 1956, Sections 21 - 24, 26. Regulations R287/4989/20.2.1976.

(E.7.2) Mission, policies and strategies for groundwater quality management

Department of Water Affairs and Forestry, November 1992. Groundwater quality management policies and strategies for South Africa.

(E.7.3) South African aquifers

Department of Water Affairs and Forestry - *Numerous geohydrological reports are available upon request, for specific aquifers and municipalities using groundwater. Directorate of Geohydrology, DWA&F, Private Bag X313, Pretoria 0001.*

Kok, T.S., 1991. The potential risk of groundwater pollution by waste disposal. *Biennial Groundwater Convention of the Groundwater Division of the Geological Society of South Africa and the Borehole Water Association of Southern Africa.*

National Groundwater Data Base for South Africa - *Information for more than 100 000 boreholes is available from the Directorate of Geohydrology, DWA&F, Private Bag X313, Pretoria 0001.*

Water Research Commission - *Many groundwater research reports, usually of a specialised nature, are available. WRC, P.O. Box 824, Pretoria 0001.*

(E.7.4) Risk assessment procedures

Environmental Protection Agency, 1985. DRASTIC - A standardsed system for evaluating groundwater pollution potential using geohydrological settings*EPA-report 600/2-85/018.*

Environmental Protection Agency, 1991. WHPA, a modular semi-analytical model for the delineation of wellhead protection areas, Version 2.0.*EPA, Office of Groundwater Protection, Washington, DC 20460.*

Environmental Protection Agency, 1991. VIRALT, a modular semi-analytical and numerical model for simulating viral transport in groundwater.*EPA, Office of Drinking Water, Washington, DC 20760.*

Parsons, R and Jolly, J., 1994. A systematic method for evaluating site suitability for waste disposal based on geohydrological criteria. *Report in preparation for the Water Research Commission.*

(E.7.5) Facilities for monitoring water quality

Borehole Water Association. Know your borehole. *Borehole Water Association of Southern Africa, P.O. Box 1338, Johannesburg 2000.*

Environmental Protection Agency, 1985. Groundwater monitoring strategy. *EPA, Office of Groundwater Protection, Washington, DC 20460.*

Everett, L.G., 1984. Groundwater monitoring: Guidelines and implementing a groundwater quality monitoring programme. *Genium Publishing Corporation, N.Y.*

Everett, L.G. 1985. Groundwater monitoring: Handbook for coal and oil shale development. *Developments in Water Science, No. 24, Elsevier, N.Y.*

Groundwater - Guideline for Boreholes. *Available from the Department of Water Affairs, Private Bag X313, Pretoria 0001.*

Weaver, J.M.C., 1992. Groundwater sampling. *Research report to the Water Research Commission. ISBN 1 874858 44 6.*

(E.7.6) Indicator variables and chemical analyses

American Public Health Association, APHA, AWWA (1985). Standard methods for the examination of water and waste water. *16th Edition. ISBN 0-87553-131-8.*